MODELLING THE IMPACT OF ANTHROPOGENIC DISTURBANCE ON WATER QUALITY IN THE COASTAL ZONE OF EASTERN GEORGIAN BAY, LAKE HURON

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MODELLING THE IMPACT OF ANTHROPOGENIC DISTURBANCE ON WATER QUALITY IN THE COASTAL ZONE OF EASTERN GEORGIAN BAY, LAKE HURON

By

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A Thesis

Submitted to the School of Graduate Studies

In Partial Fulfillment of the Requirements

For the Degree

Masters of Science

McMaster University

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McMaster University Master of Science (2016) Hamilton, Ontario (Biology)

TITLE: Modelling the Impact of Anthropogenic Disturbance on Water Quality in the Coastal Zone of Eastern Georgian Bay, Lake Huron

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SUPERVISOR: Dr. Patricia Chow-Fraser

NUMBER OF PAGES: xvi, 128

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PREFACE

This Master of Science thesis is made up of two separate chapters. These chapters have been put into context by a general introduction and conclusion. The first chapter is formatted as a manuscript to be submitted for publication in the Canadian Journal of Fisheries and Aquatic Sciences. The second chapter is formatted as a manuscript that will be submitted to management at the Township of Georgian Bay.

As the first author for both chapters, under the supervision of Dr. Patricia Chow-Fraser, I was responsible for all data analysis and preparation of the two manuscripts. I was also responsible for the laboratory processing of water samples from 2014 – 2016 and the collection of field samples from 2015 – 2016 for both projects. The remainder of the data that is not otherwise referenced in this thesis was given to me by my supervisor, Dr. Patricia Chow-Fraser.

Campbell, S. and Chow-Fraser, P. 2016. Models to predict total phosphorus in coastal embayments of eastern Georgian Bay, Lake Huron.

Campbell, S. 2016. Examining the general impact that cottage development has on coastal wetland water quality in the Township of Georgian Bay.

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GENERAL ABSTRACT

Though the water in eastern Georgian Bay is oligotrophic, some of the coastal embayments and wetlands have begun to show signs of water quality impairment that is thought to be related to human development along the shoreline. The primary objective of my thesis is to provide environmental agencies with the resources to effectively manage water quality in the coastal zone of eastern Georgian Bay. First, I evaluate the ability of the Lakeshore Capacity Model (LCM), developed for inland Precambrian Shield lakes, to predict the trophic status of coastal embayments. Finding that the LCM does not accurately predict trophic status, I develop the Anthro-geomorphic Model (AGM), which uses the level of human development and the degree of mixing between the embayment and open waters of Georgian Bay to predict embayment trophic status. Second, I explore the spatial association between densities of building, dock and road development and Water Quality Index (WQI) scores, an index designed to evaluate wetland condition, for wetlands in the Township of Georgian Bay. I found an inverse relationship between WQI scores and the density of these stressor variables inside wetland catchments, which indicates that these stressors have a negative impact on wetland water quality. I then created a series of mapping products that present building, dock and road densities, along with WQI scores for 61 wetlands in the Township of Georgian Bay, to determine how wetland water quality is spatially associated with densities of these stressor variables. I found that regions with high densities of building, dock and road development were associated with wetlands of lower quality, whereas wetlands in

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areas that had low densities of development were of higher quality. I used this information to identify areas of conservation priority for management in the Township of Georgian Bay. The results from this thesis will provide environmental managers with resources to protect the valuable coastal waters of eastern Georgian Bay.

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ACKNOWLEDGEMENTS

I would like to thank all of the people who offered me their support during my time at McMaster University. First and foremost, I would like to thank my supervisor, Dr. Pat Chow-Fraser, for her mentorship and guidance throughout the process of my Master's Degree. Pat, thank you for taking the chance of hiring me on as a research assistant to start work on these projects. Thank you to Dr. Mike Waddington for your advice and enthusiasm during committee meetings and our lab get-togethers with the McMaster University Ecohydrology Group. Thank you to Dr. Stone for sitting as chair on my defence committee. I would like to thank each of my fellow students in the PCF Lab; Chantel, Dan, James, Julia, Lindsey, Morgan and Nick for your advice during lab meetings and for being wonderful people to spend the last two years with. Thank you to all of the undergraduate students who assisted me in the lab and in the field; these projects would not have been a reality without your hard work. For the generous hospitality they have shown me, thank you to Mary Muter and Jean DeMarco for the use of their lovely cottages. I always looked forward to my time in the field knowing that we would have such great company and a comfortable place to stay.

I would personally like to thank Meredith for the never ending love and support she has shown me when I have needed it most. Thank you to my parents Mike and Jean for your love and encouragement for me to do what makes me happy.

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Finally, thank you to my brother Ian and all of my good friends for your encouragement over the past few years.

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LIST OF ABBREVIATIONS

Abbreviation	Definition	Unit
WATER CHEMISTRY		
[TP]	Total phosphorus	$\mu g \bullet L^{-1}$
[TN]	Total nitrogen	$mg \bullet L^{-1}$
(TEMP)	Temperature	°C
(DO)	Dissolved oxygen	$mg \bullet L^{-1}$
MORPHOMETRY		
A_o	Surface area	m^2
V	Lake volume	m^3
L	Length	m
B	Breadth	m
Z_{mean}	Mean depth	m
Z_{max}	Maximum depth	m
P	Perimeter	m
Z_r	Relative depth	m
D_L	Shoreline development	--
D_V	Volume development	--
A_d	Catchment or watershed area	km^2 or m^2
MODELLING		
L_T	Total areal loading rate	$mg \bullet m^{-2} \bullet y^{-1}$
R_P	Retention coefficient, fraction not lost via outflow	--
q_s	Areal water load	--
Q	Lake outlet discharge, lake outflow volume	$m^3 \bullet y^{-1}$
v	Sedimentation velocity	$m \bullet y^{-1}$
$TP_{Outflow}$	Total phosphorus at outflow	$\mu g \bullet L^{-1}$
TP_{Mean}	Seasonal mean total phosphorus	$\mu g \bullet L^{-1}$
TP_D	Total phosphorus to downstream lake	$kg \bullet y^{-1}$

GENERAL INTRODUCTION

The coastal zones of the Great Lakes are ecologically important regions containing a variety of different habitats that are home to a diverse array of plants and animals (Sierzen et al. 2012). While there is no formal definition of a coastal zone, it is generally understood as the junction where land, streams or wetlands meet open waters. It is also in this area that human activities often interact with and affect the natural environment, and the coastal zones of the Great Lakes are no exception; most human development in the Great Lakes basin is concentrated near coastal waters (Lawrence 1997; Mayer et al. 2004).

In addition to the ecological value of their coastal zones, the Great Lakes are important in terms of the natural resources they provide to sustain the populations of both Canada and the United States, as well as the economic benefits they offer to both countries. Over 30 million residents in Canada and the United States receive their drinking water from the Great Lakes (Poghosyan et al. 2014). It is also estimated that the Great Lakes recreational fishery is worth approximately 7.5 billion dollars representing an important component of the economies of both Canada and the United States (Krantzberg and Boer 2006). The Great Lakes coastal zones also offer a setting for recreational activities, such as cottaging, boating and recreational fishing, that sustain a thriving tourism economy (Vaccaro and Read 2011). For these reasons, it is important to maintain good ecological health in the Great Lakes basin, and it is concerning that there are longstanding issues of water quality impairment in many of the Great Lakes coastal zones (Michalak et al. 2013; Arhonditsis et al. 2016; Carmichael and Boyer 2016).

Threats to the Great Lakes Coastal Zone

High levels of human development are seen in the coastal areas of the Great Lakes in both the United States and Canada (Lawrence et al. 1995; Mayer et al. 2004). Human activities such as land alteration for agriculture and urbanization, deforestation and shoreline modification have had a negative impact on the water quality of the Great Lakes (Kerr et al. 2016). The severity of this impact is variable among the Great lakes, with lakes Erie and Ontario being most affected, followed by lakes Michigan, Huron and Superior (Allen et al. 2013). Water pollution has been so severe in some regions that there have been reoccurring episodes of toxic cyanobacteria blooms that have impacted access to safe drinking water in some cities (Carmichael and Boyer 2016). The impact of human development on water quality has not been limited to the open waters of the Great Lakes; certain coastal embayments (e.g. Bay of Quinte, Green Bay) and wetlands (e.g. Cootes Paradise Marsh) within the Great Lakes show particularly severe symptoms of water quality impairment (Chow-Fraser 2006, Arhonditsis et al. 2016; Lin et al. 2016; Kerr et al. 2016).

Coastal wetlands provide a number of important ecosystem services for both wildlife and humans. Regrettably, two-thirds of Great Lakes coastal wetlands have been lost since European settlement (Ball et al. 2003; Sierszen et al. 2012). The loss of coastal wetland habitat can be attributed to drainage for agricultural production, infilling for urban development and alteration for recreational developments like marinas and cottages (Dodge and Kavetsky 1995; Mayer et al. 2004). Of the coastal wetlands that remain in the southern Great Lakes, the majority are in a degraded state as a result of human disturbance within their watersheds (Crosbie and Chow-Fraser 1999; Chow-Fraser 2006). The loss of coastal wetlands directly impacts ecologically and economically significant species that depend on wetlands at critical stages during their lifecycles (Sierszen et al. 2012). It is important to note that, though wetlands in all of the Great Lakes have suffered due to human disturbance, each lake is subject to a unique set of stressors as a result of different patterns of human development that have occurred within their respective drainage basins.

Georgian Bay: A Unique Case

Georgian Bay, the most eastern arm of Lake Huron, is the largest bay in the Laurentian Great Lakes, with a surface area of approximately 15 000 km^2; this is almost equivalent to the surface area of Lake Ontario (Sly and Munawar 1988). Although Georgian Bay is connected to Lake Huron, it differs from the main basin of Lake Huron in its geological and morphological characteristics, as well as with respect to the human stressors that impact its waters (DeCatanzaro and Chow-Fraser 2011; Fracz and Chow-Fraser 2013). The shores of Georgian Bay are characterized by two distinct geological features: the Precambrian Shield which makes up the convoluted shoreline to the east and north, and the Niagara

Escarpment that forms the limestone cliffs in the west. The eastern shore has been shaped through past glaciations (McCarthy and McAndrews 2012) and differential weathering processes to create a variable coastline with many finger-like indentations that terminate in wetlands and embayments. Georgian Bay is a naturally oligotrophic body of water, primarily because the soil conditions throughout most of its watersheds are not suitable for intensive agricultural development, unlike the soil surrounding Lake Huron and the lower Great Lakes (Weiler 1988). As a result, Georgian Bay has undergone different patterns of development as compared with these other bodies of water (Weiler 1988), having minimal levels of agricultural and urban development and no large scale industrial manufacturing activities within its watershed (Sly and Munawar 1988).

The eastern shore of Georgian Bay is a popular summer vacation destination and is host to a thriving tourism economy. The human development that exists in this region consists primarily of recreational cottages, resorts and marinas, which have all become more popular (Georgian Bay Township 2014). In some areas along the eastern shore, where cottage and marina development is concentrated, some coastal embayments and wetlands have begun to show signs of water quality impairment (Gartner lee Limited 2008; Chiandet and Sherman 2014; Chow-Fraser and Croft 2015) that is thought to be related to such development (Schiefer et al. 2006). Since the stressors impacting Georgian Bay differ from those impacting the lower Great lakes and Lake Huron, appropriate management tools need to be identified for this body of water in order to assist environmental agencies in ensuring that coastal development occurs in a sustainable manner.

Need for Effective Management Tools

Coastal water quality impairment in eastern Georgian Bay is an issue of urgent concern. This is not only because the natural features of this region are being degraded, but because the entire local economy of eastern Georgian Bay is reliant on good water quality to sustain the recreational cottage, boating and tourism industries. Cottage development along the eastern shore of Georgian Bay is likely to continue due to the growing popularity of this region as a tourist destination (Georgian Bay Township 2014) and its proximity to the large urban areas of southern Ontario. Since few resources currently exist that provide environmental managers with the scientific basis for choosing the most effective tools to manage development in the unique coastal zones of eastern Georgian Bay; these resources are desperately needed.

Thesis Objectives

The overarching objective of this thesis is to develop and evaluate management tools to assess water quality in coastal embayments and wetlands of eastern Georgian Bay. In my first chapter, I evaluate the ability of the published Ontario Lakeshore Capacity Model (LCM), developed for inland Precambrian Shield Lakes of the Muskoka region, to predict trophic status in coastal embayments. Finding that the LCM is not able to reliably predict embayment trophic status, I then develop the Anthro-geomorphic Model. This alternative model uses building density and basin morphometry to predict trophic status in embayments. In addition, I investigate whether the log-linear relationship between chlorophyll-a and total phosphorus in embayments differs significantly from this relationship seen in the context of other systems, including inland Precambrian Shield lakes and the near-shore zone of Georgian Bay. This study will provide the scientific basis for choosing the best management practices to protect the coastal embayments of eastern Georgian Bay.

In my second chapter, I explore the spatial association between densities of building, dock and road development and coastal wetland water quality in the Township of Georgian Bay. I create a series of mapping products that present building, dock and road densities, along with Water Quality Index (WQI) scores, to determine how wetland quality is spatially associated with densities of these stressor variables. This analysis allowed me to identify areas where coastal wetlands are at risk of becoming degraded by these stressors, while also identifying areas that have high quality wetlands that should be preserved. The information from this study should be used by township managers to make informed decisions regarding coastal development in order to preserve the pristine water quality conditions currently found in coastal wetlands of this region.

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CHAPTER 1: Models to predict total phosphorus in coastal embayments of eastern Georgian Bay, Lake Huron

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Key words: Georgian Bay, coastal embayment, nutrient model, TP-CHL,

Laurentian Great Lakes

Abstract

The rocky shoreline of eastern Georgian Bay is marked by thousands of islands and shoals, with many indentations that terminate in embayments and coastal wetlands. Though the water in Georgian Bay is oligotrophic, a few of the coastal embayments have had recurring episodes of nuisance algal blooms and hypolimnetic oxygen depletion that may be related to excessive human development along the shoreline. Here, we evaluate the ability of the Lakeshore Capacity Model (LCM), developed for inland Precambrian Shield lakes, to predict the trophic status of coastal embayments that vary widely with respect to cottage and marina development. The LCM could only be applied to eight of ten embayments included in this study due to the large size and complexity of two watersheds; it estimated mean seasonal total phosphorus ([TP]) concentrations within ±20% of measured values for only three of the eight embayments and none of the estimates were improved by accounting for internal phosphorus loading from anoxic hypolimnia. We developed an alternate model, the Anthro-geomorphic Model (AGM), which depends only on the level of human development (number of primary residential buildings per shoreline length) and the degree of mixing between the embayment and open waters of Georgian Bay (an index that accounts for embayment morphology and distance to open Georgian Bay). The AGM estimated [TP] within ±20% of measured values for all ten sites. Compared with inland lakes and the near-shore zone of Georgian Bay, there was significantly higher chlorophyll-a concentrations per unit [TP] in embayments, suggesting that other factors must be

considered when assessing their trophic states. We recommend using the AGM instead of the LCM, because the former produced more accurate estimates and could be applied to all ten embayments without exception.

Introduction

Georgian Bay, the eastern arm of Lake Huron, is the largest bay in the Laurentian Great Lakes, with a maximum depth of 171 m (McCarthy and McAndrews 2012) and a surface area of 15 111 km^2 (Sly and Munawar 1988), which is almost 80% of the surface area of Lake Ontario (18 960 km^2; Chapra et al. 2012). It contains the largest chain of freshwater islands in the world (more than 30 000), known as the Georgian Bay Archipelago. In 2004, the Georgian Bay Archipelago was recognized as a UNESCO World Biosphere Reserve in Canada, a designation that acknowledges the tremendous native biodiversity with more than 1000 distinct habitat types that support rare and at-risk species of birds, reptiles, plants and mammals. Georgian Bay is rimmed by the Paleozoic limestone cliffs of the Niagara Escarpment to the west and the granitic rock of the Precambrian Shield to the east. Although both of these geological features influence the water chemistry of the bay, Georgian Bay owes its oligotrophic status primarily to the lack of agricultural development in the watershed of eastern Georgian Bay (Weiler 1988). As of 2012, the total phosphorus concentration ([TP]) was 4.23 μg•L^{-1} and the total nitrogen concentration ([TN]) was 0.327 mg•L^{-1}, levels only slightly higher than those of Lake Superior (Dove and Chapra 2015). Good water quality is essential to support the large number of cottagers and recreational boaters that rely on water not only for consumption but also for aesthetic enjoyment. In addition, a highly productive recreational fishery of walleye, northern pike and bass, as well as a world-class trophy fishery for muskellunge (Weller et al. 2016) also depend on good water quality in the near-shore of Georgian Bay.

Despite the overall oligotrophic status of Georgian Bay, some near-shore areas of eastern Georgian Bay are showing signs of water-quality impairment. Diep et al. (2007) conducted a survey of the water-quality conditions of eastern and northern Georgian Bay between 2003 and 2005, and found that all of the near-shore areas had very low levels of nutrients and algal biomass except in a few coastal embayments in the more populated areas, especially where embayments are somewhat separated from the open waters of Georgian Bay by a network of channels. In these areas, there have been reports of recurring episodes of hypolimnetic oxygen depletion, occurrences of nuisance algal blooms and proliferation of benthic algae (Schiefer et al. 2006; Chiandet and Sherman 2014). Such development of water-quality impairment could be attributed to a number of factors, including the effect of recent low water levels on the extent of water exchange between embayments and the main part of Georgian Bay, as well as nutrient enrichment from cottage development.

There are currently no management tools available to environmental managers to assess the relative impact of cottages on water quality of coastal embayments. Lake managers in the nearby Township of Muskoka developed the Lakeshore Capacity Model (LCM) (Lakeshore Capacity Assessment Handbook, Ministries of Ontario, 2010) to separate natural from anthropogenic phosphorus (P) inputs into lakes, so that the impact of existing and proposed cottage capacity could be modelled (Paterson et al. 2006). This model is a modified version of the mass-balance model of Dillon and Rigler (1975), and was calibrated for application in inland central Ontario Lakes to predict seasonal mean [TP] (Hutchinson et al. 1991) and then specifically for lakes on the Precambrian Shield (Paterson et al. 2006). Gartner Lee Limited (2008a) applied the LCM to one coastal embayment, Sturgeon Bay, which had exhibited symptoms of eutrophication for at least two decades (Schiefer 2003; Gartner Lee Limited 2008b). They found that the LCM performed well, as long as they also accounted for internal loading of P from the anoxic sediments.

There are good reasons for expecting the LCM developed for Muskoka lakes to apply to coastal embayments. First, inland lakes and embayments have similar types of cottage and marina development along the shoreline. Second, they both have thin soil structure and fractured bedrock that would prevent aging septic systems from performing properly and leaking into the surrounding water (Dillon et al. 1994; Dillon and Molot 1996; Robertson et al. 1998; Joy 2013). There are also reasons why the LCM may not apply to the coastal embayments. First, embayments are considerably shallower than Muskoka lakes, and some are dystrophic; the Lakeshore Capacity Assessment Handbook specifically warns against applying the LCM to shallow (Z_{mean} <5m) water bodies with high concentrations of dissolved organic carbon (Ministries of Ontario 2010). More importantly, water in coastal embayments can become mixed with the oligotrophic waters of Georgian Bay during the ice-free season, whereas water in inland lakes do not mix with water further downstream. Hence, without a proper study, there is no scientific basis for assuming that the LCM can be used to accurately predict [TP] in coastal embayments.

One of the primary goals of this study is to validate the general applicability of the LCM for coastal embayments of eastern Georgian Bay. To achieve this, we will apply the LCM (with and without accounting for internal P load) to embayments that vary with respect to cottage and marina development along their coasts. Consistent with existing practice, deviations greater than ±20% between measured and estimated [TP] concentrations will be interpreted as failed applications of the model. A second objective is to develop an alternative management model that accounts for the hydrological connection between embayments and Georgian Bay, since the degree to which water is diluted by oligotrophic Georgian Bay water would affect the nutrient status of the embayment in question. We will investigate the effect of other factors considered in the LCM such as basin morphometry, anthropogenic disturbance and landscape features, which are known to affect external and internal P loading and overall water chemistry. Lastly, we will determine if the log-linear relationship between chlorophyll-a and total phosphorus derived for embayments differs significantly from those derived for other systems, including inland Precambrian Shield lakes and the near-shore zone of Georgian Bay. This study will provide a scientific basis for choosing the best management models to protect the water quality of the ecologically important and economically valuable coastal embayments of eastern Georgian Bay.

Site Descriptions

Georgian Bay is located within the Great Lakes-St. Lawrence Forest Section (Zone L.4d) in Canada that extends from the Paleozoic sedimentary rock of the Bruce Peninsula in the south to the Sudbury-North Bay Forest zone in the north (Rowe 1972). As a result of the prevailing westerly wind patterns experienced in the Great Lakes basin, the coastal region of Georgian Bay is subject to larger amounts of precipitation than regions located further inland (McCarthy and McAndrews 2012). These conditions allow this region to host a variety of mesic tree species including sugar maple (*Acer saccharum)*, american beech *(Fagus grandifolia)*, yellow birch *(Betula alleghaniensis)*, eastern hemlock *(Tsuga canadensis)*, eastern white pine *(Pinus strobus)*, northern red oak *(Quercus rubra)* and white ash *(Fraxinus americana)*, that dominate the well-drained upland areas, and red maple *(Acer rubrum)*, black ash *(Fraxinus nigra)* and eastern white cedar *(Thuja occidentalis)* that dominate low lying swampy areas (Rowe 1972). Near the thin-soiled rocky coastline, however, there are scrubby stands of white pine *(Pinus strobus)*, trembling aspen *(Populus tremuloides)*, northern red oak *(Quercus rubra)* and white birch *(Betula papyrifera)* (Rowe 1972). The forested landscapes of inland Georgian Bay have been relatively void of human disturbance over the past century as this region has a low population density, minimal industrial development and thin acidic soil conditions that are not suitable for intensive agriculture (Weiler 1988).

The dominant geological feature in eastern Georgian Bay is the Precambrian Shield, which consists primarily of granitic bedrock that is less erodible than the limestone bedrock of the western shore. This granite bedrock has been shaped through past glaciations (McCarthy and McAndrews 2012) and differential weathering processes to create a variable coastline with many finger-like indentations that terminate in wetlands and embayments. These embayments are unique coastal systems linked to Georgian Bay through narrow connecting channels. This configuration can result in limited flushing and mixing patterns that would otherwise be common in more exposed bays. Since limited water exchange occurs between these embayments and the oligotrophic waters of Georgian Bay, much of the external nutrient load entering these systems tends to stay within the embayment. These processes result in the embayments being sheltered from Georgian Bay and functioning more like enclosed lakes than exposed coastal embayments. This is significant because when these systems experience nutrient loading, they are more susceptible to developing symptoms of eutrophication than the more exposed coastal bays of Georgian Bay or Lake Huron (Wells and Sealock 2009).

Ten embayments and one inland lake were sampled monthly from May to September from 2012 to 2015 (Table 1.1; Figures 1.1 & 1.2). Except for Woods Bay, which is too shallow to develop thermal stratification, all other sites are dimictic, becoming thermally stratified during the summer (Figure 1.3). Generally, thermal stratification becomes well established by July, although, the depth of the thermocline during the stratification season may vary among embayments, with some having relatively shallow thermocline (e.g. Cognashene Lake and North Bay) and others having a deeper thermocline (e.g. Longuissa Bay, Musquash Bay) (Figure 1.3). In spite of this, each embayment experiences a deepening of the thermocline as the season progresses, which is a common feature of water bodies in North America.

Since the LCM was developed for inland lakes, we included a deep (Z_{max}=60.80 m; Table 1.2), oligotrophic (seasonal mean [TP] of 6.56 $\mu g \bullet L^{-1}$), inland lake located in the Moon River watershed that eventually drains into Georgian Bay (Chow-Fraser 2005; Figure 1.1 and 1.2K; Table 1.1) as one of the sites we sampled to ensure we correctly applied the LCM. Blackstone Lake has a relatively low cottage density and supports both cold-water and cool-water fisheries (Schiefer 2008). We sampled the same suite of parameters in Blackstone Lake as in the embayments. We also included an unproductive embayment, Tadenac Bay, located in the Township of Georgian Bay, which is in reference condition because it has been exposed to minimal human impact over the past century. There are only 3 buildings along the shoreline, and on average during the summer, there are fewer than 10 club members staying in one of the main buildings (Figure 1.1 and Figure 1.2H). Managed by the Tadenac Club (private fishing and hunting club), the bay supports a productive fishery of northern pike, largemouth bass and smallmouth bass.

Although it is relatively shallow (Z_{mean}=5.01 m), Tadenac Bay has one deep basin (Z_{max}=29.10 m).

Musquash Bay and Woods Bay are two riverine embayments positioned at separate outflows of the Muskoka River. Musquash Bay is large and relatively deep (Z_{max}=42.86 m), and is located at the outflow of the Musquash River (Figure 1.1 and Figure 1.2D) in the Township of Georgian Bay. It is relatively undeveloped because it is only accessible by boat on a seasonal basis. By comparison, Woods Bay is located in the Township of the Archipelago at the mouth of the Moon River (Figure 1.1 and Figure 1.2J). It is shallow (Z_{mean}=3.58 m) and does not stratify during the summer (Figure 1.3). The level of human development is higher because it can be accessed by road and has a large marina.

The remaining seven embayments in our study are lacustrine, with small to moderate-sized drainage basins, and have only small streams discharging into them. Five of these are found within the Township of Georgian Bay. North Bay and South Bay are located the furthest south, close to the town of Honey Harbour (Figure 1.1; Figure 1.2E and 1.2F, respectively). They are located closest to the populated urban centers of southern Ontario and are a popular summer vacation destination. Consequently, a relatively large number of cottages are located in these embayments. Since they serve as access points to reach more remote areas of Georgian Bay, there are several marinas in the vicinity. Both embayments are also associated with high levels of residential building, road and marina development within their watersheds. North Bay and South Bay both have complex shorelines, with maximum depths of 22.74 m and 15.96 m, respectively (Table 1.2).

Situated north of Honey Harbour, there are two shallow embayments, Cognashene Lake and Longuissa Bay, which can only be accessed seasonally by boat (Figure 1.1; Figure 1.2A and 1.2C, respectively). The deep basin in Cognashene Lake is located through a constrained channel opening to Georgian Bay and has a maximum depth of 16.88 m, with a moderate level of human development along its shoreline. Longuissa Bay is also shallow (Z_{mean}=3.59 m) although it has a small depression that is sufficiently deep to establish weak thermal stratification (Z_{max}=11.89 m; Figure 1.3). It is exposed to minimal human development as it is only accessible by boat during the open-water season; however, this embayment is a popular mooring site for recreational boaters and is used heavily throughout the summer months.

Also located in the Township of Georgian Bay, Twelve Mile Bay (Figure 1.1) is situated at the end of a long open channel that extends inland from open Georgian Bay (Figure 1.2I). Its relatively small and narrow basin is sufficiently deep to develop weak thermal stratification (Z_{max}=13.98 m; Figure 1.3) throughout the summer months. Two roads run parallel along the north and south shores of Twelve Mile Bay and provide access to a moderate amount of cottage development; additionally, the community of Moose Deer Point First Nations is located inland from the southeast corner of the embayment.

Deep Bay is the only embayment in our study found in Carling Township, located just north of the city of Parry Sound (Figure 1.1 and 1.2B). It can be accessed through a narrow channel located off the northeast shore of Parry Sound Bay. This embayment has reliable year round road access which has allowed for high levels of human development along the shoreline. Deep Bay is characterised by a moderately complex shoreline (Figure 1.2B) with several deep depressions (Z_{max}=19.51 m; Figure 1.3) that develop strong thermal stratification throughout the summer season.

The northernmost coastal embayment in our study is Sturgeon Bay, located in the Township of the Archipelago (Figure 1.1 and Figure 1.2G), near the community of Pointe au Baril Station. It can be accessed through the long and narrow Brignall Banks channel. Sturgeon Bay is shallow (Z_{mean}=4.43 m) with one depression in the northernmost bay (Z_{max}=14.51 m; Figure 1.3) that is able to develop strong thermal stratification. The highly complex shoreline of Sturgeon Bay (Figure 1.2G) is heavily developed with cottages. This area supports the highest cottage development in our study and is also used as a boat access point to reach islands further out into Georgian Bay.

Methods

Water Quality Sampling

To capture seasonal variation in [TP] and chlorophyll-a ([CHL]), we sampled each embayment or lake on a monthly basis from May to September in a single year between 2012 and 2015. We were unable to sample Longuissa Bay in May due to logistical difficulties. We collected all water samples and measured water temperature (TEMP) and dissolved oxygen (DO) at the deepest point in each embayment (see Table 1.1). On each sampling occasion, we obtained physico-chemical information from surface to 1 m above the sediment surface to a maximum depth of 25 m. A YSI 6920 V2 sonde was calibrated prior to each sampling week and used to measure TEMP (°C) and DO (mg•L^{-1}) *in situ* to determine the presence and extent of hypolimnetic anoxia during the stratification season. We used a Van Dorn sampler to collect water at the midpoint of the epilimnion, metalimnion and hypolimnion in order to determine concentrations of [TP] (μg•L^{-1}); however, only samples collected from epilimnion were used to determine concentrations of [CHL] (μg•L^{-1}).

Once collected, all water samples were stored on icepacks in a cooler until they could be processed or placed in a freezer for storage and analyzed later at McMaster University. A BOD bottle was filled with HACH™ chemicals and analyzed for DO according to the Winkler method to confirm accuracy of *in situ* values from the YSI. Samples to be analyzed for [TP] were frozen and transported back to McMaster University. Unless otherwise indicated, all analyses were performed in triplicate for each variable. [TP] concentrations were determined with the molybdenum blue method (Murphy and Riley 1962) following potassium persulfate digestion in an autoclave for 50 min (120°C, 15 psi). Absorbance values were read with a Genesys™ 10 UV Spectrophotometer and final [TP] concentrations were calculated with a standard curve. Known aliquots of water were filtered through Whatman™ GC/F glass microfiber filters (0.45 μm) and subsequently used to calculate concentrations of [CHL]. Filters for [CHL] determination were folded, then wrapped in aluminum foil and placed in a freezer until they were processed. [CHL] filters were extracted in 90% reagent grade acetone in a freezer over a 24 hour period. Following extraction, samples were acidified with hydrochloric acid (0.1 M), and fluorescence was read with a Turners Design Trilogy™ Fluorometer. [CHL] measurements were collected for North Bay, South Bay and Tadenac Bay with a calibrated bbe Moldaenke FluoroProbe™ during 2015, as grab samples were not collected during intensive sampling efforts in 2012.

Trophic Status Determination

For calibration purposes, we used the trophic state classification system developed by Reckhow and Chapra (1983) to determine the trophic status for all ten embayments and the inland lake, Blackstone Lake. We will compare the trophic status of the embayments with that of Blackstone Lake, knowing that Blackstone Lake has been assessed as an oligotrophic lake in previous studies (Chow-Fraser 2005; Schiefer 2008). Seasonal mean values of [TP] collected from each stratum (epilimnion, metalimnion, and hypolimnion) were averaged to determine trophic status. Using this classification system, water bodies with [TP] <10 ($\mu g \bullet L^{-1}$) were classified as oligotrophic, and those with [TP] >20 ($\mu g \bullet L^{-1}$) were classified as eutrophic and all intermediate values (i.e. >10 ($\mu g \bullet L^{-1}$) to <20 ($\mu g \bullet L^{-1}$)) were classified as mesotrophic.

Lakeshore Capacity Model

We applied the LCM to embayments and Blackstone Lake using the components, coefficients and technical assumptions outlined by Paterson et al. (2006) and we followed instructions provided in the Lakeshore Capacity Assessment Handbook (Ministries of Ontario 2010). These two resources are the current recommended documents to guide the application of this model for lakes located on the Precambrian Shield. All calculations were tabulated with Microsoft Excel spreadsheet software and statistical analyses were performed with JMP 12 software (SAS, Cary, N.C, 2015). As instructed by the Lakeshore Capacity Assessment Handbook, we ignored upstream lakes ≤ 25 ha in surface area unless they had significant levels of human development along the shoreline.

Estimate of Total Phosphorus Concentrations

The LCM estimates seasonal mean [TP] concentrations for lakes using empirical relationships based on P budget, watershed hydrology and basin morphometry (Paterson et al. 2006; Figure 1.4; see model inputs in Table 1.3). Equation 1 in Table 1.4 was used to determine seasonal mean [TP] in water bodies being modelled. Where L_T is the total areal P load ($mg \bullet m^{-2}$) from both natural and anthropogenic sources calculated by dividing the total P load ($mg \bullet y^{-1}$) by the lake surface area (m^2), R_P represents the P retention coefficient and q_s represents the areal water load.

Natural Phosphorus Inputs

The LCM estimates P export ($kg \bullet y^{-1}$) (Eq. 2 in Table 1.4) from the watershed based on catchment area (km^2) and percentage wetland area (Paterson et al. 2006). We used the Land Information Ontario Wetland Layer (Ontario Ministry of Natural Resources and Forestry 2015) to calculate wetland area of each sub-watershed and then summed all sub-watersheds to determine total wetland area (km^2) for each embayment (Table 1.5). In addition to P-export from overland runoff, the LCM also takes into account atmospheric deposition of P onto the lake surface. We calculated atmospheric P load by multiplying lake surface area (m^2) by an atmospheric P deposition rate of 16.7 $mg \bullet m^{-2} \bullet y^{-1}$ calculated as a 17 year mean from 3 meteorological stations in central Ontario (Paterson et al. 2006).

Anthropogenic Phosphorus Inputs

The Lakeshore Capacity Assessment Handbook (Ministries of Ontario 2010) recommends that, in applying the LCM, one assume that all buildings associated with septic tanks that are situated within a 300 m buffer of the shoreline would contribute P to the water body if the soils in the watershed are thin. The soil structure of coastal Georgian Bay has been classified as dominantly coarse textured, consisting primarily of sandy loam that have good drainage, and Precambrian rock at one foot or less (Canada Department of Agriculture 1960); thus, we have assumed that any export of P from waste water systems will eventually be introduced to nearby waters. To determine the number of dwellings that would contribute P, we imported IKONOS satellite image data (1 m panchromatic images acquired in 2002, 2003, or 2008 depending on the site) into ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA) and enumerated all dwellings in these images (i.e. cottages, trailers, etc.; see Table 1.5). Google Earth™ satellite imagery taken in 2015 were also used to update the database. Structures that were not typically connected to septic systems (e.g. garages, warehouses, sheds and boathouses) were excluded from this count.

An important component of the total anthropogenic P input in the LCM is the calculation of use rates of residents in the embayments. Since we did not have the means to determine total number of seasonal, extended seasonal or permanent residences in each embayment, we categorized dwellings with reliable year-round road access as "extended seasonal", and dwellings with no reliable year-round road access as "seasonal" (Table 1.5). To calculate total anthropogenic P export, the respective seasonal usage values (Table 1.3) for each dwelling type were multiplied by the estimated per capita P contribution value of 0.66 kg•capita^{-1}•y^{-1} (Paterson et al. 2006), and then multiplied by the number of corresponding residential units.

Watershed Characteristics

We delineated watersheds using ArcGIS 10.3.1 Hydrology Toolset (ESRI™, Redlands, CA, USA) and digital elevation model (DEM; 10 m accuracy) provided by the Ontario Ministry of Natural Resources and Forestry (2013). We then used the

Waterbody layer in the Ontario Hydro Network Small Scale GIS database (Ontario Ministry of Natural Resources and Forestry 2012) to identify all lakes in the watersheds of each embayment with surface area ≥25 ha. We delineated the sub-watershed of each of these lakes so that they could be modelled individually on a watershed basis. We could not obtain data pertaining to the discharge of each lake modelled, because permanent discharge stations do not exist for all watersheds, and we did not have the resources to collect *in situ* field measures. In the absence of this information, we used Equation 3 in Table 1.4 to calculate the discharge as per recommendations by Paterson et al. (2006), where Q represented discharge ($m^3 \bullet y^{-1}$), A_d represented drainage area (m^2) and A_o represented lake surface area (m^2). Average annual long-term runoff estimates for the eastern Georgian Bay region were obtained from a Department of Fisheries and Environment (1975) annual runoff map.

Phosphorus Retention

The LCM considers the amount of P retained in each lake through P-sedimentation. Two different settling velocities may be applied depending on the extent of hypolimnetic oxygen depletion over the period of thermal stratification, with a settling velocity of 12.40 and 7.20 $m \bullet y^{-1}$ for lakes with oxic and anoxic hypolimnia, respectively (Paterson et al. 2006; Table 1.3). We collected temperature and oxygen depth profiles of each embayment to determine the appropriate settling coefficient. The reduced settling velocity applied under anoxic conditions was meant to reflect the release of P from lake sediments through internal loading or the reduced ability of P to settle out of the water column in lakes with anoxic hypolimnia (Dillon et al. 1994). It should be noted, however, that this reduced settling velocity did not correspond quantitatively to the internal P load, but was only an attempt by the LCM to account for these processes (Paterson et al. 2006). An empirical relationship was used to calculate the P retention coefficient, where v ($m \bullet y^{-1}$) represented the settling velocity (oxic or anoxic) and q_s represented the areal water load which was calculated by dividing the total lake discharge (Q; $m^3 \bullet y^{-1}$) by the lake surface area (m^2) (Eq. 4, Table 1.4). Dillon et al. (1986) empirically proved that [TP] in the open water of lakes was different from that in the outflow; therefore, equation 5 in Table 1.4 was used to estimate [TP] concentrations at the outflow of a lake. The value from this equation was then used to calculate the amount of P discharged to downstream lakes in the LCM for each embayment.

Downstream Phosphorus Input

To model each lake on a watershed basis, we calculated the amount of P flowing from one lake into another. We used the empirical relationship provided to calculate the amount of P entering downstream lakes, where TP_D was the P input to the downstream lake, $TP_{Outflow}$ was the estimated [TP] ($\mu g \bullet L^{-1}$) at the outflow of a water body and Q was the lake outflow discharge ($m^3 \bullet y^{-1}$) (Eq. 6, Table 1.4).

Internal Loading of Phosphorus

Loss of hypolimnetic oxygen can facilitate the release of nutrients from lake sediments through internal loading (Sondergaard 2001; Nurnberg and LaZerte 2004). Because internal load was not explicitly calculated in the LCM, and we presumed this process had occurred in coastal embayments that had anoxic hypolimnia, we used Nurnberg's (1984) approach to estimate internal P loading for all embayments that experienced hypolimnetic anoxia during the stratification season (Figure 1.5). The following equation was used:

Equation 1: $L_{Int.}$ = (Anoxic Area*Anoxic Period*P Release Rate) ÷ Lake Surface Area

where $L_{Int.}$ was internal P load ($mg \bullet m^{-2} \bullet d^{-1}$), anoxic area was the sediment area overlain by anoxic waters (m^2), anoxic period was the duration of anoxia in days, and P release rate was the rate at which P was released from the sediments under anoxic conditions ($mg \bullet m^{-2} \bullet d^{-1}$).

To characterize basin morphometry, bathymetric information for each site was collected *in situ* with a Lowrance™ Elite-7 HDI fish finder to produce DEMs for all embayments excluding North Bay, South Bay and Twelve Mile Bay; corresponding information for the latter three embayments were provided by H. Biberhofer, Environment and Climate Change Canada (Biberhofer unpub. data 2014). All data were imported into ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA) to create site-specific DEMs (~5 m accuracy). Vertical profiles of TEMP and DO were taken monthly (May to September) at the deep station to determine the duration and the extent of hypolimnetic anoxia (<1 mg•L^{-1}, DO) in each embayment. Since the date of fall overturn could not be ascertained in all instances, a formula developed by Nurnberg (1988) was used to estimate the date at which this event would likely have occurred. The P-release rate of 14 mg•m^{-2}•d^{-1} (representing the mean release rate for 15 anoxic lakes; Nurnberg 1984) was used to calculate the internal load for each embayment that developed anoxic hypolimnion (i.e. Deep Bay, Cognashene Lake, Longuissa Bay, North Bay, South Bay, Sturgeon Bay and Twelve Mile Bay; Figure 1.5).

Degree of Mixing with Georgian Bay

We determined the degree of mixing between embayment waters and the waters of Georgian Bay by developing an Index of Resistance to Mixing (IRM) based on morphometric parameters that are believed to contribute to the lack of water exchange between the embayment and open waters of Georgian Bay.

Equation 2: $\mathrm{IRM} = \sqrt[2]{\frac{P}{L} * Least\ Cost\ Pathway}$

where P is the lake perimeter (m) and L is the maximum length (m). Least Cost Pathway (m) represents the line of least resistance that connects the intersecting point of maximum length and maximum breadth in the embayment with the narrowest point in the outflow channel to Georgian Bay. We calculated this using the Least Cost Pathway tool in ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA).

Examination of Additional Factors

We examined a number of additional anthropogenic and morphometric factors in order to derive an alternative model that we call the Anthro-Geomorphic Model (AGM). The number of docks at each site were enumerated with the same methods described in the *Anthropogenic Phosphorus Input* section above. The total number of docks and buildings were then divided by the embayment perimeter (m) to calculate dock and building density which represented the number of structures per unit length of the shoreline (Table 1.6). Another anthropogenic stressor we considered in this study was road density. To calculate road density, we created 2 km buffers around the embayment perimeter and then used the Provincial Road Network (Statistics Canada 2014) layer to calculate length of road per unit area ($m \bullet ha^{-1}$) (Table 1.6). To examine the relative impact of building density, road density, and the IRM, as well as other morphometric variables (e.g. maximum depth, mean depth, etc. see Table 1.2) on the nutrient status of embayments, we used forward step-wise multiple regression analysis in JMP 12 (SAS, Cary, N.C, 2015).

Accuracy Assessment of Models

The precision of each model to estimate seasonal mean [TP] was evaluated to determine which model performed best when applied to coastal embayments. All predicted values that were within ±20 % deviation from measured values were considered acceptable. This criterion was established by Hutchinson et al. (1991) when they evaluated the performance of the LCM during the early stages of model development. The threshold was developed by measuring the variation in 192 epilimnetic [TP] samples taken from Harp Lake, Huntsville, Ontario, over a nine-year period. They found that the coefficient of variation was approximately 20%, which therefore, represented the amount of variation to be expected in long-term studies of [TP] concentrations in central Ontario lakes. The Ontario Ministry of the Environment and Climate Change continues to use this criterion as a threshold to determine whether an estimated value should be accepted or rejected when applying the LCM (Ministries of Ontario 2010).

Following Pineiro et al. (2008), we evaluated the performance of each model by regressing measured values against estimated values and assessing how close they were to a perfect measured = estimated fit (i.e. slope of 1 and intercept of 0). Blackstone Lake was excluded from this regression analysis since we were only interested in how the model performed on coastal embayments. We also calculated average absolute error ($\mu g \bullet L^{-1}$) and average percent error between measured and estimated values. These measures of performance were used collectively to evaluate the accuracy of predictions produced by each model.

Data for Comparison of Total Phosphorus – Chlorophyll-a Relationship

We compared our TP-CHL regression equation with those in published and unpublished studies of similar aquatic systems because of the high variability among results in previous literature (Prairie et al. 1989; Havens and Nurnberg 2004). We assembled [TP] and [CHL] data from several sources: 1) Diep et al. (2007) who sampled 134 locations in the near-shore zone of eastern and northern Georgian Bay between 2003 and 2005, inclusive; 2) Zimmerman et al. (1983), who sampled 24 inland lakes located in central Ontario; 3) Experimental Lakes Area (ELA) Freshwater Institute (Findlay et al. unpub. data) who sampled 8 lakes in north western Ontario and 4) Molot and Dillon (1991), who sampled 15 central Ontario lakes from 1976 to 1987.

Results

Embayment Trophic Status

As expected, Blackstone Lake was classified as oligotrophic (Table 1.7). Based on Reckhow and Chapra's (1983) trophic state classification system we found that Longuissa Bay, Musquash Bay and Tadenac Bay were classified as oligotrophic (TP; <10 $\mu g \bullet L^{-1}$), while Cognashene Lake, Woods Bay, Twelve Mile Bay, North Bay, Deep Bay and South Bay were classified as mesotrophic (TP; >10 $\mu g \bullet L^{-1}$ to 20 $\mu g \bullet L^{-1}$; Table 1.7). Sturgeon Bay was the only embayment to be classified as eutrophic (TP; >20 $\mu g \bullet L^{-1}$).

Lakeshore Capacity Model

We applied the LCM to Blackstone Lake as a check on the validity of our calculations, knowing that this inland lake has been confirmed by previous studies, as well as the current study, as being oligotrophic. The model produced an estimated [TP] concentration of 6.96 $\mu g \bullet L^{-1}$, which was in close agreement with the measured value of 6.56 $\mu g \bullet L^{-1}$ (Table 1.8). The absolute error resulting from this model was 0.04 $\mu g \bullet L^{-1}$ with a percent error of 6.10, which is well within the acceptable ±20 % deviation from our measured value (Table 1.8). This comparison confirmed that our assumptions and calculations used to calculate [TP] for Blackstone Lake were correct.

The LCM could only be applied to eight of the ten embayments sampled, because the watersheds of Woods Bay and Musquash Bay were too large and complex to be analysed during the course of this study. Terrestrial runoff accounted for the largest source of P input in the LCM and was influenced by the size of the catchment area and the proportion of wetlands in the drainage basin, with Deep Bay having the largest input (533.75 $kg \bullet y^{-1}$) and Cognashene Lake having the smallest (25.81 $kg \bullet y^{-1}$) (Table 1.9). By comparison, the anthropogenic P load was the second largest contributor of P to embayments, with Sturgeon Bay having the largest input (271.50 $kg \bullet y^{-1}$) and Tadenac having the smallest input (2.34 $kg \bullet y^{-1}$). Contribution from atmospheric P deposition was generally small, and was dependent on the surface area of the water body; Sturgeon Bay had the largest input (94.75 $kg \bullet y^{-1}$) while Longuissa Bay had the smallest (7 $kg \bullet y^{-1}$). The seasonal mean [TP] predicted by the LCM for three of the embayments fell within ±20 % deviations of the measured values (Table 1.8). The best-fit linear regression between measured and estimated [TP] produced a weak relationship with correlation coefficient of 0.42, slope of 0.75 and an intercept of 3.62 (Figure 1.6A; Table 1.10). These slope and intercept values compare poorly against the expected values of one and zero. The average absolute error was 3.14 $\mu g \bullet L^{-1}$ and the average absolute percent error was 22.40 for this model.

Given that Gartner Lee Limited (2008a) was able to improve the LCM for Sturgeon Bay by incorporating an estimate of internal P load, we followed this practice and calculated an internal P load for all sites that exhibited hypolimnetic anoxia (<1 $mg \bullet L^{-1}$) during the period of thermal stratification (i.e. Cognashene Lake, Deep Bay, Longuissa Bay, North Bay, South Bay, Sturgeon Bay and Twelve Mile Bay; see Figure 1.5). Calculations of P from sediment re-mineralization varied considerably among embayments, from a minimum of 3.41 $kg \bullet y^{-1}$ in Longuissa Bay to a maximum of 569.02 $kg \bullet y^{-1}$ in Deep Day (Table 1.9). Accounting for additional inputs of [TP] from internal loading yielded acceptable estimates for only two of the eight embayments (Tadenac Bay, Sturgeon Bay; Table 1.8). Regression between estimated and measured [TP] produced a weak relationship with a correlation coefficient of 0.13, slope of 0.18 and an intercept of 9.71 (Table 1.10; Figure 1.6B), which deviated considerably from the expected slope of one and intercept of zero. The average absolute error with the inclusion of internal load increased to 9.87 $\mu g \bullet L^{-1}$, and had an average absolute percent error of 76.85 (Table 1.10). Clearly, incorporating an estimate of internal P load into the LCM did not improve the predictive ability of the model for all of our sites.

Anthro-geomorphic Model

We wanted to develop a predictive model using additional landscape and morphometric variables (e.g. IRM, surface area (m^2), volume (m^3), breadth (m), mean depth (m), max depth (m), perimeter (m), catchment size (m^2), percent wetland in catchment area, road density, dock density and building density). Preliminary explorations revealed that variables reflecting the degree of anthropogenic disturbance along the shoreline were significant predictors of seasonal mean [TP] in embayments. These variables included building density (number of residential structures per unit length of shoreline (m); n = 10; r^2 = 0.79; P=0.0006), road density (road length per unit area of watershed (ha); n = 10; r^2 = 0.73; P=0.0018) as well as dock density (number of docks per unit length of shoreline (m); n = 10; r^2 = 0.61; P=0.0069). We reasoned that lack of mixing between the oligotrophic water of Georgian Bay and the water in embayments should increase mean [TP] in embayments. Therefore, irrespective of the anthropogenic P load, we expected embayments with low IRM scores to have low [TP] due to greater water exchange, and embayments with high IRM scores to have high [TP] as a result of a lesser degree of water exchange (Table 1.6). This expectation was upheld when we regressed seasonal mean [TP] against the IRM values and found a significant positive relationship (n = 10; r^2 = 0.40; P = 0.0489).

Next, we used multiple stepwise regression analysis to explore the best combination of multivariate models to predict [TP]. Both dock density and road density were removed from the model because of their high correlation with building density. We retained building density as a variable because it is a more direct measure of nutrient input than the other two variables given that the greatest anthropogenic source of P input into the embayment is from septic systems associated with the buildings along the lakeshore. When we regressed [TP] against building density and IRM, we found no significant interaction; the regression equation that resulted is as follows:

Equation 3: Mean TP = 3.670 + 895.792 * Building Density + 0.031 * IRM

(n = 10; r^2 = 0.91; P = 0.0002)

This regression model was highly significant and explained 91% of the total variation in [TP] values. The AGM also predicted seasonal mean [TP] within ±20% of measured [TP] for all ten embayments (Table 1.8). The AGM produced the least amount of error with an average absolute error of 1.20 ($\mu g \bullet L^{-1}$) and an average absolute percent error of 9.97 (Table 1.10). The regression between measured and expected values was highly significant, with a correlation coefficient of 0.91, a slope of 0.99 and an intercept of 0.01, which were almost in perfect agreement with the expected slope of one and intercept of zero (Figure 1.6C; Table 1.10).

Total Phosphorus – Chlorophyll-a Relationship

We found the following log-linear relationship between mean seasonal epilimnetic [TP] and [CHL] in coastal embayments (Figure 1.7a).

$$\textbf{Equation 4:}\quad \log CHL = -0.157 + 0.853 * \log TP$$

$$(n = 10;\ r^2 = 0.74;\ P = 0.0013)$$

We plotted this relationship together with other TP-CHL relationships assembled from unpublished and published studies (Figure 1.7b). Sites for this comparison included four groups: the 10 embayments included in this study, eastern Georgian Bay, northern Georgian Bay and inland Precambrian Shield lakes (see Methods for sources). We found no significant differences among geographic groups with respect to the slope of each regression line ($P = 0.843$; ANCOVA; Table 1.11). Holding [TP] constant, however, we found that the y –intercepts were all statistically different from each other. Sites in northern Georgian Bay had the lowest [CHL], followed by eastern Georgian Bay, then inland lakes, while our embayments had the highest values of [CHL] per unit of [TP] (Figure 1.7b).

Discussion

One of our objectives was to assess the appropriateness of using the LCM to differentiate between natural and anthropogenic sources of P for coastal embayments of eastern Georgian Bay. We know that the LCM produced accurate estimates of [TP] for inland lakes of the Precambrian Shield (Paterson et al. 2006; Hutchinson et al. 1991) and this was confirmed with the model calculation for Blackstone Lake, an inland lake on the Precambrian Shield with an outflow that discharges into Georgian Bay. The estimated [TP] for Blackstone Lake calculated using the LCM, and based on our assumptions outlined in the *Methods* section, was within 6.10 % of measured values. When we applied the LCM to embayments, however, we found the model over-estimated [TP] by ±20 % for six of eight embayments. Incorporating internal P load did not improve the predictive ability of the model, but resulted in even higher over-estimates for sites that developed anoxic hypolimnia. By comparison, the AGM we developed, based on the density of primary residential buildings along the shoreline of embayments as well as basin morphometry, successfully predicted [TP] within ±20 % deviation of measured values for all ten embayments. It also produced the least amount of error associated with slope and intercept estimates corresponding to the regression of measured against estimated values.

Accuracy of the AGM depended heavily on our ability to include sites that spanned the entire gradient of [TP] and IRM values. Tadenac Bay and Musquash Bay (<0.0001 and 0.0029, respectively) both had extremely low building densities (number of residential structures per unit length of shoreline (m)), which were orders of magnitude lower than those in Sturgeon Bay and South Bay which had the highest building densities (0.0106 and 0.0109, respectively). Therefore, we could say with confidence that Tadenac Bay was in reference condition for embayments, inasmuch as Blackstone Lake was in reference conditions for inland lakes. The IRM scores varied by five-fold, with Longuissa Bay having the lowest score of 54 and Sturgeon Bay having the highest score of 261 (Table 1.6). The high [TP] in Sturgeon Bay reflects a combination of high cottage density as well as high IRM score, that presumably exacerbated the P enrichment in this embayment since the extensive network of channels prevented the embayment water from being diluted by the oligotrophic water of Georgian Bay.

The LCM was developed and calibrated with six inland lakes (Dillon et al. 1986; Paterson et al. 2006) that had small surface areas (A_{mean}=55 ha) with deep basins (mean Z_{max}=32 m), whereas our eight embayments were generally large (A_{mean}=178 ha) and relatively shallow (mean Z_{max} value of 18 m). The difference in morphometry between the calibration lakes and embayments are likely the cause of some of the variation found in model results due to differing lake processes. Differences in flushing rate may also be a reason why the LCM did not yield acceptable predictions of [TP] for our embayments. Another reason for the difference in performance of the LCM may be related to how the P settling velocity had been empirically determined and the choice of using oxic versus anoxic rates. Inland lakes that had been used to measure settling rates were deep and well-stratified and had uniform basin morphometry (Kirchner and Dillon 1975; Dillon and Kirchner 1975). By contrast, the coastal embayments sampled in this study were shallow, though they typically had one deep basin that could develop thermal stratification (Table 1.2; Figure 1.3). Even when anoxic conditions developed in the hypolimnia, the proportion of bottom sediments overlain by anoxic waters was small (a low of 1.09 % in Longuissa Bay to high of 21.45 % in Deep Bay; mean of 10.76 %). We attempted to correct for over-estimates of internal loading by re-calculating the LCM using anoxic settling velocities for only the portion of the sediments that were overlain by anoxic waters. Unfortunately, this did not improve the overall predictive power of the LCM.

Another dilemma we encountered when applying the LCM was using both settling rate and calculation of internal load. We assume that the settling rate velocity is meant to approximate release of nutrients into the water column under anoxic conditions (Dillon et al. 1994). Therefore, when we also calculated internal load using the anoxic factor, we were essentially doubling the contribution of internal load. That may explain why the LCM produced gross overestimates when internal load was added to the model calculation (Figure 1.6B; Table 1.8).

As a general comment, use of the LCM has a number of disadvantages compared with the AGM. First, the LCM requires a large number of variables, which necessitate time-consuming calculations and/or onerous and expensive data gathering efforts. In order to use the LCM, data must be collected for census information, geospatial land use layers, and seasonal temperature-oxygen depth profiles of all lakes in a watershed. In particular, we found it very difficult to obtain data on the seasonality of residences, which is considered too sensitive to release by government agencies (Ministries of Ontario 2010). Model calculations also required a suite of proprietary geospatial data layers (e.g. wetland cover, digital elevation models) that may be costly to obtain. In addition, knowledge of and access to a geographic information system is necessary to calculate landscape variables required for successful implementation of the model. Costly and time consuming collection of field measurements may also be required to determine the oxygen regime of each lake modelled in a watershed. Therefore, in addition to empirically demonstrating that the AGM is superior to the LCM for predicting mean seasonal [TP] for our embayments, we also found the AGM relatively easy to apply compared to the LCM.

Since these coastal embayments produce a larger amount of [CHL] per unit [TP] compared with other systems (Figure 1.7b), lake managers run the risk of underestimating the amount of [CHL] produced from increases or reductions in [TP]. We would therefore recommend that environmental managers use the TP-CHL relationship presented in this study when determining changes in primary production as a result of change in P load. This will allow managers to accurately estimate changes in algal biomass from increases or reductions in P input resulting from management efforts. We do not know why the intercept for embayments in the comparison of TP-CHL relationships is significantly higher, but it may be related to inefficient grazing pressure (Hansson 1992; Sarnelle 1992; Mazumder 1994) or a higher proportion of biologically available P (Schindler 1978; Hoyer and Jones 1983; Butkus 1988).

Use of the AGM for Management

The trophic state classification system developed by Reckhow and Chapra (1983) used 10 $\mu g \bullet L^{-1}$ to separate oligotrophy from mesotrophy, and 20 $\mu g \bullet L^{-1}$ to separate mesotrophy from eutrophy. Using this classification system, we can assign

Longuissa Bay, Musquash Bay, and Tadenac Bay to the oligotrophic category (7.39, 7.85 and 8.24 $\mu g \bullet L^{-1}$, respectively), Sturgeon Bay to the eutrophic category (20.96 $\mu g \bullet L^{-1}$) and the remaining six embayments to the mesotrophic category (see Table 1.7). None of the coastal embayments in the oligotrophic category experienced any symptoms of cultural eutrophication. By comparison, all of the embayments in the mesotrophic category developed noticeable hypolimnetic oxygen depletion, with the exception of Woods Bay. Finally, Sturgeon Bay, the only embayment in the eutrophic category, exhibited more severe symptoms of cultural eutrophication that included oxygen depletion in the hypolimnion, reduced water clarity in the epilimnion, and several instances of harmful and/or nuisance algal blooms over the past decade.

Unless environmental managers can alter the degree of water mixing between the embayment and Georgian Bay, building density along the shoreline is the only variable that can be manipulated to maintain good water quality in these coastal embayments. In order to provide managers with a guide for preserving water quality in the ten embayments studied, we used the AGM to determine the building density thresholds that correspond to predicted [TP] of 10, 15 and 20 $\mu g \bullet L^{-1}$ for each embayment (Table 1.13). To avoid development of hypoxic hypolimnia, [TP] in coastal embayments should remain below 10 $\mu g \bullet L^{-1}$; a threshold that a majority of the embayments in this study have already surpassed. In fact, even if no buildings were present, the AGM estimates that Sturgeon Bay would have a [TP] concentration of 11.5 $\mu g \bullet L^{-1}$, lending support to paleolimnological studies conducted in Sturgeon Bay that indicate it was naturally mesotrophic even before European contact (Gartner Lee Limited 2008a).

In this study we show that the LCM does not produce accurate estimates of seasonal mean [TP] in coastal embayments and therefore should not be used as a management tool to determine the trophic response to human development in these systems. We recommend that the AGM be used instead because it produced more accurate estimates of [TP] and could be applied to all ten embayments without exception. We found that embayments produced significantly higher [CHL] per unit of [TP] when compared to other geographic regions in Georgian Bay and inland Precambrian Shield lakes. It is therefore important that environmental managers use the TP-CHL relationship presented in this paper when evaluating changes in primary production that would result from an increase or a reduction of P loading to an embayment. An issue that merits scientific exploration is the role that nitrogen limitation might play in phytoplankton growth, community structure and the potential for the development of toxic cyanobacteria blooms in coastal embayments. The ability of some cyanobacteria to fix nitrogen allows a number of these algae species to periodically become dominant under nitrogen-limiting conditions (Paerl et al. 2001). This process can be exacerbated in water bodies that have long residence times, surface water temperatures exceeding 20 °C and develop thermal stratification during the summer season (Reynolds and Walsby 1975; Paerl 1988; Shapiro 1990), all of which are characteristics that many coastal embayments possess. Understanding these processes might become essential under future climate change scenarios where we expect to see more variable water levels and higher average temperatures, that when combined with nutrient enrichment, could increase the occurrence of nuisance or harmful cyanobacteria blooms in the coastal embayments of Georgian Bay.

Acknowledgements

I would first like to thank all of the graduate and undergraduate members of the Chow-Fraser Lab, past and present, who made this project possible. Most importantly, thank you to Dr. Patricia Chow-Fraser for her guidance and mentorship throughout this project. I would like to thank Julia Rutledge for always taking time out of her busy schedule to share her wisdom and expertise relating to water chemistry. Thank you to Lindsey Boyd for her continued support and assistance both in the lab and in the field. I would also like to thank Mary Muter, Jean DeMarco, Andy and Barb Metelka, and Paul and Judi Perchenson for allowing us to use their cottages throughout the summers of 2012 to 2015. Thank you to Arthur Zestepa for his assistance in collecting water samples and the use of his equipment in the field. I would also like to thank Hans Biberhofer for providing me with bathymetric information used in this study. Finally, thank you to Environment and Climate Change Canada for funding this project through the Lake Simcoe South-eastern Georgian Bay Clean-up Fund.

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Table 1.1: Location of ten coastal embayments and Blackstone Lake, an inland lake. Latitude and longitude correspond to the location in each embayment or lake where we collected water samples and obtained physico-chemical readings *in situ.*

Site name	Code	Latitude	Longitude
Blackstone Lake	BL	45.22278	-79.87613
Tadenac Bay	TD	45.05830	-79.97540
South Bay	SB	44.87635	-7978615
North Bay	NB	44.89099	-79.79197
Cognashene Lake	CG	44.95328	-79.98950
Twelve Mile Bay	TW	45.08321	-79.94811
Deep Bay	DB	45.39490	-80.22375
Musquash Bay	MU	44.94828	-79.85066
Sturgeon Bay	ST	45.61325	-80.43147
Woods Bay	WB	45.13774	-79.98938
Longuissa Bay	LG	44.96137	-79.88864

Table 1.2. Summary of basin morphometry for ten embayments and Blackstone Lake. A_o=surface area (m^2), V=volume (m^3), L=maximum length (m), B=breadth (m), Z_{mean}=mean depth (m), Z_{max}=maximum depth (m), Z_r=relative depth (m), D_L=shoreline development, D_V=volume development.

Site name	A_o ($10^6 m^2$)	V ($10^6 m^3$)	L (m)	B (m)	Z_{mean} (m)	Z_{max} (m)	P ($10^3 m$)	Z_r (m)	D_L	D_V
Blackstone Lake	5.083	101.534	3983	1587	19.97	60.80	37.86	2.39	4.75	0.99
Tadenac Bay	3.931	19.695	3684	1239	5.01	29.10	37.59	1.30	5.35	0.52
South Bay	1.494	4.389	1403	891	2.94	15.96	17.82	1.16	4.11	0.55
North Bay	1.886	9.180	2137	907	4.87	22.74	22.53	1.47	4.63	0.64
Cognashene Lake	0.965	2.590	1416	576	2.66	16.88	13.13	1.52	3.77	0.47
Twelve Mile Bay	1.189	6.516	3823	461	5.48	13.89	16.47	1.13	4.26	1.18
Deep Bay	2.664	15.880	4390	760	5.96	19.51	24.50	1.06	4.23	0.92
Musquash Bay	3.201	40.293	3326	1398	12.59	42.86	22.21	2.12	3.50	0.88
Sturgeon Bay	4.988	22.109	2646	1346	4.43	14.51	33.10	0.58	4.18	0.92
Woods Bay	3.630	13.003	2781	1966	3.58	13.42	20.30	0.62	3.00	0.80
Longuissa Bay	0.355	1.276	1477	347	3.59	11.89	5.30	1.77	2.53	0.91

Table 1.3: Sources of information used to apply the Lakeshore Capacity Model (LCM) in this study.

Type	Model Component	Value	Source and Data Type
Source of input	Atmospheric Phosphorus Deposition Rate	16.7 mg•m^{-2}•y^{-1}	Paterson et al. 2006
	Land Information Ontario Wetland Layer		ArcGIS shapefile from Ontario Ministry of Natural Resources and Forestry 2011
	Ontario Hydrology Network - Small Scale Waterbody		ArcGIS shapefile from Ontario Ministry of Natural Resources and Forestry 2012
	Extended Seasonal Unit	1.27 capita•y^{-1}	Paterson et al. 2006; satellite imagery
	Seasonal Unit	0.69 capita•y^{-1}	Paterson et al. 2006; satellite imagery
	Resort Unit	1.18 capita•y^{-1}	Paterson et al. 2006; satellite imagery
	Trailer Park Unit	0.69 capita•y^{-1}	Paterson et al. 2006; satellite imagery
	Campground / Tent Trail Unit	0.37 capita•y^{-1}	Paterson et al. 2006; satellite imagery
	Youth Camp Guest	125 g•capita^{-1}•y^{-1}	Paterson et al. 2006; satellite imagery
	Per Capita Phosphorus Contribution	0.66 kg•capita^{-1}•y^{-1}	Paterson et al. 2006; satellite imagery
Retention	Oxic Phosphorus Settling Velocity	12.4 m•y^{-1}	Paterson et al. 2006
	Anoxic Phosphorus Settling Velocity	7.2 m•y^{-1}	Paterson et al. 2006
Hydrology	Runoff Estimate	0.400 m•m^{-2}•y^{-1}	Canada Department of Fisheries and the Environment 1975
	Provincial Digital Elevation Model - Version 3.0		ArcGIS Raster File from Ontario Ministry of Natural Resources and Forestry 2013

Table 1.4: List of equations used in the Lakeshore Capacity Model (LCM). 1) Equation to estimate seasonal mean total phosphorus ($\mu g \bullet L^{-1}$); 2) Equation to estimate terrestrial total phosphorus export ($kg \bullet y^{-1}$); 3) Equation to estimate lake discharge ($m^3 \bullet year^{-1}$); 4) Equation to estimate phosphorus retention coefficient; 5) Equation to estimate total phosphorus concentration ($\mu g \bullet L^{-1}$) at lake outflow and 6) Equation to estimate total phosphorus export to downstream lakes ($kg \bullet y^{-1}$). Please see table presenting glossary of symbols at the beginning of this document.

Equation #	Formula	Source
1	$TP_{Mean} = L_T \bullet (1-R_P) \bullet (0.956 \bullet q_s)^{-1}$	Dillon et al. 1986
2	$TP = \text{Catchment Area} \bullet (0.47 \bullet \%\ \text{Wetland Area} + 3.82)$	Paterson et al. 2006
3	$Q = (A_d + A_o) \bullet \text{Mean Annual Runoff}$	Paterson et al. 2006
4	$R_p = v \bullet (v + q_s)^{-1}$	Paterson et al. 2006
5	$TP_{Outflow} = 0.956 \bullet TP_{Mean}$	Paterson et al. 2006
6	$TP_D = TP_{Outflow} \bullet Q$	Paterson et al. 2006

Table 1.5: Data used to calculate anthropogenic and natural terrestrial phosphorus loading for the Lakeshore Capacity Model (LCM) for eight embayments and Blackstone Lake. Seasonal=# primary seasonal residential dwellings; Extended=# primary extended seasonal residential dwellings; Trailer=# residential trailer units; Campground=# campsites; Wetland=wetland area in watershed (km^2) and Watershed=watershed area (km^2).

Site name	Seasonal	Extended	Trailer	Campground	Wetland	Watershed
Blackstone Lake	43	116	0	0	4.68	107.04
Cognashene Lake	70	0	0	0	0.23	4.68
Deep Bay	66	166	0	0	4.28	90.66
Longuissa Bay	6	0	0	0	0.42	5.06
North Bay	130	28	89	0	1.58	10.21
South Bay	54	99	42	0	1.76	9.13
Sturgeon Bay	111	240	0	81	9.52	74.74
Tadenac Bay	3	0	0	0	5.67	50.13
Twelve Mile Bay	0	125	0	0	1.96	12.73

Table 1.6: Summary of Building Density (# buildings • shoreline length $(m)^{-1}$), Dock Density (# docks • shoreline length $(m)^{-1}$) and Road Density (length of road (m) • ha^{-1}) within a 2 km buffer of the shoreline, and the Index of Resistance to Mixing (IRM), which is calculated by Eq.: 2 in Methods.

Site name	Building Density	Dock Density	Road Density	IRM
Blackstone Lake	0.00419	0.00400	5.87	n/a
Longuissa Bay	0.00112	0.00112	0.00	54.04
Twelve Mile Bay	0.00759	0.01591	7.54	57.27
North Bay	0.01096	0.00768	7.55	77.32
Musquash Bay	0.00288	0.00297	0.00	84.50
Woods Bay	0.00443	0.00907	3.78	124.96
Deep Bay	0.00948	0.00617	11.95	127.47
South Bay	0.01094	0.01886	15.81	131.25
Tadenac Bay	0.00008	0.00016	1.62	131.84
Cognashene Lake	0.00533	0.00571	0.00	133.51
Sturgeon Bay	0.01061	0.01777	7.44	261.08

Table 1.7: Mean seasonal (±SE; $\mu g \cdot L^{-1}$) total phosphorus [TP] (samples collected from all strata) and mean seasonal epilimnetic (±SE; $\mu g \cdot L^{-1}$) Chlorophyll-a [CHL] for ten embayments and Blackstone Lake. In all cases, using Reckhow and Chapra's (1983) trophic state classification system, sites with [TP] <10 ($\mu g \cdot L^{-1}$) were interpreted as being oligotrophic, those with [TP] >10 ($\mu g \cdot L^{-1}$) to <20 ($\mu g \cdot L^{-1}$) were interpreted as being mesotrophic, and all sites with [TP] >20 ($\mu g \cdot L^{-1}$) were interpreted as being eutrophic.

Site	Year Sampled	TP ($\mu g \cdot L^{-1}$)	CHL ($\mu g \cdot L^{-1}$)	Trophic Status
Blackstone Lake	2013	6.56 (± 0.54)	1.90 (± 0.18)	Oligotrophic
Longuissa Bay	2015	7.39 (± 0.70)	4.48 (± 0.34)	Oligotrophic
Musquash Bay	2015	7.85 (± 0.94)	3.78 (± 0.38)	Oligotrophic
Tadenac Bay	2012	8.24 (± 0.57)	5.15 (± 1.29)	Oligotrophic
Cognashene Lake	2014	10.60 (± 0.89)	4.86 (± 0.83)	Mesotrophic
Woods Bay	2015	11.11 (± 2.72)	4.04 (± 0.74)	Mesotrophic
Twelve Mile Bay	2014	13.00 (± 1.11)	5.77 (± 0.75)	Mesotrophic
North Bay	2012	13.41 (± 0.66)	7.62 (± 1.17)	Mesotrophic
Deep Bay	2014	18.41 (± 2.32)	11.72 (± 3.32)	Mesotrophic
South Bay	2012	18.85 (± 2.13)	6.66 (± 0.80)	Mesotrophic
Sturgeon Bay	2015	20.96 (± 4.45)	6.59 (± 0.97)	Eutrophic

Table 1.8: Summary of measured seasonal mean total phosphorus [TP] ($\mu g \bullet L^{-1}$) and estimates of [TP] calculated by the Lakeshore Capacity Model (LCM), the Lakeshore Capacity Model + Internal Loading (LCM+IL) and by the Anthro-geomorphic Model (AGM). Statistics to evaluate model performance are percent deviations (% Deviation) between estimated and measured values. N/A=value unavailable because LCM and LCM+IL could not be applied to these embayments due to the extremely large size and complexity of their respective watersheds. * denotes that estimated value exceeds ±20% of measured value (Hutchinson et al. 1991).

Site	Measured TP	Estimated TP			% Deviation		
		LCM	LCM + IL	AGM	LCM	LCM + IL	AGM
Blackstone Lake	6.56	6.96	6.96	N/A	6.10	6.10	N/A
Cognashene Lake	10.60	9.72	30.88	12.58	-8.30	191.32*	18.68
Deep Bay	18.41	13.54	23.55	16.11	-26.45*	27.92*	-12.49
Longuissa Bay	7.39	9.77	10.46	6.35	32.21*	41.54*	-14.07
Musquash Bay	7.85	N/A	N/A	8.77	N/A	N/A	12.99
North Bay	13.41	17.15	35.02	15.89	27.89*	161.15*	18.49
South Bay	18.85	19.67	28.67	17.54	4.35	52.10*	-6.95
Sturgeon Bay	20.96	14.42	17.65	21.27	-31.20*	-15.79	1.48
Tadenac Bay	8.24	7.38	7.38	7.83	-10.44	-10.44	-4.98
Twelve Mile Bay	13.00	17.99	27.89	12.24	38.38*	114.54*	-5.85
Woods Bay	11.11	N/A	N/A	11.52	N/A	N/A	3.69

Table 1.9: Summary of sub-compartments of phosphorus load (kg•y^{-1}) for each water body calculated by the Lakeshore Capacity Model (LCM). Internal phosphorus loading was calculated independently with Nurnberg's (1984) formula and included as an additional source in the LCM. ATM=Atmospheric phosphorus load, RUNOFF=Terrestrial phosphorus load, UPSTR=Phosphorus load from upstream lakes/wetlands, ANTHRO=Anthropogenic phosphorus load, INTERNAL=Phosphorus load from sediments, and TOTAL=Total phosphorus load.

Site name	ATM	RUNOFF	UPSTR	ANTHRO	INTERNAL	TOTAL
Blackstone Lake	88.79	282.21	233.88	116.81	0.00	721.69
Cognashene Lake	13.39	25.81	0.00	31.88	154.69	225.77
Deep Bay	46.37	533.75	20.09	169.20	569.02	1338.42
Longuissa Bay	7.00	38.95	0.00	2.73	3.41	52.09
North Bay	26.19	113.06	0.00	123.20	273.36	535.81
South Bay	23.25	117.61	0.00	126.70	122.53	390.09
Sturgeon Bay	94.75	503.62	82.33	271.50	245.84	1198.04
Tadenac Bay	55.31	246.71	74.41	2.34	0.00	378.77
Twelve Mile Bay	23.80	135.51	0.00	104.78	145.31	409.4

Table 1.10: Summary of regression analysis relating measured total phosphorus [TP] (μg•L^{-1}) against estimated [TP] by three management models. LCM=Lakeshore Capacity Model, LCM+IL=Lakeshore Capacity Model + Internal Load, and the AGM=Anthro-Geomorphic Model. $Error_{mean}$= mean absolute error (μg•L^{-1}), $Error_{mean\%}$=mean absolute % error. The expected slope and intercept in all cases are 1 and 0, respectively.

Model	$Error_{mean}$	$Error_{mean\%}$	r^2-value	Slope	Intercept
LCM	3.14	22.40	0.42	0.75	3.62
LCM+IL	9.87	76.85	0.13	0.18	9.71
AGM	1.20	9.97	0.91	0.99	0.01

Table: 1.11: Comparison of TP-CHL relationships calculated for data corresponding to different lakes located on the Precambrian Shield within Ontario. Data for ten embayments of eastern Georgian Bay are from this study. Data for Eastern GB and Northern GB were collected by Diep et al. (2007) during synoptic surveys conducted between 2003 and 2005 at 134 nearshore locations in eastern and northern Georgian Bay; data are means of duplicate samples collected in spring, summer and fall from the epilimnion. Data for ELA (Experimental Lakes Area) and LEWG (Lake Ecosystem Working Group) were obtained from Chow-Fraser et al. 1994; these are means of epilimnetic samples collected at least monthly from lakes. Data obtained from Molot and Dillon 1991 represent means of epilimnetic samples collected at least monthly from lakes.

Location	Equation	n	r^2	SE	P	Source
Embayment GB	Log CHL-a = -0.157 + 0.853*Log TP	10	0.74	0.18	0.0013	This study
Eastern GB	Log CHL-a = -0.610 + 1.114*Log TP	90	0.71	0.07	<0.001	Diep et al. 2007
Northern GB	Log CHL-a = -0.655 + 1.034*Log TP	44	0.83	0.06	<0.001	Diep et al. 2007
Inland Lakes	Log CHL-a = -0.501 + 1.083*Log TP	47	0.48	0.19	<0.001	Chow-Fraser et al. 1994 (ELA; LEWG); Molot and Dillon 1991

Table 1.12: Building densities (# buildings•shoreline (m)$^{-1}$) predicted by the Anthro-Geomorphic Model that correspond to three trophic states as determined by total phosphorus [TP] (μg•L^{-1}) in the 10 coastal embayments.

Site name	**10 μg•L^{-1}**	**15 μg•L^{-1}**	**20 μg•L^{-1}**	**Existing # dwellings**
Cognashene Lake	34	107	179	70
Deep Bay	69	204	340	232
Longuissa Bay	28	57	87	6
Musquash Bay	94	217	340	64
North Bay	101	226	350	247
South Bay	48	146	245	195
Sturgeon Bay	*	129	313	351
Tadenac Bay	102	326	552	3
Twelve Mile Bay	84	176	276	125
Woods Bay	59	171	283	90

*The AGM predicts that background concentration for Sturgeon Bay would be 11.46 μg•L^{-1} even when building density is zero.

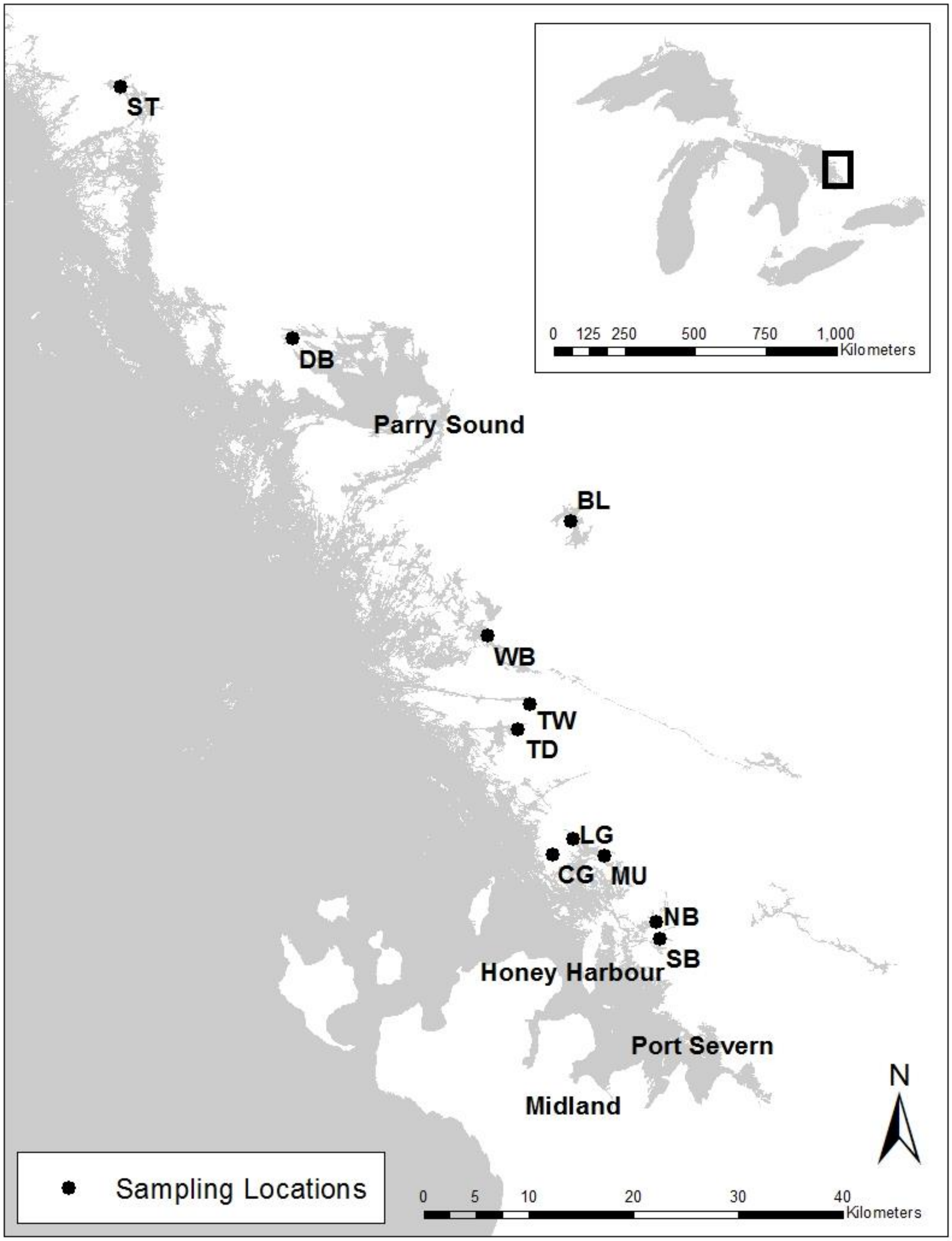


Figure 1.1: Location of ten coastal embayments in eastern Georgian Bay and the inland lake, Blackstone Lake (BL) within the Georgian Bay drainage basin

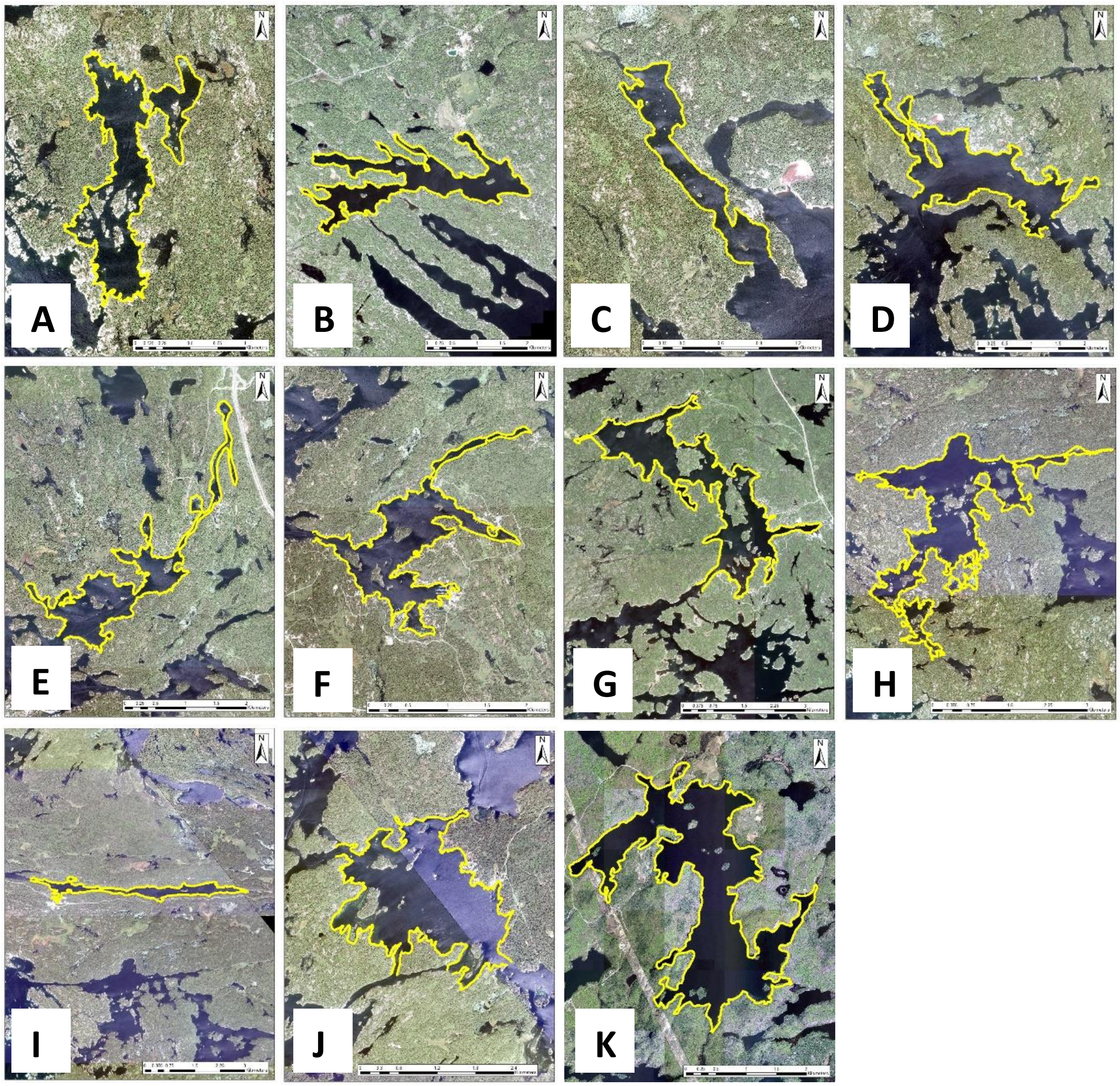


Figure 1.2: IKONOS satellite imagery of study sites acquired during 2002 and 2003. A) Cognashene Lake, B) Deep Bay, C) Longuissa Bay, D) Musquash Bay, E) North Bay, F) South Bay, G) Sturgeon Bay, H) Tadenac Bay, I) Twelve Mile Bay, J) Woods Bay and K) Blackstone Lake.

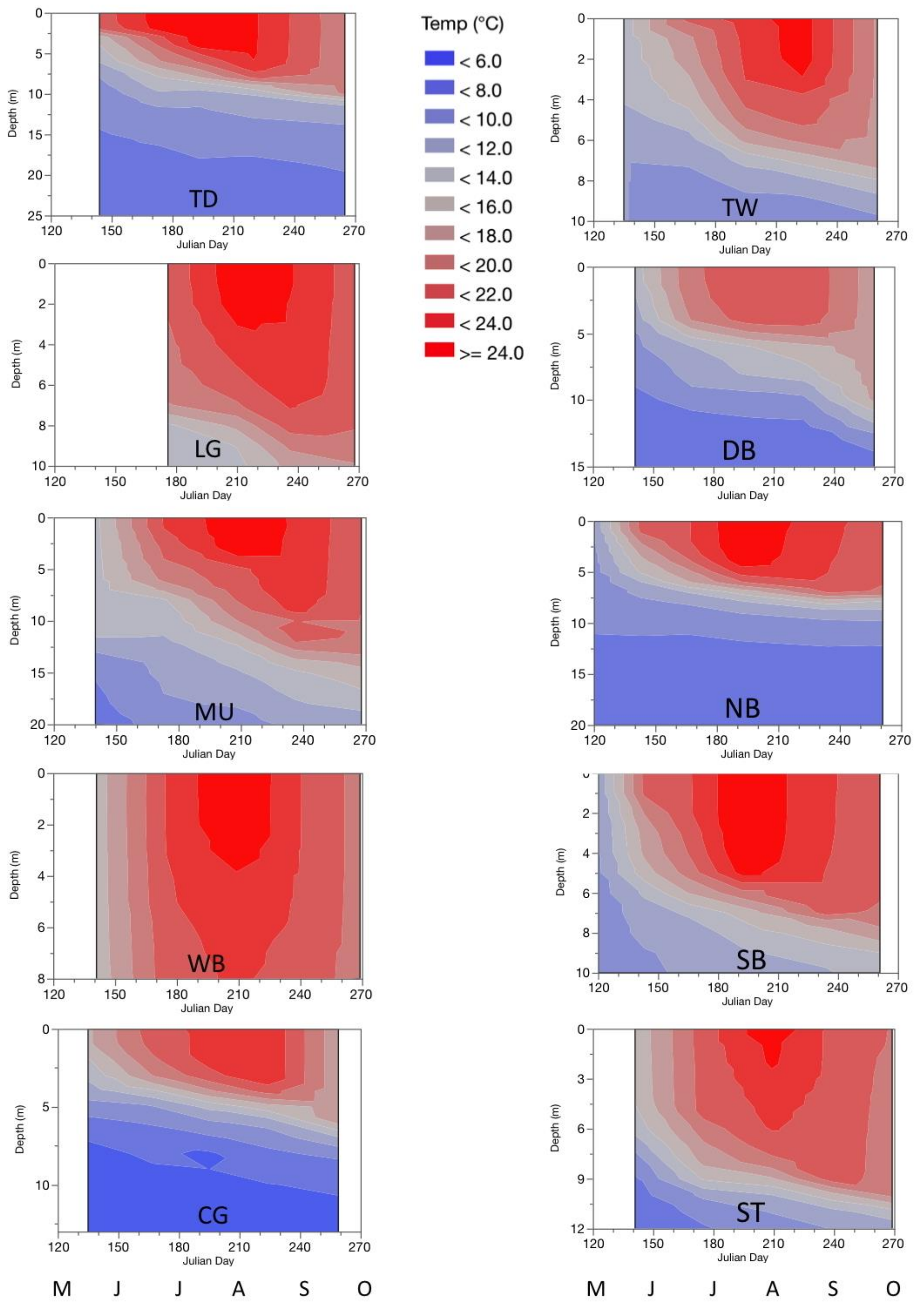


Figure 1.3: Temperature isopleths for each embayment in this study.

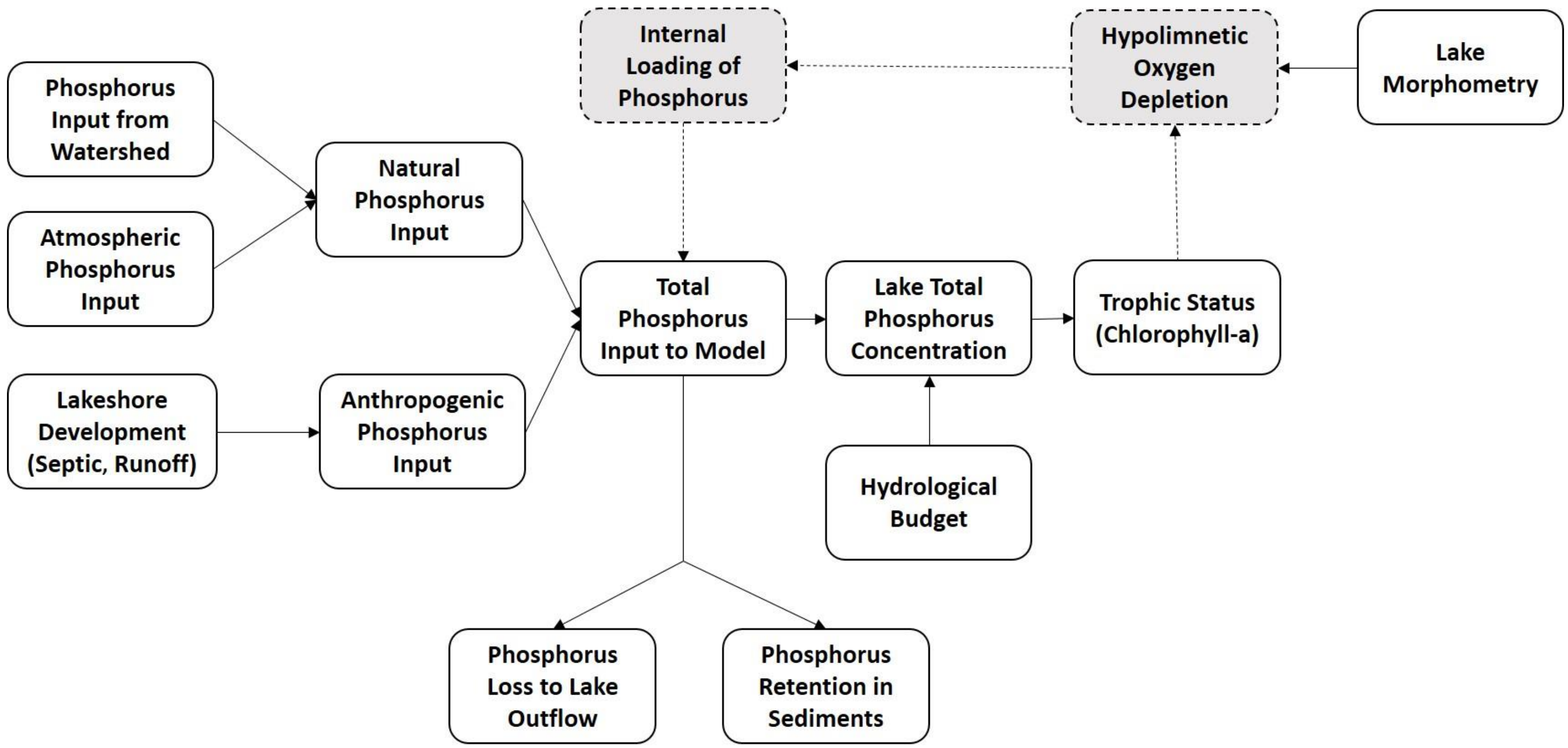


Figure 1.4: Conceptual model (modified from Paterson et al. 2006) showing components of the Lakeshore Capacity Model (LCM). White boxes and solid lines represent original model components and the grey boxes and dashed lines represent additional calculation of internal load (i.e. release of phosphorus from anoxic sediments).

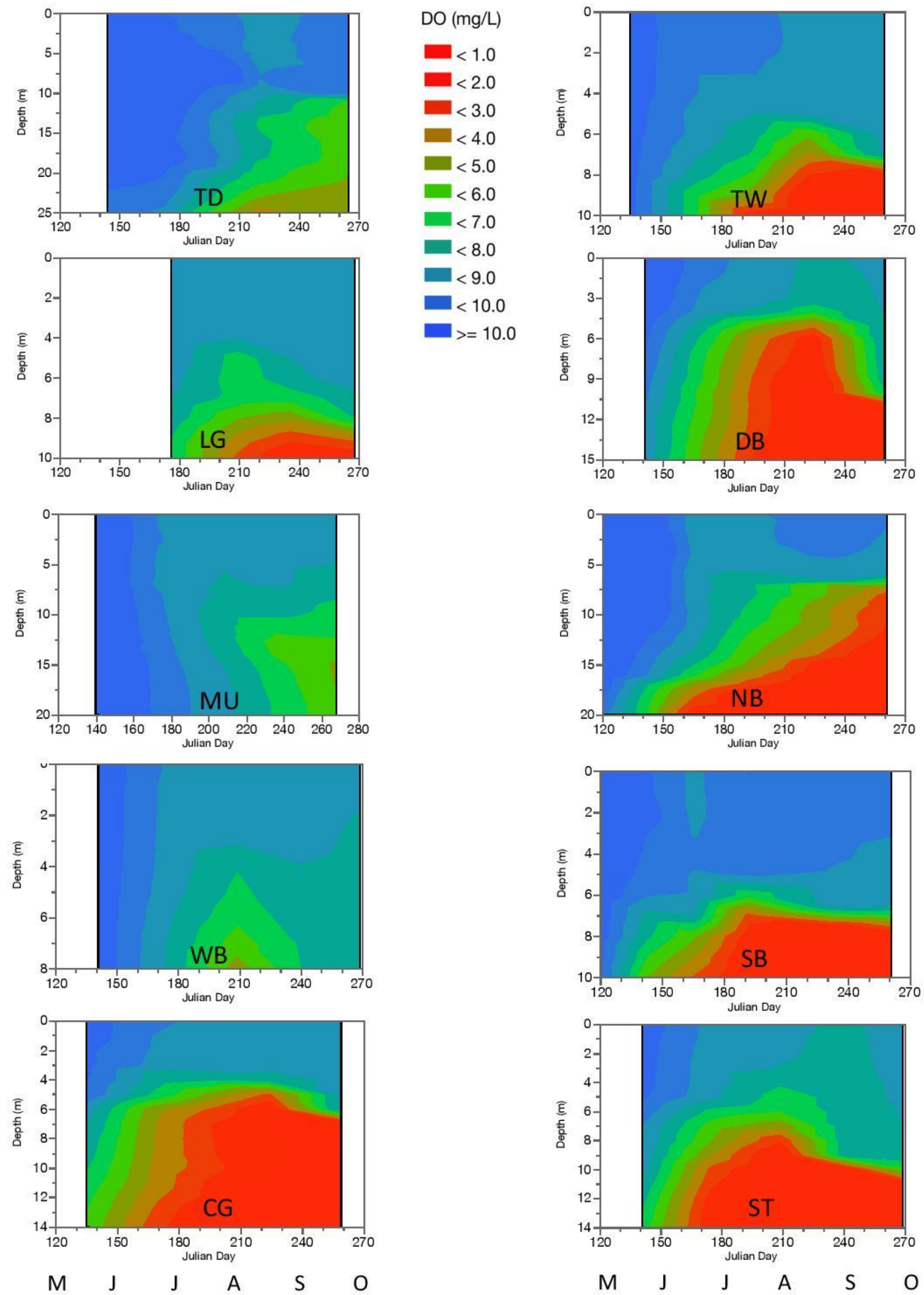


Figure 1.5: Dissolved oxygen (DO) isopleths for each embayment in this study

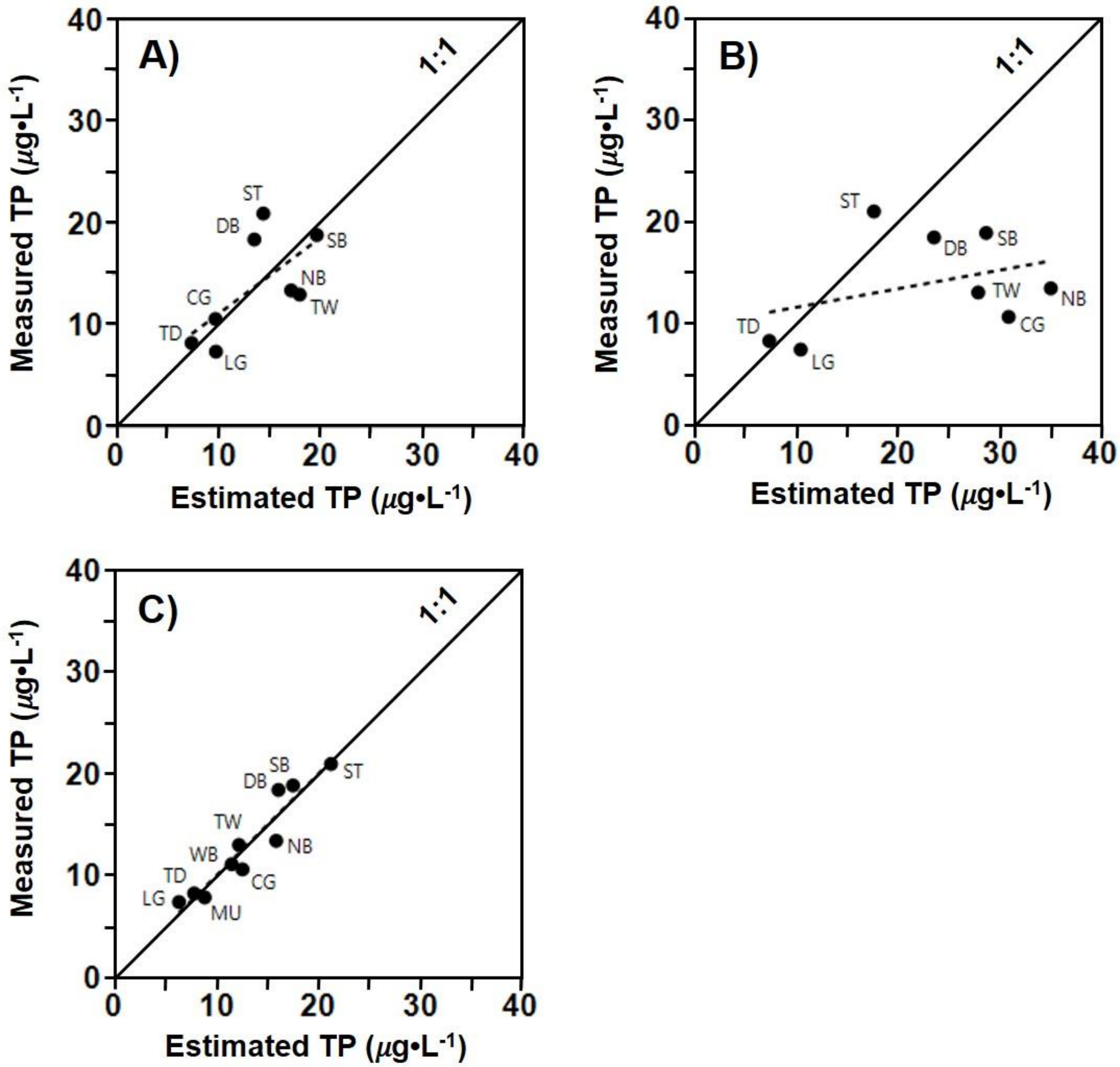
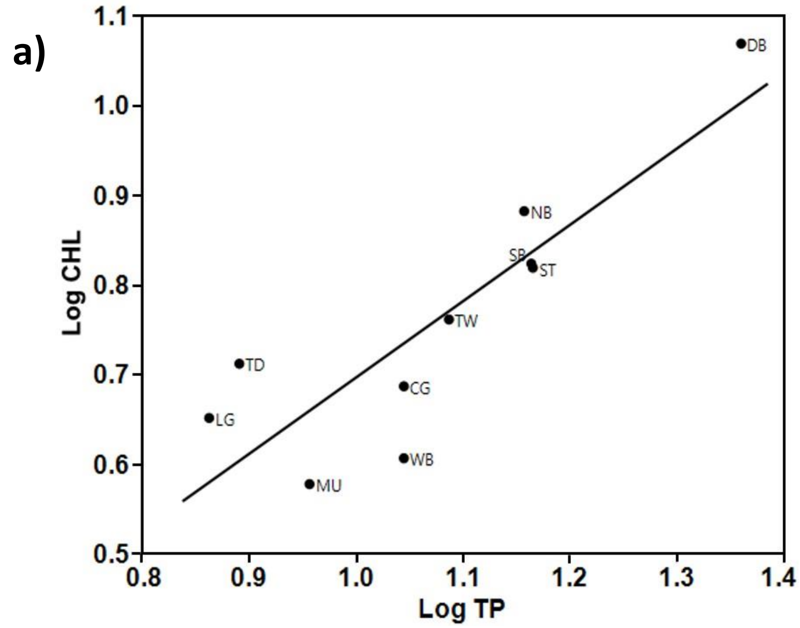


Figure 1.6: Measured vs. estimated scatterplots for A) Lakeshore Capacity Model (LCM), B) Lakeshore Capacity Model with inclusion of internal loading (LCM+IL) and C) Anthro-geomorphic Model (AGM). Solid line is the line of unity (measured=estimated). Dotted line represents best-fit linear regression of the data.



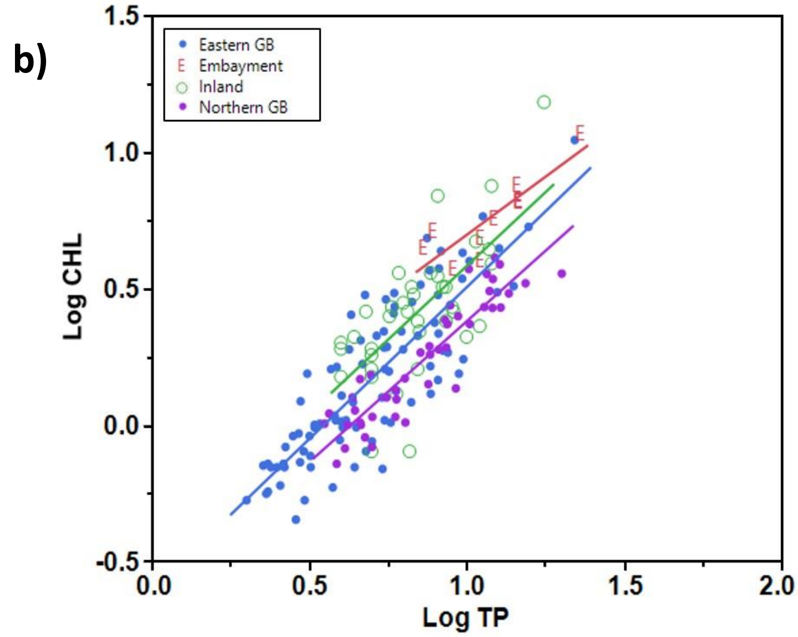


Figure 1.7: Plot of mean seasonal epilimnetic $\log_{10}$CHL (μg•L^{-1}) vs. mean seasonal epilimnetic $\log_{10}$TP (μg•L^{-1}) for A) of ten embayments in this study and B) comparison of ten embayments in this study with data from other studies. See Table 11 for information relating to data sources.

Chapter 2: Examining the general impact that cottage development has on coastal wetland water quality in the Township of Georgian Bay

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Key words: Georgian Bay, coastal wetland, anthropogenic disturbance, Water Quality Index, Laurentian Great Lakes

Abstract

Georgian Bay is home to some of the most pristine coastal wetlands in the Great Lakes basin. This is largely because most of its watershed is not suitable for agricultural production; therefore, Georgian Bay has undergone different patterns of development than experienced in the lower Great Lakes (e.g. minimal agricultural and urban development). The human development that does occur in this region consists primarily of recreational cottages and marinas. In some areas, where these developments occur in high densities along the shoreline, coastal wetlands have shown signs of water quality deterioration. The main objective of this study is to explore the spatial association between densities of building, dock and road development, and Water Quality Index (WQI) scores, an index designed to evaluate wetland condition in the Great Lakes, for 61 coastal wetlands in the Township of Georgian Bay. First, I found an inverse relationship between WQI scores and building, dock and road densities inside wetland catchments, indicating that these stressors do have a negative impact on wetland water quality. I then created a series of mapping products that present building, dock and road densities, along with WQI scores for 61 sites, to determine how wetland water quality is spatially associated with densities of these stressor variables. I found that regions with high densities of building, dock and road development were associated with wetlands of lower quality, whereas areas that had low densities of these stressors typically had wetlands of high quality. I used this information to identify areas of conservation priority for management in the Township of Georgian Bay. This study should be

used by township managers to make informed decisions regarding coastal development to preserve the pristine water quality conditions currently found in coastal wetlands of this region.

Introduction

Coastal wetlands provide many valuable services for both humans and wildlife including sediment and nutrient retention, water storage and groundwater recharge, and provide habitat for ecologically diverse flora and fauna (Keddy et al. 2009). Despite these benefits, destruction of these economically and ecologically important systems continues in the Great Lakes basin (Detenbeck et al. 1999). It is estimated that, since European settlement, over 70% of the original wetland area in southern Ontario has been lost due to human activities (Ball et al. 2003; Ontario Ministry of Natural Resources and Forestry 2015). The loss of coastal wetland habitat can be attributed to drainage for agricultural production, infilling for urban development and alteration for recreational development like cottages and marinas (Dodge and Kavetsky 1995). Of the coastal wetlands that do remain in the southern Great Lakes, the majority are in a degraded state due to human disturbance within their catchments (Crosbie and Chow-Fraser 1999; Chow-Fraser 2006).

The continued degradation and destruction of coastal wetlands in the Great Lakes due to human activities has led to a large body of literature devoted to identifying stressors that impact the condition of coastal wetlands (Crosbie and Chow-Fraser 1999; Danz et al. 2005; Danz et al. 2007; Chow-Fraser 2006; Trebitz et al. 2007; Morrice et al. 2008). These studies have found negative correlations between wetland water quality and anthropogenic stressors including land use (e.g. agricultural and urban), population density, atmospheric deposition and point source pollution (Crosbie and Chow-Fraser 1999; Chow-Fraser 2006; Trebitz et al. 2007; Morrice et al. 2008). The study of these stressors to quantify the impact that human disturbance has on wetland water quality is appropriate for the lower Great Lakes because these regions are home to high human population densities and agricultural development. In the upper Great Lakes, however, these approaches might not be appropriate due to the absence of these types of stressors.

Eastern Georgian Bay is one area where factors such as agricultural and urban development, population density, atmospheric deposition and point source pollution may not be appropriate sources of human disturbance on wetland water quality. There is neither high population density nor intensive agricultural or urban development in this region (Campbell 1974). While eastern Georgian Bay was once the site of intensive resource extraction, this is no longer the case (Campbell 1974; Sly and Munawar 1988; Weiler 1988). Furthermore, the thin acidic soil conditions of inland Georgian Bay are not suitable for agricultural production (Weiler 1988); thus, the region's forested landscapes have seen minimal amounts of human disturbance over the past century. Presently, the eastern shore of Georgian Bay has low population densities and is relatively undeveloped in comparison to areas around the lower Great Lakes; yet some of the wetlands, particularly located along the south-east coast, have shown signs of water quality deterioration (Chow-Fraser and Croft 2015). It is therefore necessary to identify the unique anthropogenic stressors that are impacting wetland water quality in eastern Georgian Bay.

The human development that exists in eastern Georgian Bay consists primarily of cottages, resorts and marinas, which have all become more popular over the past century (Township of Georgian Bay 2014). Recreational cottage development can impact coastal water quality through the loading of primary nutrients via septic system effluent (Joy 2013). Generally, septic systems are effective at treating incoming waste water; however, on Precambrian Shield landscapes, septic systems are constructed atop thin soil structures underlain by granite bedrock that can prevent these systems from functioning properly (Scalf et al. 1977; Wilhelm et al. 1994). Septic systems have therefore been identified as the largest source of anthropogenic nutrient input on Precambrian Shield landscapes (Dillon et al. 1994; Robertson et al. 1998; Joy 2013). An additional problem associated with coastal building development is nutrient loading from fertilized lawns and gardens into nearby waters during precipitation events (Engel and Penderson 1998). When excess nutrients enter the waters of coastal wetlands, undesirable algal growth can occur, not only reducing the aesthetic appeal of coastal areas, but also their ecological integrity.

Recreational cottage development is often accompanied by several other associated anthropogenic stressors. Shoreline modification through the construction of docks and piers is commonly associated with development in coastal areas. The construction of docks or the alteration of the shoreline with retaining walls or artificial beaches can negatively impact both submergent and emergent vegetation (Beauchamp et al. 1994; Radomski and Goeman 2001; Thom et al. 2005). The installation of these structures can shade out aquatic plants and increase shoreline erosion by altering natural currents (Beauchamp et al. 1994). Coastal areas with high dock densities are also associated with increased boat traffic; such traffic can cause the re-suspension of bottom sediments through the turbulence created by propellers (Engel and Pederson 1998). As in the lower Great Lakes, most human development in eastern Georgian Bay is also associated with road infrastructure that provides access to the waterfront.

Intensified road development is known to be associated with related stressors such as increased housing development, population density, and a decrease in natural landscape features (Hawbaker et al. 2004; Wolter et al. 2006), all of which can have a negative impact on wetland water quality (Trebitz et al. 2007; Morrice et al. 2008). Even though road density in Georgian Bay is much lower than that in the lower Great Lakes (DeCatanzaro and Chow-Fraser 2010), road development can still have a negative relationship with wetland water quality (DeCatanzaro et al. 2009) and is a significant threat for the coastal region of eastern Georgian Bay because it can lead to cottage and marina development in previously undisturbed areas. Roads can also alter the hydrology of streams because of their impervious surfaces and fragment natural habitats, as well as increase water turbidity through sedimentation in their runoff (Nelson and Booth 2002).

In light of the above, I believe that three significant anthropogenic stressors impacting water quality in the coastal wetlands of eastern Georgian Bay are recreational building, dock and road development; however, not all anthropogenic stressors influencing water quality are physical features of the landscape, and water levels are no exception. Historically, water levels in Lake Huron-Georgian Bay have fluctuated on multi-year cycles (~10 every years; Hanrahan et al. 2010). During the period from 1999 to 2013, however, Georgian Bay experienced a period of prolonged low water levels. More recently, water levels have rebounded above the historic average and have remained elevated for two years (2014 – 2016). A study by Boyd and Chow-Fraser (unpub. data 2016) found a statistically significant increase in mean Water Quality Index (WQI; explained below) scores for wetlands in Georgian Bay sampled during the period of prolonged low water levels to the recent period of high water levels. They also found that there were significantly lower total phosphorus ($\mu g \bullet L^{-1}$) and total suspended solids ($mg \bullet L^{-1}$) concentrations in these wetlands during the high water period compared to the low water period. These differences are thought to be attributable to the dilution of wetland waters with the open waters of Georgian Bay. It is therefore important to consider the influence that water levels might have on coastal wetland water quality when using long-term datasets that span both of these water level periods.

The Water Quality Index (WQI) developed by Chow-Fraser (2006) is a tool that can be used to monitor and track changes in wetland water quality in response to human activities. The WQI was designed specifically for coastal wetlands of the Great Lakes and has been found to be significantly correlated with the amount of altered land within wetland catchments (Chow-Fraser 2006; DeCatanzaro et al.

2009). The index combines 12 water quality parameters to generate a relative score of wetland condition ranging from -3 to +3, with low values indicating poor condition and high values indicating better condition. The range in values can be separated into the following six categories to objectively rank wetland condition: highly degraded (-3 to -2), very degraded (-2 to -1), moderately degraded (-1 to 0), good (0 to +1), very good (+1 to +2) and excellent (+2 to +3). The WQI allows environmental managers to categorize wetlands based on varying states of degradation and monitor changes in water quality over time.

The main objective of this study is to explore the spatial association between densities of building, dock and road development, and coastal wetland quality in the Township of Georgian Bay. This analysis will allow me to identify areas where coastal wetlands are at risk of becoming degraded due to these types of development, as well as areas that are home to high quality wetlands that should be preserved. I use the WQI scores for 40 coastal wetlands sampled during the summers of 2015 – 2016, and after controlling for the effect of water level change, I incorporate an additional 21 sites into my analysis that were sampled during the low water period, in order to create a dataset of 61 sites within the boundary of the Township of Georgian Bay. I then regress WQI scores against building, dock and road densities within the primary catchment of each wetland, in order to determine whether these stressors do in fact negatively impact coastal wetland water quality. Finally, I create a series of maps that show the spatial pattern in WQI scores with building, dock and road densities. This is one of a few studies that directly link recreational activities to water quality impairment in coastal wetlands of eastern Georgian Bay and should allow township managers to make better planning decisions to preserve the pristine water quality conditions currently found in coastal wetlands of this region.

Methods

Study Area

The inland forests of eastern Georgian Bay have been relatively free of human disturbance since the collapse of the logging industry in the early 1900's (Weiler 1988). As a result of the historical logging of this region, the forests of eastern Georgian Bay are considered to be second growth (Weiler 1988). Sugar maple (Acer saccharum), american beech (Fagus grandifolia), yellow birch (Betula alleghaniensis), eastern hemlock (Tsuga canadensis), eastern white pine (Pinus strobus), northern red oak (Quercus rubra) and white ash (Fraxinus americana) dominate the interior forests, while near the thin-soiled rocky coastline there are scrubby stands of iconic white pine (Pinus strobus), trembling aspen (Populus tremuloides), northern red oak (Quercus rubra) and white birch (Betula papyrifera) (Rowe 1972). The soils of eastern Georgian Bay are dominantly classified as course textured, having good drainage and Precambrian rock at one foot or less (Canada Department of Agriculture, 1960, Weiler 1988), with soil structures becoming thinner near the coastline.

The coastline of eastern Georgian Bay has been shaped by past glaciations and differential weathering processes to form a convoluted shoreline (McCarthy and McAndrews 2012) that has many finger-like embayments that terminate in coastal wetlands. The variable morphology of the coastline provides shelter from wind and wave energy, allowing for the development of many small coastal wetlands that form larger complexes when in close proximity to each other (Midwood et al. 2012). These wetlands are quite diverse in typology ranging from exposed to sheltered and from lacustrine to riverine. Many of these systems have low nutrient concentrations due to the open connection they share with the oligotrophic waters of Georgian Bay and the relatively undisturbed natural conditions in their catchments (DeCatanzaro and Chow-Fraser 2011).

Our study site is located in the Township of Georgian Bay, which is situated along the south-east shore of Georgian Bay, at the entrance to the 30 000 Islands Archipelago (Figure 2.1). This region is ecologically important, having high levels of biodiversity and a highly productive recreational fishery of walleye, northern pike and bass, as well as a world-class trophy fishery for muskellunge (Weller et al. 2016). There are two main communities located along the coastline within the Township of Georgian Bay; Port Severn is positioned to the south, at the mouth of the Trent Severn Waterway, and Honey Harbour is located further north along the coastline. These communities have permanent populations of 657 and 268, respectively (Township of Georgian Bay 2014). The Township of Georgian Bay is located close to the populated urban areas of southern Ontario and is a popular summer vacation destination with 86% of its total population classified as seasonal residents and 13.5% classified as permanent (Township of Georgian Bay 2014). As a result, there is a relatively large number of cottages and marinas located along the coastline. This area has the highest proportion of properties that are only accessible by water (55%) in the District of Muskoka and the lowest proportion of property accessible by roads (14%); therefore, residents of the Township of Georgian Bay are reliant on boats for transportation with 95% of households owning one or more vessels (Township of Georgian Bay 2014). The Township of Georgian Bay was chosen for this study because it has high levels of cottage and marina development, as well as abundant coastal wetlands situated along its coastline, some of which might be at risk of becoming degraded as a result of lakeshore development.

Water Quality Index Data

Sites included in this study were selected from among a larger dataset of WQI sampling sites located throughout Lake Huron. I imported these data into ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA) in order to select all sites inside the geographic boundary of the Township of Georgian Bay (Figure 2.1). A total of 61 sites were identified within this region, with data covering a sampling period that ranged from 2003 to 2016 (Figure 2.1; Table 2.1). It is important to note that 40 of these 61 sites were sampled at least once during 2015-2016 and of these, 20 were sampled in 2016 specifically for inclusion in this project. I chose to include only the most recent WQI sampling data for the 40 sites that were collected in 2015 and 2016. Data from the remaining 21 sites, which were sampled between 2003 and 2010, were included after I adjusted for the effect of water level differences between the prolonged low water period that ended in 2013 and the more recent high water period.

Controlling for Water Level Influences on WQI Scores

A study by Boyd and Chow-Fraser (unpub. data 2016) found that WQI scores collected during the low water period were significantly lower compared to WQI scores collected during the high water period in Georgian Bay. This poses a problem in that, if water level differences were not controlled for, I would have risked falsely associating the lower WQI scores as being a result of lakeshore development in sites sampled during low water years. In order to control for the change in water level, I needed to examine a set of WQI sampling sites at which data were gathered during both the low water level period and the high water level period, and determine the amount by which the mean of WQI scores increased between these two periods. Once I had determined the exact increase in mean WQI scores, I could control for this effect by adding the mean increase to each of the 21 sites for which data were only collected during the low water level period.

I defined the low water level period in Georgian Bay as spanning from 1999 to 2013 and the high water level period as from 2014 – 2016 (Figure 2.3). There were 17 wetlands among the 61 WQI sampling sites in the Township of Georgian Bay that had been sampled during both water level periods (DV, GY, GI, HRM, LSP, MNC, ME, MO, MS, NB, OB, OJ, QI, RB, TD1, TD2 and TB; see Table 2.1). I used the data from these 17 sites to calculate the difference between mean WQI scores from the low and high water level periods. A paired t-test was first used to determine if there was a significant difference in mean WQI scores between the two water level periods; the increase in mean WQI scores between the two periods was then added to the scores of the 21 sites sampled during the low water level period. Controlling for changes in water level allowed me to use a more robust dataset including all 61 WQI sampling sites inside the Township of Georgian Bay.

Calculation of Development Densities

I wanted to determine if buildings, docks and roads had a negative impact on wetland water quality, in order to justify including these variables in mapping products that demonstrate the spatial association between these stressors and WQI scores. I quantified densities of development as the number of “development units” (i.e. buildings, docks, or road length (km)) per catchment unit area (ha) or shoreline perimeter (m). I used the McMaster Coastal Wetland Inventory (MCWI), which is an inventory of coastal wetlands in eastern Georgian Bay (Midwood et al. 2012), to determine the physical characteristics of each wetland included in this study. I imported the MCWI into ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA) to measure the wetland shoreline perimeter and the area of each of the 61 wetlands. Total wetland area was calculated as the area of both low-marsh and high-marsh zones combined. Wetland shoreline perimeter was calculated as the total shoreline length (km) of each wetland (Figure 2.2). I delineated the local catchments of each wetland using

ArcGIS 10.3.1 Hydrology Toolset (ESRI™, Redlands, CA, USA) and a digital elevation model (DEM) with 10 m accuracy (Ontario Ministry of Natural Resources and Forestry 2013). I defined local catchment as the area drained by the wetland (Figure 2.2).

To enumerate the number of buildings and docks inside each wetland catchment, I imported high resolution (1 m panchromatic) IKONOS satellite image data acquired in 2002, into ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA). All residential structures (e.g. cottages, trailers) located within the local catchment of each wetland were enumerated. In addition to IKONOS imagery, I used Google Earth™ satellite imagery taken in 2015 to update the database and to ensure that features were properly enumerated in the event of seasonal changes in imagery acquisition between these two data sources. Building density was then calculated as the number of buildings per watershed unit area (ha). The number of docks along the wetland perimeter were enumerated with the same methods as building density. Dock density was calculated as the number of docks per wetland perimeter (m). To calculate road density, I imported the Provincial Road Network Layer (Ontario Ministry of Natural Resources and Forestry 2012) into ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA) and used the Geo-processing Clip Tool to calculate road length (km) per unit area (ha) of the watershed.

Creation of Stressor Density Maps

I imported geodatabases that accounted for the number of buildings and docks located along the shoreline of the Township of Georgian Bay into ArcGIS 10.3.1 (ESRI™, Redlands, CA, USA) to create maps illustrating the spatial distribution of these features along the coastline. I used the Point Density Spatial Analyst tool to calculate raster layers that represented the density of buildings within a 300 m buffer of the shoreline, as well as dock density along the shoreline. I then used the Line Density Spatial Analyst tool to calculate a road density raster layer using the Provincial Road Network Layer (Ontario Ministry of Natural Resources and Forestry 2012) that represented road density throughout the entire Township of Georgian Bay.

Building density was calculated as the number of building units in a 1000 m radius of each 10 x 10 m^2 raster cell. The building geodatabase that was used only included building development within a 300 m buffer from the shoreline; therefore, this map only represents building density along the coastline. Dock density was calculated with the same method and represents the number of dock units in a 1000 m radius of each 10 x 10 m^2 raster cell. Road density was calculated as the length of road (km) in a 1000 m radius of each 10 x 10 m^2 raster cell. These layers were then overlain on a map with WQI scores to make general observations relating to water quality conditions and proximity to these stressors.

Results

Difference in WQI Scores between Water Level Periods

Consistent with the findings of Boyd and Chow-Fraser (unpub. data 2016), I found that there was a significant difference between the mean WQI score for data collected during the low water level period (M = 1.508, SD = 0.29) and the mean WQI score for data collected during the high water level period (M =1.895, SD = 0.28) for the 17 sites compared (paired t-test, t = 3.89, P = 0.0005; Figure 2.4). This difference represented a mean increase in WQI scores of 0.387 from the low water period to high water period. In order to control for the effect of water level change on WQI scores for sites sampled during the low water period, I added the difference (0.387) to the WQI scores of each of the 21 sites (BBB, BS2, BST, CD, GI2, HS, LG, LY1, MB2, MF, MOB2, MPN, MPS, NB1, OCL, PMP, RB1, RB2, RB3, ROS, TWB; Table 2.1). By controlling for water level differences I was then able to include into my analysis the 21 sites that were sampled during the low water period.

Landscape Features, Cottage Development and WQI Scores

Watershed area for the 61 coastal wetlands included in this study varied considerably, ranging from 3.95 ha to 1750.59 ha, with an average area of 119.29 ha (Table 2.1). Wetland area (low-marsh + high-marsh) varied from a minimum of 0.08 ha to a maximum of 39.86 ha, with an average area of 7.59 ha. Human disturbance inside wetland catchments was also variable, with 22 wetlands having no development and 39 having at least one form of development (e.g. buildings, docks or roads). WQI scores for the 61 coastal wetlands were relatively high, ranging from +0.440 to +2.868, with an average of +1.947 (Table 2.1; Figure 2.5). Overall, wetlands in the Township of Georgian Bay did not show signs of serious water quality degradation; 26 sites classified as "excellent", 34 classified as "very good" and one classified as "good".

Correlation between WQI Scores and Building, Dock and Road Densities

I wanted to explore the relationship between coastal wetland WQI scores and densities of development (building, dock and road) to determine if these features do in fact have a negative impact on wetland water quality. I hypothesized that coastal wetlands with greater densities of development in their primary catchments and along their shorelines would have lower WQI scores than wetlands that have little or no such development. Using best-fit linear regression, I found significant negative relationships between WQI scores and building density (n = 61; r^2 = 0.14; P = 0.0025; Figure 2.6a), dock density (n = 61; r^2 = 0.19; P = <0.0005; Figure 2.6b) and road density (n = 61; r^2 = 0.13; P = 0.0047; Figure 2.6c). These results are consistent with my hypothesis that greater densities of development in the primary watersheds of coastal wetlands have a negative impact on water quality, and provide justification for using these variables to model the spatial associations between WQI scores and anthropogenic stressors.

Spatial Association between Features of Cottage Development and WQI Scores

Examining the density maps, it is apparent that each stressor (i.e. building, dock or road) is present at higher densities in the southern portion of the Township of Georgian Bay near Port Severn and Honey Harbour. Building, dock and road densities decreased as latitude increased with pockets of high densities located only at the end of Twelve Mile Bay (Figures 2.7, 2.8 and 2.9). It is also clear that WQI scores followed a similar pattern; WQI scores were lowest in the southern portion of the township and generally increased moving north along the shoreline. Thus, poor quality wetlands were generally found in areas that had high densities of development, and high quality wetlands were generally found in areas with lower densities of development. More specifically, lower quality wetlands appeared to be concentrated along the coast from Port Severn in the south to Honey Harbour in the north, with an additional cluster of lower quality sites at the end of Twelve Mile Bay (Figures 2.7, 2.8 and 2.9). LY1, a site situated in the centre of Honey Harbour, had the lowest WQI score in this study (WQI: >0.50; Figure 2.5; Table 2.1) and is surrounded by the highest densities of dock and building development, as well as high road densities.

Conversely, coastal wetlands of higher quality were generally located along the stretch of shoreline north of Honey Harbour, extending to Tadenac Bay (Figure 2.5). There is a cluster of high quality sites located in Tadenac Bay (WQI: >1.5; Figure 2.5), which is an embayment situated south of Twelve Mile Bay, and a grouping of high quality sites located in Musquash Bay, which is north of Honey Harbour. Another grouping of high quality sites is also found at the northernmost extent of the township, at the mouth of the Moon River (WQI: >2.5; Figure 2.5). All of these high quality sites are located where road access does not reach the shoreline (i.e. there are no roads) and are surrounded by low densities of building and dock development. It is clear from examining the spatial associations between densities of building, dock and road development that wetlands near high concentrations of these stressors are generally of lower quality than wetlands subject to low or no densities of development.

Discussion

Coastal wetland water quality in the Township of Georgian Bay is generally in exceptional condition compared to wetlands located in the lower Great Lakes (DeCatanzaro et al. 2010; Cvetkovic and Chow-Fraser 2011); all WQI scores for the township fall within the “good” (0 to +1), “very good” (+1 to +2) and “excellent” (+2 to +3) ranges, regardless of whether water level differences are controlled for. Despite this, I have demonstrated that there is an inverse relationship between WQI scores and densities of building, dock and road development (Figure 2.6). This relationship indicates that the human development in this area has a negative impact on water quality in coastal wetlands. Examination of the spatial distribution of densities of building, dock and road development supports the hypothesis that wetlands of lower quality are generally located in close proximity to high densities of development. These negative associations should be of great concern to local residents and township managers who rely on the preservation of good water quality for consumption, as well as recreational enjoyment.

The negative association between high densities of development and lower quality wetlands allows certain wetlands within the Township of Georgian Bay to be identified as being vulnerable to symptoms of water-quality impairment if development continues to increase. Three main coastal areas require particular attention. The first region is the coastline that extends from Port Severn to Maceys Bay just south of Honey Harbour, which is subject to a high density of building development along the shoreline and a high density of road development (Figure 2.10). The wetlands along this stretch of shoreline are generally of lower quality as compared to areas that are exposed to minimal development (e.g. Tadenac Bay). The second area requiring attention is the area surrounding the town of Honey Harbour, which has the highest building and dock densities, as well as dense road development, and also sites of lower quality (Figure 2.7, 2.8, 2.9 and 2.10). The third area requiring attention is located at the end of Twelve Mile Bay near the community of Moose Deer Point First Nations. This region is subject to moderate building and road densities (Figures 2.7 and 2.9) as well as high levels of dock development due to the presence of several large marinas in the area (Figure 2.8 and 2.10), and is home to three sites of lower quality.

It is important to note that areas that have high building and dock densities are also areas that have reliable year round road access. Additionally, coastal wetlands that are of high quality are generally located in areas that are not accessible by road (e.g. Tadenac Bay, Musquash Bay, Moon River; Figures 2.9 and 2.10). Although WQI scores did not produce a strong regression with road density inside the primary catchments of coastal wetlands (Figure 2.6c), I would suggest that the development of roads along the Georgian Bay coastline is the determining factor of the amount of development in an area. This is because the development of road infrastructure is often associated with related stressors such as housing development, population density and a decrease in natural landscape features (Hawbaker et al. 2004; Wolter et al. 2006). It is important that township managers understand this association so that they may make informed decisions as to whether or not they will allow road infrastructure to be extended to the coastline in more remote regions of the Township of Georgian Bay. The extension of such infrastructure is likely to encourage rapid cottage development along the coastline, which may result in the degradation of high quality coastal wetlands within the township.

It is clear that wetlands in the Township of Georgian Bay that are located near the densely populated regions of Port Severn and Honey Harbour are of lower quality than wetlands located in more remote, sparsely populated areas. These patterns are particularly troubling because the area surrounding Port Severn and Honey Harbour is home to spawning habitat for the ecologically significant and economically important muskellunge (Weller et al. 2016), whose population is thought to be threatened by increased cottage and marina development, as well as inconsistent water level fluctuations (Leblanc et al. 2014). Coastal wetlands in the Township of Georgian Bay also provide critical habitat for rare and threatened species, such as the Blanding's turtle, during certain life stages (Markle and Chow-Fraser 2014). It is therefore important that the quality of these habitats are maintained to sustain reproductive populations of these and other ecologically significant species.

An additional factor that must be considered by township managers is the effect that water levels have on WQI scores. The findings of this study, which are consistent with the findings of Boyd and Chow-Fraser (unpub. data 2016), show that there was a significant increase in WQI scores between the low water period and the high water period in Georgian Bay (Figure 2.4). This information should be taken into consideration when township managers use water quality monitoring data as the basis for management decisions. The increase in WQI scores between these two water level periods could cause managers to falsely conclude that development of the lakeshore is not having a negative impact on water quality. If new development projects are allowed to move forward on this basis, this could result in further degradation of wetlands.

Management Implications

Information in this study should be used by managers of the Township of Georgian Bay as a resource to make informed decisions regarding the management of coastal wetlands within township boundaries. The maps that I have presented have been used to identify key areas where water quality may be at risk of further deterioration if development of the lakeshore is allowed to continue; I have also identified wetlands of high quality that should be given conservation priority by township managers (Figure 2.10).

One of the important patterns shown in this study is that all high quality wetlands were located in areas where there is no road access to the coastline. It seems that the development of road infrastructure to the coastline in the Township of Georgian Bay facilitates higher concentrations of recreational cottage development (i.e. Twelve Mile Bay, Honey Harbour Port Severn; Figure 2.10), which is consistent with other studies (Hawbaker et al. 2004; Wolter et al. 2006). Therefore, road development should not be extended to additional regions along the coastline in the Township of Georgian Bay.

The locations in the Township of Georgian Bay that I have shown to have high development densities and wetlands of lower quality should be given high priority by township managers in terms of conservation efforts. Although the WQI scores of these sites indicate that they are currently considered to have relatively good water quality, continued lakeshore development could degrade these wetlands to a point where they do not perform the ecosystem services that are valuable to both humans and wildlife alike. A prime example of these ecosystem services is seen in the coastal wetlands near Honey Harbour and Port Severn (Figure 2.10), which have been identified as important spawning and nursery habitat for muskellunge. These coastal wetlands are thus needed to sustain the world-class trophy fishery for this species in Georgian Bay (Leblanc et al. 2014; Weller et al. 2016). As shown in this example, the preservation of coastal wetland health is not only important because of the ecological benefits of high quality wetlands, but also for their economic benefits.

Maintaining good coastal water quality is essential for both the ecological and economic well-being of the Township of Georgian Bay. A large portion of the township's economy is reliant on the cottage and boating industries that not only depend on coastal waters for consumption, but also for aesthetic and recreational enjoyment. Although wetland water quality in the Township of Georgian Bay is generally in exceptional condition, this study has shown that features associated with cottage development (i.e. buildings, docks and roads) have a negative impact on coastal wetland water quality. Further, the maps provided indicate that wetlands in close proximity to regions with high densities of lakeshore development are generally of lower quality. This information should be considered by township managers before making decisions regarding coastal development in the Township of Georgian Bay. The continuation or intensification of development in these regions could further degrade these wetlands and inhibit their ability to perform the critical services that are important for both humans and wildlife.

Acknowledgements

I would first like to thank Alex Furukawa for all of the preliminary work he completed for this project in his undergraduate thesis; Alex, your research formed the foundation for this project. I am also grateful for the guidance and mentorship of my supervisor, Dr. Patricia Chow-Fraser. I am particularly appreciative that she took the time to review this manuscript. Thank you to Lindsay Boyd for all of her help in the field, as well as in the lab processing water samples. Thank you to Gillian Fraser, Shah Ameri, James Marcaccio, Dan Weller and Nick Luymes for their assistance in the field. I would like to thank Mary Muter and Jean Demarco for the use of their lovely cottages. Finally, thank you to all of the Chow-Fraser lab members, both past and present, for contributing to the ever growing WQI database that was used in this project.

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Table 2.1: List of 61 coastal wetlands, along with wetland codes, location (latitude and longitude), n: sampling events, Water Quality Index (WQI) scores, summary of cottage development densities (Building Density: primary residential structures • watershed area (ha)$^{-1}$; Dock Density: docks • shoreline perimeter (m)$^{-1}$; Road Density: road length (km) • watershed area (ha)$^{-1}$) and landscape characteristics (Wetland Area (ha); Shoreline Perimeter (km); Watershed Area (ha)) for each coastal wetland sampled for WQI in the Township of Georgian Bay. Note: * represent sites controlled for the effect of water level.

Wetland Code	Latitude	Longitude	n	WQI	Building Density	Dock Density	Road Density	Wetland Area	Shoreline Perimeter	Watershed Area
BBB*	45.03004	-79.98555	1	2.821	0.0000	0.0000	0.0000	6.43	1.30	308.09
BS2*	44.85023	-79.86250	1	1.765	0.2116	0.0022	0.0000	2.52	0.48	9.45
BST*	44.88303	-79.86918	1	2.099	0.0000	0.0000	0.0000	10.05	0.46	85.38
CD*	45.04691	-79.98742	1	1.787	0.0000	0.0000	0.0000	7.24	0.54	68.01
DV	45.04762	-80.00144	2	1.785	0.0000	0.0000	0.0000	3.80	3.42	16.57
GBT 01	45.08812	-80.04480	1	1.169	0.4292	0.0043	0.0217	7.48	0.78	51.25
GBT 03	45.04294	-79.98991	1	2.326	0.0000	0.0000	0.0000	3.20	1.36	27.42
GBT 12	45.08265	-79.92751	1	2.057	0.0000	0.0000	0.0184	7.55	1.44	300.18
GBT 15	44.99543	-79.95271	1	1.436	0.0000	0.0000	0.0000	3.79	0.60	15.75
GBT 17	45.01336	-79.91174	1	1.852	0.0000	0.0000	0.0000	4.52	1.43	127.61
GBT 22	44.95901	-79.91196	1	2.701	0.0000	0.0000	0.0000	3.96	0.81	160.75
GBT 23	44.95680	-79.86123	1	1.403	0.0000	0.0000	0.0000	2.36	0.59	36.09
GBT 25	44.89011	-79.86346	1	2.235	0.0315	0.0009	0.0000	8.33	1.13	63.58
GBT 28	44.86894	-79.84777	1	1.865	0.2836	0.0033	0.0000	4.44	0.54	14.10
GBT 30	44.85257	-79.83461	1	2.006	0.4514	0.0000	0.0000	10.41	0.25	8.86
GBT 31	44.87122	-79.85341	1	1.899	0.0890	0.0000	0.0000	2.63	0.40	22.46
GBT 33	44.89133	-79.80402	1	1.968	0.0445	0.0061	0.0000	12.84	0.92	157.41
GBT 39	44.78334	-79.73915	1	1.641	0.3374	0.0014	0.0000	8.61	1.73	5.93
GBT 41	44.79957	-79.72628	1	1.950	0.0000	0.0000	0.0701	5.86	0.75	4.89
GBT 42	44.95197	-79.85162	1	2.365	0.1448	0.0020	0.0000	6.97	0.54	13.81
GBT 43	44.95052	-79.84019	1	2.445	0.0000	0.0000	0.0000	6.01	0.35	249.05

Wetland Code	Latitude	Longitude	n	WQI	Building Density	Dock Density	Road Density	Wetland Area	Shoreline Perimeter	Watershed Area
GBT 48	44.78725	-79.75720	1	1.760	0.0000	0.0000	0.0000	0.93	1.08	12.44
GBT 49	44.80130	-79.76312	1	1.554	0.5560	0.0038	0.0112	6.80	1.87	98.92
GBT 50	44.84198	-79.81953	1	2.402	0.7622	0.0044	0.0000	30.87	5.61	10.50
GBT 55	44.90139	-79.78426	1	1.435	0.2262	0.0030	0.0386	8.56	1.46	39.79
GI	44.78560	-79.74665	2	1.717	0.0000	0.0000	0.0000	18.26	1.02	15.04
GI2*	44.77892	-79.74610	1	2.868	0.0523	0.0000	0.0000	13.72	0.61	38.21
GY	44.92019	-79.81687	1	1.949	0.1915	0.0119	0.0000	0.75	1.32	31.33
HRM	45.08681	-79.99732	1	1.669	0.1083	0.0000	0.0103	3.47	0.72	55.39
HS*	44.94457	-79.86375	1	1.807	0.0000	0.0000	0.0000	0.43	0.46	3.95
LG*	44.96911	-79.89476	2	2.707	0.0025	0.0000	0.0000	9.77	3.42	400.46
LSP	44.97825	-79.92967	1	1.530	0.0000	0.0000	0.0000	3.73	2.42	57.40
LY1*	44.86974	-79.81767	2	0.440	2.1031	0.0394	0.0654	7.28	0.78	39.47
MB2*	44.80441	-79.76686	1	1.002	0.3049	0.0010	0.0170	0.51	1.36	272.22
ME	45.07054	-80.05065	1	1.612	0.0743	0.0010	0.0000	15.31	0.60	26.94
MF*	45.10710	-79.93031	1	2.690	0.0000	0.0000	0.0000	0.08	1.43	103.94
MNC	45.06198	-79.94571	1	2.439	0.0286	0.0000	0.0085	7.50	0.81	1750.59
MO	45.01289	-79.94363	2	2.119	0.0000	0.0000	0.0000	12.02	0.59	496.42
MOB2*	45.12552	-80.00972	1	2.536	0.0881	0.0015	0.0000	18.77	1.13	11.35
MPN*	44.81541	-79.78986	1	2.064	0.8632	0.0123	0.0471	2.78	0.25	13.90
MPS*	44.81297	-79.79282	1	1.671	2.2233	0.0066	0.0311	2.07	0.40	10.79
MS	44.81164	-79.78093	1	2.151	0.4044	0.0056	0.0361	5.05	1.73	37.10
NB	44.89257	-79.78781	1	1.833	0.0474	0.0055	0.0000	3.09	0.75	126.54
NB1*	44.89654	-79.79397	1	1.594	0.1676	0.0058	0.0102	7.25	0.54	95.47
OB	44.79788	-79.73443	1	2.012	0.7685	0.0121	0.0266	13.94	5.61	377.35
OCL*	44.95624	-79.93316	1	2.120	0.0000	0.0000	0.0000	2.47	1.46	30.62
OJ	44.88784	-79.85737	2	1.941	0.1022	0.0023	0.0000	2.03	0.87	19.56
PMP*	45.29767	-80.08439	1	2.420	0.0000	0.0000	0.0090	2.56	2.24	35.96
PO	45.05269	-80.00192	1	1.754	0.0000	0.0000	0.0000	39.86	1.32	73.17

Wetland Code	Latitude	Longitude	n	WQI	Building Density	Dock Density	Road Density	Wetland Area	Shoreline Perimeter	Watershed Area
QI	44.83615	-79.81256	1	1.452	0.2107	0.0020	0.0000	10.98	1.30	28.48
RB	44.85553	-79.83168	2	1.950	0.5437	0.0046	0.0000	16.61	0.48	18.39
RB1*	44.86850	-79.83112	1	2.283	0.6380	0.0077	0.0000	2.44	0.46	10.97
RB2*	44.86269	-79.83784	1	1.651	0.3940	0.0037	0.0000	9.63	0.54	30.46
RB3*	44.85780	-79.84208	1	1.841	1.0080	0.0083	0.0000	5.06	7.05	28.77
ROS*	44.99517	-79.92303	1	2.159	0.0061	0.0000	0.0000	4.36	0.56	164.23
TB	44.87037	-79.85922	2	1.562	0.1221	0.0009	0.0000	14.67	0.92	171.94
TD01	45.03534	-79.99205	1	2.217	0.0000	0.0000	0.0000	14.67	1.73	17.49
TD02	45.04285	-79.98498	1	2.279	0.0000	0.0000	0.0000	10.98	0.75	4.29
TD03	45.05426	-79.96496	1	2.461	0.0000	0.0000	0.0000	0.80	0.54	40.86
TWB*	45.08881	-80.01054	1	1.818	0.0000	0.0000	0.0035	3.27	0.35	394.93
VE	44.84123	-79.78009	2	1.762	0.1495	0.0018	0.0146	4.22	1.08	334.53

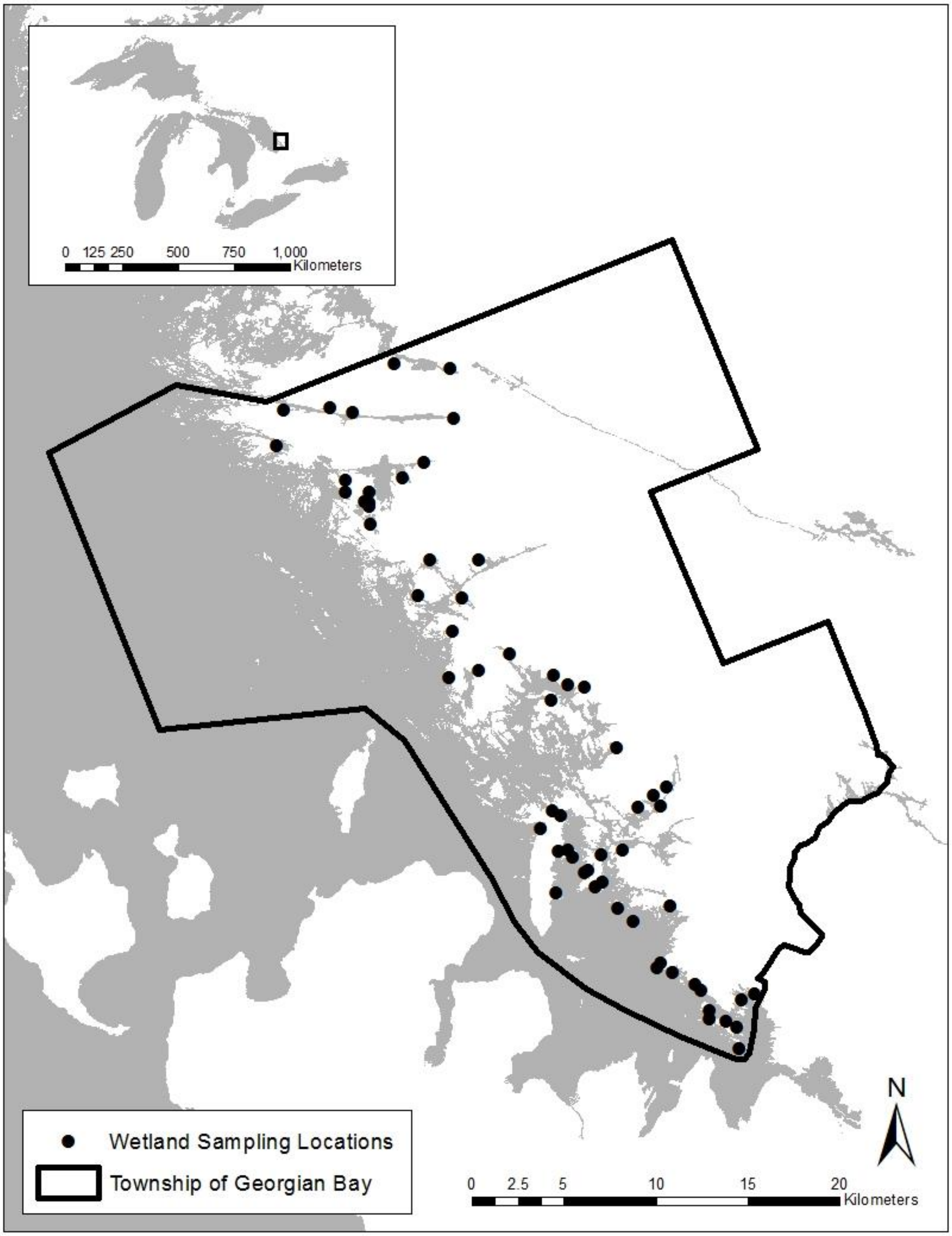

Figure 2.1: Location of coastal wetlands in the Township of Georgian Bay sampled from 2003 – 2016 for the Water Quality Index (WQI)

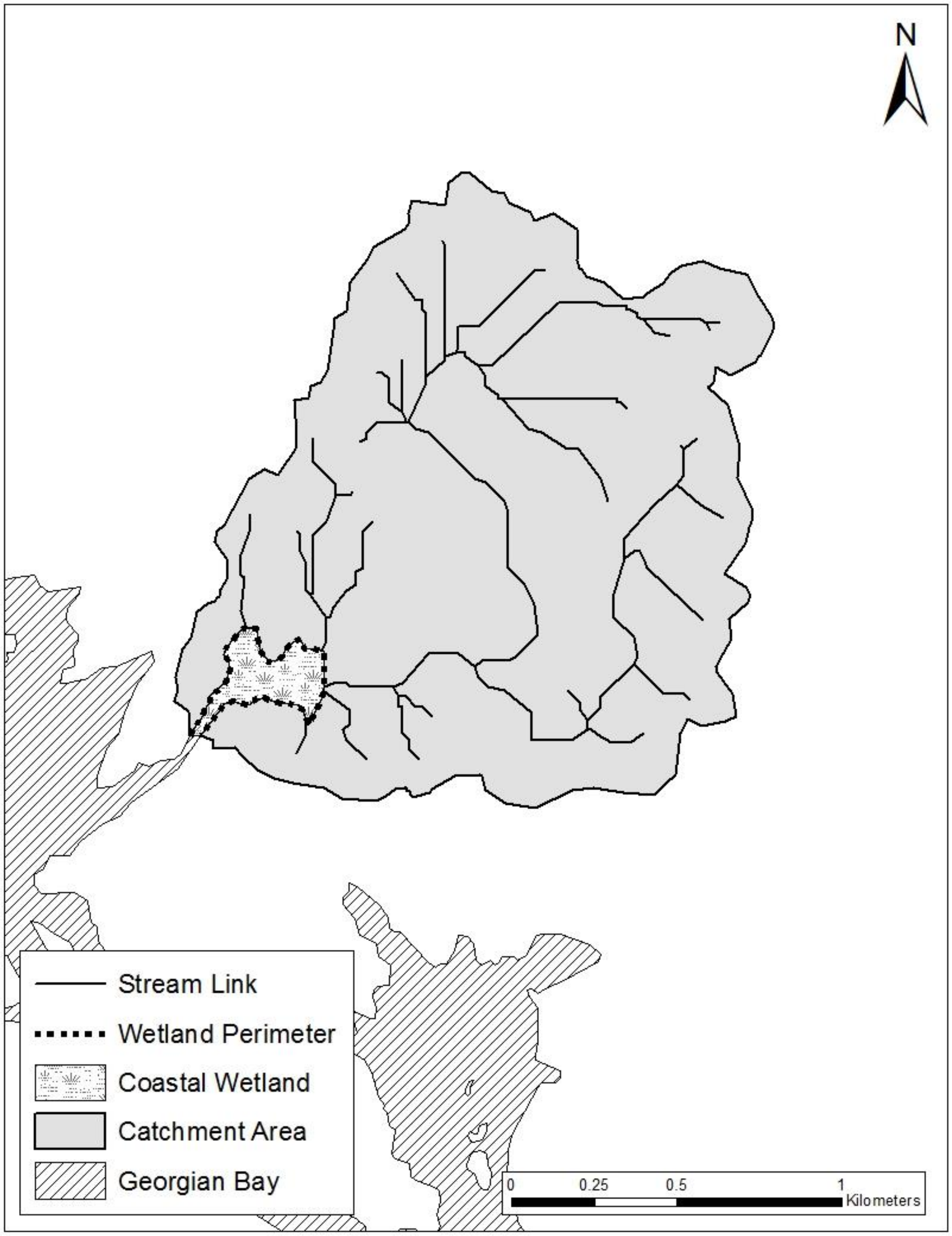

Figure 2.2: Map illustration of a primary coastal wetland catchment area (area that drains to wetland).

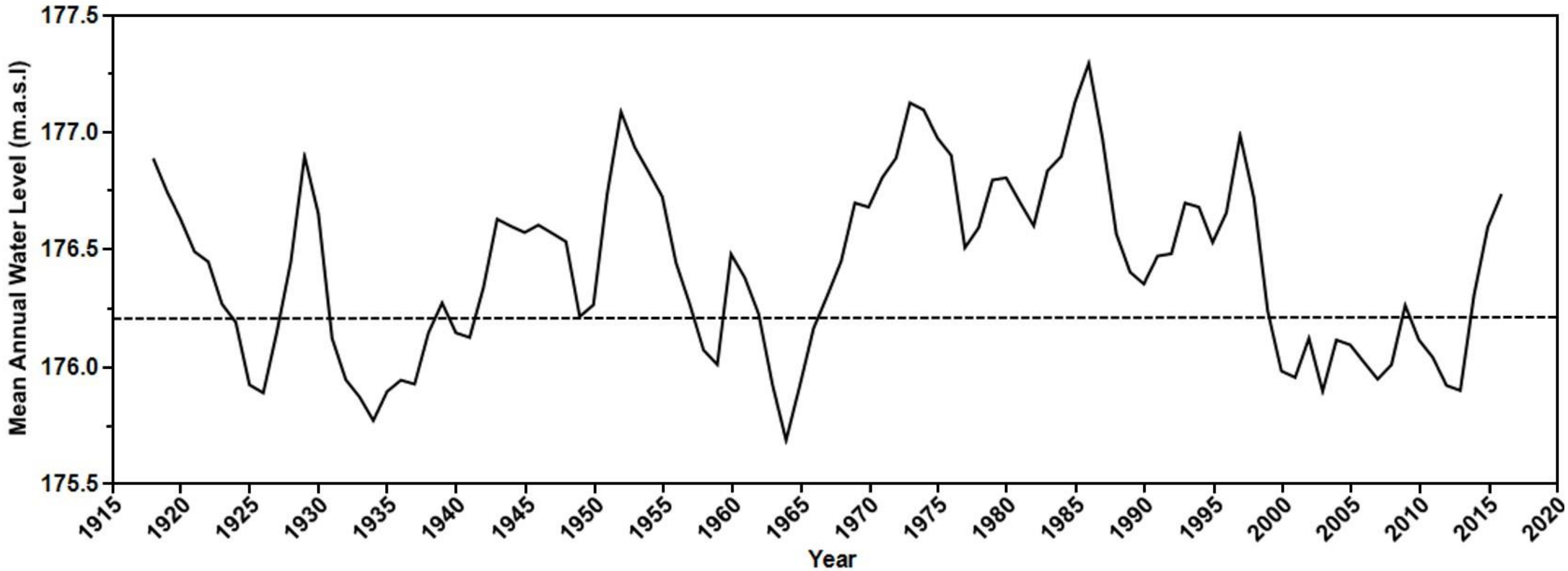

Figure 2.3: Mean annual Georgian Bay water levels through time (1918 – 2016). Dashed line indicates the long-term mean.

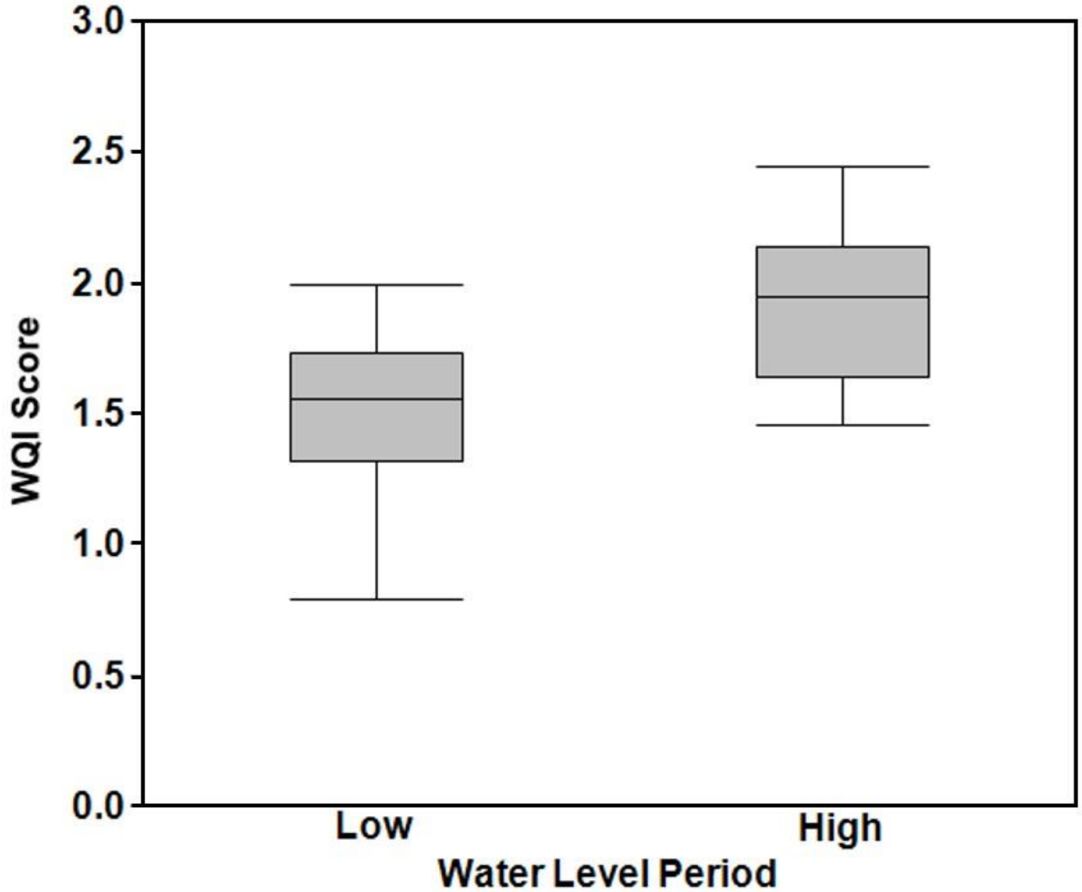

Figure 2.4: Box plot illustrating the difference in Water Quality Index (WQI) scores between low and high water level periods for 17 sites located in the Township of Georgian Bay. Solid line represents the mean.

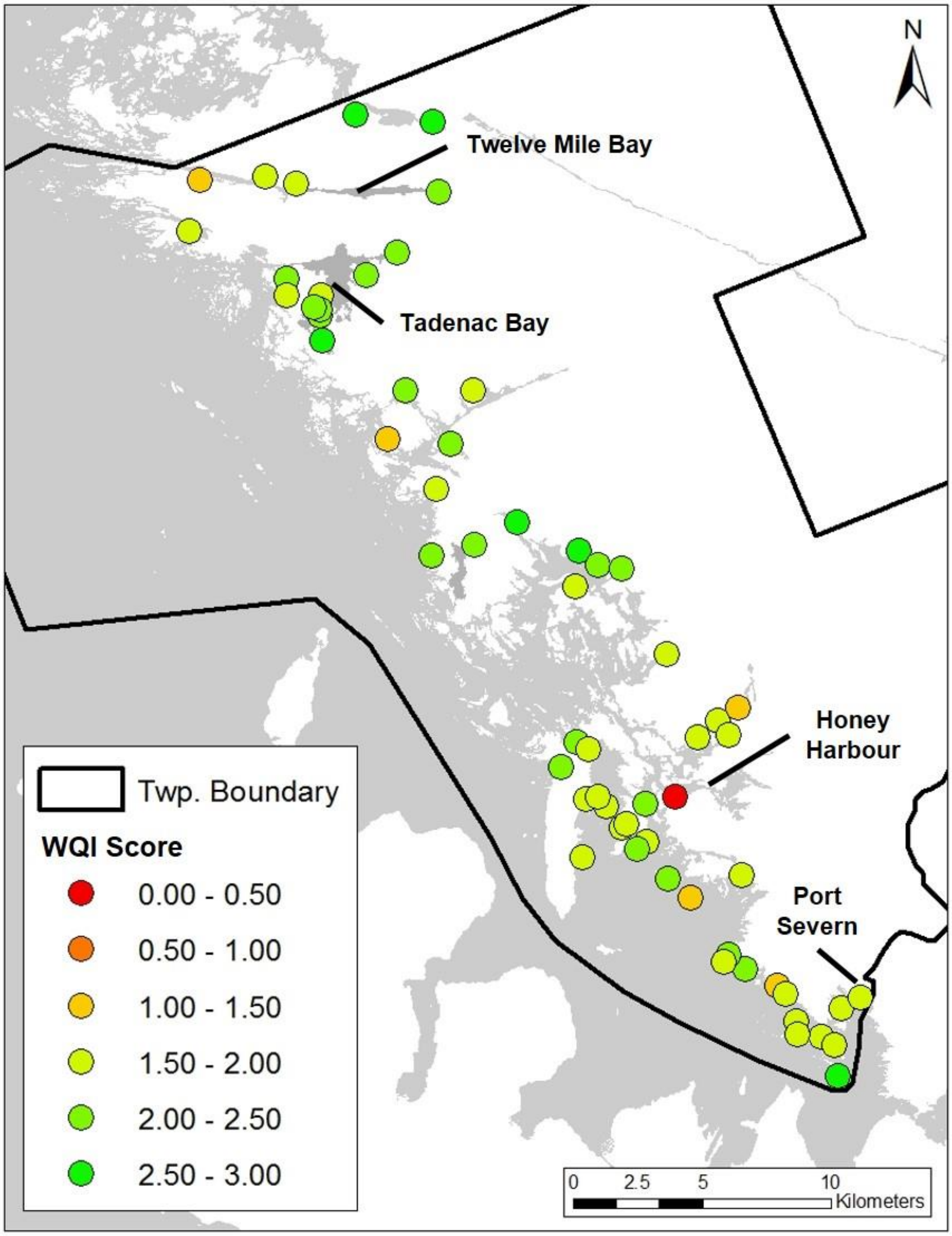

Figure 2.5: Map displaying Water Quality Index (WQI) scores in the Township of Georgian Bay

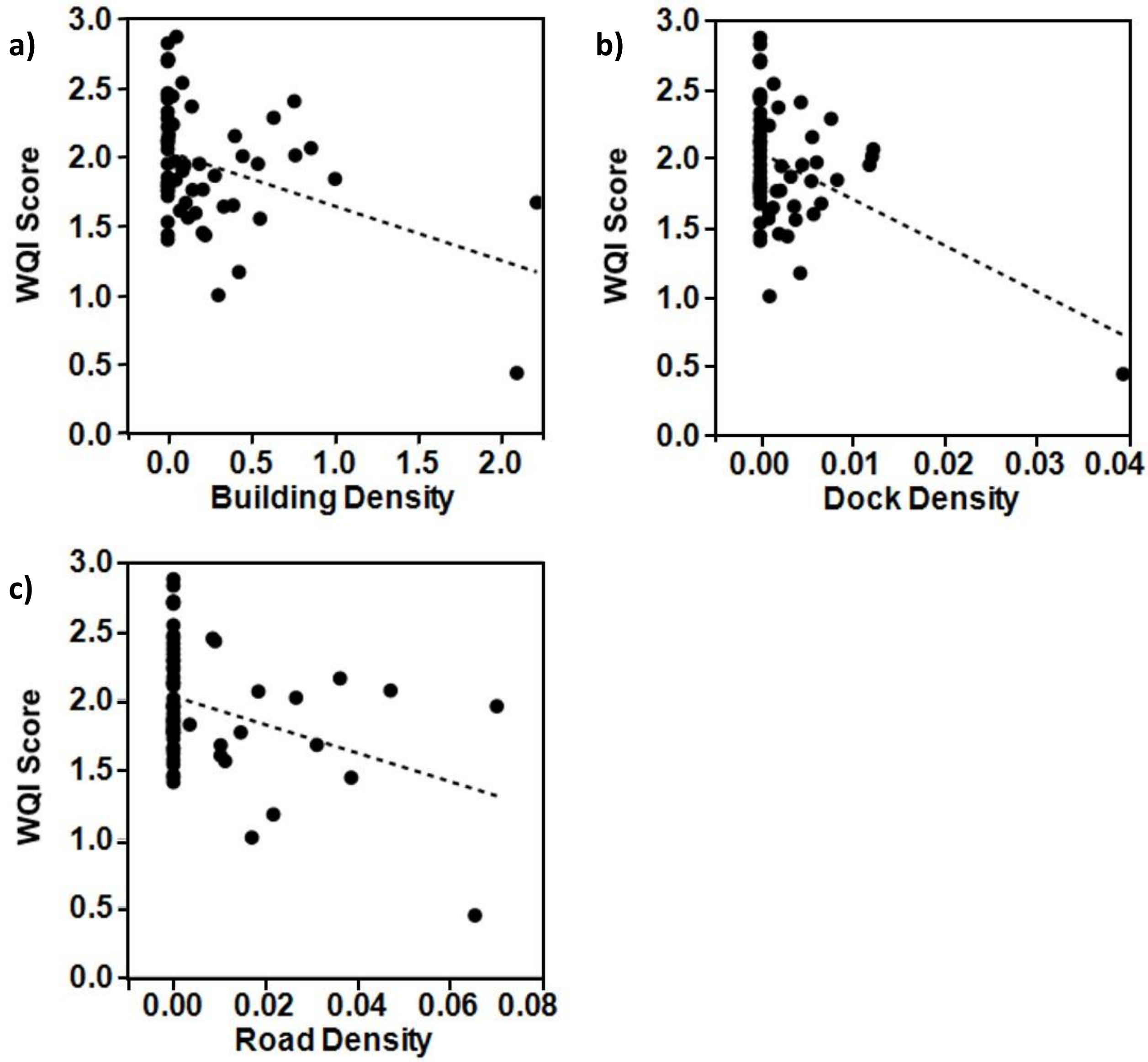

Figure 2.6: Scatter plots representing the relationship between Water Quality Index (WQI) scores and densities of human development. a) Building Density: primary residential structures • unit watershed area $(ha)^{-1}$; b) Dock Density: docks • unit wetland perimeter $(m)^{-1}$ and c) Road Density: road length (km) • unit watershed area $(ha)^{-1}$. Dashed lines represent best-fit linear regression. Note: WQI scores for sites sampled during the low water period have been controlled for water level differences (see Table 1).

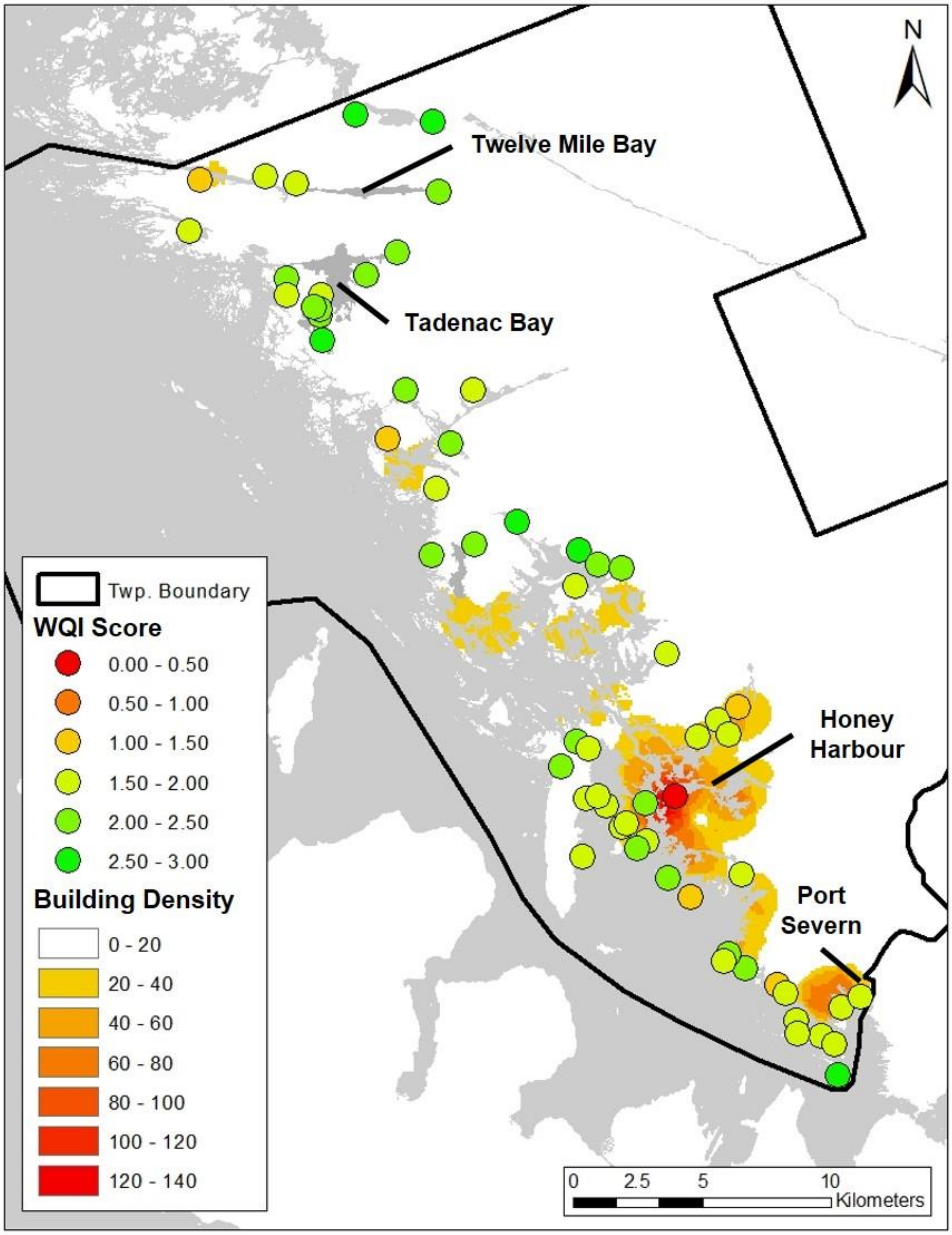

Figure 2.7: Map displaying Water Quality Index (WQI) scores and building density within 300 m from the shoreline. Building density represents number of building units within a 1000 m radius of each 10x10 m^2 raster cell.

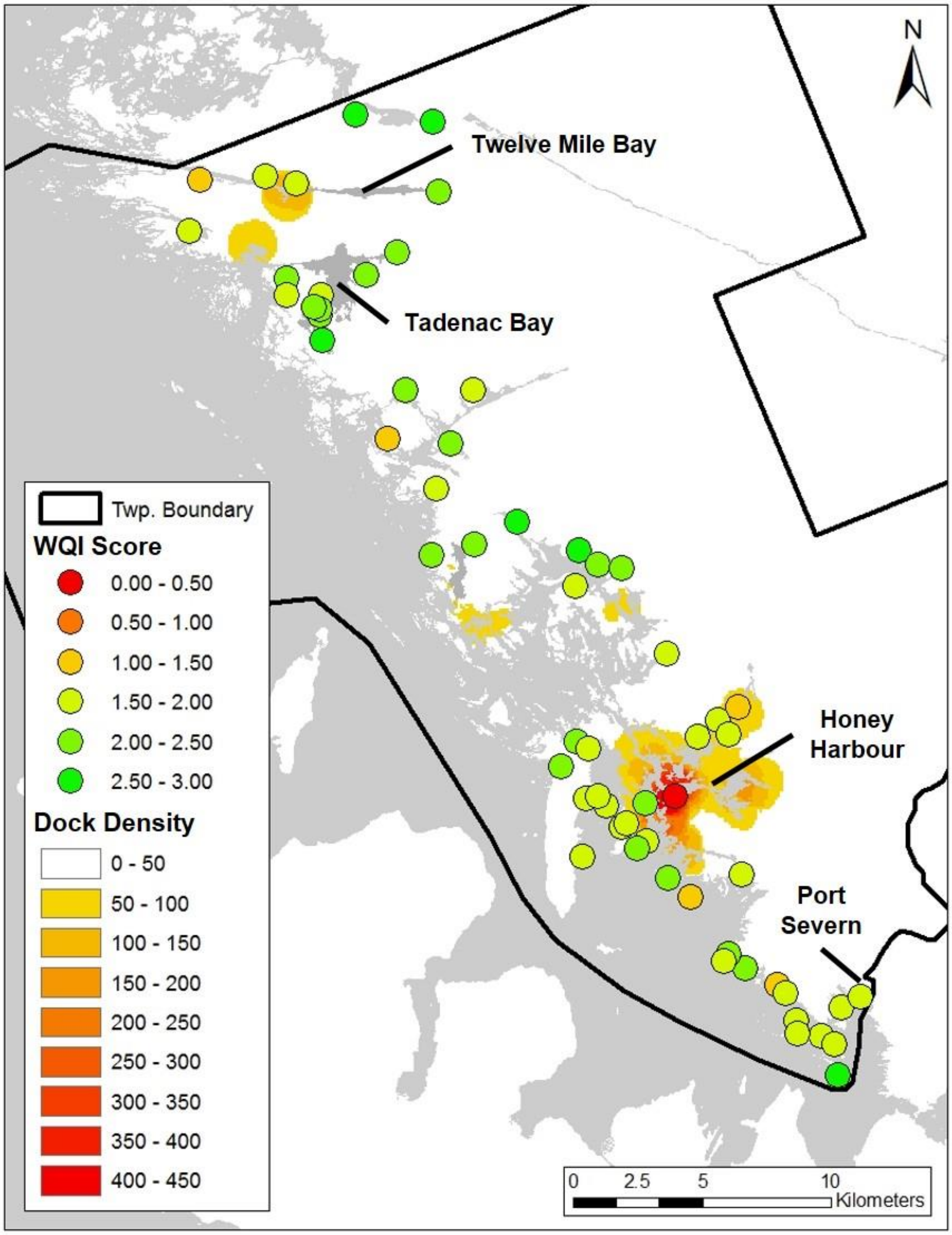

Figure 2.8: Map displaying Water Quality Index (WQI) scores and dock densities along the shoreline. Dock density represents number of dock units within a 1000 m radius of each 10x10 m^2 raster cell.

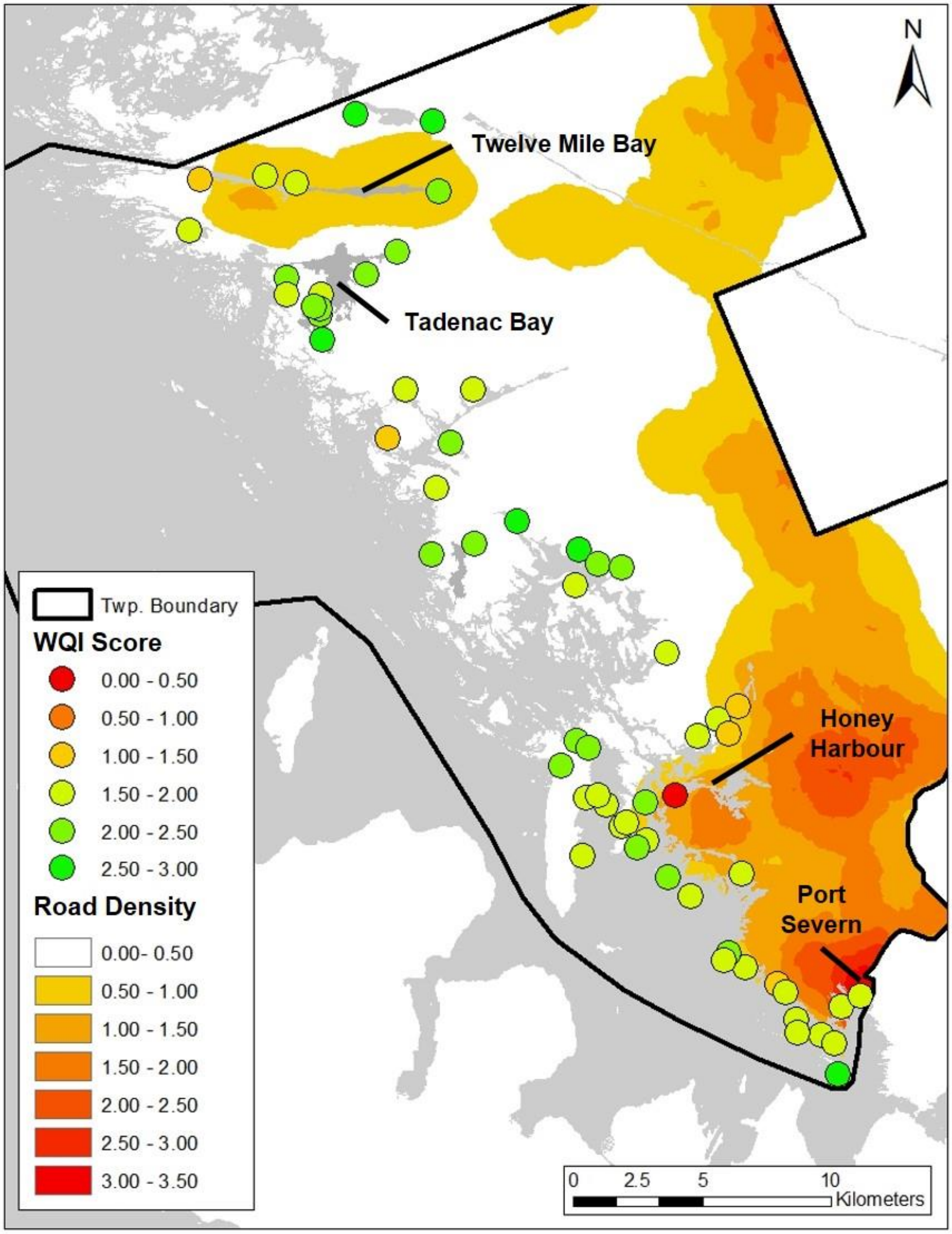

Figure 2.9: Map displaying Water Quality Index (WQI) scores and road densities in the Township of Georgian Bay. Road density represents road length (km) within a 1000 m radius of each 10x10 m^2 raster cell.

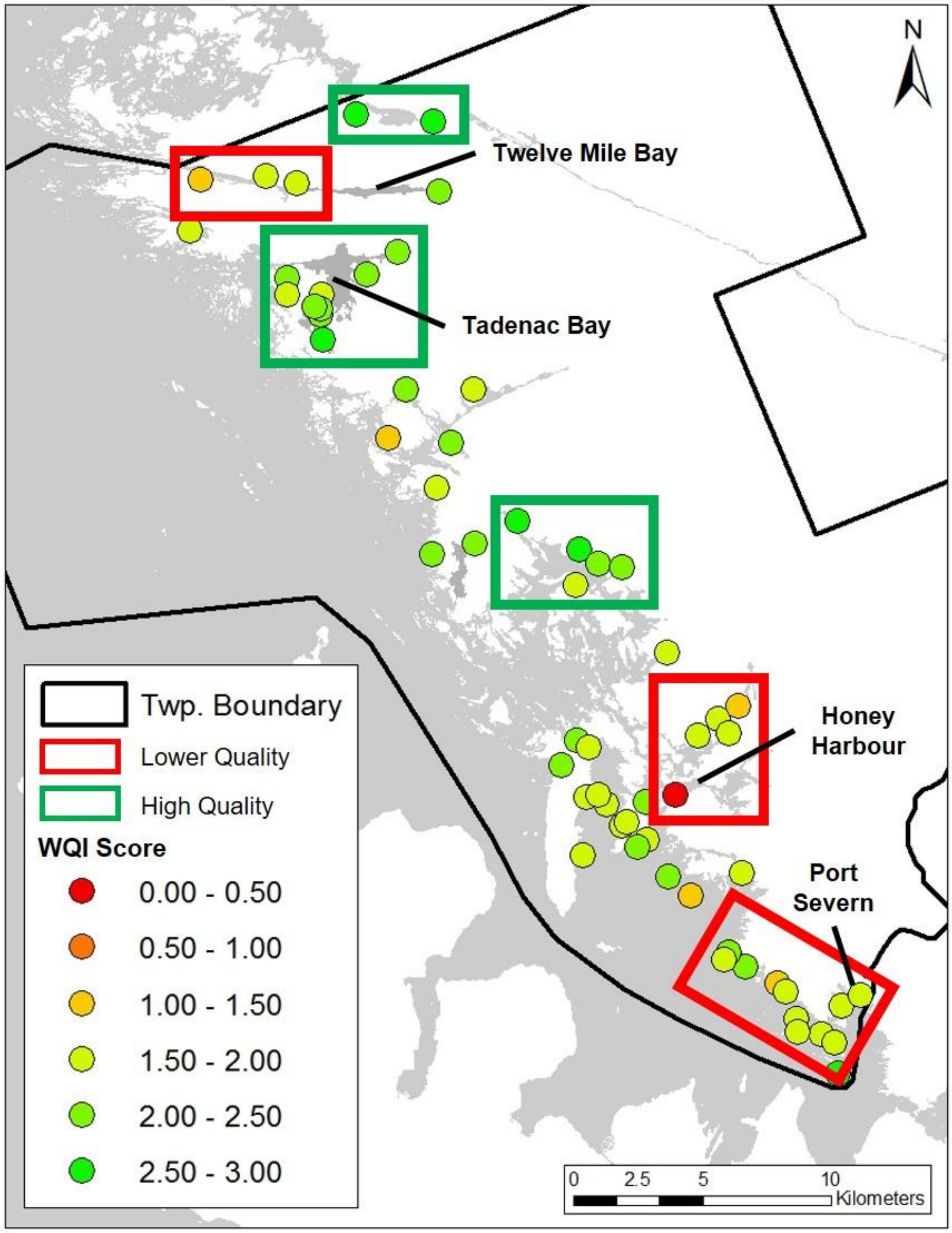

Figure 2.10: Map displaying Water Quality Index (WQI) scores and areas of interest for township managers regarding the quality of coastal wetlands in the Township of Georgian Bay.

GENERAL CONCLUSION

Although the coastal waters of eastern Georgian Bay are considered to be in excellent condition, I have shown in this thesis that in many regions along the shoreline, where recreational cottage and marina development are concentrated, coastal water quality has become degraded; this finding is consistent with other studies conducted in eastern Georgian Bay (Schiefer et al. 2006; Gartner Lee Limited 2008; Chiandet and Sherman 2014). These results solidify the need to develop a toolkit that environmental managers can use to ensure that future lakeshore development in this region is undertaken in a sustainable and responsible manner. The studies completed in this thesis are intended to provide environmental managers with the appropriate resources to manage water quality in coastal embayments and wetlands located in eastern Georgian Bay.

In Chapter 1, I found that the Lakeshore Capacity Model (LCM), developed for inland Precambrian Shield lakes, did not produce accurate estimates of total phosphorus [TP] when applied to coastal embayments; furthermore, the calculation was not improved by the inclusion of internal phosphorus load from anoxic hypolimnia. It is for these reasons that I would recommend that the LCM not be used as a management tool to predict trophic status in coastal embayments of eastern Georgian Bay. As an alternative, I recommend that the Anthro-geomorphic Model (AGM), developed in this thesis, be used instead, as it produced more accurate estimates of [TP] and could be applied to all ten embayments without exception.

With respect to lakeshore development in coastal embayments, the AGM can be used by environmental managers as a guide for preserving water quality in the ten embayments included in this study. I used the AGM to determine the building density thresholds that correspond to predicted [TP] of 10 $\mu g \bullet L^{-1}$ (oligotrophic), 15 $\mu g \bullet L^{-1}$ (mesotrophic) and 20 $\mu g \bullet L^{-1}$ (eutrophic) for each embayment (Table 1.13; Reckhow and Chapra 1983). In order to avoid hypolimnetic hypoxia, it is important that coastal embayments maintain a [TP] concentration of <10 $\mu g \bullet L^{-1}$, which is a threshold that a majority of the embayments included in the study have surpassed. It is therefore important that environmental managers ensure that building development in the three embayments that remain in an oligotrophic state (i.e. Longuissa Bay, Musquash Bay, and Tadenac Bay) stay within the appropriate building threshold to preserve current water quality conditions. For the remaining embayments that have already begun to show signs of water quality impairment, the AGM can be used as reasoning for environmental managers to prevent any additional development that might exacerbate this impairment.

In Chapter 2, I found that stressors associated with cottage development (i.e. buildings, docks and roads) have a negative impact on coastal wetland water quality, as measured by the Water Quality Index (WQI). Through the creation of mapping products that present how building, dock and road densities were spatially associated with wetland quality, I found that regions with high densities of these stressors were associated with wetlands of lower quality, whereas areas that had low development densities were associated with wetlands of high quality. I also found that a majority of the higher quality wetlands in the Township of Georgian Bay were located in remote regions that were not accessible by road.

I used maps that present development densities to identify areas that should be given priority by township managers in terms of conservation efforts (Figure 2.10). First, wetlands of high quality that are located in more remote locations of the Township of Georgian Bay should be protected from future development in order for the township to maintain an abundance of high quality coastal wetlands. This could be effectively achieved by ensuring that no new road development be permitted to reach remote regions of the coastline, as the extension of such infrastructure allows for high concentrations of cottage development to occur. Second, regions that have high densities of development that are associated with wetlands of lower quality should be managed very carefully. Although these sites are considered to have relatively good water quality, as indicated by their positive WQI scores, increased lakeshore development could degrade these wetlands to a point where they no longer perform valuable ecosystems services, such as providing important spawning and nursery habitat for economically important fish species like the muskellunge. It is essential that environmental managers ensure that no new large-scale development projects occur in these vulnerable regions. Information from this study should be used by township managers to make more informed decisions regarding future coastal development.

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