The relationship between water quality and riparian zone land-use and macrophyte removal permits issued between 2013-2020 in the Rideau Canal Waterway

by

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Abstract

In the Rideau Canal Waterway, eastern Ontario, residents are required to obtain permits from Parks Canada for macrophyte removal in front of their property for recreational purposes. The goal of this thesis is to determine the strongest environmental predictors of permit number and density in the Rideau Waterway to promote sustainable management of this system. Multiple linear regression models indicated that the best predictors of permit number were percent crop and pasture in the catchment, Secchi depth, algal blooms, catchment area, and total phosphorus (adjusted R^2 value of 0.94 and p-value of 1.952e-05). The strongest predictors of permit density are percent crop and pasture, deviation from circle, status of lakes, chloride, catchment area, and catchment to lake volume (adjusted R^2 value of 0.779 and p-value of 0.006). The results of this thesis research indicate that macrophyte removal permits are issued most often in nutrient rich lakes with greater crop and pasture in their catchments.

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Chapter 1: Introduction

Macrophytes are aquatic plants and include aquatic spermatophytes, pteridophytes and bryophytes (Srivastava et al., 2008). Submerged macrophytes in lakes and rivers play an important role in the ecosystem, providing habitat for invertebrates and fish species, influencing aquatic nutrient cycles, and contributing to the structure and functioning of the freshwater hydrologic regime (Chambers et al., 1999a; Journal et al., 2018). Macrophytes are used as an indicator of aquatic ecological health, as they reflect the quality of physical, chemical and biological habitat conditions (Ogdahl & Steinman, 2015; Wojciechowski et al., 2018).

Excessive growth of macrophytes in larger rivers is known to cause a variety of issues, including the depletion of oxygen at night, accumulation of organic matter, reduced water flow, and can impede navigation, fishing, and recreational use (Journal et al., 2018; Wojciechowski et al., 2018). In eastern Ontario, excessive macrophyte growth is due to several factors including both natural and human-induced factors (Parks Canada, 2017). These factors include nutrient run-off and enrichment (i.e. phosphorus) to waters from agricultural and neighbouring urban areas, cutting vegetation from the water (i.e. shoreline cutting, boat propellers), and introduction of zebra mussels (Parks Canada, 2017; Xiao-e Yang et al., 2008).

Lake systems can also suffer from a large loss of macrophytes, caused by shading from algal blooms driven by eutrophication (Journal et al., 2018; Kaj Sand-Jensen & Borum, 1991). These losses are often the result of activities associated with industrialization and urbanization, such as dredging and pollution, that harden shorelines and alter the sediments (Albert & Minc, 2004; Ogdahl & Steinman, 2015). This can be seen in the Laurentian Great Lakes, where saw

mill and industrial activities have caused impairment to river mouth and near-shore ecosystems (Allan et al., 2013; Larson et al., 2013). These activities often lead to a loss of macrophyte communities through changes in shorelines gradients and substrate quality (Gabriel & Bodensteiner, 2012), which can result in a reduction of ecosystem services that macrophytes' provide, including nutrient cycling, sediment stabilization, and essential habitats for a variety of species (Ogdahl & Steinman, 2015). Therefore, the consideration of macrophytes in response to changes in aquatic ecosystems is important when developing political and management strategies for aquatic environments (Journal et al., 2018).

Currently, a number of methods are used that aim to control and remove macrophytes ionvolving physical, chemical or biological removal (Wojciechowski et al., 2018). Physical methods are widely being used in Ontario by motorized machinery, such as excavators, to physically remove or cut vegetation (Carpenter & Adams, 1978; Hussner et al., 2017; Wojciechowski et al., 2018). The impacts of using machinery to mechanically manage and remove macrophytes have led to limnological changes, such as oxygen depletion, increased nutrients in the water column, increased turbidity, and oscillations in pH (Wojciechowski et al., 2018). The use of other methods including pesticides and other chemicals in fertilizers can also increase the risk of algal blooms and may result in the hardening of shorelines.

Despite the increasing number of aquatic habitat restoration projects worldwide, few studies have focused on the environmental conditions that influence people to remove macrophytes. A key indicator of macrophyte removal is the request of permits from managing authorities to remove macrophytes in the area in front of their properties. These macrophyte removal permits and subsequent macrophyte removal can have several effects on aquatic

ecosystem health in relation to water quality and land-use. The management of aquatic ecosystems is of particular concern in Ontario, Canada. Eastern Ontario is anticipated to experience rapid population growth in the upcoming decades (Journal et al., 2018). Development pressures are expected to have an impact on rivers and lakes in Ontario, however, the impacts on macrophytes are not well understood (Journal et al., 2018). Although there have been several lake restorations projects in Ontario, most studies fail to make macrophytes a focal point of restoration and monitor and evaluate the removal of macrophytes in relation to aquatic ecosystem health. Evaluation and documentation of aquatic ecosystem health in relation to macrophyte removal is important to fill research gaps, better inform management practices to protect ecosystems, and improve monitoring strategies.

The objective of my thesis is to determine which water quality and land use variables, if any, are strong predictors of the number and density of macrophyte removal permits issued by Parks Canada on the Rideau Waterway. To meet this objective, I will take a regional-scale approach in examining a number of water quality parameters and various land uses associated with the permitted areas to determine if there are significant predictors of permit distribution on the Rideau Waterway, to help inform future management practices and to help identify areas susceptible to excessive macrophyte growth along the shorelines, that in turn may help inform management actions and practices.

1.1 Research Objectives

The objective of my thesis is to assess the links between macrophyte removal in the Rideau Canal Waterway in relation to water quality and land use. The research objectives are:

a) What is the spatial pattern of aquatic vegetation removal permits in the Rideau Waterway?
b) Are water quality and land use variables important predictors of the total number and density of macrophyte removal permits issued on a lake?

Chapter 2: Literature Review

2.1 Significance of Macrophytes

The growth forms of macrophytes are classified into four groups: emergent macrophytes (e.g., *Typha latifolia* L.), floating leaved macrophytes (e.g., *Nuphar lutea* (L.) Sm.), free-floating macrophytes (e.g., *Eichhornia crassipes* (Mart.) Solms) and submerged macrophytes (e.g., *Myriophyllum spicatum* L.) (Srivastava et al., 2008). Most macrophytes are rooted and constitute as a living link between sediments and the water (Carpenter & Lodge, 1986). Macrophytes grow in shallow waters between the shoreline and open waters dominated by plankton, and can intercept or modify material flows from land to the pelagic zone (Carpenter & Adams, 1978).

Macrophytes play an important role in maintaining good water quality in shallow lakes (Kuiper et al., 2017). They serve as a substrate for epiphytic organisms and provide habitat for small fish and invertebrates (Chambers et al., 1999a). Macrophyte beds serve as long term sinks for organic material (Benoy & Kalff, 1999), and as short terms sources of nutrients and metals to water bodies (Jackson et al., 1994; Landers, 1982; Rooney & Kalff, 2000). Aquatic macrophytes in littoral zones of lakes have two fundamental properties which make them significant as limnological indicators (Melzer, 1999). They react slowly and progressively to changes in nutrient loads, functioning as integrators of environmental conditions to which they are subjected to, and can serve as long-term indicators with high spatial resolution (Melzer, 1999). Taller macrophytes modify water columns, trap fine particles by reducing light, and can reduce water currents (Chambers et al., 1999a).

Submerged macrophytes have major effects on productivity and biogeochemical cycles in freshwater because they inhabit important interfaces in both stream and lake ecosystems (Carpenter & Lodge, 1986). Submerged macrophytes form an ecotone between land and water, and vertically between sediments and water (Srivastava et al., 2008). Submerged macrophytes have an important role in carbon and nutrient flux within predominantly shallow lakes that dominate landscapes globally (Rooney & Kalff, 2000). There is a positive feedback between aquatic plants and water clarity, in which macrophytes enhance their growing conditions (Kuiper et al., 2017; Scheffer et al., 1997; van Donk & van de Bund, 2002). These mechanisms cause a tendency of the system to resist changes in external conditions, such as promoting a clear water state (Kuiper et al., 2017; Scheffer, 1997). They can maintain a clear water state by various mechanisms including bicarbonates utilization, uptake of nutrient ions, and providing refuge for harvesters of phytoplankton (Srivastava et al., 2008). Rooted submerged macrophytes can take up nutrients from both the sediment pore water (Barko & Smart, 1980) and from overlying water (Ozimek et al., 1993; Srivastava et al., 2008). In temperate lakes, such as those in the eastern Ontario region, the production and decomposition cycle of macrophytes, and associated biogeochemical fluxes, follow an annual cycle (Carpenter, 1980).

Many lakes possess alternative stable states over a range of environmental conditions (Kuiper et al., 2017). Understanding the importance and significance of macrophytes in lakes is important for management purposes to stabilize water states and ensure a healthy ecosystem. Considering the diverse role of macrophytes in aquatic ecosystems, the management of macrophytes based solely on abundance is unlikely to preserve ecological integrity (Chambers et al., 1999b). The potential for ecosystem change as a result of macrophyte removal, including

increases in abundance of planktonic algae or colonization by an invasive plant species should be considered before decision-making processes (Chambers et al., 1999b).

2.1.1 Source Versus Sink

Macrophytes act as sinks for particulate matter and sources of dissolved phosphorus (P) and organic carbon (Carpenter & Lodge, 1986). The assessment of macrophyte effects on nutrient and carbon ycles are dependent on the time scale of the study (Carpenter & Lodge, 1986). For example, during phases of active growth, macrophytes in streams act as sinks for particulate matter during the growing season, and export organic matter when decaying (Carpenter & Lodge, 1986; Dawson, 1980; Howard-Williams, 1981). In the early stages of macrophyte decay, leaching of dissolved organic carbon is rapid (Carpenter & Lodge, 1986; Godshalk & Wetzel, 1978; Otsuki & Wetzel, 1974). Organic matter that is leached from decaying macrophytes can be readily metabolized by suspended bacteria (Carpenter & Adams, 1978; Carpenter & Lodge, 1986). Phosphorus released by decaying macrophytes is rapidly absorbed by phytoplankton, leading to an increase in chlorophyll concentrations (Carpenter & Lodge, 1986; Landers, 1982). Nitrogen (N), in contrast to phosphorus, is typically released more slowly during carbon decay (Boston & Perkins, 1982; Brock, 1984; Carpenter & Adams, 1979; Carpenter & Lodge, 1986; D. S. Nichols & Keeney, 1973).

On an annual scale, most macrophytes are a net source of particulate organic matter (Carpenter & Lodge, 1986; Howard-Williams, 1981). During periods of decaying the plants become a net source (Carpenter & Lodge, 1986; Landers, 1982). A number of data sets provide estimates of annual net phosphorus fluxes to and from macrophyte beds (Carpenter & Lodge, 1986; Howard-Williams, 1981). Howard-Williams & Allanson (1981) determined that in a

Potamogeton pectinatus L. stand, the phosphorus cycle was relatively closed (Carpenter & Lodge, 1986). The authors concluded macrophytes released phosphorus assimilated by periphyton and epiphyton decomposed on littoral sediments that effectively retain phosphorus (Carpenter & Lodge, 1986; Howard-Williams & Allanson, 1981). In a *Myriophyllum spicatum* L. stand where net fluxes were directly measured, the littoral zone was a significant source of dissolved phosphorus, exceeding the total phosphorus (TP) loading from the watershed during the summer period (Carpenter & Lodge, 1986; Howard-Williams & Allanson, 1981).

Macrophyte community composition can also influence the magnitude of phosphorus and organic carbon fluxes from the littoral zone (Carpenter, 1983; Carpenter & Lodge, 1986). In oligotrophic lakes, macrophyte biomass is low and can turn over slowly. Therefore, surficial sediments are oxidized and retain phosphorus making net fluxes to and from the littoral zone small (Carpenter & Lodge, 1986). In contrast, in eutrophic lakes, high macrophyte biomass is supported with high turnover rates. Surficial sediments are therefore reduced, and release phosphorus making fluxes to and from the littoral zone large (Carpenter & Lodge, 1986).

2.2 Biogeochemical Influences of Submerged Macrophytes

Factors that influence the biomass and distribution of submerged macrophytes have been well studied (Rooney & Kalff, 2000). Among lakes, macrophyte biomass and distribution can vary with littoral slope (Duarte & Kalff, 1986), latitude (Duarte & Kalff, 1987), water transparency (Chambers & Kaiff, 1985; Dale, 1986), and sediment characteristics (Anderson & Kalff, 1988; Rooney & Kalff, 2000). A majority of the studies on the distribution and biomass of macrophytes focus on the factors that vary in space within growing seasons (Rooney & Kalff, 2000).

Submerged macrophytes translocate and excrete phosphorus and other nutrients. The amount of nutrient accumulation in a waterbody is dependent on the physiological capacity for further uptake and the biomass of aquatic macrophytes, which varies depending on the species (Srivastava et al., 2008). Phosphorus is an important limiting nutrient in lakes and waterbodies. Roots are an important avenue for the entry of phosphorus into macrophytes (Barko & Smart, 1980; Carignan & Kalff, 1980; C. S. Smith, 1978). Phosphorus is translocated to shoots, where it may enter lake water through the release of living or decaying shoots (Carpenter & Lodge, 1986). Phosphorus, in contrast to organic carbon, is not released from living submerged macrophytes at significant ecological rates (Carpenter & Lodge, 1986). Smith (1978) demonstrated that release rates of phosphorus are less than 5% of uptake rates throughout the year for *Myriophyllum spicatum* L.

The metabolism of submerged macrophytes also has a strong influence on dissolved inorganic carbon and pH (Carpenter & Lodge, 1986). Macrophytes remove inorganic carbon from water by both assimilation into organic matter and precipitation as carbonate salts on the leaves (Carpenter & Lodge, 1986; Wetzel, 1960). In soft waterbodies, sediment pore waters are a major source of dissolved inorganic carbon, and up to 95% of the carbon fixed by macrophytes enters from the roots (Carpenter & Lodge, 1986; Sand-Jensen & Søndergaard, 1979; Sand-Jensen & Søndergaard, 1978, p.; Wium-Andersen, 1971). Where dense submerged macrophytes are present, oxygen changes as large as 8 mg l^{-1} can occur in the water (Carpenter & Lodge, 1986; Ondok et al., 1984).

Submerged macrophytes are more effective in oxygenating the water than floating-leaved macrophytes (Carpenter & Lodge, 1986; Pokorný & Rejmánková, 1983). For example, mats of

duckweed can decrease oxygen concentrations by preventing re-oxygenation from the atmosphere (Carpenter & Lodge, 1986; Morris & Barker, 2011). In a river with dense macrophytes, it was shown that daily oxygen dynamics were strongly dependent on macrophyte biomass (Carpenter & Lodge, 1986; Kelly et al., 1983). Daily oxygen fluxes in dense standing *Myriophyllum spicatum* L. beds were twice as great as those in an adjacent harvest plot (Carpenter & Gasith, 1978; Carpenter & Lodge, 1986). Submerged vascular macrophytes also oxidize their rhizospheres (Carpenter & Lodge, 1986). Photosynthetically-produced oxygen diffuses to roots through the aerenchyma, and then diffuses across the epidermis into the sediment (Carpenter & Lodge, 1986; Oremland & Taylor, 1977; Sand-Jensen et al., 1982; R. D. Smith et al., 1984). The degree to which sediments are oxidized varies depends on the balance between oxygen release by the macrophytes and by chemical and microbiological processes in the sediment (Carpenter & Lodge, 1986). Rhizosphere oxidation has significant biogeochemical consequences (Carpenter & Lodge, 1986).

2.3 Nutrient Enrichment

During the second half of the twentieth century submerged macrophytes have disappeared from many shallow lakes in temperate regions due to external nutrient loading from anthropogenic activity (Gulati & Donk, 2002; Körner, 2002; Kuiper et al., 2017). Lakes can switch from a clear-water state, dominated by macrophytes, to a turbid-water state with fewer to no plants, which can be prone to cyanobacterial blooms (Kuiper et al., 2017; Scheffer et al., 1993). As a result, management efforts have been devoted to the restoration of aquatic plant communities, through the reduction of external nutrient loading, especially phosphorus (Cullen & Forsberg, 1988; Hilt et al., 2006; Jeppesen et al., 2005; Kuiper et al., 2017). Phosphorus is an essential element for plant growth and its input has long been recognized as essential to maintain

economically viable levels of crop production. Although nitrogen and carbon are also essential to the growth of aquatic biota, a lot of attention has been focused on phosphorus inputs, because phosphorus is often the element limiting growth of algal development in aquatic environments (Brogan et al. 2001). Excessive phosphorus inputs to aquatic ecosystems can lead to cyanobacterial blooms, thus triggering ecological imbalances and a series of environmental problems including toxic algal blooms, increased anoxia, loss of biodiversity, and loss of aquatic plant beds (Carpenter et al., 1998; Elser & Bennett, 2011; Li et al., 2018; Turner et al., 1995).

Changes in nutrient levels and light conditions caused directly or indirectly by anthropogenic activities can greatly influence the abundance of macrophytes (Chambers et al., 1999a). Watershed land use patterns directly influence water quality, through changes in salinity, sedimentation, and nutrient loading (Chambers et al., 1999a). Carpenter et al. (1998) stated that nutrient enrichment has become a serious problem, degrading aquatic ecosystems and impairing the use of fresh water for drinking, industry, agriculture, recreation, and many other purposes. The main sources of nutrient loading to Canadian freshwater ecosystems include municipal and rural wastewater, manure and fertilizer in agricultural runoff, and industrial discharges. A study by Evans et al. (1996) examined Lake Simcoe in Southern Ontario which has undergone environmental changes that are consistent with increased phosphorus inputs due to human activity, resulting in a loss of cold-water fish habitat.

It has been recommended by Chambers et al. (1999b) that on a landscape level, authorities should consider nearby riparian zone land-use patterns and if they directly or indirectly affect nutrient and sediment loading. A better understanding between aquatic habitats at the landscape level is necessary for the effective management of macrophytes in freshwater

systems (Chambers et al., 1999a). The findings of this thesis can help to inform the management of lakes along the Rideau Canal Waterway, specifically with reference to permits requested and issued to shoreline property owners, distributed by Parks Canada in the eastern Ontario region (Evans et al., 1996; Ginn, 2011).

2.3.1 Agricultural Fertilizers

The application of fertilizers can increase nutrient levels and accelerate eutrophication (Brown 1985. Dressing et al. 2016). Farming systems have intensified over time, and in recent years, it has become evident that the increase in losses of N and P from agricultural land has detrimental effects on water quality and the environment (Hart et al. 2004). In the past two decades, Hart et al. (2004) have demonstrated that there has been a large increase in research regarding the issues surrounding the losses of P to surface and groundwater. Fertilizer application can be described as acute temporary sources, or chronic sources. The high concentrations of P in recently applied sources can elevate dissolved P in surface runoff and leachate to greater concentrations (Kleinman et al., 2011). High inputs of P applied to soils prone to runoff during heavy rainfall periods or before flood irrigation events produce the greatest potential for acute transfers of applied P to nearby bodies of water (Kleinman and Sharpley 2003, Nash et al. 2000, Withers et al. 2003, Kleinman et al. 2011). While the contribution of surface applied P sources to runoff diminishes during weeks after applications, Kleinman et al., (2011) note that the contribution of P and soil erosion and dissolution persist over time. Soil erosion presents the greatest concern to most P mitigation programs (Kleinman et al., 2011). When poorly monitored, erosion-related losses of P can greatly threaten crop production. In recent years there has been recognition that the release and mobilization of dissolved P from

agricultural soils is becoming more of a contributor to eutrophication that seen before (Kleinman et al., 2011).

2.3.2 Urban Runoff

Nearly 50 year ago, Cowen & Lee (1976) suggested that one of the adverse effects of urbanization in North America is the increase in the volume of urban stormwater runoff that is discharged into lakes and streams. In addition, (Carpenter et al., 1998) highlight that significant amounts of P and N can also enter surface waters from nonpoint sources including construction sites, runoff of lawn fertilizers and pet waste, as well as inputs from unsewered developments. Construction sites are also a large area for concern in urban nonpoint sources of pollution. Considering construction sites may often occupy a relatively small percentage of land area, erosion rates can be very high, making the nonpoint pollution yield very high (Carpenter et al., 1998). Urban point sources of water pollution including sewage and industrial discharges are often significant, and therefore are intensively managed (Carpenter et al., 1998). Phosphorus is of particular concern and interest in urban stormwater runoff due to the role of eutrophication (Cowen & Lee, 1976). Consequently, as Cowen & Lee (1976) argued, a better understanding of P in urban areas and the relative contributions of P is needed to develop and maintain effective management.

2.4 Land Use

Human land use is a major factor that influences non-point sources of pollution in watersheds and impacts the delivery of excess nutrients to bodies of water (Baker et al., 2001; Correll, 1996; O'Neill et al., 1997). Changes in land cover and land management practices are considered key influencing factors behind the alteration of hydrological systems, which impacts

runoff and water quality (Bai et al., 2010; Huang et al., 2013; Tong & Chen, 2002). Several studies in the last few decades have identified the various factors that affect freshwater quality (Hossain, 2017; Jeppesen et al., 2005; Ngoye & Machiwa, 2004; Novotny, 1996; Palmer et al., 1998). Different types and patterns of land use within catchments influence nutrient availability and can impact downstream water quality and primary production through runoff and overland flow (Gorman et al., 2014; Lee et al., 2009; Sun et al., 2018). Downstream water quality and macrophyte abundance can be linked to: a) the proportion of urban or industrial land (Sass et al., 2010; Tong & Chen, 2002; White & Greer, 2006); b) the proportion of agricultural land which can influence nutrient loading (Gorman et al., 2014; Knoll et al., 2003), and c) the type of agricultural land (Arbuckle & Downing, 2001; Sun et al., 2018). The littoral zones of lakes may experience patterns of nutrient and pollutant concentrations cause by natural or artificial inflows, or by non-point sources (Melzer, 1999; O'Neill et al., 1997).

As already noted, shifts in aquatic environments have been linked to both agriculture (Crosbie & Chow-Fraser, 1999; Dodson et al., 2005; Egertson et al., 2004; Heegaard et al., 2001; Rasmussen & John Anderson, 2005; Virola et al., 2001) and urban land use practices (Alexander et al., 2008; Arts, 2002; Bowen & Valiela, 2001; Hauxwell et al., 2003; S. A. Nichols & C. Lathrop, 1994; Sass et al., 2010). Nutrients and sediments in runoff from agricultural land use can cause eutrophication which can result in declines in submerged aquatic plants (Crosbie & Chow-Fraser, 1999; Egertson et al., 2004; Gleason et al., 2003; Rasmussen & John Anderson, 2005; Sass et al., 2010). Negative effects can range from aquatic plant communities shifting to predominantly floating and emergent species to the collapse of the macrophyte community (Sass et al., 2010). Urban perturbations to aquatic ecosystems include impacts that vary from agricultural land such as sewage leakage, road salts and contaminants in runoff, and nutrient

pollution from landscaping (Findlay & Houlahan, 1997; Jonathan W. Moore et al., 2003; Rasmussen & John Anderson, 2005; Sass et al., 2010; L. Wang et al., 2003). Additional negative impacts have been linked to land development near shoreline ecotones (Alexander et al., 2008; Houlahan et al., 2006; Hrabik et al., 2005; Jonathan W. Moore et al., 2003; Radomski & Goeman, 2001; Sass et al., 2010). Due to varying urban development types and impacts, various studies have attempted to describe urban impacts as a possible cause, but have found it difficult to link urban land use to specific causal impacts on aquatic macrophyte communities (Dodson et al., 2005; Rasmussen & John Anderson, 2005; Sass et al., 2010; Toivonen & Huttunen, 1995). These results suggest that further research is needed to examine the relationships between natural and anthropogenic landscape features and macrophyte cover (Cheruvelil & Soranno, 2008).

2.5 Lakeshore Development and Urbanization

During the 20-21st century, water bodies have received many alterations and pressures which are found to be strongly related to the intensive use of lakeshore and waterfront development (Furgała-Selezniow et al., 2022; Schmieder, 2004). Human habitation and increases in shoreline development has a cumulative effect on water quality and biota of aquatic ecosystems (Engel & Pederson, 1998; Radomski & Goeman, 2001). Lakeshore development includes new houses, fertilized lawns, gravel driveways, and septic systems. Although aquatic communities can resist small changes in water quality, the cumulative effects of lakeshore alterations and additions can lead to significant ecosystem responses (Engel & Pederson, 1998). In theory, dense algal blooms can shade-out underlying rooted plants in deep water and ultimately reduce the area of plant habitat for fishes, invertebrates, and diving ducks (Engel & Pederson, 1998). Loss of native plants, in turn, can result in invasions by turbidity-tolerant exotic plants (Engel & Pederson, 1998). Lakeshore urbanization often follows a progression: an

underdeveloped lake is colonized by several homes on septic systems. This is followed by increasing residential development, land and riparian zone clearing, and loss of coarse woody debris, until the maximum human population density is reached (Moore et al., 2003).

Residential development of shorelines often leads to reduced aquatic vegetation and changes in water quality, as property owners alter nearshore areas to create more aesthetically desirable shoreline properties (Beck et al., 2013; Christensen et al., 1996; Jennings et al., 2003; Marburg et al., 2006; Radomski & Goeman, 2001). The effects of shoreline development on lakes may be most evident near docks as homeowners remove aquatic macrophytes and wooded structures for easier access to the shoreline (Beck et al., 2013). A greater understanding of the effects of residential development at varying spatial scales and the effects on macrophyte community structures could improve the protection of shoreline habitats (Beck et al., 2013). Recent research that has focused on identifying the effects of shoreline development consistently describe a negative relationship between shoreline development and aquatic habitat quality (Beck et al., 2013; Bryan & Scarnecchia, 1992; Christensen et al., 1996; Gaeta et al., 2011; Radomski & Goeman, 2001). The study by Beck et al. (2013) supports these findings by determining that shoreline development can have lake-wide cumulative effects on macrophyte assemblages and identified a negative response of macrophytes to shoreline development. The authors found that emergent-floating and sensitive species richness was negatively correlated with increasing shoreline development, and the total submerged species richness was positively associated (Beck et al., 2013). This suggests that the vulnerability to shoreline development could be explained by depth (Beck et al., 2013). Emergent and floating-leaf species colonize in shallow regions of lakes and are preferentially removed by waterfront property owners due to shoreline proximity (Beck et al., 2013; Radomski & Goeman, 2001). Therefore, deeper lakes

with low watershed development were particularly at risk in the study by Beck et al. (2013), where more shoreline development was associated with fewer emergent-floating and sensitive species. Removal of vegetation may have cumulative lake-wide effects on richness metrics and can alter the distribution of macrophyte species within lakes (Beck et al., 2013). Possible management efforts can focus on the protection of riparian areas and discouragement of aquatic macrophyte removal to prevent the negative effects associated with shoreline development (Beck et al., 2013).

2.5.1 Riprap and Seawalls

The installation of riprap and seawalls can cause an increase in siltation and nutrient enrichment of lake water through debris fall and erosion (Engel & Pederson, 1998). Soil washing from construction sites can contain a mix of particles and varying textures (Engel & Pederson, 1998). Nutrients carried by particles can increase algal blooms and water quality can continue to deteriorate after construction (Engel & Pederson, 1998). Soil can also wash into lakes when waves erode the base of seawalls (Dai et al., 1977; Engel & Pederson, 1998; Krull, 1969). Vertical or inclined seawalls can sometimes create an undertow from breaking waves that can scour lake bed, whereas riprap can deflect wave energy to minimize wave scour (Engel & Pederson, 1998). As a result, the increased water turbulence can cause silt and algae in suspension, which can cause an increase in water turbidity and shading of tiny submersed macrophytes (Engel & Pederson, 1998). Water turbidity can also be increased when lakeshore development leads to an increase in boating, where clay and silt are kept in suspension depending on the composition of the bottom of the lake and the nature and frequency of the boats (Engel & Pederson, 1998; Yousef et al., 1980). Nutrients in shallow sediments can also rise into

the water column with the passing of boats, including P which can stimulate growth of attached or planktonic algae (Engel & Pederson, 1998; Murphy & Eaton, 1983).

The removal and destruction of plant habitat, riprap and seawall construction can all have widespread ecological effects (Engel & Pederson, 1998). Consequently, the loss of underwater foliage creates sites for more exotic species. For example, shoot fragments of Eurasian watermilfoil could take root and grow on disturbed lake sites and spread through stolons and new shoot fragments (Engel & Pederson, 1998), replacing beds of native plants. Turions of curly-leaf pondweed (*Potamogeton crispus* L.) can also sprout on disturbed sites (Engel, 1990; Sastroutomo et al., 1979). Immobile rooted aquatic plants are affected by riprap and seawall construction. Increased erosion during construction, can burry underwater shoots and smother seeds, resting buds, and shoot fragments in lake sediment (Engel & Pederson, 1998; Kautsky, 1987). As water hits a base of a shoreline structure, unrooted plants (i.e. coontail (*Ceratophyllum demersum* L.*))* and weakly rooted plants (i.e. American elodea (*Elodea canadensis* Michaux)) are likely to drift away (Engel & Pederson, 1998; Sculthorpe, 1967), leaving species that mat the lake bed with roots and stolons.

2.5.2 Piers

The addition of piers to lakeshores can also have an immense impact on plant growth (Engel & Pederson, 1998). During construction, plants can be uprooted, and the pier can continue to shade underwater foliage. Piers on wave-washed shores can form pockets that collect sediment, allowing for some aquatic plants to take root (Engel & Pederson, 1998). Species such as water lilies (Nymphaeaceae) and free-floating duckweeds (Lemnaceae) can thrive behind boathouses and between piers if there isn't significant boat activity (Engel & Pederson, 1998).

The addition of piers can increase boat pressure, and consequently can damage plants from direct contact with boat hulls and/or motor propellers as well as indirectly from wakes of passing boats (Engel & Pederson, 1998; Liddle & Scorgie, 1980). As plants disappear from boating lanes, they can become uprooted or shredded, and can then grow slowly in water that is muddied by heavy boat traffic on a lake (Engel & Pederson, 1998). Aquatic plants differ in their resistance to flow (Haslam, 1978) and damage from boat wakes (Engel & Pederson, 1998). Floating-leaf plants are more damaged than submersed or emersed ones, as boat waves diminish with depth. Well-rooted submersed plants, such as Eurasian watermilfoil, are less able to be dislodged passing motorboats than unrooted ones, such as coontail (Engel & Pederson, 1998). Plants with pliable stems and short growth are less damaged by boats than those with brittle stems and taller growth, such as curly-leaf pondweed (Engel & Pederson, 1998).

2.5.3 Wooded Areas

Most deposits of coarse wood debris can protect lakeshores. Debris can block waves and ice action that pass over the lake bed and can prevent seeds from sprouting and/or shoots from rooting (Engel & Pederson, 1998). Debris in lakes collect sediment and can become coated in algae and detritus (Engel & Pederson, 1998; Harmon et al., 1986). However, the removal of woody debris can expose lake sediment to sunlight, improving plant growth and habitat for fish species (Engel & Pederson, 1998). A study by Buchan & Padilla (2000) and Cheruvelil & Soranno (2008) examines Eurasian watermilfoil invasions using landscape-level variables. The authors (Buchan & Padilla, 2000; Cheruvelil & Soranno, 2008) found that the amount of forest land cover within a catchment is consistently negatively related to the presence of Eurasian watermilfoil.

2.5.4 Vegetative Buffers

Aquatic macrophytes act as vegetative buffers, that filter soil and dissolved nutrients moving downslope (Engel & Pederson, 1998). Shallow buffers extend parallel to the shore and can stretch from less than 23 metres to the width or entire shorelines (Engel & Pederson, 1998). Deep buffers extend perpendicular to the shore to join uplands, helping with nutrient filtration and soil detention (Engel & Pederson, 1998).

2.6 Invasive Species

Biological invasion of non-native macrophytes in aquatic ecosystems is a well-known phenomenon. Exotic species invasions are becoming a common occurrence, and are often linked to a decrease in the relative abundance and richness of native species in communities (Buchan & Padilla, 2000). In the past 15 years, researchers have started to focus on these species on aquatic communities and ecosystem dynamics (Schultz, 2012). It was suggested by Schultz (2012) that information on invasive macrophytes along with environmental data could be beneficial to create models to better predict the impacts of macrophyte invasion. However, the effects of invasive macrophytes on trophic dynamics on a smaller scale are less well-known, and more research is essential to define system-level processes (Schultz, 2012). For example, Hogsden et al. (2007) examined the population of *Cabomba caroliniana* (Carolina Watershed) which now covers extensive littoral areas in the shallow waters of Kasshabog Lake in Ontario. Little is known about the broader ecological implications of its introduction and establishment (Schultz, 2012). Additionally, *C. caroliniana* is changing the macrophyte community composition of lakes, having an impact on epiphytic algae, and creates new habitats for some macroinvertebrates (Schultz, 2012). Further studies are required to determine the extent of these ecological impacts.

Eurasian watermilfoil (*Myriophyllum spicatum* L.) has also been present in Ontario's waters since the 1960s. Eurasian watermilfoil exists on every continent except for Antarctica, and is native to Europe, Asia, and Northern Africa (Buchan & Padilla, 2000; Couch & Nelson, 1985). The earliest confirmed specimen in North America was collected in 1942, and increased its range in subsequent decades to include the Eastern part of the continent as far north as Ontario and Quebec provinces of Canada, and the western part of the continent of North America as far north as Vancouver Island and the Okanagan Valley (Buchan & Padilla, 2000). This species has led to many forms of mechanical, chemical, and biological controls to manage the nuisance growth of milfoil (Borrowman et al., 2014). New concerns have arisen surrounding increased invasiveness and resilience through the hybridization of Eurasian watermilfoil with the native northern watermilfoil (Borrowman et al., 2014).

2.7 Eutrophication and Algal Blooms

Biological effects of eutrophication include the increased growth of planktonic algae, cyanobacteria, and macrophytes. (Brogan et al. 2001). In freshwater lakes, eutrophication results in excessive growth of phytoplanktonic algae and cyanobacteria (blue-green) algae (Brogan et al. 2001). Several species of cyanobacteria can release noxious taste and/or odor-causing effects that can have negative impacts on the public and the drinking water industry (Wagner & Adrian, 2009; S. B. Watson et al., 2008; Winter et al., 2011). Many authors (Brogan et al. 2001, Breen et al. 2017, Yang et al. 2008) have highlighted that eutrophication can interfere with the quality-dependent uses of the waterway for aquatic habitats, recreational use, and as drinking water sources. For example, studies have shown that recent taste and odor events have occurred in Lake Ontario, which had major impacts on the drinking water provided to a large population, causing a widespread public reaction (Wagner & Adrian, 2009; Winter et al., 2011). Excessive

growth of phytoplanktonic algae and cyanobacteria (blue-green) algae is a result of eutrophication.

Although algal blooms can be a natural phenomenon, they have become a greater issue over the past few decades due to eutrophication, and in terms of extent and public perception (Anderson et al., 2002). Algal blooms are of particular concern in freshwater systems because of the potential of many species to produce toxins in the water that can impact both human and animal health (Winter et al., 2011). Many regions around the world share a common concern with nuisance growths of algae and algal blooms in lakes and rivers. For example, there have been reports of toxic cyanobacteria blooms in the Great Lakes since the late 1990s (Anderson et al., 2002; Winter et al., 2011). More specifically, there has been an increasing number of incidents of harmful algal blooms that threaten the integrity of water bodies in the Ontario Rideau Canal Waterway.

It is well known that human activity near lakes can have significant effects on the state of lakes. When nutrient-rich pollution enters a lake, this can cause toxic algal blooms, resulting in decreased oxygen levels, a loss of diversity, fish mortality and degradation of water quality (Zikey, 2022). Other human activities that promote nuisance growth of algae include acidification (Turner et al., 1995) and the introduction of non-native species (Anderson et al., 2002; Higgins et al., 2005; Vanderploeg et al., 2001). Typically in shallow eutrophic lakes, if macrophytes are absent, cyanobacteria dominates (Zastepa et al., 2014). It is important to note that submerged macrophytes can inhibit the growth of algae, and can also play a role in assisting in maintaining the health of aquatic systems (L. Wang et al., 2003; Wu et al., 2016). Macrophytes have the capability of limiting nutrient run-off in aquatic systems by taking up nutrients on the

shoreline as they enter the lake, before they enter the open water zone (Zikey, 2022). Submerged macrophytes can also compete directly with algae for nutrients and light (S. Wang et al., 2020). However, at a smaller lake scale where management occurs, strong year to year variations in cyanobacterial blooms remain challenging to explain and predict (Pick, 2016).

2.8 Public Perception of Macrophytes and Shoreline Aesthetics

People have differing views on how they value lakeshores (Engel & Pederson, 1998). Residents may relate shoreland beauty to settings of scattered trees and lakeside lawns, whereas others believe developed shorelines to be unnatural and not aesthetically pleasing (Engel & Pederson, 1998; Macbeth, 1989). The existence of aquatic plants on personal property can be viewed positively or negatively by property owners. Some may view lakeshore vegetation as favourable for angling but unfavourable for swimming (Engel & Pederson, 1998). Plant cover can affect the appearance of shoreline structures and how riparian owners envision "natural scenic beauty" (Engel & Pederson, 1998).

In a study by Larned et al. (2006), the authors examined a Papanui Stream in Christchurch, New Zealand, and conducted a neighbourhood mail survey to assess opinions about positive and negative aspects of a rehabilitation project to determine whether macrophytes were prominent in perceptions of stream health. The results of the study determined that there were two common perceptions about macrophytes. First, the survey participants stated that aesthetic value can be increased by the addition of macrophytes a low cover level, however, when macrophyte cover obstructs the view of flowing water, aesthetic value is reduced (Larned et al., 2006). Second, there was little perceived difference between the aesthetic value of native versus non-native macrophytes. Respondents states that frequent weeding to maintain low levels

of macrophyte cover was desirable regardless of whether the macrophytes were native or non-native species (Larned et al., 2006). The removal of aquatic plants by lakeshore property owners is an action that may lead to various impacts on lake ecosystems (Schroeder & Fulton, 2013). Authors Radomski & Goeman (2001) suggest that property owners are often likely unaware that their actions can cause negative impacts to an ecosystem's health. It is suggested by Schroeder & Fulton (2013) that owners may rationalize their decisions to remove aquatic plants from their lakeshore to improve access for recreation and desired aesthetic appearance. The ecological impacts of macrophyte loss may not be seen immediately due to the lag between plant removal and the appearance of ecological consequences (Jennings et al., 1999; Liu et al., 2007; Schroeder & Fulton, 2013).

In order to better understand the resilience of lake ecosystems and the impacts of plant removal on aquatic ecosystems, organizations such as lake associations need to better understand factors that predict the protection and removal of macrophytes by lake property owners (Schroeder & Fulton, 2013). Having a greater understanding of lakefront property owners' behaviour and motives could be informative and could improve the field of conservation. For example, if people seek to live on developed lakeshore with a lower abundance of emergent of floating-leaf vegetation, habitat loss and consequences of development could be inflated (Radomski & Goeman, 2001). Evidence of lake vegetative loss in relation to shoreline development is needed to substantiate previously completed work (Radomski & Goeman, 2001).

2.9 Mechanical Removal of Freshwater Macrophytes and Water Quality

Over the past 20 years, aquatic macrophyte control programs have developed across Canada (Chambers et al., 1999). These programs are most notable in British Columbia and

Ontario, in response to the invasion of Eurasian watermilfoil (Chambers et al., 1999). These programs are directed at improving water resource values by humans, and therefore, focus on reducing or eradicating weed beds that reduce aesthetics, interfere with recreation such as swimming and boating, impede navigation traffic, etc. (Chambers et al., 1999). There are, however, ecosystem consequences of extreme changes in macrophyte assemblage. For freshwater macrophytes, complaints of excessive biomass have served as the trigger for management action (Chambers et al., 1999).

Mechanical harvesting has become the preferred management strategy for nuisance macrophytes (Brooker & Edwards, 1975; Carpenter & Gasith, 1978). In the past, control of nuisance vegetation has been dealt with using chemicals (Wile, 1978). Mechanical removal was introduced as an alternative method that would be less damaging environmental effects (Wile, 1978). However, the impacts and effects of harvesting on aquatic environments are not well understood. The assessment of environmental impacts of macrophyte harvesting must consider changes in water quality, caused by disruption of littoral zones (Carpenter & Gasith, 1978). Although mechanical removal of macrophytes is the most commonly used method for eradication, it can also cause undesirable disturbances in aquatic communities (Wojciechowski et al., 2018). One of the most immediate physiochemical effects of mechanical macrophyte management techniques is the mobilisation of sediments (James, 2013). In situations where hand raking or excavators are used, bed sediment is almost always disturbed (James, 2013). In addition, when a harvester or cutter boat is used, the volume of sediment that is disturbed can vary depending on the depth of the waterway, the depth at which the plants are cut, and the amount of sediment that is trapped among the plant material (James, 2013). During harvesting of macrophytes, the stems are cut and the plants float to the surface where these are collected and

removed (Carpenter & Gasith, 1978). During this process, surficial sediments may be resuspended and dissolved material may leach from damaged stems (Carpenter & Gasith, 1978). The re-suspension of sediment and release of sequestered nutrients (P) combined with temporary reduced uptake of nutrients by plants, can trigger a waterbody to reach a state of hyper-eutrophication and flush other potential contaminants downstream (Quilliam et al., 2015).

Another immediate effect of mechanical managements technique is the release of solutes from the cutting or maceration of plant material (James, 2013). In sites where significant biomass and/or area of macrophytes are cut and where water velocity is minimal, the release of solutes could result in changes to water chemistry in the waterway (James, 2013). This was demonstrated in a New Zealand study where in a Waikato Drain, along with a short-term increase in turbidity, Wilcock et al., 1998) observed an increase in ammonia and decreases in dissolved reactive P and nitrate levels (James, 2013). Studies that have investigated the impacts of removing macrophytes have also identified limnological changes such as shifts in the composition and abundance of zooplankton (Choi et al., 2014; Maceina et al., 1992), oxygen depletion, increases in turbidity, oscillations in pH, and export of P and N generated from the bottom of the lake (Crossetti & Bicudo, 2008; Granéli & Solander, 1988; Waterman et al., 2011; Wojciechowski et al., 2018; Young et al., 2004).

There are also environmental costs associated with macrophyte removal which can lead to various levels of ecosystem disturbance, particularly if it promotes the spread of invasive plant species (Dorahy et al., 2009; Quilliam et al., 2015). The presence of macrophytes in eutrophic waters can inhibit the growth of algae by reducing the availability of light and nutrients (Quilliam et al., 2015). Therefore, the removal of large standing macrophytes in eutrophic

waterbodies could lead to faster growing algae and could shift the ecosystem to phytoplankton dominance (Quilliam et al., 2015; Sayer et al., 2010). Studies have reported the relationship between aquatic macrophytes and phytoplankton, such as the study of Boyd et al. (1971). In lakes where phytoplankton production is inhibited by secretion of substances produced by macrophytes or by competition for nutrients, removal of the vegetation could potentially result in an increase in algal production (Wile, 1978). This was demonstrated in a study by Neel et al. (1973) in Lake Sallie, Minnesota, where elevated algal densities following macrophyte removal after the first year or harvesting operations were documented. Contrastingly, in 1973, a plant harvesting program was initiated in Chemung Lake by Wile (1978). Following 2 years of preliminary studies, an experimental aquatic study was carried out. Wile (1978) concluded that an increase in algal biomass was not observed following harvesting. These contrasting results, among others, should be supported by further investigation to determine the relationship between algal densities and biomass following harvesting and macrophyte removal.

Other changes in ecosystem structure resulting from the use of chemical and mechanical measures to control macrophytes may lead to conditions that promote algal cyanobacteria growth and dominance in eutrophic lakes (Kleinman et al., 2011). Many recreational lakes in Ontario can experience excessive growth of aquatic plants, particularly Eurasian watermilfoil (Wile, 1978). Some of the most common effects of such excessive growth is the accumulation of organic matter, which increases the dispersion of pathogenic agents (Thomaz et al., 2009; Wojciechowski et al., 2018). In addition to changing ecosystem function, Eurasian watermilfoil can become a nuisance by forming dense mats on the surface of waterbodies that reduce open area in littoral zones, and wash up on shorelines (Buchan & Padilla, 2000). This can reduce the aesthetic appeal of lakes and areas available for swimming and boating, impacting local

navigation, fishing, and recreational uses (Buchan & Padilla, 2000; Thomaz et al., 2009; Wojciechowski et al., 2018).

There is a lack of quantitative data on the effectiveness of various best practice techniques to minimise the negative ecological impacts of mechanical and chemical macrophyte control. Although previous studies have documented that many lakes physio-chemical and morphometric features are related to macrophyte biomass and species composition, there is a lack of research that examines how these features are overall related to macrophyte cover (Cheruvelil & Soranno, 2008). It is important to better understand how and what variables predict lake macrophytes by includes lakes across Ontario including natural and anthropogenic features in this study. For example, studies by Duarte & Kalff (1987) and Rooney & Kalff (2000) have shown that the maximum depth of macrophyte growth is negatively related to the latitude of lakes across large geographic regions.

Chapter 3: Study Region

3.1 Introduction

The Rideau Canal is a waterway located in Ontario, Canada, extending from the City of Kingston in the south to the City of Ottawa in the north with a total distance of 202 kilometres (125 miles) (K. Watson, 2007). The canal was built between 1826 and 1831, presently using 45 lock stations, allowing vessels drafting up to five feet (1.52 metres) to navigate the route (K. Watson, 2007). During the late 1800s, two additional locks were built to connect the Town of Perth, via the Tay River, to the Rideau Canal (K. Watson, 2007). It was initially built as a military project to provide a connection between Montreal and Kingston, and is now being used mainly for recreational purposes over the last century (Robb, 2018). The Rideau Canal is a National Historic Site, and in 2000, the Rideau Canal Waterway system was designated as a Canadian Heritage River, and in 2007, it was designated a United Nations Educational, Scientific and Cultural Organization (UNESCO) World Heritage Site (LeBlond, 2009). Since construction in 1832, the shores of the Rideau Canal Waterway have been altered to support logging, farming, mining, and milling operations (LeBlond, 2009).

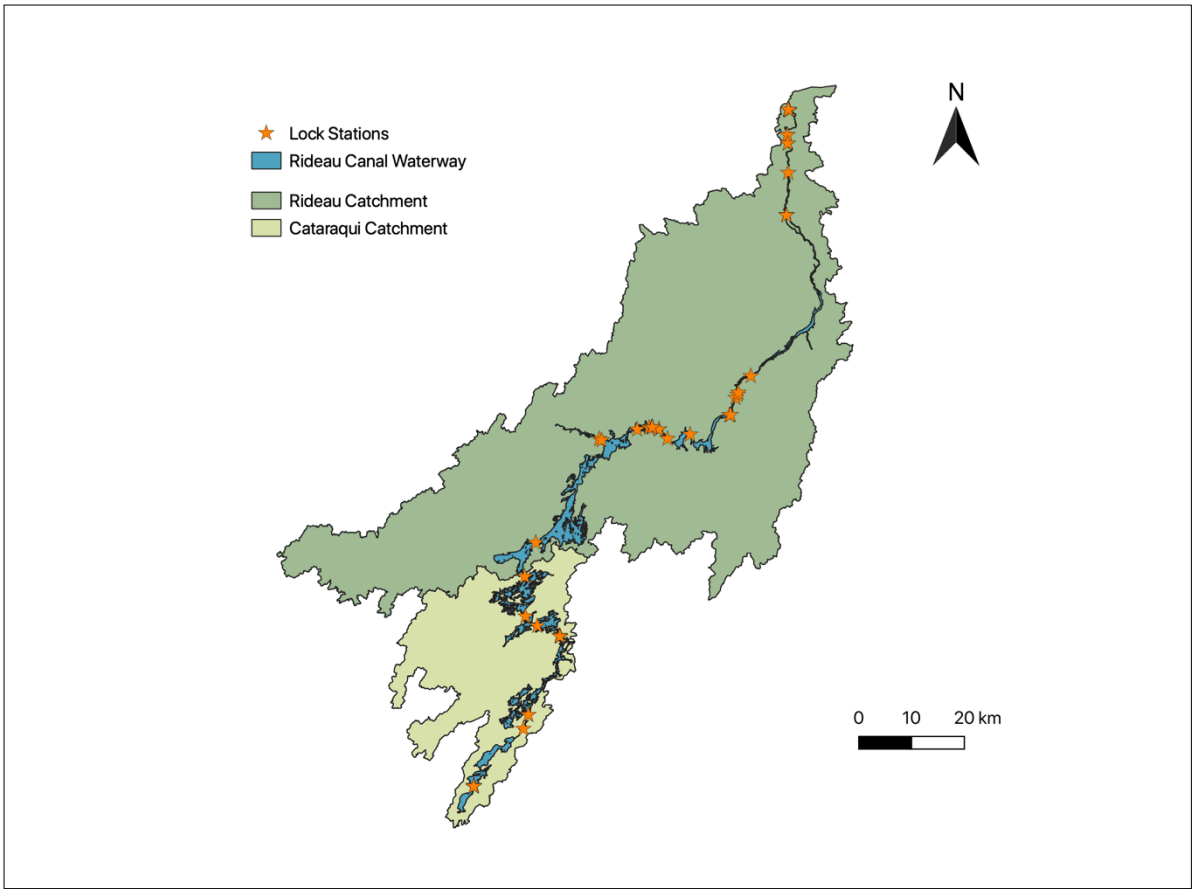

Figure 3- 1: Map of the Rideau Canal Waterway in eastern Ontario. Lock stations located on the waterway are represented by orange stars. The Rideau and Cataraqui Watersheds are divided into subwatersheds outlined in black. Map created in QGIS 3.4 using geospatial data acquired from Parks Canada and the RVCA.

3.2 Geography of the Rideau Canal Waterway

The Rideau Canal route crosses a major watershed divide with the Rideau River watershed to the north, which flows to the Ottawa River, and the Cataraqui River watershed to the south, which flows into the eastern end of Lake Ontario and then into the St. Lawrence River (K. Watson, 2007). The inundated landscape along the Rideau Canal Waterway is a result of the 47 masonry locks at 21 lock stations and 52 dams that were constructed (K. Watson, 2007). Canal structures such as locks and canal cuts represent less that 10 percent of the waterway (K. Watson, 2007). These connect a series of lakes and rivers to form the navigation network (K. Watson, 2007). Some of the lakes along the route pre-date the canal construction, others were formed by flooding when rivers sections were dammed (K. Watson, 2007).

South of Smiths Falls, the Rideau Canal consists of a series of 14 interconnected lakes that are typically used for recreational purposes and commercial fishing (LeBlond, 2009). There are 31 named lakes in the Rideau Lakes sub-watershed. The Upper Rideau Lake drains in two directions through the cut that connects it to Newboro Lake in the Cataraqui River section of the Canal and through the dam at Narrows Locks into Big Rideau Lake (*Rideau Lakes Subwatershed Report 2014*, 2014). The lakes for this study were selected based on data availability and macrophyte removal permits issued by Parks Canada. Figure E-1 demonstrates the Rideau Canal Waterway, with the lakes included in the study, in blue. Climate and geology are the two main contributing factors to the geography of the Rideau Waterway including the Rideau River, the Gananoque River, the Cataraqui River, and the Rideau Lakes (K. Watson, 2007). The southern part of the Rideau Route (Rideau Lakes/Cataraqui River region) contains exposures of old rocks of the Canadian Shield, known as the Frontenac Axis (K. Watson, 2007). The north and south portion of this area are younger sedimentary rocks (K. Watson, 2007). Although the underlying

geology, which helped to shape the Rideau River, dates back to one billion years ago, the most recent ice age has significantly shaped the superficial features of the Rideau region that are seen today (K. Watson, 2007). Within the Frontenac Axis, glaciers gouged out units of rocks of softer crystalline limestone, leaving behind depressions that filled with water to become some of the Rideau Lakes (K. Watson, 2007). As the ice retreated, the landscape underwent progressive changes. The retreating glaciers left behind deposits of glacial till, boulders, gravel, sand, silt and clay (K. Watson, 2007).

The bedrock of the entire region was depressed from the weight of the glaciers (K. Watson, 2007). In the northern sections of the Rideau Waterway, the glaciers depressed the bedrock to below sea level. As the glaciers retreated, salt water from the Atlantic Ocean flooded in to create the brackish Champlain Sea. Within the southern Ontario region, the earlier retreat of the glaciers, and the direction of the retreat, has resulted in more mature development of the rivers in that area (K. Watson, 2007). The Gananoque and Cataraqui River systems both have more mature development than the Rideau River, where isostatic rebound slowed the development of the river (K. Watson, 2007). The Rideau River is underlain by soft silt and clay sediments, deposited by the Champlain Sea on top of glacial till, which lies on top of flay-lying sedimentary rocks (K. Watson, 2007). This gives the Rideau River section a different appearance than the central and southern Rideau Waterway, most particularly the almost complete absence of lake development, a poorly developed stream drainage system, and many swamps (K. Watson, 2007). The central and southern portions of the Rideau Waterway are characterized by an increase in topographic expression and extensive lake development (K. Watson, 2007). This extends south on the Rideau Waterway to Kingston Mills, which makes the southmost exposure of the Canadian Shield on the Rideau Route (K. Watson, 2007).

The Rideau Waterway in the pre-flooding era spanned three watershed, the Rideau River watershed, the Gananoque River watershed and the Cataraqui River watershed (K. Watson, 2007). The main difference in the pre-flooding era was that the central Rideau lakes (Newboro, Clear, Indian, Opinicon and Sand) drained into the Gananoque River, not to the Cataraqui River, via the White Fish River (K. Watson, 2007). Prior to European settlement, the Rideau Route region was heavily forested (K. Watson, 2007). The heavy forest cover meant that water retention in the surrounding lands were far greater at the time of the canal construction than it is today (K. Watson, 2007). As forests were cut down for merchantable timber and/or to create farmland, the water retention of the land decreased (K. Watson, 2007). When the canal was built in 1826-1831, there was sufficient watershed capacity that reservoir lakes were not required (K. Watson, 2007). As the forests were removed and water retention decreased, mill dams on watershed lakes played a large role in maintaining reservoir capacity (K. Watson, 2007). By 1865, more water retention was required, and the first government dam was constructed at the outlet of Eagle Lake (K. Watson, 2007). Other government dams, leading to the reservoir system that is in place today, soon followed (K. Watson, 2007). Present day reservoir lakes include Canoe, Kingsford, Devil, Buck, Loughborough, Bobs, and Wolfe Lakes (K. Watson, 2007).

3.3 Institutional History of the Rideau Canal Waterway

From 1856 to 1936, various jurisdictions were responsible for the Rideau Canal as a commercial transportation corridor (Bergman et al., 2022). In 1925, the Canadian federal government recognized the Rideau Canal was as a National Historic Site of Canada (Bergman et al., 2022; Charron et al., 1982). From 1936 to 1972, Transport Canada held primary responsibility for managing the Rideau Canal (Bergman et al., 2022; Charron et al., 1982). In 1972, Parks Canada became the primary stewards of the Rideau Canal, where there was a

notable shift towards recreational use (Bergman et al., 2022; Charron et al., 1982). The Canadian federal government and Ontario provincial government signed an agreement in 1975 outlining a plan for development and land use as well as establishing jurisdictional responsibilities (Bergman et al., 2022; Charron et al., 1982). It was determined that the federal government was to manage navigation, water flows, recreation, historical and cultural heritage, and natural resource management was divided among municipal, provincial, and federal organizations (Bergman et al., 2022).

The governance of the Rideau Canal today is complex because it combines the actions of different levels of government, as well as community groups, nongovernmental organizations, and businesses (Bergman et al., 2022; Lemos & Agrawal, 2006). Parks Canada has ownership and jurisdiction over all in-water and shoreline works built in, on, or over the system (Parks, 2017). The waterway bed is administered by Parks Canada, and as such, Parks Canada has a role in the management of aquatic vegetation. Adjacent lands and the watersheds are privately owned or under other jurisdictions, notably municipalities and the provincial government (Bergman et al., 2022).

Today, the Rideau Canal encompasses 13 municipalities and counties, and the Cataraqui River watershed and the Rideau River watershed as well as their respective Conservation Authorities, the Cataraqui Conservation Authority and the Rideau Valley Conservation Authority (RVCA). Governance of the waterway also includes several First Nations (Bergman et al., 2022). Parks Canada engages with the Algonquins of Ontario, the Haudenoshaune (Mohawks of Bay of Quinte, and the Mohawk Council of Akwesasne), Alderville First Nation, and the Williams Treaties First Nations Signatories, whose territories intersect with the Rideau Canal (Bergman et

al., 2022). Research on the various perspectives and interests of stakeholders that are involved in management is necessary to better understand the complexities of the social and ecological systems, as well as improve management and governance (Bergman et al., 2022).

Volunteer residents and cottage owners around the Rideau Canal Waterway are passionate and enthusiastic about the waterway and have created various lake associations. Local groups on the Rideau Canal Waterway play various roles as stewards in communities along the waterway by educating residents through a number of methods, including newsletters and conferences, participation opportunities in community science, monitoring lakes, and promoting environmental awareness (Bergman et al., 2022; Rees, 2014). Lake associations play an important role with one another, as well as with other community groups, scientists, regional groups (i.e., Federation of Ontario Cottagers' Associations), conservation authorities, and government entities such as municipalities and Parks Canada.

3.4 Ecosystem Management of the Rideau Canal Waterway

Parks Canada plays an important role in maintaining the aquatic ecosystem of the Rideau Canal Waterway. The Rideau Canal Waterway has been managed by Parks Canada since 1972, as part of the Canada-Ontario-Rideau-Trent-Severn-Agreement (CORTS) (Charron et al., 1982). Parks Canada oversees cutting and removing excessive aquatic vegetation from the navigation channel (Parks, 2017). Parks Canada owns mechanical harvesting equipment and operates a program of aquatic plant removal at many locations along the Rideau Canal Waterway (Parks, 2017). On the canal stretches of the system, Parks Canada directly removes macrophyte biomass by mechanical harvest when it is impeding with navigation. Should a waterfront owner on any of the lakes that make up the system wish to remove aquatic vegetation directly from their property,

a permit from Parks Canada is required. Parks Canada issues permits that allow property owners to remove macrophytes over a 10-metre by 30-metre area adjacent to their shoreline (Parks, 2017).

However, numerous other federal, provincial, and municipal agencies also play a role in the management of the Rideau Canal Waterway. One important partner is the RVCA. This authority is one of Ontario's 36 Conservation Authorities. Under Ontario's Conservation Authorities Act, the RVCA is responsible for advancing conservation, restoration, development and management of natural resources on the watershed (*Biology and Ecology*, 2020). The RVCA has gathered information on surface water quality since the 1970s and continues to do so. RVCA has worked to protect the health and integrity of the watershed through environmental monitoring and reporting, on the ground restoration and stewardship, conservation, and work with other municipalities, levels of government, and academia (*Biology and Ecology*, 2020). Together, supported by research and monitoring, these multiple uses constitute a practical example of managing a reservoir ecosystem (Charron et al., 1982).

During the summer during growing periods, collaborative and volunteer-based monitoring and reporting programs have been implemented and run for the Rideau Canal Waterway and the surrounding waterways. The RVCA staff collects information on the physical conditions of waterways, including vegetation, wetlands, shorelines, and land uses (*Biology and Ecology*, 2020). This information is used to determine the conditions of streams, lakes, and rivers over the watershed to better inform management decisions for future purposes (*Biology and Ecology*, 2020). These projects rely on conservation authority staff who monitor samples, including homeowners on the watershed. This project aims to further explore the Eastern Ontario

data from Parks Canada, and several groups and volunteer-based organizations including RVCA using GIS data and build on it to identify what variables influence of shoreline or near shore macrophyte removal in the Rideau Canal Waterway in relation to water quality and riparian zone land-use. The types of data that will be used for the purpose of this project include hydrological data, nutrient data, land-use data, algae data, and permit distribution data.

3.5 Ecological Implications of Construction and Operations

Prior to construction, the surrounding the Rideau Canal Waterway was composed of mixed woodland and wetland (Bergman et al., 2022; Karst & Smol, 2000). The construction of the Rideau Canal profoundly altered local ecology. Prior to the construction of the Rideau Canal, three watersheds made up the Rideau Route: the Rideau River, Gananoque River, and the Cataraqui River (Bergman et al., 2022). The Gananoque River and the Cataraqui River were heavily altered and transformed from a delta-like wetland into a constructed river, forming a connection between the upper Rideau and lower Cataraqui watershed (Bergman et al., 2022). Several lakes were created (i.e. Colonel By Lake); others were greatly expanded (i.e. Bobs Lake and Opinicon Lake) during construction (Bergman et al., 2022). The magnitude of ecological change appears to vary between lakes, with different biological indicators in deeper lakes (i.e. Big Rideau Lake) suggesting that there may be less change to these compared to the more shallow lakes, such as Lower Rideau Lake (Bergman et al., 2022; FORREST et al., 2002).

Since European settlement began in the early 1800's, lakes within the Rideau Canal Waterway have received increased nutrient loading from various anthropogenic sources including logging, settlement, flooding, agriculture, industry, and finally cottage development (Forrest et al., 2002). During the canal construction, lakes and rivers were flooded by

approximately two to three metres to create navigable depths for transportation purposes (Forrest et al., 2002). It was likely that increased hydrological changes and increased nutrients in the canal lakes was a result of the flooding of lowland areas (Forrest et al., 2002). The construction of dams and locks altered the natural hydrology of the lakes and rivers in the Rideau waterway system by controlling the inflow and outflow of upstream lakes (Forrest et al., 2002). Although there are occasional repairs to the dams and locks, water levels within the canal lakes are maintained within annual water-level restrictions (~1m) to ensure navigation depths, water supply, hydroelectricity and runoff dilution requirements are met (Forrest et al., 2002). The hydrological controls on lakes in the Rideau Canal Waterway system may have also increased nutrient concentrations by augmenting natural residence times, which provides for the lakes to accumulate nutrient in the water column, and release P and other nutrients from sediments (Forrest et al., 2002).

Post-construction sedimentary records suggest gradual increases in nutrients, that correspond with increasing housing development in the region during the early 20th century (Bergman et al., 2022; Forrrest et al., 2002). On a global scale, canal systems are widely recognized as channels for invasive species (Bergman et al., 2022; Kim & Mandrak, 2016; Lin et al., 2020). Key invasive species include Eurasian watermilfoil (*Myriophyllum spicatum* L.) which has spread rapidly through connected waterways (Bergman et al., 2022; Borrowman et al., 2014). The connection of the watersheds allows for new corridors and habitats, which can result in invasions and encourages the replacement of local biotas with non-indigenous species (biotic homogenization) (Bergman et al., 2022). In the Rideau Canal Waterway, the level of ecological aquatic connectedness is understudied.

3.5.1 Waterfront Land-use and Development

A large portion of settlement in the Rideau Canal Waterway area came following the construction of the canal. During the construction of the canal, trees were cut for miles on each side of the canal (Forrest et al., 2002; Passfield, 1982). By the 1860s, two-thirds of the canal's surrounding area had been cleared for agricultural and settlement purposes (Forrest et al., 2002). By the 1880s, several industries became established along the canal including mills, cheese factories, maple sugar factories, and extraction operations (Forrest et al., 2002). The resulting industrial discharges likely added nutrient to the Rideau Canal Waterway system (Forrest et al., 2002). Following the 1890s, tourism increased and took on a major role in the Rideau area (Forrest et al., 2002). Cottage development began in the 1930s and has since increased along the canal route (Forrest et al., 2002). To date, a number of canal communities, as well as rural and seasonal residents, are situated along the Rideau Canal Waterway system (Forrest et al., 2002).

Canals in close proximity to human settlement provide for rich recreational opportunities (Bergman et al., 2022). This can lead to a social-ecological tension between shoreline development and conservation of local biodiversity and ecological systems (Bergman et al., 2022). Parks Canada has several regulatory tools for guiding in-water and shoreline works. These tools are meant to work in conjunction with land-use planning guidelines and permits from various Conservation Authorities and municipalities that guide riparian and upland development (Bergman et al., 2022). In more rural areas, a large portion of development is from waterfront landowners who are typically seasonal cottage users. Common waterfront developments include docks, boathouses, erosion controls, and aquatic vegetation management (Bergman et al., 2022). Problems associated with shoreline development can range from aesthetic value and interest in

maintaining historic character, to concerns regarding freshwater biodiversity and ecosystem health (Bergman et al., 2022).

Parks Canada is heavily involved in the development of municipal shoreland policies, and review of land use and development activities along the Rideau Canal Waterway (Parks Canada, 2005). Since the 1950s, many sections of shoreland on the waterway have evolved from natural and agricultural use to intensive cottage and suburban development (Parks Canada, 2005). Not only is the waterway historically important, but it also supports an essential tourism-based industry for the region, which is valued at approximately $24 million annually (Parks Canada 2005). The effects of waterfront development have caused a dramatic change in the character and state of the natural environment and has reduced the quality and diversity of riparian ecosystems (Parks Canada, 2005). Parks Canada's primary interest in land uses adjacent to the waterway and waterway lands is the enhancement of the natural, cultural, and scenic values of the waterfront lands (Parks Canada, 2005). Therefore, the potential impacts of construction and alterations of in-water and shoreline works, buildings, and boating activities on the natural environment is of primary concern (Parks Canada, 2005).

Parks Canada encourages municipalities and other agencies to contribute to the protection of the waterway through municipal planning policies (Parks Canada, 2005). Parks Canada has the legal mandate under the Ontario Planning Act as both the reviewing agency and landowner to provide input to the development of all municipal plants and planning decisions including private land-use development. Additionally, Parks Canada plays a large role in the review of municipal planning policies, official plans and shoreland development proposals (Parks Canada, 2005). However, there are insufficient resources to educate or consult with landowners, developers and

municipalities on their role in protecting the waterways and the use for waterfront development (Parks Canada, 2005). Although there is a general acceptance of the need to protect the natural values of the waterway, there is a need to identify waterfront lands of natural value and to protect them through official plant designation and private land stewardship. This provides a great opportunity and basis to examine the natural states of the aquatic ecosystems in relation to land-use and waterfront development. Further research is required to better understand how the alteration of shoreline habitats affect water quality, and how to use this knowledge to address conflicting social and ecological values of aquatic vegetation (Bergman et al., 2022).

Chapter 4: Methods

4.1 Permits

Data were provided by Parks Canada containing information on 85 permits that were issued between 2013-2020 on the entire Rideau Waterway (Figure E-1). Aquatic macrophyte removal permits are valid for five years. These data were loaded into QGIS 3.4 for analysis. The size of area for vegetation removal under the permits distributed ranged from 3 metres by 15 metres to 1524 metres by 1524 metres.

4.2 Water Quality Data

Water quality in the Rideau Canal Waterway varies spatially and is closely monitored by municipalities, health units, lake associations, citizen science groups, Conservation Authorities, the Ontario Ministry of the Environment, Conservation and Parks (MOECP), and Parks Canada (Bergman et al., 2022). Datastream is an open access hub for sharing water data and was consulted to determine what databases contained data relevant for the project study area. Water quality data were obtained from the Provincial (Stream) Water Quality Monitoring Network (PWQNM), and the Ontario Lake Partner Program (LPP). The PWQNM measures water quality in rivers and streams across Ontario (Ontario Ministry of Environment, Conservation and Parks, 2022). More than 400 locations are currently monitored in partnership with Ontario's Conservation Authorities, participating municipalities and provincial parks (Ministry of the Environment, Conservation and Parks, 2022). Participating partners collect water samples on approximately a monthly basis and deliver samples to the Ministry's laboratory where they are analyzed for a suite of water quality indicators (Ministry of the Environment, Conservation and Parks, 2022). The Ontario Lake partner program provides TP, chloride, and calcium

concentration data, and water clarity data for hundreds of Ontario's inland lakes (Ontario Ministry of Environment, Conservation and Parks, 2022). The data are collected annually through volunteer monitoring efforts, known as community science (Ontario Ministry of Environment, Conservation and Parks, 2022).

To determine the relationship between macrophyte removal and water quality, physical and chemical variables were examined from 2015 to 2022. Water quality variables that were examined in this study are Secchi depth, total phosphorus, calcium, and chloride. The mean value of each water quality variable by lake from 2015 to 2022 was used for the analysis. The mean value of each water quality variable was determined by calculating the average value of various samples within a year within an individual lake. The annual averages were then combined to produce a final mean value. TP was log transformed using the function log() in R to reduce the skewness of the variable. It is important to note that Loon Lake and Lower Rideau Lake did not have data available for 2015 to 2022, therefore, an average from the year 2014 was used for each of the lakes for the analysis.

4.3 Land Use Data

The RVCA conducted a littoral zone mapping project that was completed in 2019 for the 31 lakes within the Rideau Canal Waterway sub-watershed including Big and Upper Rideau lakes. The Cataraqui Region Conservation Authority also provided a land use classification shapefile that was used for analysis of land-use data for the study sites. To develop a map of developing riparian-zone usage in the study area, spatial datasets of land use in each of the Rideau Canal Waterway sub-watersheds from the RVCA and the Cataraqui Region Conservation Authority were compiled. Land use within each of the sub-watersheds was examined to

determine if there is a relationship between land use and permit distribution. QGIS was used to complete the analysis of percentages of each land-use type within the sub-watershed.

Land use was examined using the NAD83 / Ontario MNR Lambert projection in QGIS for this study. Land cover was classified into ten major land cover types in the Rideau Waterway and Cataraqui Region– wooded area (hedgerow, island, plantation, treed), unevaluated wetland, evaluated wetland, crop and pasture, aggregate site (pit or quarry), settlement (estate, pervious commercial, industrial, pervious homestead, residential, townhome), transportation (rail, major or minor road, unpaved), meadow, water (buffer around wetland, lake, pond, river), and unclassified land. The land use variable was determined by dividing the area of the lake's sub-catchment by the area of the land cover in the sub-catchment.

4.4 Physical Variables

The physical characteristic variables examined for the purpose of this study were mean depth, catchment to lake area, catchment to lake volume, catchment area, number of permits, number of permits per km shoreline, number of algal blooms, nearest neighbour, distance matrix algal blooms, deviation from circle, mean distance of permits, maximum depth, shoreline length, unclassified land, lake volume, and surface area. The mean depth, maximum lake depth, shoreline length, volume, and lake surface area for each lake were determined using data provided by RVCA and the Cataraqui Region Conservation Authority. The catchment to lake area was determined by dividing the sub-catchment area by the lake area. The sub-catchments were delineated by RVCA and the Cataraqui Region Conservation Authority.

Catchment to lake volume was measured by dividing the sub-catchment area by the lake volume. The number and distribution of permits was determined by Parks Canada from 2013-2020. Permits per kilometre shoreline was determined by dividing the lake's shoreline length in kilometres by the number of permits distributed on the given lake. The number of algal blooms were measured Parks Canada during 2010 and 2011.

A Nearest Neighbor Analysis (NNA) was completed to examine algal blooms in 2010 and 2011 in relation to the nearest macrophyte removal permit issued by Parks Canada in the Rideau Canal Waterway from 2013 to 2020. The NNA uses the QGIS tool called distance matrix which locates the closest point from the algal blooms layer from the macrophyte removals layer issued by parks Canada. Given the locations of all permits distributed by Parks Canada, the nearest algal bloom from both 2010 and 2011 can be identified for each location where a permit was distributed. For the Nearest Neighbour Analysis, the permit distribution shapefile layer was the input layer. Algal blooms layer was the target point layer. There were 70 lakes with algal blooms in total, and 12 lakes where algal blooms were present where no permits were distributed (Table D-1). In lakes where multiple permits are distributed, a separate variable was used whereby the average distance from each permit to all algal blooms in the lake was calculated.

The status of lakes was used as a categorical variable. Information provided by Parks Canada and K. Watson (2007) was used to classify lakes that were flooded or man-made as "created", and lakes that existed prior to the construction of the canal were classified as "regulated." The shoreline deviation of each lake was calculated for each waterbody in the Rideau Canal Waterway system. The level of shoreline measurement is referred to as the shoreline development index. The shoreline development index measures the deviation of the

size of the lake from a circular pattern. Hutchinson & Edmondson (1957) presented an index of the development of shoreline that is defined as the ratio of the length of lakeshore to the circumference of a circle of equal area to the lake (Osgood, 2005). The shoreline development index (DL), is calculated as follows, where L is the length of the lake shoreline and A is the lake's surface area (Osgood, 2005):

$$DL = L/(2\ (\pi A)1/2)$$

DL is a unit-less index, so L and A must have compatible units (for example, m and m^2, respectively) (Osgood, 2005). A DL for a circle has a value of 1.0, and the value increases as the shape of the lake surface deviates from that of a circle (Osgood, 2005). The shoreline development index is used to compare lake shapes and to evaluate the impact of factors associated with shorelines (Osgood, 2005). For example, as stated by Osgood (2005), the relative amount of littoral area increases with DL. The mean distance of permits was calculated to by measuring the average distance between all permits distributed on the Rideau Canal Waterway using QGIS.

4.5 Statistical Analysis

4.5.1 Correlation Matrix

A correlation matrix of the water quality and land use variables was produced using the 'corrplot' package (Wei & Simko, 2021) in R. The corrplot package was used as a visual explanatory tool to identify patterns in the data (Figure 5-1). The correlation matrix was used to eliminate variables that were highly correlated prior to analysis. A correlation coefficient of 0.7 and -0.7 was used as a cut-off, to determine which variables would be used for the analysis. The

following variables were not included for the purpose of this analysis; calcium, maximum depth, volume, shoreline length, surface area, catchment area to lake area, nearest neighbour, distance matrix algal blooms, unclassified, settlement, transportation, water, unevaluated wetland, evaluated wetland, and meadow.

4.5.2 Dendrogram

A hierarchical cluster analysis was used to determine the major divisions between the study lakes based on their water quality and land-use data (Galili, 2015). This analysis allows for a visualization of the similarities of study lakes based on the explanatory variables and produces clusters of lakes that are most similar in their environmental parameters. The 'dendextend' package (Galili, 2015) was used to produce a visualization of clustering of the data by lake.

4.5.3 Multiple Linear Regression

The statistical analyses were performed in R Studio version 3.6.1 (R Core Team, 2020). The scale() function in R was used in the pre-processing of the data . Scaling normalized the dataset using the mean value and standard deviation, to ensure that differences in units and scale were accounted for prior to modelling and analysis.

The relationship between water quality variables and physical characteristic variables were examined with permits and permits per km shoreline as the response variables. Multiple linear regression models were selected to predict both the number of total permits issued and the number of permits issued per kilometre shoreline that explained the most amount of variability in the data, while being the most precise models. Stepwise model selection was performed based on Akaike Information Criteria with the function stepAIC() in the R package 'MASS' (Venables & Ripley, 2002) to compare and select the best multiple linear regression models. The stepwise

selection was used as it combines forward and backwards selections, starting with no predictors and sequentially adding the most contributive predictors. It ensured that the assumptions of multiple linear regression were met.

Analysis of variance (ANOVA) tests were carried out using the anova() function in R. A statistical significance level of 0.05 was used in this analysis to test for significance. The package 'mlbench' (Leisch & Dimitriadou, 2021) was used to test the final model against predicted values and observed values.

4.5.4 Heat Map

Parks Canada issues permits that allow property owners to remove macrophytes over a 10-metre by 30-metre area perpendicular to the shoreline. A heat map was created by loading a shapefile of macrophyte permits issued by Parks Canada into QGIS 3.4 (Figure E-5). Areas that may be harvested are maximum 50% of the water frontage to a maximum of 10m (30 feet) wide. The heat map radius was set to 500 metres to demonstrate finer details and variation in point density. The heat map uses a colour-coded system to demonstrates areas of high density where permits have been issued to shoreline property owners by Parks Canada from 2013-2020 on the Rideau Canal Waterway. The map used the EPSG 3161 projection and is shown in kilometres.

Chapter 5: Results

5.1 Permits

There was a total of 85 permits distributed by Parks Canada on the Rideau Canal Waterway from 2013-2020. 30 permits were distributed in the Rideau watershed, and 55 permits were distributed in the Cataraqui watershed. The highest density of permits distributed on the Rideau Canal Waterway were in the southern portion of the waterway (Figure E-4, Figure E-5).

5.2 Water quality

The mean total phosphorus value across all lakes was 22.15 µg/L. The highest total phosphorus value was 61.35 µg/L in Cranberry Lake and the lowest total phosphorus value was 7.95 µg/L in Big Rideau Lake. The mean Secchi depth value across all lakes was 3.38 m. The most shallow Secchi depth value was 0.83 m in Colonel By Lake and the deepest value was 5.14 m in Sand Lake. The mean chloride value across all lakes was 6.93 mg/L. The highest chloride value was 12.03 mg/L in Upper Rideau Lake and the lowest chloride value was 0.70 mg/L Lower Rideau Lake.

5.3 Land Use

The average percentage of land use across all catchments were as follows: wooded area – 45.72 % (varied between 67.5-20.3%); unevaluated wetland – 1.04% (varied between 6.4-0%) evaluated wetland – 4.96% (varied between 19.4-1.2%); crop and pasture – 22.85% (varied between 55.8-4.9%); aggregate site – 0.04% (varied between 0.3-0%); settlement – 1.14% (varied between 6.3-0%); transportation – 0.58% (varied between 3-0%); meadow – 0.24% (varied between 2.3-0%) ; water – 23.33% (varied between 34.5-15%); and unclassified land – 0.14% (varied between 1.4-0%). The highest percentages of land use across the catchments of

the relevant lakes were wooded areas (45.72%), crop and pasture (22.85%), and water (23.33%). The highest percentages of land use of crop and pasture within a catchment, were located in Colonel By Lake (55.8% of catchment, Kingston Mills Catchment), Benson Lake, Indian Lake, Loon Lake, Newboro Lake (22.8% of catchment, catchment Above Chaffeys Locks). Colonel By Lake was classified as a created lake, whereas the other lakes were classified are natural, but classified as regulated lakes.

5.4 Statistical Analysis

5.4.1 Correlation Matrix

Several physical and chemical variables were highly correlated ($r>0.7$) (Figure 5-1); therefore, a single representative variable was selected from each highly correlated group to be included as a potential explanatory variable in the linear regression analyses.

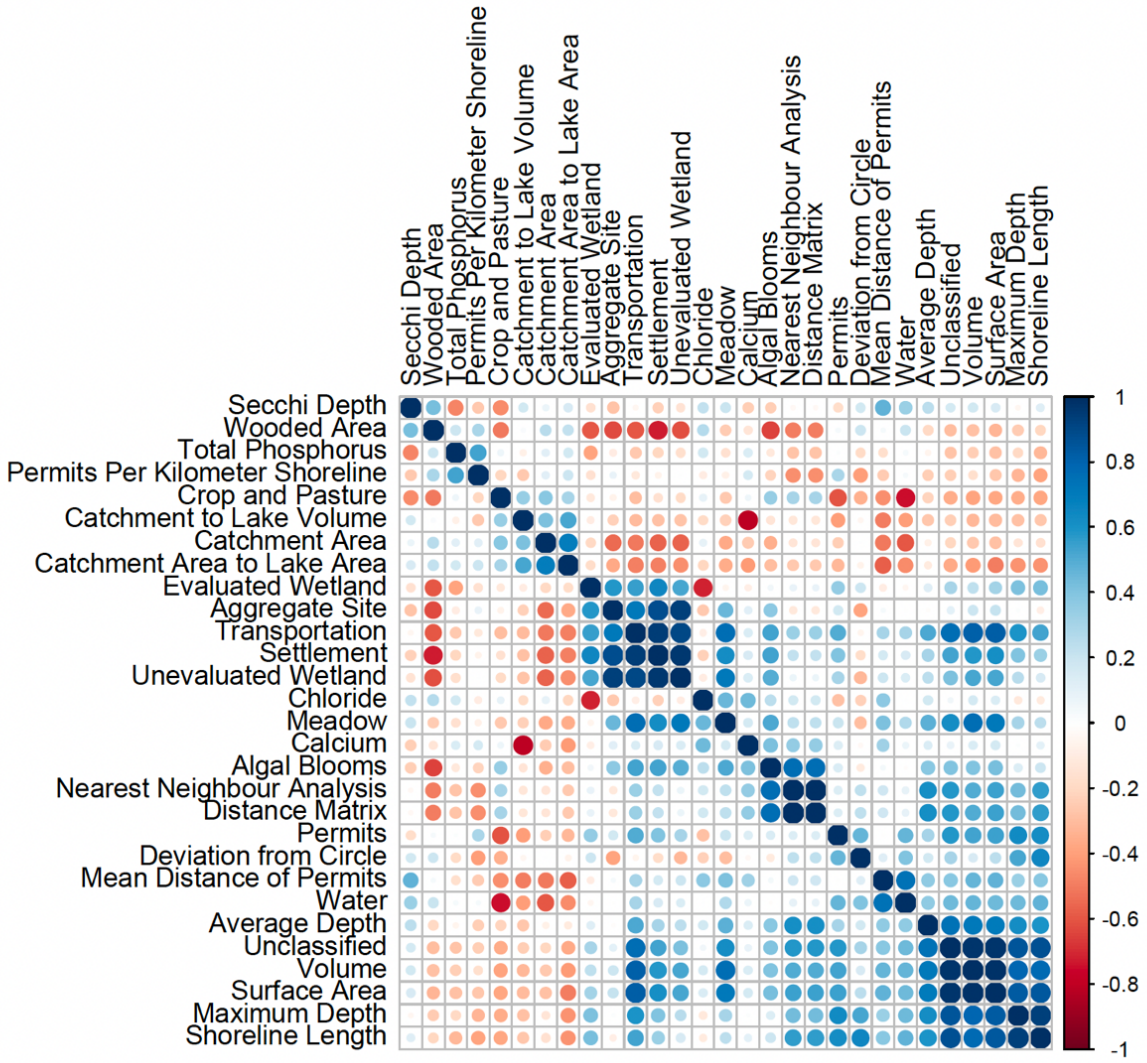

Figure 5- 1: Correlation matrix demonstrating the correlation among chemical and physical properties. Comparisons of physical and chemical properties are displayed with a colour gradient to denote Spearman's correlation coefficients. Positive correlations are indicated in blue and negative correlations are indicated in red.

5.4.2 Dendrogram

The dendrogram clustered Big Rideau Lake alone in the primary node (Figure 5-2), suggesting that this lake had the most dissimilar environmental data to the other study sites. Big Rideau Lake had 15 macrophyte removal permits distributed on the lake and had the largest shoreline length and surface area (100 km^2) of all the study sites. Big Rideau Lake also had a high deviation from circle value of 8.46 which would indicate that its shoreline has many bays, and consequently had a high number of permits issued on the waterbody. Big Rideau Lake also had 4 reported algal blooms from 2010-2011. Big Rideau Lake is classified as a regulated waterbody. Lower Rideau Lake and Upper Rideau Lake were located in the next node. Lower Rideau Lake is classified as a regulated waterbody, has 9 permits issued, has a deviation from circle value of 4, and had 4 reported algal blooms from 2010-2011. The Upper Rideau Lake is classified as regulated, had 6 permits distributed, had a shoreline deviation from circle value of 1.45, and had 7 reported algal blooms. Colonel By Lake, Loon Lake, and Sand Lake were in the next node. Colonel By Lake had 1 permit issued, had a deviation from circle value of 3.41, had 8 reported algal blooms and was classified as a created waterbody. Loon Lake had 1 permit issued, a deviation from circle value of 4.38, 0 reported algal blooms and was classified as regulated waterbody.

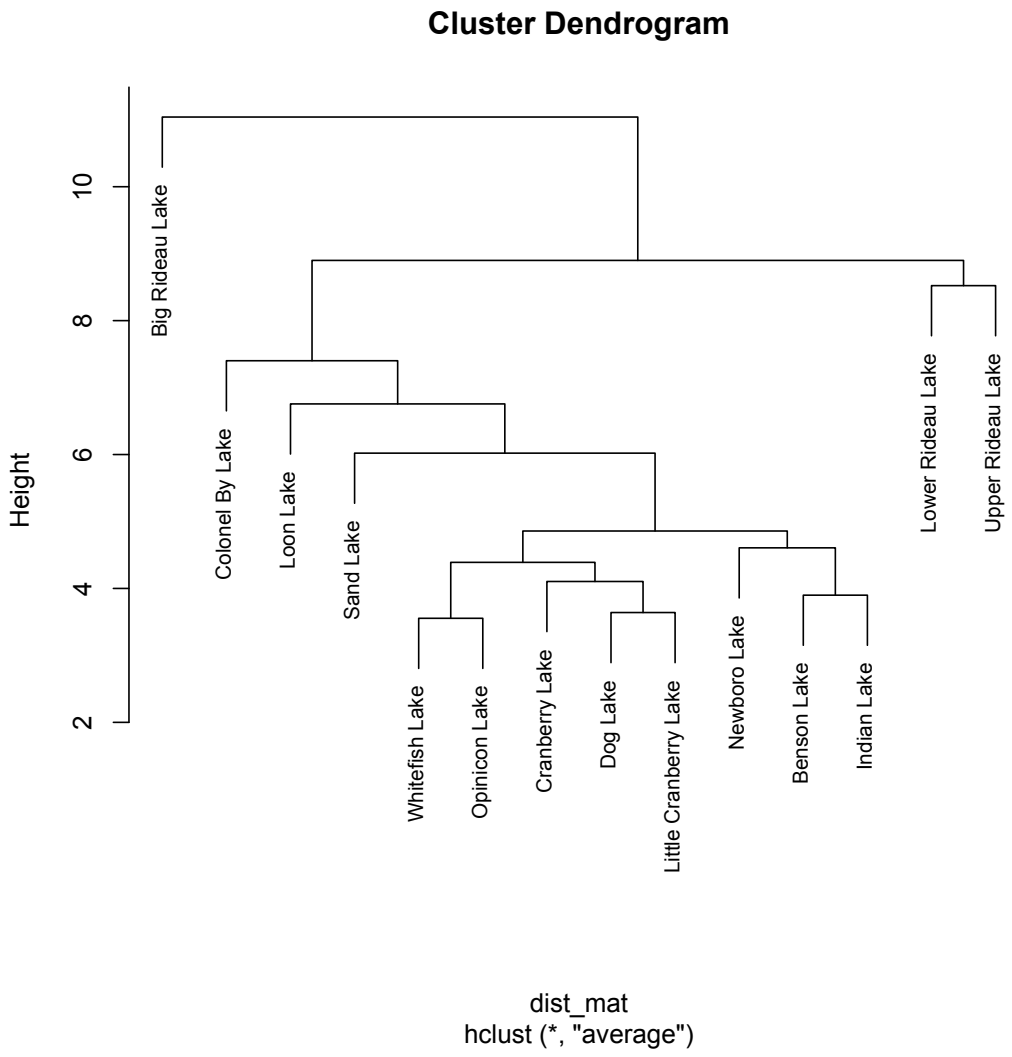

Figure 5- 2: Dendrogram of dataset using dendextend package in R. This dendrogram demonstrates the hierarchical cluster analysis for the classification of lakes in the Rideau Canal Waterway used in this study. The height represents the scaled distance at which each fusion of the clusters were made.

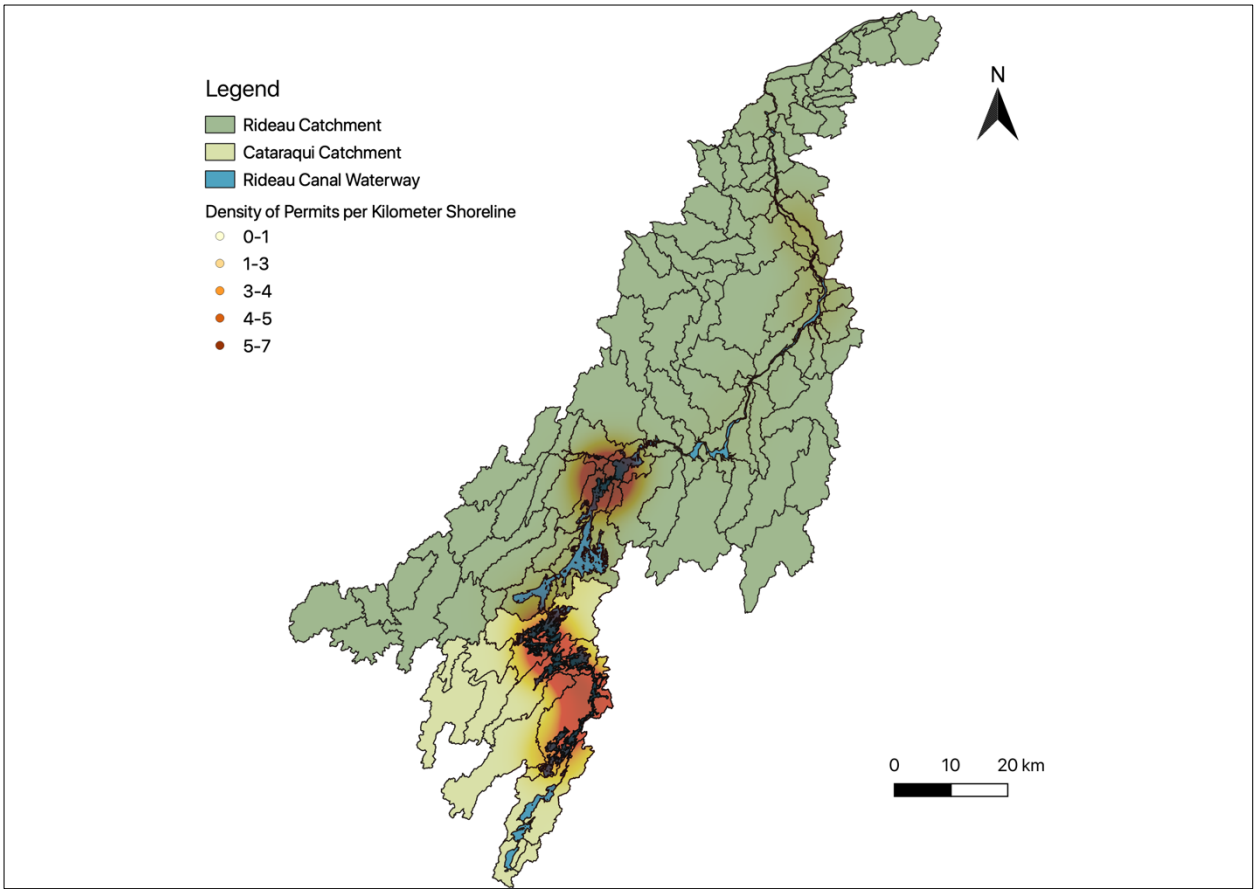

Figure 5- 3: Heat map of the density per kilometre of permits distributed on the Rideau Canal Waterway. Map created in QGIS 3.4 using geospatial data acquired from Parks Canada and the RVCA.

5.4.3 Permits Model

The multiple linear regression model selected to identify predictors of permits, included crop and pasture, Secchi depth, algal blooms, catchment area and TP (Appendix A, A.1).

Permits = 25.2 + (-0.4758)crop.and.pasture + (-2.758)Secchi + (0.8485)Algal.blooms + (4.901e-08)Catchment.Area + (-2.435)log(TP)

The model had an adjusted R^2 value of 0.94 and a p-value of 1.952e-05 indicating that it is statistically significant. The predicted versus observed plot showed the effect of the model, comparing it to the observed data (Figure 5-3). The model was a good fit, as shown by the points falling closely around the fitted line.

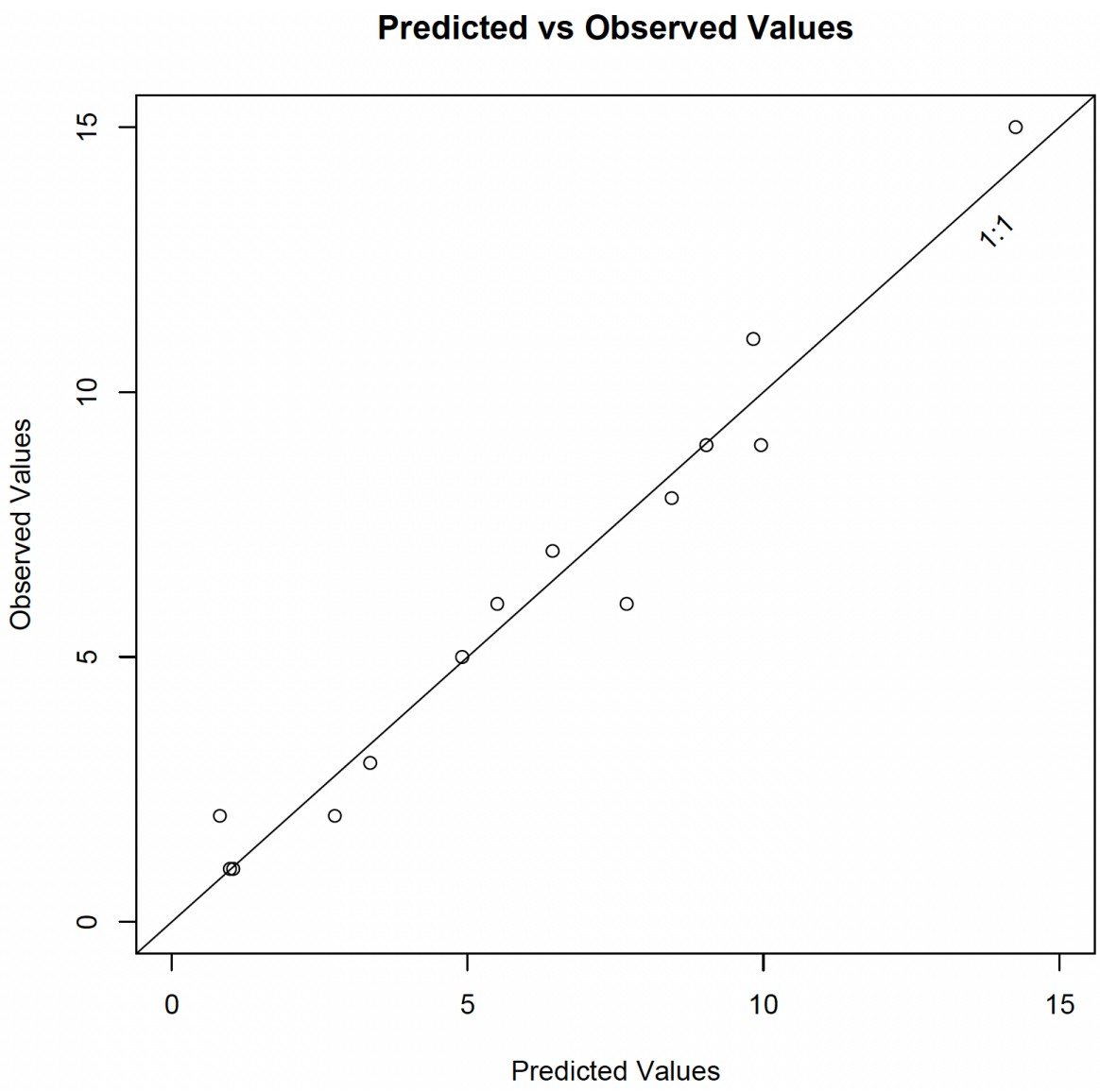

Figure 5- 4: Plot of actual versus predicted values for the permits model described in section 4.10. Model has an adjusted R^2 value of 0.94 and a p-value of 1.952e-05. A one-to-one line is included in the plot, demonstrating that the model is a good fit.

A secondary simpler multiple regression model was selected that contained fewer variables. This model contained the variables crop and pasture, Secchi depth, algal blooms, and catchment area (Appendix A, A.2). An ANOVA was carried out between the original model and the secondary model. The variability explained between the two models was not significantly different based on the ANOVA.

Permits = 3.10 + (-0.2151)crop.and.pasture + (-0.6361)Secchi + (1.26)Algal.blooms + (4.135e-08)Catchment.area

Spearman rank correlation was used to determine the relationship between the permits per kilometre shoreline variable and the model's predictor variables. The variable crop and pasture was a predictor of both permits and permits per kilometre shoreline. Permits and crop and pasture had a negative correlation value of -0.642. When examining the other predictor variables of the multiple linear regression using permits as the response variable, catchment area had a correlation value of -0.351 indicating that as catchment area decreases, the number of permits increases. Secchi depth had a negative correlation of -0.298. This could indicate that lower Secchi depth values leads to an increase in permits distributed. Algal blooms had a correlation of 0.138 indicating that perhaps increased algal blooms, which is a symptom of increasing nutrients or eutrophication, can result in higher macrophyte biomass, therefore, influencing the number of permits being requested to Parks Canada. This could also be a result of a decline in desired shoreline aesthetics. TP has a correlation of 0.004, indicating that as TP levels increase, the number of permits issued by Parks Canada also increases. Nutrients such as phosphorous can stimulate the growth of planktonic algae, which can also result in a decline in a desired shoreline aesthetic, leading to an increase in permits issued.

5.4.4 Permits Per Kilometre Shoreline Model

The multiple linear regression model selected to identify predictors of permits per kilometre shoreline included crop and pasture, deviation from circle, status, chloride, catchment area, and catchment to lake volume (Appendix A, A.3).

Permits per km shoreline = 0.586+ (-0.00599)crop.and.pasture + (-0.03886)Deviation.from.circle + (-0.137)Status +(-0.0168)Chloride + (9.301e-10)Catchment.Area +(-2.951e-09)Catchment.to.Lake.Volume

The model has an adjusted R^2 value of 0.779 and a p-value of 0.006 indicating that it is statistically significant, and that the model explains 77% of the variability within the data. All assumptions of multiple regression were met. The package 'mlbench' (Leisch & Dimitriadou, 2021) was used to test the original model against predicted values and observed values (Figure 5-4). The predicted versus actual plot showed the effect of the model, comparing it to the null model. The model was a good fit, as it the points are close to the fitted line.

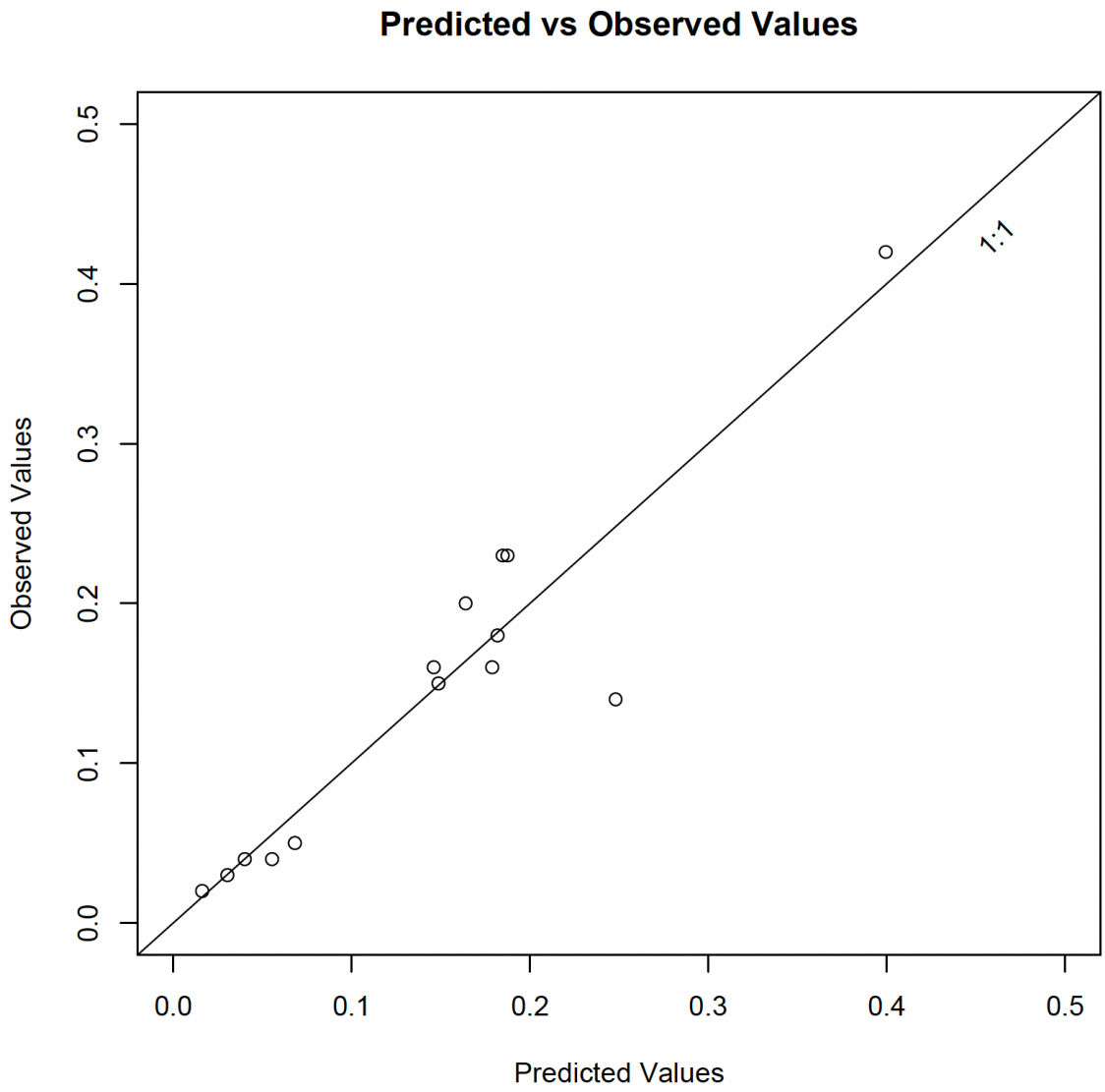

Figure 5- 5: Plot of actual versus predicted values for the permits per kilometre model described in section 4.10. Model has an adjusted R^2 value of 0.779 and a p-value of 0.006. A one-to-one line is included in the plot, demonstrating that the model is a good fit.

A secondary, simpler multiple regression model was selected that contained fewer variables. This model contained the variables crop and pasture, deviation from circle, status, and chloride (Appendix A, A.4).

Permits per km shoreline = 0.633 + (-0.005629)crop.and.pasture + (-0.03589)Deviation.from.circle + (-0.14807)Status + (-0.012991)Chloride

This model had an adjusted R^2 value of 0.618 and a p-value of 0.011. This model explains 60% of the variability. An ANOVA was carried out between the original model and the secondary model. The ANOVA test had an output Pr(>F) value of 0.061, determining that the alternate model wasn't a drastically better fit than the original model. The variability between the two models is not significant when running the ANOVA. Therefore, the model can be simplified by removing the two variables: catchment to lake volume and catchment area. Although a simpler model still has good predictability, the original model containing 6 variables is still the optimal model.

Spearman rank correlation coefficient was used to determine the relationship between the permits per kilometre shoreline variable and the model's predictor variables. Deviation from a circle had a negative correlation of -0.115. The status of lakes could not be compared using Spearman's correlation value as it is a categorical variable. Chloride had a negative correlation of -0.119, indicating that as chloride decreases, the permits distributed per km shoreline increases. Catchment to lake volume had a positive relationship with permits per km shoreline, indicating that as the catchment to lake volume increases, the explanatory variable does as well.

Chapter 6: Discussion

Macrophytes provide refuge to many species of fish and invertebrates and offer a variety of ecosystem services (Jeppesen et al., 2012; O'Hare et al., 2018). It is critical to better understand what is influencing landowners and waterfront users to request macrophyte removal permits from by Parks Canada. Water quality and land use data were examined to determine if any parameters were good predictors of macrophyte removal permits issued by Parks Canada from 2013-2020 on the Rideau Canal Waterway. The results of the linear regression model indicate that the best predictors of permits issued by Parks Canada over the 7 years using multiple regression were crop and pasture, Secchi depth, algal blooms, catchment area, and TP. In addition, the best predictors of permits per kilometre shoreline using multiple regression are crop and pasture, deviation from circle, status of lakes, chloride, catchment area, and catchment to lake volume. The two models were proven to be statistically significant and have implications for future natural resource management and ecosystem health practices moving forward.

6.1 Crop and Pasture

In Eastern Ontario, excessive macrophyte growth is linked to several factors including nutrient run-off and enrichment to waters from agricultural and neighbouring urban areas, and cutting vegetation from the water (Parks Canada, 2017; Xiao-e Yang et al., 2008). Crop and pasture and catchment area were both predictor variables of permits and permits per km shoreline. It is possible that permit distribution (macrophyte removal) is therefore primarily a result of enrichment to waters from nearby agricultural lands and urban areas, as crop and pasture is an explanatory variable within the model. These findings suggest that if there is a high percentage of land use of crop and pasture, this could influence nutrient runoff and loading,

therefore, causing an abundance of macrophyte growth, and potentially eutrophication, ultimately leading to more permits for removal being requested by shoreline users. This is consistent with the history of the Rideau Canal Waterway. As mentioned earlier, by the 1860's, two thirds of the canal's surrounding area was cleared for agricultural and settlement purposes (Forrest et al., 2002). As a result, there were likely nutrient additions to the Rideau Canal Waterway system over a prolonged period. The application of fertilizers can increase nutrient levels and accelerate eutrophication (Brown 1985. Dressing et al. 2016). These findings are supported by other studies in the field that state that downstream water quality and macrophyte abundance can be linked to: a) the proportion of urban or industrial land (Sass et al., 2010; Tong & Chen, 2002; White & Greer, 2006); b) the proportion of agricultural land which can influence nutrient loading (Gorman et al., 2014; Knoll et al., 2003), and c) the type of agricultural land (Arbuckle & Downing, 2001; Sun et al., 2018). These findings are also consistent with studies that have identified that shifts in aquatic environments have been linked to both agriculture (Crosbie & Chow-Fraser, 1999; Dodson et al., 2005; Egertson et al., 2004; Heegaard et al., 2001; Rasmussen & John Anderson, 2005; Virola et al., 2001) and urban land use practices (Alexander et al., 2008; Arts, 2002; Bowen & Valiela, 2001; Hauxwell et al., 2003; S. A. Nichols & C. Lathrop, 1994; Sass et al., 2010). This is also supported by Forrest et al. (2002), because by the 1860's, two-thirds of the Rideau Canal Waterway's surrounding area was cleared for agricultural and settlement purposes, (Forrest et al., 2002) followed by several industries being established along the canal in the 1880's, followed by cottage development beginning in the 1930's which has ever since increased along the canal route.

These findings are also consistent with those of Szpakowska et al. (2022), where a canonical correspondence analysis showed that farming, had the greatest influence on water

quality and vegetation in small waterbodies. The analysis specifically showed that arable land located in the buffer zone, extending to 100m form the shoreline had the greatest influence. This effect may be related to the retention of nutrients in soil, caused by sorption complex. Nutrient and sediment runoff from agricultural land use can have major impacts on aquatic plant communities, causing them to shift predominantly to floating and emergent species (Sass et al., 2010). This could likely cause a shift to less aesthetically pleasing shorelines for riparian zone landowners and cottage owners, which can influence and potentially increase the number of permits requested and distributed on the waterway.

6.2 Catchment Area and Catchment to Lake Volume

It was also determined that catchment area is a good predictor of both permits and permits per kilometre shoreline. In a study by Szpakowska et al. (2022), the authors concluded that the catchment areas of waterbodies significantly affect small aquatic ecosystems. Specifically, as the distance from the shoreline increases, the influence of the catchment area decreases (Szpakowska et al., 2022). This offers an opportunity for future research to examine the relationship between catchment area size in relation to the number of permits being distributed on lakes within a canal system and the potential implications on aquatic ecosystems and macrophyte communities.

It is also possible that smaller lakes are more influenced by surrounding catchment land cover, and therefore are likely to be more impacted by agricultural land use and agricultural runoff. Catchment to lake volume can also act as a good predictor, as this measurement indicates whether lakes shallow or deeper in nature. Future research could be conducted to determine if there a strong trend demonstrating whether permits are distributed in shallow or deeper lakes.

6.3 Water Quality Variables

Secchi depth, TP, and chloride were all identified as statistically significant predictors of permits and permits per km shoreline on the Rideau Canal Waterway. These findings align with the study by Lauridsen et al. (2015), where the authors identified that macrophyte assemblage richness in the north and central European lakes was related to Secchi depth and chlorophyll *a*, which also aligns with the findings of a study by Søndergaard (1983). Secchi depths were correlated and deemed an important predictor for macrophyte richness (Lauridsen et al., 2015), as they measure the amount of suspended material (algae, microscopic organisms, and sediment) within a water column. This is consistent with the findings of this study and suggest that the transparency and amount of suspended material within a water column influence permit distribution along the Rideau Canal Waterway. Long term monitoring of Secchi depth and potentially chlorophyll *a* can be carried out to further explore if these parameters continue to act as predictors of macrophyte removal permits being issued.

TP was found to be another predictor variable of permit distribution on the Rideau Canal Waterway. Submerged macrophytes can translocate and excrete P, along with other nutrients. Phosphorus is a nutrient that is translocated to macrophyte shoots, where it can enter waterbodies through either the released of living or decaying shoots (Carpenter & Lodge, 1986). The lakes with the highest percentages of P over the study period are Cranberry Lake, Dog Lake, Little Cranberry Lake, and Colonel By Lake. It is important to note that all 4 of these lakes have their status as “created”, therefore, they were created following the flooding of the canal. The lakes with the highest percentages of TP over the course of the study are all located in the Cataraqui portion of the Rideau Canal Waterway. It is also important to note that respectively, these lakes have 6, 8, 7 and 1 permit distributed by Parks Canada on the lakes. Of the 85 permits that were

distributed, 22 permits were distributed on these four lakes alone. This supports the statement by Forrest et al. (2002) that there were increased hydrological changes and increased nutrients in the canal lakes as result of the flooding of lowland areas. This also supports their statement (Forrest et al., 2002) that the construction of dams and locks altered the natural hydrology of the lakes and rivers in the Rideau waterway system by controlling the inflow and outflow of upstream lakes. This also likely resulted in an increase of nutrients in the lower portion of the canal system in the Cataraqui region. In contrast, lower levels of TP could indicate that macrophytes are absorbing the nutrients, therefore, increasing macrophyte abundance, therefore, influencing shoreline owners to request permits from Parks Canada. It was also determined that chloride is a good predictor of permits per kilometre shoreline. Moving forward, close monitoring of nutrient levels within the Rideau Canal Waterway lakes should be carried out over time to determine if there is a strong relationship between water quality and macrophyte removal permits being issued over time.

6.4 Algal Blooms

Within the Rideau Canal Waterway, cyanobacteria blooms can occur throughout most of the system if conditions are favourable, although blooms are most commonly found in the southern sector of the waterway where wetlands were flooded to create lakes during the construction of the canal (Karst & Smol, 2000) and where phosphorus concentrations are consistently high (>35 µg/L). The findings of Karst & Smol, (2000) are interesting and complement the findings of this study, as algal blooms were identified as a predictor of permits distribution on the Rideau Canal Waterway, and there was clear evidence of higher concentration of permits for macrophyte removal distributed in the lower, southern portion of the Rideau Canal Waterway (Figure E-4, Figure E-5).

Shoreline habitants may view algae as a threat to shoreline aesthetic and associate it with macrophyte growth, however, it may be unclear to shoreline owners that macrophytes can reduce algal blooms under eutrophic conditions (Bakker et al., 2010). The highest numbers of algal blooms on lakes occurred on Colonel By Lake (8 algal blooms), Upper Rideau Lake (7 algal blooms), Big Rideau Lake (5 algal blooms), and Whitefish Lake (5 algal blooms) (Table D-1). It is important to note that these lakes with the highest reported number of algal blooms also had a significant number of permits being issued on the lake over the study period.

This offers an opportunity for future studies to determine if there is a higher number of permits issued on lakes with algal blooms versus lakes with no reported algal blooms, and if the removal of macrophytes can result in a higher frequency of algal blooms. This would require close monitoring of algal blooms over time. Further research is required to determine what management actions, such as an increase in water flow, or perhaps closer monitoring of macrophyte removal, can reduce cyanobacteria blooms.

6.5 Deviation from Circle

Deviation from circle was identified as a predictor variable of permits per kilometre shoreline and suggests that the amount that a lake deviates from a circle, could indicate that it is more likely to have macrophyte removal permits submitted. The more a lake shoreline deviates from a circle, it could imply that there are more bay areas and closed off shoreline, leading to less water flow and nutrient build up. This can lead to algal blooms in bay areas, and as a result, could influence lake residents to submit requests for macrophyte removal permits on lakes in the Rideau Canal Waterway.

6.6 Status of Lakes

It was determined that the status of a lake (created versus regulated) is a good predictor variable of permits per kilometre shoreline. Of the 14 lakes examined for this study, 5 lakes were classified as created and 9 regulated. In the study by Forrest et al. (2002), the authors compared limnological changes associated with 19th century canal construction and other catchment disturbances in four lakes within the Rideau Canal system. It was determined that the magnitude of ecological change appears to have differed across lakes, with biological indicators in deeper lakes, like the Big Rideau Lake, suggesting less change in comparison to shallower lakes, such as Lower Rideau Lake. These findings, along with the preliminary findings of this study, offer an opportunity for further research to determine if the magnitude of ecological change over time In the Rideau Canal Waterway vary across lakes that are created versus regulated.

Moving forward, more research can be conducted and collected to determine if permits issued on the Rideau Canal Waterway are more frequently submitted on created versus regulated lakes on the waterway. This could help to inform natural resources management practices and would also help conservation authorities to target educational programs and initiatives for shoreline users within different parts of the waterway to learn more about the significance of macrophytes in aquatic ecosystems, and the impacts of macrophyte removals in lake ecosystems.

Chapter 7: Conclusions

This thesis explored the environmental influences associated with macrophyte removal permits requested by shoreline owners and issued by Parks Canada along the Rideau Canal Waterway. The Rideau Canal Waterway has become an important waterway for social, ecological, and economical reasons. There has become a need to identify the relationships between macrophyte removal permits being distributed on the waterway, and possible associated with natural and anthropogenic landscape features in and water quality. The main objective of this study was to examine and quantify water quality parameters and natural and an anthropogenic land use predictors of permit distribution on the Rideau Canal Waterway. In Chapter 5, it was determined that the best predictors of permit distribution were crop and pasture, Secchi depth, algal blooms, catchment area, and TP. It was also determined that the best predictors of permits distributed per kilometre shorelines were crop and pasture, deviation from circle, status of lake, chloride, catchment area, and catchment to lake volume.

The findings of this thesis stress the importance of investigating the potential associations of macrophyte removal with water quality and watershed land use. The Rideau Canal Waterway is a complex waterway where both environmental and human impacts play a role in overall aquatic ecosystem health. The results of this study support previous findings that further research is needed to examine the relationships between natural and anthropogenic landscape features and water quality with macrophyte cover (Cheruvelil & Soranno, 2008). Macrophyte removal permits distribution should be explored and maintained to determine the impacts on aquatic macrophyte assemblages and communities. Therefore, it is essential that further efforts are made to quantify and identify the associations and impacts of water quality and land use on

macrophytes and how this influences shoreline owners and removal of macrophytes in the Rideau Canal Waterway to help identify further management actions. Future research focussed on quantifying the impacts of macrophyte removal and water quality can be used to support natural aquatic ecosystems.

The findings of this study can help to better inform and provide resources to consult with landowners, developers, and municipalities on their role in protecting the Rideau Canal Waterway and the impacts of waterfront development (Parks Canada, 2005). Research findings have indicated that stakeholders want more opportunities to participate in governance and inform environmental and natural resource policy (Bergman et al., 2022; Mistry, 2020). Considering stakeholders' passion and enthusiasm for the Rideau Canal Waterway and their involvement in community science research, further research should be conducted on how to further stakeholders' knowledge about the impacts of macrophyte removals and the implications on lake ecosystems. There is a need for more opportunities to improve governance and monitoring that will allow for a more holistic approach for management of the Rideau Canal Waterway System (Bergman et al., 2022).

In the meantime, management efforts can be made to better collaborate with shoreline owners to better inform lake users of the impacts of macrophyte removal on aquatic ecosystems. As anthropogenic pressures in the Rideau Canal Waterway continue to increase, further social and ecological disturbances are also expected to increase, such as loss of habitat and biodiversity, toxic algal blooms, and management of land use and infrastructure (Bergman et al., 2022). As shown in a number of studies mentioned in this thesis, human practices including shoreline development, land use, and macrophyte removal can reduce resilience to stressors (Bergman et

al., 2022). There are, however, opportunities for research, community science, and government coordination to increase resilience and promote positive change within the Rideau Canal Waterway system (Bergman et al., 2022).

There are opportunities to address gaps in social-ecological interactions such as aquatic plant management, to incorporate more ecological knowledge into plant management, at both the shoreline scale and catchment scale to enhance ecosystem habitats. There is also a great opportunity for continued work on improving water quality throughout the Rideau Canal Waterway system.

Appendices

Appendix A - Statistical Code and Output Summary

A.1 Model 1

full.model1 <- lm(Permits ~ crop.and.pasture + Secchi + Algal.blooms + Catchment.Area + log(TP), data = permits2)

Coefficients:	Estimate	Std. Error	T value	Pr(>|t|)
(Intercept)	2.524e+01	2.799e+00	9.017	1.83e-05 ***
Crop.and.pasture	-4.758e-01	3.612e-02	-13.173	1.05e-06 ***
Secchi	-2.758e+00	3.093e-01	-8.914	1.99e-05 ***
Algal.blooms	8.485e-01	1.349e-01	6.292	0.000235 ***
Catchment.Area	4.901e-08	8.144e-09	6.018	0.000317 ***
Log(TP)	-2.435e+00	6.161e-01	-3.952	0.004226 **

A.2 Model 2

full.model2 <- lm(Permits ~ crop.and.pasture + Secchi + Algal.blooms + Catchment.Area, data = permits2)

Coefficients:	Estimate	Std. Error	T value	Pr(>|t|)
(Intercept)	3.101e+00	2.841e+00	1.092	0.28673
Crop.and.pasture	-2.051e-01	8.311e-02	-2.468	0.02186 *
Secchi	-6.361e-01	4.885e-01	-1.302	0.20632
Algal.blooms	1.260e+00	3.515e-01	3.584	0.00165 **
Catchment.Area	4.135e-08	1.904e-08	2.172	0.04091 *

A.3 Model 3

full.model3 <- lm(permits.per.km.shoreline ~ crop.and.pasture + Deviation.from.circle + Status + Chloride + Catchment.Area + Catchment.to.Lake.Volume, data=permits2)

Coefficients:	Estimate	Std. Error	T value	Pr(>|t|)
(Intercept)	5.860e-01	8.411e-02	6.966	0.000218 ***
Crop.and.pasture	-5.990e-03	1.496e-03	-4.003	0.005167 **
Deviation.from.circle	-3.886e-02	8.043e-03	-4.831	0.001897 **
Status	-1.371e-01	3.091e-02	-4.435	0.003025 **
Chloride	-1.683e-02	5.596e-03	-3.008	0.019714 *
Catchment.Area	9.301e-10	3.512e-10	2.648	0.033034 *

A.4 Model 4

full.model4 <- lm(permits.per.km.shoreline ~ crop.and.pasture + Deviation.from.circle + Status + Chloride, data=permits2)

Coefficients:	Estimate	Std. Error	T value	Pr(>|t|)
(Intercept)	0.632648	0.107886	5.864	0.00024 ***
Crop.and.pasture	-0.005629	0.001774	-3.173	0.01131 *
Deviation.from.circle	-0.035899	0.010434	-3.441	0.00738 **
Status	-0.148070	0.040371	-3.668	0.00517 **
Chloride	-0.012991	0.007116	-1.826	0.10118

Appendix B - Summary of Variables

Table B- 1: Summary of variables used for this study.

Variables	Minimum Value	Maximum Value	Median	Mean	Standard Deviation
TP (µg/L)	7.95	61.35	13.97	22.15	15.77
Secchi Depth (m)	0.83	5.14	3.73	3.38	1.34
Calcium (m)	12.3	28.44	24.58	24.14	3.76
Chloride (mg/L)	0.70	12.03	7.08	6.93	2.76
Number of Permits	1.00	15.00	6.00	6.07	4.12
Mean Depth (m)	2.10	12.50	3.60	5.06	3.21
Maximum Depth (m)	2.60	110.00	11.35	21.52	28.27
Volume (m^3 x106)	3.68	726.80	40.65	117.98	215.00
Shoreline Length (m)	14300.00	300000.00	44400.00	60790.00	72000.00
Surface Area (m^2)	1940000.00	100000000	7180000.00	17430000.00	2817000.00
Catchment Area (m^2)	27700544.00	170198354.00	135208740.00	118609076.10	47905515.28
Catchment Area to Lake Area	989837.1	81434619.1	17413573.80	28543042.3	29304711.60
Catchment to Lake Volume	122202.10	46249552.70	3101648.10	7324622.08	11984957.50
Deviation from Circle	1.45	8.46	5.08	4.80	1.96
Algal Blooms	0	8	2.5	2.71	2.76
Nearest Neighbour	0	2816	562.51	817.73	945.43
Mean Distance for Permits (m)	0	1928.15	323.86	572.76	562.86
Distance Matrix Algal Blooms	0	2815.83	562.43	817.23	945.69
Unclassified (%)	0	1.4	0	0.14	0.40
Aggregate Site (% landcover)	0	0.3	0	0.036	0.09
Settlement (%)	0	6.3	0	1.14	2.20
Transportation (%)	0	3	0	0.58	1.16
Water (%)	15	34.5	24.2	23.33	5.14
Unevaluated Wetland (%)	0	6.4	0	1.04	2.25
Evaluated Wetland (%)	1.2	19.4	3.15	4.96	5.17
Wooded Area (%)	20.3	67.5	47.2	45.72	11.90
Crop and Pasture (%)	4.9	55.8	20.4	22.85	11.80
Meadow (%)	0	2.3	0	0.22	0.635
Permits per Kilometre Shoreline	0.02	0.42	0.16	0.15	0.11

Appendix C - Sample of Data

Table C- 1: Sample of data collected for the study. Water quality data ranges in the lakes from 2013 to 2020.

Lake Name	TP (µg/L)	Secchi Depth (m)	Calcium (m)	Chloride (mg/L)	Permits (#)	Mean Depth (m)	Max Depth (m)	Catchment Area	Deviation from Circle	Algal Blooms	Crop and Pasture (%)	Status of Lake
Cranberry Lake	44.97	2.68	25.5	7.39	6	2.6	5.5	135208740	1.58	0	20.4	Created
Dog Lake	41.13	1.69	21.72	7.48	8	5.79	50	135208740	5.36	0	20.4	Created
Little Cranberry Lake	61.35	2.99	25.4	7.67	7	6.59	5.4	135208740	6.22	3	20.4	Created
Whitefish Lake	13.37	3.72	24.69	7.09	11	2.1	7.6	135208740	6.59	5	20.4	Created
Benson Lake	13.83	3.23	23.76	4.68	5	2.7	12	170198354	4.8	0	28.1	Regulated
Indian Lake	12.3	4.82	24.62	7.06	3	10	2.6	170198354	3.15	3	28.1	Regulated
Loon Lake	11.44	4.8	12.3	4.56	1	2.7	8.2	170198354	4.38	0	28.1	Regulated
Newboro Lake	14.1	4.3	28.44	10.94	2	3.2	23.8	170198354	5.91	1	28.1	Regulated
Opinicon Lake	13.75	3.73	24.5	6.59	9	2.5	10.7	75873594	5.74	0	4.9	Regulated
Sand Lake	13.36	5.14	24.53	6.76	2	5.2	14.3	27700544	6.19	2	14.7	Regulated
Big Rideau Lake	7.95	3.81	24.45	5.29	15	12.5	110	106780463	8.46	5	10.3	Regulated
Lower Rideau Lake	18.3	1.32	24.52	0.7	9	2.8	23	51827272	4	4	24	Regulated
Colonel By Lake	25.68	0.83	27.28	8.73	1	4	6.1	116336753	3.41	8	55.8	Created
Upper Rideau Lake	18.5	4.22	26.23	12.03	6	8.1	22	60380064	1.45	7	16.2	Regulated

Appendix D - Algal Blooms

Table D- 1: Table demonstrating lakes with algal blooms that occurred from 2010 to 2011, monitored through a citizen science program coordinated by Parks Canada.

Lake Name	Number of Reported Algal Blooms
Colonel By Lake	8
Upper Rideau Lake	7
Whitefish Lake	5
Big Rideau Lake	5
Big Rideau Lake	4
Little Cranberry Lake	3
Indian Lake	3
Sand Lake	2
Newboro Lake	1
Cranberry Lake	0
Dog Lake	0
Benson Lake	0
Loon Lake	0
Opinicon Lake	0

Appendix E - Maps

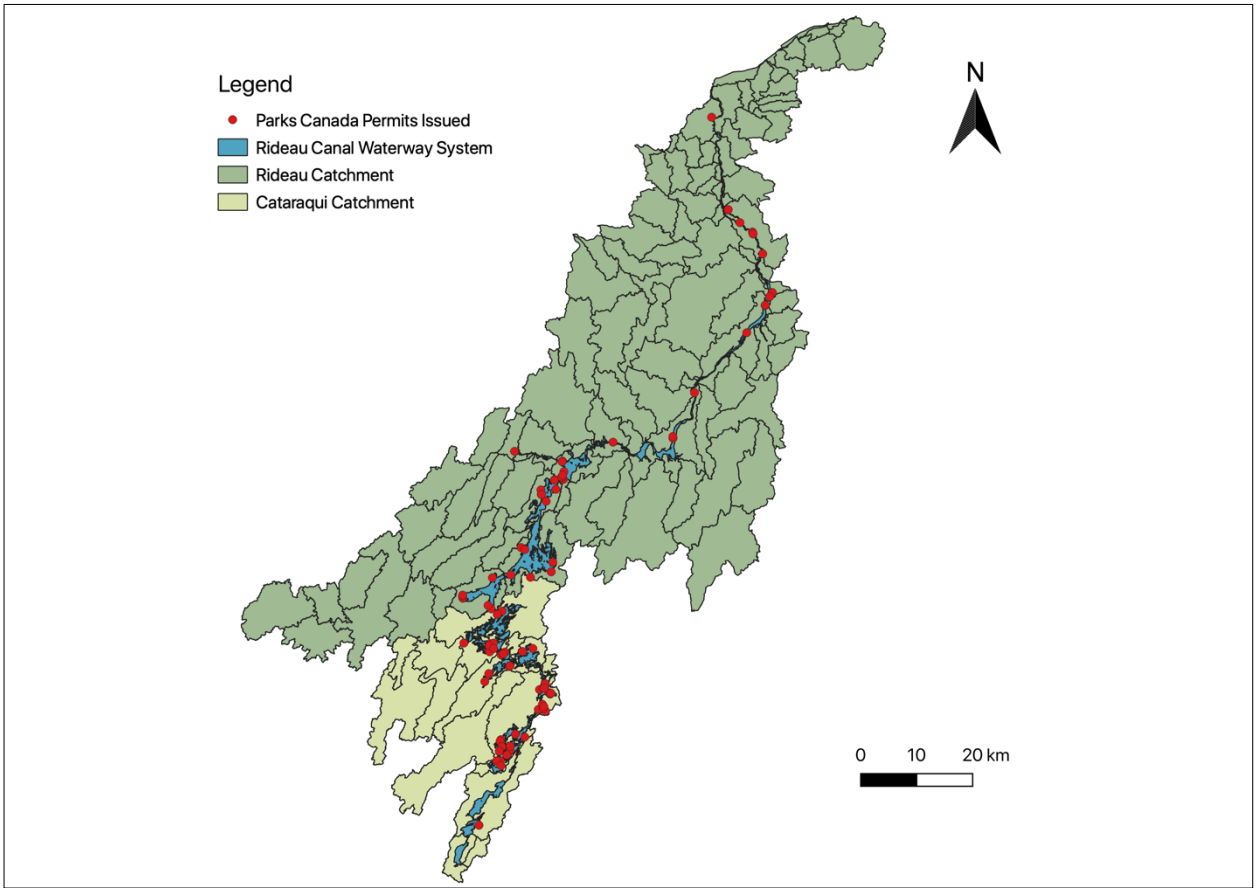

Figure E- 1: Map of macrophyte removal permits issued by Parks Canada on the Rideau Canal Waterway in Ontario, from 2013-2020 within the Rideau and Cataraqui watershed. The Rideau and Cataraqui Watersheds are divided into subwatersheds outlined in black. Map created in QGIS 3.4 using geospatial data acquired from Parks Canada and the RVCA.

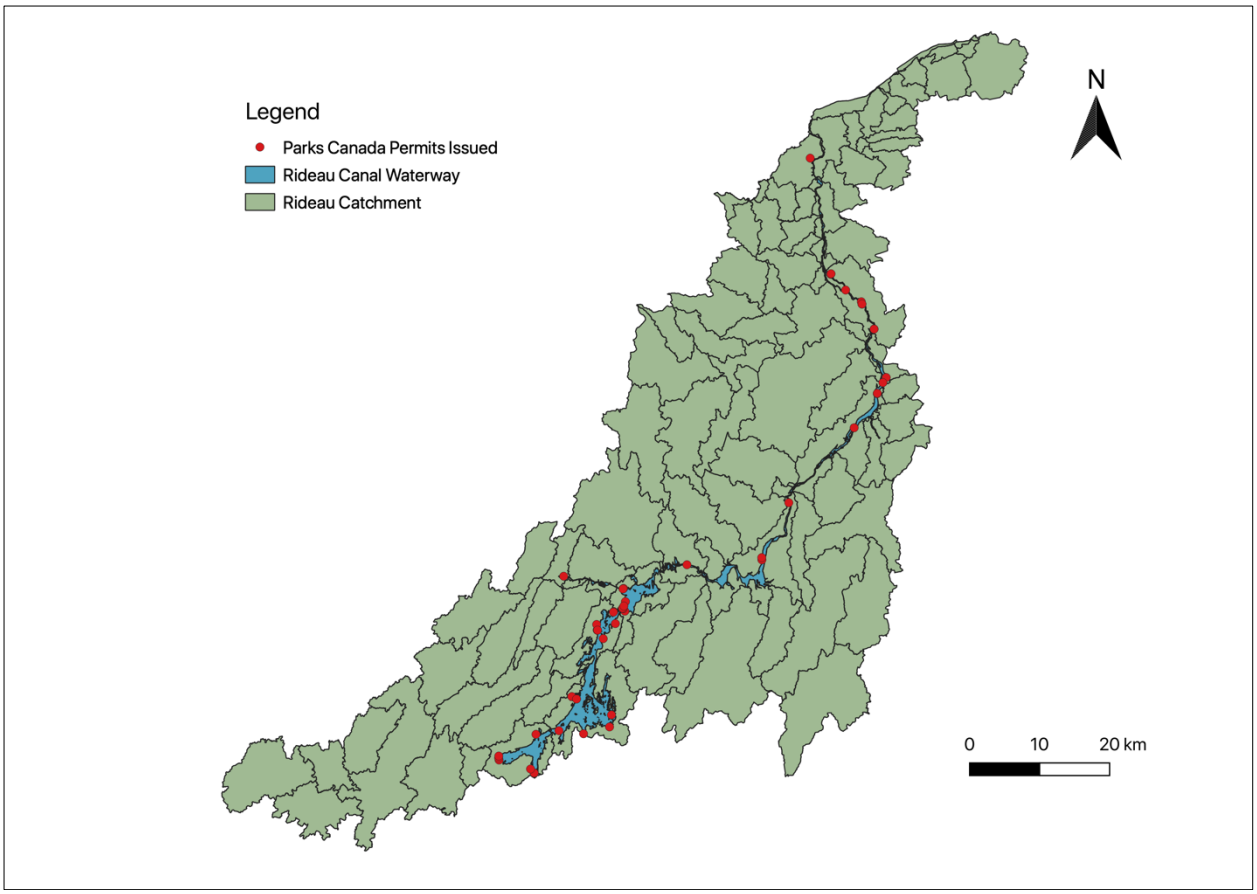

Figure E- 2: Map of macrophyte removal permits issued to shoreline users by Parks Canada in the Rideau watershed from 2013-2020. The Rideau Watershed is divided into subwatersheds outlined in black. Map created in QGIS 3.4 using geospatial data acquired from Parks Canada and the RVCA.

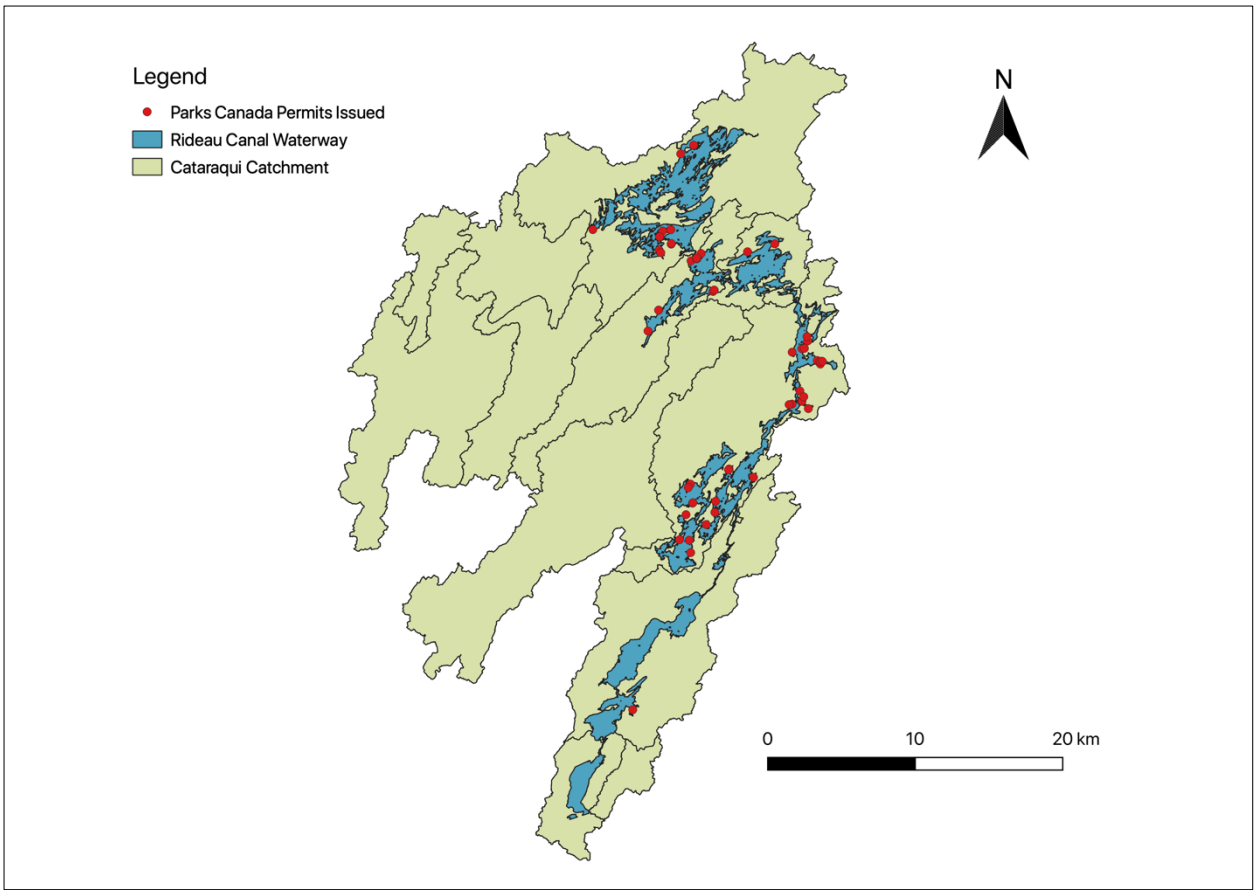

Figure E- 3: Map of macrophyte removal permits issued to shoreline users by Parks Canada in the Cataraqui watershed from 2013-2020. The Cataraqui Watershed is divided into subwatersheds outlined in black. Map created in QGIS 3.4 using geospatial data acquired from Parks Canada and the RVCA.

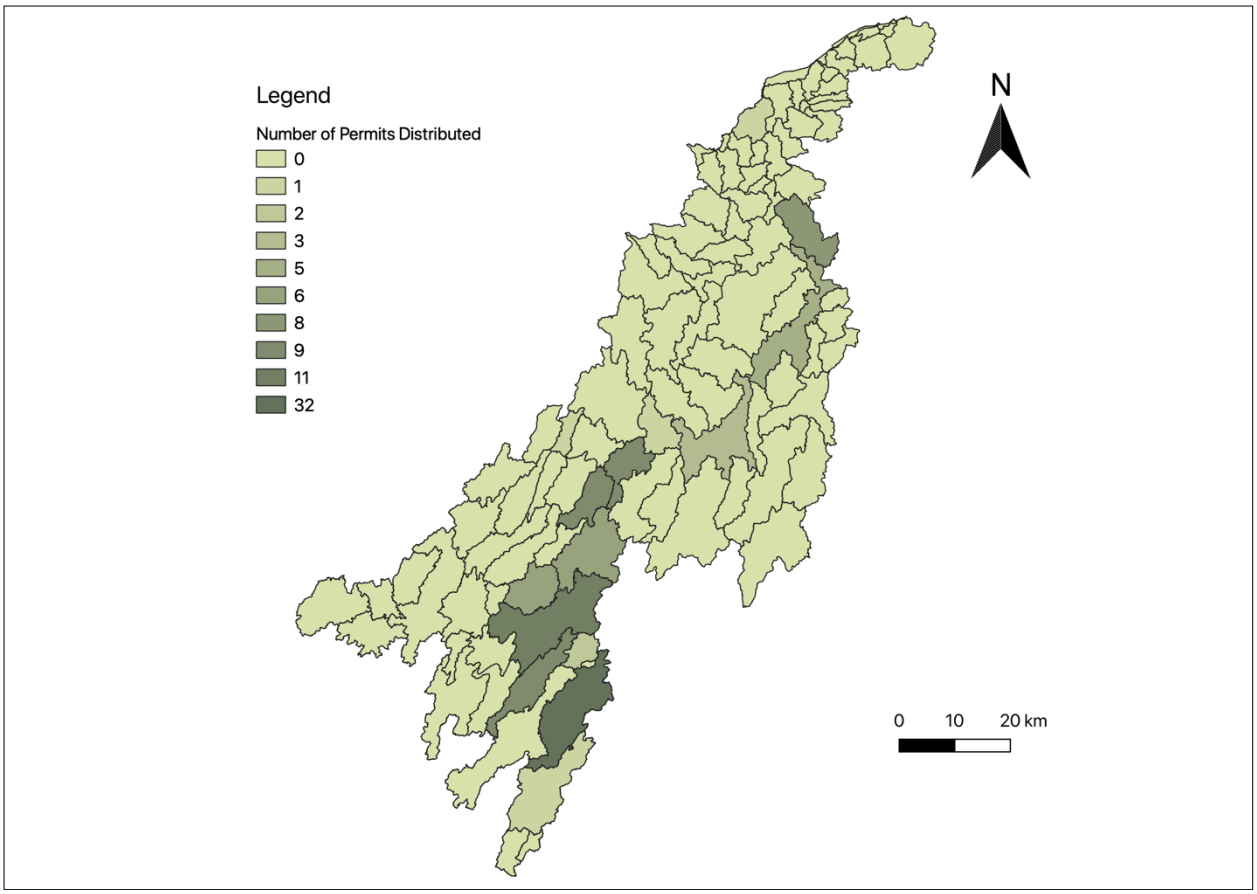

Figure E- 4: Map demonstrating the density of the number of permits distributed by Parks Canada from 2013-2020 in the Rideau Canal Waterway, segmented by catchments. The figure demonstrates that there is a higher concentration of permits distributed in the lower portion of the Rideau Waterway, specifically in the Cataraqui region. Map, count points in polygon of catchments in the Rideau and Cataraqui catchments. Map created in QGIS 3.4 using geospatial data acquired from Parks Canada and the RVCA.

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