A PRELIMINARY ASSESSMENT OF THE INFLUENCE OF ADJACENT LAND-USE ON GROWTH-FORM COMPOSITION OF VEGETATION IN SMALL NEWFOUNDLAND BASIN BOGS

by © Jean Elizabeth Granger

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Abstract

Anthropogenic land-use in wetland-adjacent landscapes has been demonstrated to alter wetland ecology in several ways, including changes to vegetation composition. However, comparatively little of this kind of research has been dedicated to bog wetlands specifically. The purpose of this study was to examine if vegetation growth-form composition in small basin bogs is influenced by different types of adjacent land-use throughout the St. John's region of Newfoundland, Canada. The results provide evidence that overall vegetation composition in small basin bogs are different depending on adjacent land-use (pasture, urban, or natural) and graminoid growth-form vegetation specifically decreases in bogs next to pasture land-use. Additional studies of a similar nature, particularly those implementing remote sensing methods, may provide further evidence to strengthen this relationship in the future.

Acknowledgments

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I would also like to thank C-Core who provided me with office space, and those who gave me the opportunity to partake in the mapping and inventory of wetlands in Newfoundland, from which some of the data for this study was obtained. The inventory project was undertaken with the financial support of the Government of Canada through the federal Department of Environment and Climate Change, and the Research & Development Corporation of Newfoundland and Labrador. This wetland inventory was completed with contributions from Ducks Unlimited Canada, Government of Newfoundland and Labrador Department of Environment and Conservation, and Nature Conservancy Canada. I would like to thank these organizations for the generous financial supports and providing such valuable datasets.

Table of Contents

List of Tables

List of Figures

List of Abbreviations

ANOSIM	Analysis of Similarities
ANOVA	One-Way Analysis of Variance
CI	Annual Crop Inventory 2016
CNWWG	Canadian National Wetlands Working Group
CWCS	Canadian Wetland Classification System
EPA	Environmental Protection Agency
EPA-WA	Environmental Protection Agency Wetland Conditional Assessment
IBI	Indices of Biotic Integrity
LDI	Land Development Intensity Index
MBE	Maritime Barrens Ecoregion
NDVI	Normalized Difference Vegetation Index
NIR	Near-Infrared
NL	Newfoundland and Labrador
NLDFL	Newfoundland and Labrador Department of Fisheries and Land Resources
NLDME	Newfoundland and Labrador Department of Municipal Affairs and Environment
NMDS	Non-Metric Multidimensional Scaling
PDW	Policy for Development in Wetlands
PERMANOVA	Permutational Multivariate Analysis of Variance
PFT	Plant Functional Types
PSR	Pressure-State-Response
RE	Red-Edge
SA	Atlantic Subarctic Wetland Region
SJMA	Pressure-State-Response
USA	United States of America
VI	Vegetation Indices

CHAPTER 1: INTRODUCTION

1.1 Context

Wetlands can be defined as "land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment" (National Wetlands Working Group 1997, p.1). Globally, wetlands cover between five to ten percent of the lands surface and provide benefits to all forms of life (Kingsford et al. 2016). Such benefits include habitat for many species, filtration for improved water quality, and nutrient cycling (Hanson et al. 2008; Mahdavi et al. 2017a). The provision of such services has resulted in wetland monikers such as "nature's kidney's" and "biological supermarkets". These names are in dramatic contrast to a historical context in which wetlands were extensively damaged and destroyed due to human-related developments, resulting in an estimated 54% to 70% of the world's wetlands being lost or impaired since at least the early 1900s (Davidson 2014; Kingsford et al. 2016).

In more recent times the desire to protect and mitigate wetland ecosystems has grown out of a greater understanding of the services these ecosystems provide, the historic and current loss and damage of wetlands around the world, and projections of a growing human population and a changing climate (Hanson et al. 2008; Erwin et al. 2009). The growing concern for wetlands has, at least since 1970s (Erwin et al. 2009), resulted in numerous organizations and groups (government, academia, conservation groups, etc.) acquiring and applying knowledge of wetlands in the form of policies, management, tools and campaigns to improve and protect global, regional, and/or local wetland resources and services (Environment of Canada 1991; Hanson et al. 2008; Ramsar Convention Secretariat 2010).

The services wetlands provide are made possible via the natural functions these ecosystems carry out as a result of complex, interacting relationships amongst climate, geomorphology, biogeochemistry, hydrology, and flora and fauna (Mitsch and Gosselink 2000). These relationships can easily be disturbed and degraded through the influence of natural regimes like fire (Foster and Glaser 1986; Norton and De Lange 2003; Feldmeyer-Christie et al. 2009) and anthropogenic activity (Brinson and Rheinhardt 1996; Reiss 2006; McLaughlin and Cohen 2013). As a result, much research has been dedicated to understanding the multitude of ways in which these disturbances can impact and impair wetland ecology, wetland function and the provision of wetland services. Such research is often incorporated into methods for assessing current wetland condition, detecting early signs of decreasing condition, and to developing, improving and assessing the effectiveness of management and policy (Mita et al. 2007; Moges et al. 2016; Stuber et al. 2016).

There are countless ways in which anthropogenic activity may influence wetland ecology. The problem of understanding the numerous and complex influences is only amplified in the face of limited budgets and time that organizations can contribute. As such, many governments develop conservation policies that aim only to directly prevent wetland loss (Houlahan et al. 2006; Rubec and Hanson 2009), with little to no constraints placed on wetland-adjacent lands, despite the evidence that suggests that wetland ecology can be impacted not only by direct development but by anthropogenic activity outside of wetlands borders in the adjacent landscapes (Houlahan and Findlay 2004; Fernandes et al. 2011; Patenaude et al. 2015). In more recent times, some organizations and governments have begun to incorporate landscape-level information into policies and management tools (Government of Alberta 2013; Ontario Ministry of Natural Resources and Forestry 2017; Tiner 2018). For example, in the United States of America (USA) the Environmental Protection Agency Wetland Conditional Assessment (EPA-WA) which is used to guide state policy development, includes landscape-scale information to assess wetland condition (Miller et al. 2006; U.S. EPA 2011; Nestlerode et al. 2014).

Landscape-scale anthropogenic activity has been shown to impact wetlands in a variety of direct, secondary, and cumulative ways (Brown and Vivas 2005). Many wetlands (marsh, swamp, etc.) are associated with hydrological systems and therefore may be influenced by direct pollutant inputs to water systems (Haidary et al. 2013). For example, catchments with greater amounts of agricultural development results in greater phosphorus and nitrogen levels in associated streams, which in turn feed into associated wetlands (Amiri et al. 2012). Greater quantities of impervious cover in catchments and adjacent landscape have been associated with decreased habitat quality and alterations to ground-water levels (Clapcott et al. 2011; Haidary et al. 2013; Miller et al. 2016). Additionally, roads can re-direct water flow and produce complex pollutants that can influence water quality, vegetative growth, and faunal biodiversity (Houlahan et al. 2006; Bignal et al. 2007; Margriter et al. 2014; Sengbusch 2015).

It is important that landscape-scale information applied in the context of wetland policy and management be ecologically meaningful, i.e., the landscape-scale information is known to significantly impact some aspect of wetland ecology (Brown and Vivas 2005; Haidary et al. 2013; Margriter et al. 2014). Though landscape can and does impact wetland ecosystems, the extent is not always known (Rooney and Bayley 2011) and may differ depending on the region, the type of wetland, the wetland feature, and the aspect of landscape being examined (Struber et al. 2016). If meaningful associations between wetlands and landscapes are not established and considered, policies and methods protecting wetlands may be ineffective or misleading (Rooney and Bayley 2011). As such, relationships between different aspects of wetland and landscape have been studied and developed in different parts of the world and for different types of wetlands (Hychka et al. 2007; Margriter et al. 2014; Guidugli-Cook et al. 2017).

1.2 Problem

Most of the research studying the relationships between adjacent landscapes and wetlands have been conducted within the USA, for use within specific states, and for wetlands defined through USA wetland classification systems (Cowardin et al. 1979; Brinson 1993; Smith et al. 1995). There are comparatively fewer of this type of research in Canada, specific to individual provinces and specific to wetlands defined by the Canadian Wetland Classification System (CWCS; National Wetlands Working Group 1997). See Table 1.1 for a list of the classes defined by the CWCS. This should be taken into consideration given that most provincial-based wetland policies do not place restrictions on anthropogenic development adjacent to wetlands (Houlahan et al. 2006; Newfoundland and Labrador Department of Municipal Affairs and Environment [NLDME] 2018). Of additional consideration is that most of the USA-based landscape assessment methods were developed for non-peat forming wetlands (Chipps et al. 2006; Hychka et al. 2007; Martinez-Lopez et al. 2014; Miller et al. 2016). In Canada, peat-forming wetlands include bogs, fens, and swamps (National Wetlands Working Group 1997). Peat, or soils comprised of poorly decomposed plant material, builds extensively in peatlands as a result of the limited ability of microbes to decompose litter in conditions of saturated waters, low oxygen, and higher acidity produced by peatland mosses (Moore 1989; Warner and Asada 2006). Non-peat forming wetlands include some swamps, marsh, and shallow water (National Wetlands Working Group 1997). Canada has extensive amounts of peatland that have great potential to be under pressure from anthropogenic land-use, both directly and indirectly. Though studied less than their freshwater and coastal wetland cousins, some evidence has shown that adjacent landscape or the products of adjacent landscape, such as increased nitrogen deposition and water table alteration, can influence bog wetland ecology in a variety of ways that have implications on the provision of wetland services (Bignal et al. 2007; Pellerin et al. 2008; Pasquet et al. 2015; Sengbusch 2015).

Table 1.1. Wetland Classes as defined by the Canadian Wetland Classification System (National Wetlands Working Group 1997).

Wetland Class	Class Description
Bog	Ombrotrophic peatland dominated by Sphagnum moss species.
Fen	Minerotrophic peatland dominated by graminoid species and brown mosses.
Marsh	Minerotrophic wetland with periodic standing water or slow-moving water, dominated by graminoids, shrubs, forbs, and emergent plants.
Shallow Water	Minerotrophic wetland where water is up to 2m deep for most of the year, and where there are less than 25% of emergent plants or woody plants.
Swamp	Peatland or mineral wetland dominated by woody vegetation.

An opportunity for Canadian-based research on the impact of adjacent landscape on bog wetlands exists in the province of Newfoundland and Labrador (NL) where research to inventory and map NL wetlands has produced the first up-to-date maps of wetlands in the province (Amani et al. 2017; C-Core 2017; Mahdavi et al. 2017b). This project has produced maps for multiple locations across NL, including the St. John's Metropolitan area (SJMA) region where bog wetlands are the most extensive of all wetland classes (C-Core 2017). Table 1.2 presents information on the distribution of wetlands in the SJMA and Figure 1.1 shows some examples of the classes of wetlands found in NL. All NL wetlands are managed at the provincial government level by the Policy for Development in Wetlands (PDW) which, while restricting development within wetlands, does not specify restrictions on lands adjacent to wetlands despite the extensive evidence that showing such development can and does influence wetlands and the quality of services said wetlands provide. Additionally, this policy notes that, given the extensive nature of wetlands, with peatlands specifically referenced, there is room for development to meet needs if impacts are minimized (NLDME 2018).

Given that the SJMA has already experienced extensive loss of wetlands and wetland functions (Slaney 2006; Ren 2014), the population of the SJMA is projected to increase over the next 15 years (City of St. John's 2014; Simms and Ward 2017), agriculture land-use in the area is expected to increase (Newfoundland and Labrador Department of Fisheries and Land Resources [NLDFL] 2019), and given there is the potential in NL (based on the outlines of the PDW) for development to continue adjacent to wetlands, there is a greater need to understand the various ways in which adjacent anthropogenic activity can impact these ecosystems. Such impacts may in turn have implications of the quality of wetland functions and provision of services in NL. By gaining a better understanding of the ways in which landscape may alter bog wetlands in NL, there is a potential to justify the implementation of landscape restrictions in governmental policy, and a potential for such knowledge to be incorporated into a future wetland condition assessment methodology. Additionally, this research may contribute to the limited existing research on how adjacent landscapes can impact aspects of bog wetland ecology and in turn, bog services.

Table 1.2. Wetland coverage of the St. John's Metropolitan Area (C-Core 2017).

Wetland Class	% of Total Wetland Area	% of Total Study Area
Bog	36	16.00
Fen	21	9.15
Swamp	32	14.38
Marsh	3	1.52
Shallow Water	8	3.76

Figure 1.1 Wetland classes in Newfoundland, Canada: (a) bog, (b) fen, (c) swamp, (d) marsh, and (e) shallow water.

1.3 Research Objectives

The goal of this thesis is to examine and potentially establish a relationship between bog-adjacent anthropogenic landscapes and bog vegetative growth-form via a preliminary study of Newfoundland bogs. This research builds on evidence provided by previous studies of non-bog wetlands that demonstrate the impacts of adjacent landscape on wetland vegetation, and in turn the impacts on the ability of those wetlands to provide ecosystem services. The results of this research may establish a basis for the continued study of bog vegetation and the impacts of adjacent anthropogenic land-use in a place such as NL, where policies do not consider or incorporate anthropogenic impacts outside of a wetland's boundary. Additionally, by measuring vegetation at the scale of growth-forms, it is possible in this study to test the potential for remotely sensed vegetation indices (VI) to measure changes in bog vegetation growth-forms, with the potential of being used in lieu of fieldwork. The specific objectives of this thesis are to:

- Determine if anthropogenic activity in bog-adjacent landscapes is associated with bog vegetation composition at the scale of growth-form.
- Determine the scale at which anthropogenic activity in land-use
- Determine if remotely sensed vegetation indices (VI) can be used to measure differences in bog vegetation composition.

1.4 Questions

The overarching question this research asks is “Does anthropogenic activity (presence and type) in bog-adjacent landscapes influence bog vegetation composition measured at the growth-form level?” Specific questions addressed are:

- Does overall vegetative growth-form composition in bogs adjacent to natural, urban, and pasture landscape differ significantly?
- Does the measured cover of individual growth-forms in bogs adjacent to natural, urban, and pasture landscapes differ?
- How does the quantity of anthropogenic land-use and the scale at which land-use is quantified influence the strength of associations between growth-form composition and adjacent land-use in bog wetlands?
- Do remotely sensed VI have the potential to quantify changes in bog growth-form composition as a result of environmental change due to adjacent anthropogenic land-use in the same manner as field data?

1.5 Conceptual Framework

This research was conducted within the context of The Pressure-State-Response (PSR) feedback framework, which is built upon the understanding that anthropogenic activity, both direct and indirect, can exert pressures on ecosystems, both locally and globally (Organization for Economic Co-operation and Development 1993). Those pressures that cause a change in the state of an ecosystem, both in terms of the environment and the quality of ecosystem services, can result in response in the form of better policy, management, or legislation protecting said ecosystem from said pressures (Organization for Economic Co-operation and Development 1993). The PSR has often been applied in the context of wetland ecosystems (Evans et al. 2014; Chen 2016; Sun et al. 2016; see Figure 1.2) and contextualizes (Watzin et al. 2005) the development and application of well-known wetland conditional assessment methodologies (U.S. EPA 2002a; 2002b).

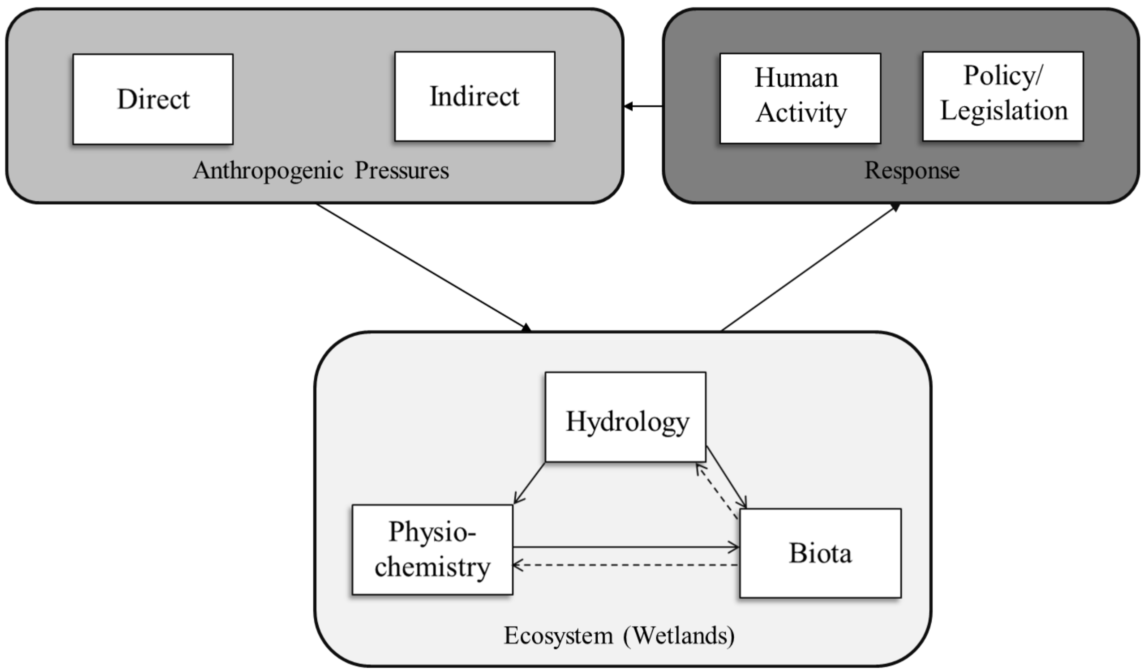

Figure 1.2. Pressure-State-Response framework in the context of wetland ecosystems (adapted from Organization for Economic Co-operation and Development 1993 and Mitsch and Gosselink 2000).

In the context of wetlands, direct anthropogenic pressures can include land-use development directly within wetlands (Laroche et al. 2012), farming of a wetland's resources (Evans et al. 2014), and trampling (tourist or All-Terrain Vehicle-related; Laroche et al. 2012; Tsai et al. 2012). Indirectly, wetlands can be impacted by adjacent land-use (i.e., pollutant discharge, pollutant deposition, and water table alteration via impenetrable surfaces or run-off) (Houlahan et al. 2006; Evans et al. 2014; Stuber et al. 2016), or by global anthropogenic activity (i.e., climate change) (Erwin et al. 2009; Evans et al. 2014; Churchill et al. 2015; Robroek et al. 2015). These various pressures can then alter the state of a wetland by influencing one or more of the interacting facets of wetland ecosystems that contribute to the state, including hydrology, physio-chemistry, and floral and faunal components (Mitsch and Gosselink 2000). Because beneficial wetland functions and services are a result of the total interacting wetland state, changes due anthropogenic pressures can change the quality or ability of a wetland to produce such functions and services (Brown and Vivas 2005; McLaughlin and Cohen 2013; Evans et al. 2014). This may in turn influence anthropogenic changes in the form of changing behaviour, policy, or legislation to protect or preserve said wetland ecosystems.

It is challenging to understand the full extent of pressures, states, and necessary responses to ensure the protection of wetland ecosystems and the services they provide (Sun et al. 2016). As such, various wetland-based assessments may focus on one, or multiple portions of the PSR framework (Sun et al. 2016), establishing relationships between a specific pressure and a specific facet of wetland-state (Laroche et al. 2012; Stuber et al. 2016), or a specific facet of wetland-state and the provision of services (Evans et al. 2014). In the context of this study, there is a focus on establishing the existence, or lack, of a relationship between a specific type of pressure (adjacent landscape) and a specific facet of wetland state (bog growth-form composition). This derivation of the PSR model relevant to this study can be seen in Figure 1.3. Relationships between adjacent-landscape pressures and wetland vegetation have been extensively established through research and implemented in policy and management around the globe (Fennessy et al. 2007; Mita et al. 2007; Moges et al. 2016).

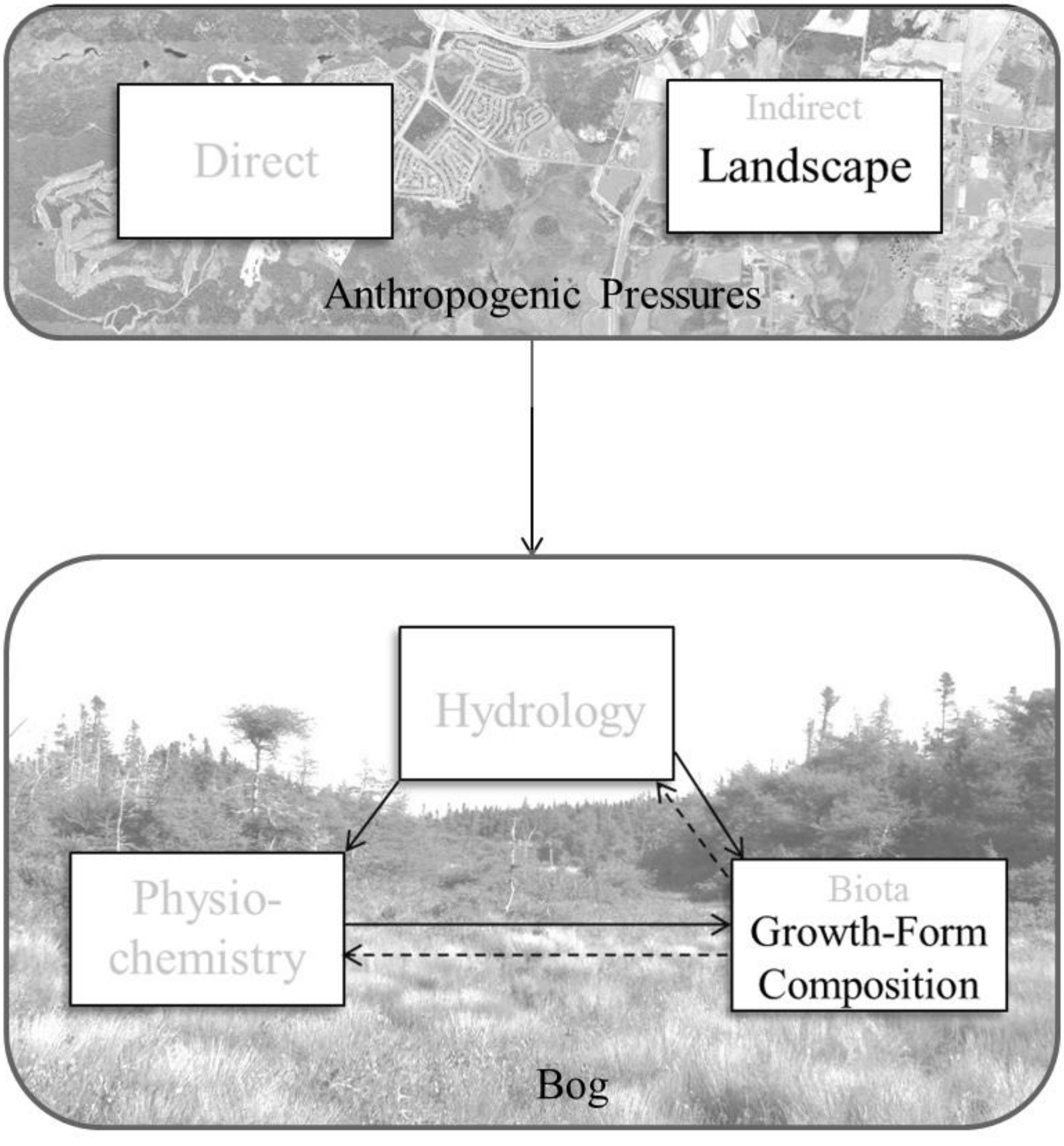

Figure 1.3. The influence of adjacent landscape on bog growth-form composition in the context of the Pressure-State-Response framework (adapted from Organization for Economic Co-operation and Development 1993 and Mitsch and Gosselink 2000).

1.6 General Study Design

The study design was based on other studies of wetland-landscape relationships. These studies all shared similar methods which required selecting the type(s) of wetlands of interest, appropriate scale, landscape units to measure, specific wetland sites, measurable aspects of wetlands ecology, and statistical tests for examining the relationships between measures of landscape and measures of wetland ecology. The specifics of these choices typically vary based on purpose, location, and wetland ecology and type. Table 1.3 presents a list of various wetland-landscape studies and associated methods.

Table 1.3. A brief summary of the methods used by a selection of studies examining the impacts on landscape on wetland ecology.

Study	Scale	Landscape	Wetland Measures	Statistics	Wetland Type
Hale et al. 2004	Watershed	%Barren %Forest %Pasture %Urban %Wetlands Etc	Benthic Biomass, fish species, and various other benthic measurements	NMDS, ANOSIM, Logistic Regression	Estuaries
Haidary et al. 2013	Catchment	% Agriculture % Forest % Grassland % Urban	Various water quality measurements	Spearman Rank Correlation, Kruskal-Wallis Test	River and stream-associated wetlands
Miller et al. 2016	500m and 1km	%Agriculture %Barren %Developed %Pasture Etc.	Various classes of measurements including vegetation: % cover woody, % cover *Carex*, % cover graminoid etc.	T-Tests, Random Forest Regression	Freshwater wetlands
Moges et al. 2016	Catchment	Urban Agricultural Forest	Various vegetation measurements including: % cover graminoid, % cover perennials, % cover shrub etc.	Spearman Rank Correlation, Mann-Whitney U	Hydrologically-connected wetlands
Patenaude et al. 2015	A range between 0.4km to 2.4km	% Wetland % Impervious Cover	Macro invertebrate and Vegetation measurements including: % cover of vegetation, and Floristic Quality Index	GLM, NMDS	Shallow open water wetlands
Stuber et al. 2016	100m	%Developed % Herbaceous Pasture %Pine Forest %Road %Row Crop Etc.	Macrophyte measurements including: % cover species, % cover herbaceous, % cover obligate wetland	NMDS, MRPP	Geographically Isolated Wetlands

The studies listed in Table 1.3 choose to focus their research on a specific type of wetland verses wetlands in general. Often, studies such as these distinguish wetland types by following a wetland classification system. In the U.S., most wetlands are defined following the Cowardian standard (Cowardin et al. 1979). In Canada, wetlands are often defined following the CWCS (National Wetlands Working Group 1997). The CWCS defines five major classes as seen in Table 1.1 and further separate these classes on the basis form. The accuracy and strength of the relationship between adjacent landscape and wetland ecology may depend on the classification method of choice and the narrowness of which the wetlands of interest have been defined.

Wetland-landscape studies often apply measures of vegetation to assess a wetland's response to activity in adjacent landscapes. Vegetation provides several advantages over other measurable aspects of wetland ecology. Vegetation has been shown to be sensitive to various types of environmental change including anthropogenic activity (U.S. EPA. 2002a; Miller et al. 2006), pollution and changes in hydrology (Adamus and Brandt 1990; Nevel et al. 2005; Wilson and Bailey 2012). Vegetation can also be grouped and classified in several biologically meaningful ways (species, genus, functional group) that allow for easy sampling and a simplified method for predicting and modeling ecosystem response to environmental change (Chapin et al. 1996; Robroek et al. 2015). Additionally, vegetation is present in all wetlands, is (generally) immobile, and is easily sampled (Bedford 1996; U.S. EPA. 2002a; Miller et al. 2006). As a result, vegetation is often a preferred choice for describing wetland ecology.

Many of the studies described in Table 1.3 apply the concept of "reference wetland" (Brinson and Rheinhardt 1996). Generally, reference wetlands are under no anthropogenic influence and are considered "pristine" representations of wetlands (Jones et al. 2015). Pristine wetlands act as a type of experimental control against which wetlands under the influence of anthropogenic influence are compared (Jones et al. 2015). However, these reference wetlands are rarely "pristine" given the state of climate change and globally-cycled pollutants. As a result, reference wetlands, are the next best alternative (Faber-Langendoen et al. 2012). In studies of wetlands and the landscape, reference wetlands often contain little to no anthropogenic activity within their landscapes (Miller et al. 2016; Stuber et al. 2016).

This research incorporates aspects of the methods described in Table 1.3 as well as the concept of reference wetlands, with some modifications made specifically for bog vegetative growth-forms, for bog wetlands in general, and for the study area. Additionally, this study chose to focus on a specific bog form (basin bogs) verses bogs in general, as basin bogs are a common bog form in the study area and bogs of different forms have different structure, vegetation composition and relationship to the surrounding run-off (Wells 1981; National Wetlands Working Group 1997) The methods section in Chapter 3 provides more detail.

1.7 Study Area

The study area is located on the Avalon Peninsula, in the eastern part of NL centered at 47°31'54.68" N, 52°47'27.19"W (Figure 1.4). Making up the northeastern part of the

Avalon Peninsula, the study area includes the most developed and populous area of the province, the St. John's Metropolitan Area (SJMA; Statistics Canada 2011), with a population of 209,955 (Statistics Canada 2017). The SJMA has roughly 255.9 people per square kilometer (Statistics Canada 2017), exceeding the average provincial population density of 3.9 (Statistics Canada 2016).

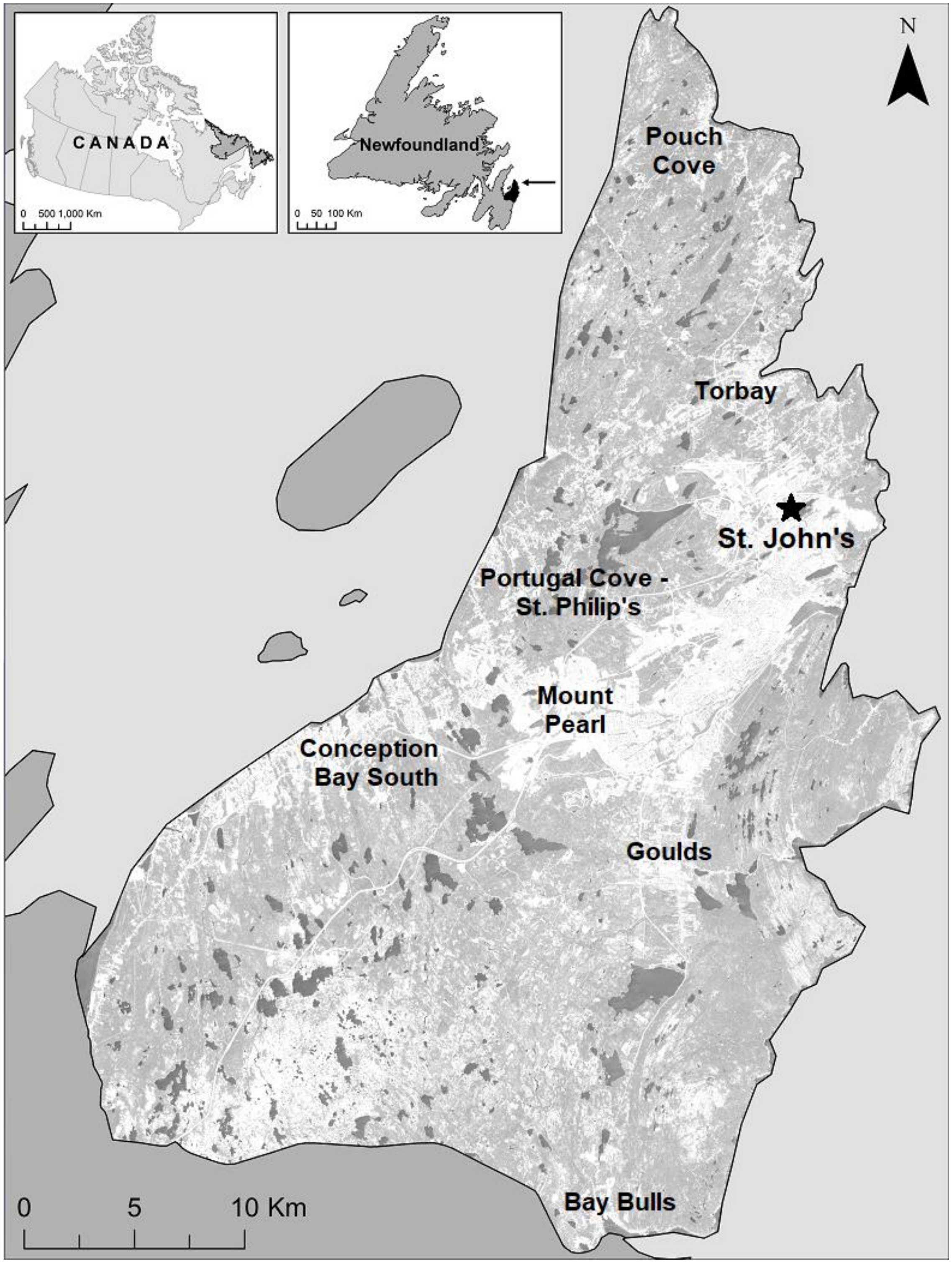

Figure 1.4. Study Area located in the St. John's Metropolitan Area of Newfoundland and Labrador.

The study area is located within the Boreal Forest Biome and more specifically within the Maritime Barrens Ecoregion (MBE; Ecological Stratification Working Group 1996) and the Atlantic Subarctic Wetland Region (SA; National Wetlands Working Group 1986), as seen in Figure 1.5 and Figure 1.6 respectively. The MBE is a region defined generally based on similar vegetation and soil characteristics (South 1983), and the SA is a region defined generally based on similar wetland development and distribution (Zoltai 1979). The MBE is characterized by an oceanic climate of cool and foggy summers, mild and wet winters, and annual precipitation reaching over 1250 mm (South 1983). Natural land-cover within the MBE is dominated by balsam fir stands and extensive low nutrient landscapes such as heath- and peatlands (South 1983). Within the SJMA specifically, natural landscapes are largely replaced by anthropogenic land-use including the densest urban land-use in the province, and small amounts of farm and pasture land due to the lack of extensive arable land in the province (NLDFL 2017).

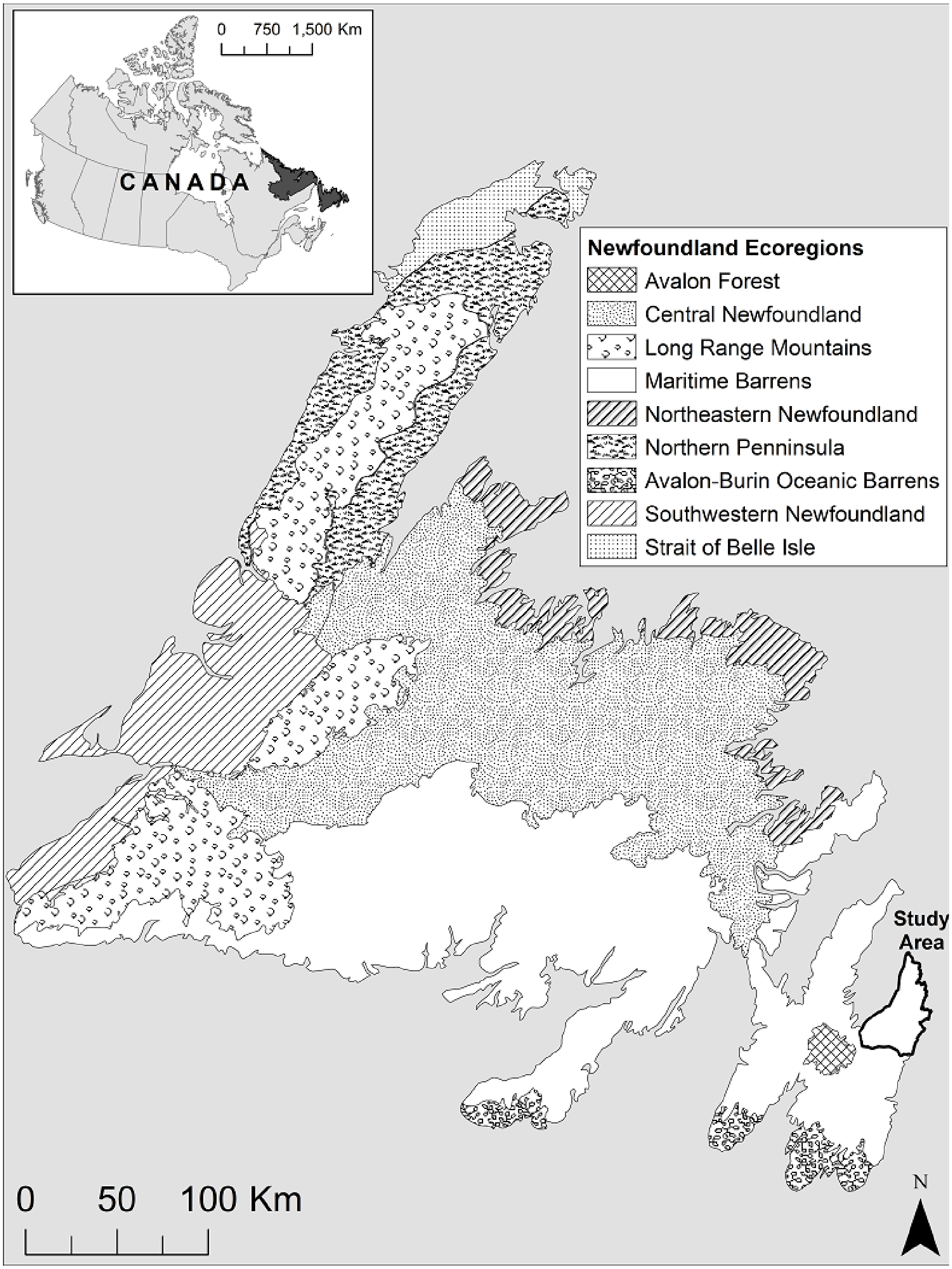

Figure 1.5. Newfoundland Ecoregions (adapted from Ecological Stratification Working Group 1996).

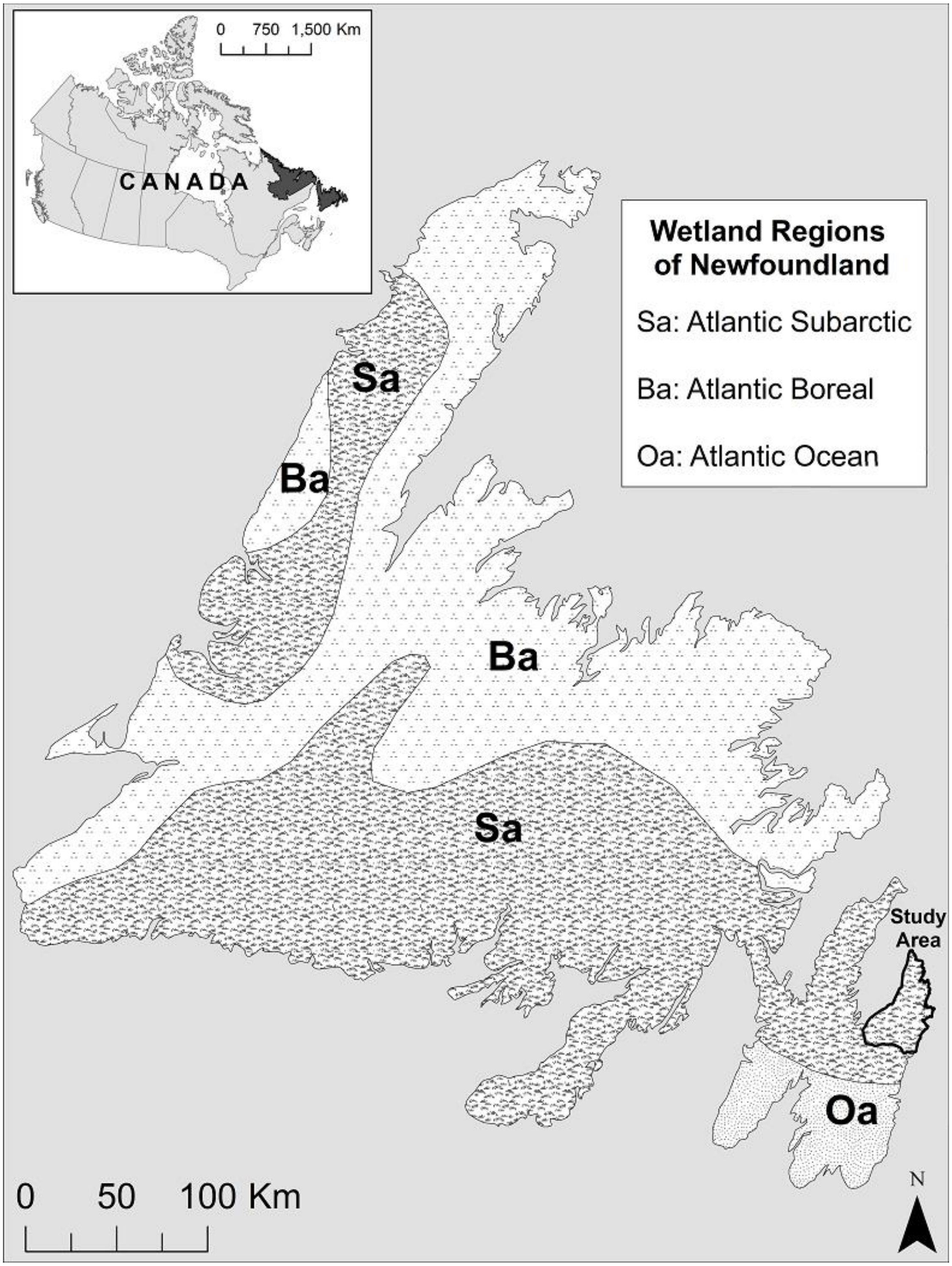

Figure 1.6. Newfoundland Wetland Regions (adapted from National Wetlands Working Group 1986).

As a wetland region, the area is dominated by peat-forming wetlands of basin bogs, slope bogs, domed bogs, blanket bogs and fens (Wells 1981). The dominance of such peatlands is largely the result of the wet climate, poorly drained soils, and contributions of acidic seepage waters from siltstone, slates and volcanic rock (Pollett 1967; Zoltai 1979; Wells 1981), all common requirements for peat formation (discussed in Chapter 2). Non-peat forming wetlands, such as marsh, are comparatively rare (Pollett 1967). In the recent past, these peatlands have been mostly used for agricultural and forestry research, community pastures, and peat moss harvesting (Wells 1981). Wetlands within the study area are largely managed by the provincial government through the PDW, whereby development within wetlands in approved or denied based on consideration of impacts on water quality, hydrological functions, and terrestrial and aquatic habitats (NLDME 2018). A recent wetland inventory of the SJMA reports bog and fen as being the most dominant wetland classes and marsh as the least common (see Table 1.2; C-Core 2017), which corresponds to previous descriptions of wetlands in the area (Pollet 1967; Wells 1981).

1.8 Thesis Organization

This thesis is laid out in four chapters. Chapter 1 acts as an introduction to both the context and purpose of this research with specific discussion of adjacent landscape-wetland relationships and bog wetlands in NL. Chapter 2 consists of an extensive literature review that begins with discussion of wetlands in general and ends with specific discussion of bog wetlands and the ways that bog wetland growth-forms may be influenced by adjacent landscape and landscape-associated influences. Chapter 3 is presented in manuscript format and discusses the methods and the results of a preliminary analysis of the influence of adjacent landscape on bog growth-form composition. Finally, Chapter 4 provides the conclusion of the work of this thesis, discussing the results and implications, and providing recommendations for future research.

1.9 Co-authorship Statement

The research questions and the study design were conceived by myself, Dr. Carissa Brown, and Dr. Bahram Salehi. I conducted the field work and analysis in consultation with Dr. Carissa Brown.

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CHAPTER 2: LITERATURE REVIEW

The following chapter is a general review of bog wetlands and how anthropogenic activities in bog-adjacent landscapes has the potential influence bog ecology and specifically influence bog vegetation composition.

2.1 Bog Wetlands

Bogs a class of wetlands characterized by a substantial accumulation of peat and oligotrophic (low nutrient content) conditions (National Wetlands Working Group 1997). Peat is soil-like material that contains at least 30% organic matter (partially decomposed plants) which accumulates as the result of a water-saturated, low oxygen environment (Moore 1989; Joosten and Clarke 2002; International Peat Society 2019). Such environments occur in poorly drained landscapes and in regions with a perhumid climate, where the average rate of precipitation is greater than that of the average evaporation and where temperatures allow for plant growth but minimal decomposing activity by microorganisms (Moore 1989; Joosten and Clarke 2002). Under such conditions plants grow, die, become litter, and because of the low oxygen in the saturated soils, decomposition by microbes proceeds at reduced rates (Joosten and Clarke 2002). Although peat builds in bogs, fens, and swamps, bogs differ in that peat has accumulated to such a level that the bog surface becomes mostly separated from all sources of water (ground and surface water) except for that of atmospheric inputs (National Wetlands Working Group 1997). Bog wetlands are often referred to as ombrotrophic as they are low in dissolved minerals as a result of a lack of surface water input (National Wetlands Working Group 1997).

2.1.1 Bog Vegetation

An important aspect of bog development and functioning is vegetation composition (Dorrepaal et al. 2005; Ward et al. 2013; Kuiper et al. 2014). Because water input (e.g., precipitation) in bogs are low in nutrients, bogs form an environment for species with relatively high stress (e.g., low nutrient) ecological niches (Dyderski et al. 2016). Generally, these are species that can obtain their nutrients from non-water sources (e.g., carnivorous plants), and species that can form symbiotic relationships with nutrient-fixers (Dorrepaal 2007; Mitsch et al. 2009).

In many studies of bog and peatland vegetation classification of vegetation species has been based on, or derived, from that defined by Chapin et al. (1996) in a study of growth-forms in arctic tundra ecosystems. Like peatlands, arctic tundra ecosystems are low in nutrients and species diversity, and thus, the two share many similarities in terms of growth-form presence and distribution (Chapin et al. 1996). Chapin et al. (1996) describe a branching classification of tundra growth-forms (Figure 2.1). The classification was supported through a cluster analysis of 37 tundra species via numerous traits such as: response to climate (e.g., northern latitudinal limit), response to disturbance (e.g., seed dormancy, dispersal distance), resource acquisition (e.g., plant height, nitrogen fixation), and nutrient use and competitive balance (e.g., leaf mass, woodiness, leaf width) (Chapin et al. 1996). This approach has been applied in a multitude of studies of bog vegetation and bog response to anthropogenic influence (e.g., Dorrepaal 2007; Ward et al. 2013; Robroek et al. 2015; Wu and Blodau 2015).

Studies of growth-forms, sometimes referred to as plant functional types (PFT), are of interest because such groupings are known to: (1) easily be associated with ecosystem functioning, and (2) respond to environmental changes (both at small- and landscape-scales; Sieben and le Roux 2012). In peatlands growth-forms have been found to be associated with formation and nutrient cycling (Robroek et al. 2016), and to change in response to alterations in environmental conditions (Bubier et al. 2003; Buttler et al. 2015). Recent benefit of the analysis of bogs at the growth-form level is the ability of remote sensing imagery can capture detail at the scale of growth-form (Harris et al. 2015; Guo et al. 2017). The terms "growth-form" and "plant functional type" (PFT) are used interchangeably by Chapin et al. 1996 and other studies of peatland vegetation composition. Generally, PFT is used as a collective term (Sieben and le Roux 2012) referring to the numerous ways in which species can be grouped, not only including growth-form but other physical or chemical traits, such as leaf longevity, shoot length, and biomass (Dronova et al. 2012; Sieben and le Roux 2012). Here, the term growth-form is used rather.

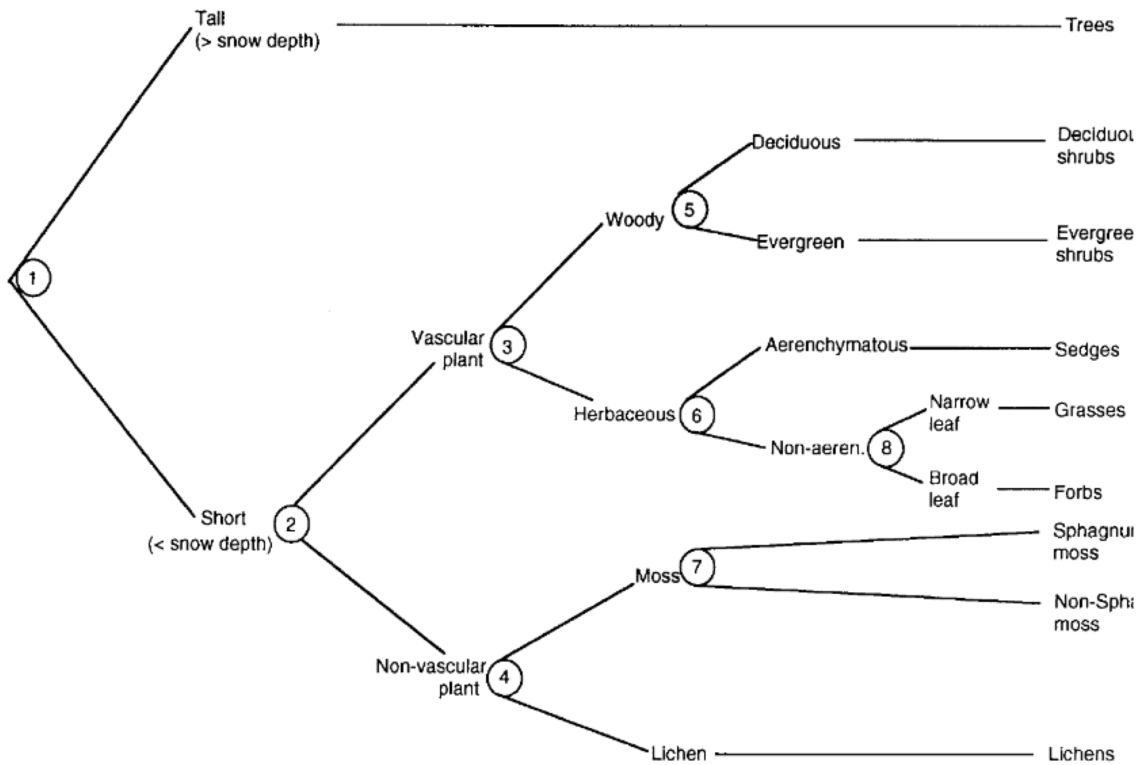

Figure 2.1. Growth-Form classification of tundra vegetation frequently applied to bog wetlands (Chapin et al. 1996).

2.1.2. Bog Functions and Services

Like other wetlands, bogs carry out natural functions that in turn provide some beneficial services to humans at both local and global scales (Joosten and Clarek 2002; Hanson et al. 2008; Kimmel and Mander 2010). Such functions may be hydrologically-, bio-geochemically-, or habitat-based and the potential of bogs to carry out such functions can range from very- to minimally- capable (Hanson et al. 2008). Table 2.1 lists a variety of functions and related services carried out by bog wetlands. A frequently discussed service of bog wetlands is their potential role in climate regulation and change. This is because peatlands, including bogs, store 1/3 of the globes carbon in the soil and are sensitive to warming associated with climate change (Zhuang et al. 2006; Joosten 2010). Peatlands are projected to become an increasing source of atmospheric methane and carbon as climates continue to increase (Zhuang et al. 2006; Joosten 2010; Gill et al. 2017). Emission of carbon and methane have the potential to contribute to future ongoing warming of the climate (Gill et al. 2017). Because bog functions and services, such as climate regulation, are not only the result of a single feature of a bog, but of all the interacting parts (chemistry, hydrology, species diversity, vegetation composition, and climate; Mitsch and Gosselink 2000, p.109), changes to any of these features may influence others and ultimately have the potential to influence bog functions and therefore, the services humans derive from these ecosystems (Luan and Wu 2015; Wang et al. 2015).

Table 2.1. Bog functions and services (Joosten and Clarke 2002; Hanson et al 2008; Kimmel and Mander 2010).

Service	Function
Fibre and Fuel	Peat production
Food and Medicine	Habitat for historically medicinal plants and various foods including berries.
Climate Regulation	Peat build-up is a major form of carbon storage, with potential to act as a carbon source.
Biodiversity	Unique habitat for niche species for plants and animals.
Groundwater Recharge	Sphagnum mosses can release and absorb water in amounts many times their size.

2.2. Canadian Bogs

Canada contains an estimated 24% of the world's wetlands (National Wetlands Working Group 1997) 170 million hectares of which is peatland (Poulin et al. 2004). It is estimated that about 20 million hectares of peatlands have been lost in the last 200 years (Environment Canada 1991). In Canada, the major wetland related policy that guides the protection of these ecosystems is the Federal Policy on Wetland Conservation, established in 1991. The stated objective of this policy is to "promote the conservation of Canada's wetlands to sustain their ecological and socio-economic functions, now and in the future" (Environment of Canada 1991). A specific goal of the policy includes the maintenance of wetland functions and values. Notably, this policy can only be applied to wetlands located on federal lands. As a result, on privately- or provincially-owned lands, wetland conservation and management is dependent on provincial governments (Austin and Hanson 2007). Provincial-based policies all vary in terms of their specific goals and restrictions (Austin and Hanson 2007), with those that have specific conservation objectives to those with only incidental relationships to wetlands. For example, in NL the only wetland related policy is one that is focused on permitting or not permitting development in wetlands rather than on conservation specifically (Austin and Hanson 2007; NLDME 2018).

In Canada, the national estimate of wetland extent states that there is ~150 million hectares (1.5 million km^2) of wetlands (National Wetlands Working Group 1997; Reimer 2009; Mahdavi et al. 2017). A provincial break-down of wetlands, last updated and published in 1988 by the Canadian National Wetlands Working Group (CNWWG 1988), can be seen in Table 2.2. Based on estimates of land-area, Manitoba, Ontario, NL, and Saskatchewan have the greatest extents of wetlands, the majority of which are comprised of peatlands.

Table 2.2. Estimates of provincial wetland extent as stated by the CNWWG (1988).

Province	% of land area that is peatlands	% of land area that is wetlands
Alberta	20	21
British Columbia	1	3
Manitoba	38	41
New Brunswick	2	8
Newfoundland and Labrador	17	18
Northwest Territories (includes Nunavut)	8	9
Nova Scotia	3	3
Ontario	25	33
Prince Edward Island	1	1
Quebec	9	9
Saskatchewan	16	17
Yukon Territory	3	3

2.2.1 Newfoundland Bogs

Previously it had been estimated 18% of the land area of NL is made up of wetlands, the majority of which are peatlands though a recent remotely-sensed inventory of NL wetlands (CNWWG 1988; Mahdianpari et al. 2018) estimates a much greater coverage. Beginning in the 1950s, peatlands in NL were drained for the development of several pastures, and in the 1980s, were used to produce vegetables (NLDFL 2017b). Recently, peatlands have been stripped of peat, ditched and drained for the development of cranberry farms (NLDFL 2017; NLDME 2019). There is an estimated 290 acres of cranberry farms in NL currently, and an estimated total of 465 acres expected by 2020 (NLDFL 2016).

The most common peatland forms in eastern Newfoundland, which encompasses the study area for this research, includes ribbed fen, slope fen, domed bog, blanket bog, slope bog and basin bog (Wells 1981). These forms are differentiated on the basis of surface formation, relief, and proximity to water bodies, and different bog forms have different characteristic vegetation compositions (Wells 1981). Table 2.3 describes the characteristics of the most common bog forms in eastern NL.

Table 2.3. Bog forms common in Eastern Newfoundland, as defined by the CWCS (National Wetlands Working Group 1997).

Bog Form	Form Description
Basin	Bogs confined to a basin with a flat surface and water is received via precipitation and run-off from the immediate surroundings.
Slope	Bogs that occur on and are level with slopping terrain.
Domed	Bogs at least 500m in diameter with a convex surface and drainage radiates from the centre to the edges.
Blanket	Expansive bogs with a uniform surface and lack of pools that cover gentle slopes on hillsides and valleys.

At the provincial government level, wetlands including bogs are managed by the PDW (NLDME 2018). The PDW is the only wetland-related policy in the province (though there is provincial legislation which regulates ATV use within wetlands; NLDFL 2019), and is not specifically conservation-based (Austen and Hanson 2007; NLDME 2018), but relates directly to anthropogenic development within wetlands, moderating and allowing development in such a way so that development will not “adversely affect water quality, water quantity, hydrologic characteristics or functions, and terrestrial and aquatic habitats of the wetland” (NLDME 2018).

Directly citing the extensive nature of wetlands in NL with specific reference to peatlands, the policy states that there is room for development to occur as needed. However, there has yet to be a province-wide assessment of wetland loss (Austen and Hanson 2007) and it has been predicted that the population and housing development in the study area is to increase over the next 15 years (City of St. John's 2014; Simms and Ward 2017). Additionally, the region which includes the study area is the only region in the province that is expected to have a future population increase (Simms and Ward 2017). This is of relevance to the human population within this area because the quality and quantity of bog functions and services derived from these ecosystems may be altered by wetland loss or degradation (Austen and Hanson 2007; Kimmel and Mander 2010). The monetary values of the loss of such services in other parts of the world have been reported in several previous studies (Michell 2017; Narayan et al. 2017). In addition to the direct services bogs make available to populations, NL bogs provide important habitat to at-risk species. For example, the vulnerable Rusty Blackbird (*Euphagus carolinus*) uses bogs and other forested wetlands for breeding (NLDFL 2017a).

2.3 Anthropogenic Landscapes and Wetland Ecology

NL wetland policy clearly acknowledges the potential for the development (e.g., housing and agriculture; NLDME 2018; City of St. Johns 2014) of wetlands including bogs despite the province having no inventory to track wetland loss over time. Similarly, the condition of those wetlands that do remain is largely unknown, as there has been no notable work towards the development of a province-based wetland assessment procedure to evaluate and track the changes in wetlands under anthropogenic influence. Anthropogenic land-use conversion both within and without wetland boundaries is frequently described as the greatest threat to wetland loss and quality of wetlands services both currently and in the future (Millennium Ecosystem Assessment 2005). As a result, there has been increasing demand to understand the variety of ways in which anthropogenic land-use can impact bog ecosystems at the scale of both the wetland and the wetland landscape (Jacobs et al. 2010).

Adjacent landscapes have increasingly been studied under the premise that human activities in the surrounding landscape can have direct, secondary, and cumulative effects on the ecology (processes, community composition, etc.) of ecosystems (Brown and Vivas 2005). Wetlands impacted by adjacent landscapes have resulted in eutrophication, monotypic vegetation stands (Zedler and Kercher 2005), and changes in typical faunal composition (Uzarski et al. 2017). The negative impacts on wetlands has resulted in the increasing incorporation of landscape scale information in the development wetland policy, assessment tools and indicators of wetland service health (U.S. EPA. 2002a).

Wetland vegetation is amongst the most widely used indicators of anthropogenic influence on wetland ecology (Wilson and Bailey 2012). There are several studies of wetlands of different types assessing the impact of the landscape on wetland condition (Hychka et al. 2007; Johnston et al. 2009; Tsai et al. 2012; Miller et al. 2016). When studying wetland vegetation in the context of anthropogenic influence, it is sometimes convenient or necessary to group species that share similar physical, chemical, or biological traits, or responses to environmental manipulation or change (Baxendale et al. 2016), for example growth-form (Chapin et al. 1996). One of the most appealing aspects of using growth-form in studies of wetland changes in response to anthropogenic activity is that often in the field, growth-forms are easier to identify, capable of being resolved by and estimate compared to that of individual species (U.S. EPA. 2002a). Additionally, growth-forms are most easily associated with wetland functioning (Houlahan et al. 2006; Jagerbrand et al. 2012; Nishimura and Tsuyuzaki 2015; Hedwall et al. 2017) and can be captured by remote sensing methods (Harris et al. 2015; Guo et al. 2017).

Much of the studies of wetlands at the scale of landscape tend to focus on specific wetland classes in specific geographical areas because the sensitivity of vegetation to human influence can vary by region, by wetland type, and by stressor. This means that wetland-landscape relationships in novel geographical areas and across new wetland classes should be established before they are applied. Notably, most studies of wetlands and landscape follow the USA wetland classification standards Cowardin et al. 1979; Brinson 1993; Smith et al. 1995). Rooney and Bailey (2011) is one of the few exceptions, where they found that landscape quality to a scale of >1500m had an impact on vegetation and chemical composition of shallow water wetlands in Alberta.

Additionally, most studies have tended to focus on those wetlands that have direct and obvious relationships to the greater watershed through waterflow connectivity, such as marsh or shallow water systems (Hychka et al. 2007; Rooney and Bayley 2011; Martinez-Lopez et al. 2014; Miller et al. 2016). These wetlands are directly influenced by the quality of upstream water inputs, which in turn are influenced by pollution in the form of run-off from the catchment landscape (Porcella and Middlebrooks 1971; Zedler and Kercher

2005). Few studies specifically consider wetlands that have unclear connections to their surrounding uplands, such as geographically isolated wetlands (Stuber et al. 2016) or bogs.

2.3.1 Anthropogenic Influences on Bog Wetland Vegetation

Growth-form composition and abundance in bogs has been shown change overtime because of alterations to environmental conditions as a result of anthropogenic activity. Environmental (naturally occurring and controlled) changes that have been shown to influence bog growth-form composition includes: additions of certain nutrients such as nitrogen (Nishimura and Tsuyuzaki 2015; Wu and Blodau 2015; Hedwall et al. 2017), water table alteration (Kleinebecker et al. 2010; Churchill et al. 2015; Luan and Wu 2015), growth-form removal (Ward et al. 2013; Robroek et al. 2015), and litter quality (Dorrepaal 2007; Ward et al. 2015). Different growth-forms have been shown to respond variably to these changes (Nishimura and Tsuyuzaki 2015; Udd et al. 2015). Changes to typical bog growth-form composition as a result of such environmental changes have been found to have implications on such bog services as greenhouse gas cycling and carbon storage (Maljanen et al. 2010; Ward et al. 2013; Luan and Wu 2015; Robroek et al. 2015) water quality (Evans et al. 2014) and water retention (Robroek et al. 2010).

Environmental conditions can be the result of changes in the landscape. For example, nitrogen pollution in run-off and in the atmosphere can be the result of surrounding fertilized agricultural landscapes or roadway and emissions. Additionally, the composition of pollutants in the atmosphere within and around urban areas can be complex containing assorted pollutants originating from homes, roadways, industrialized areas, sewage treatment plants, and garbage dumps that can act as both point and non-point sources of pollutants. (Pinho et al. 2008; Koch et al. 2016). Urban areas may also generate urban heat islands (Adam 2017; Munzie et al. 2014). Catchment development also has the potential to alter the height of water tables by re-directing water sources from one area to another and decreasing water penetration into soil via solid surfaces (Bignal et al. 2007; Pasquet et al. 2015). Though the study of specific landscape types and amounts on bog wetlands and bog growth-forms are limited, specific adjacent anthropogenic land-use features including roads, industrial features, and agriculture can alter bog vegetative composition via pollutants and ground-water level alteration (Bignal et al. 2007; Pellerin et al. 2008; Pasquet et al. 2015; Sengbusch 2015).

2.3.1.1 Bryophytes (Mosses)

Mosses as a group are often used as bio-indicators of the presence of anthropogenic air pollutants such as heavy metals, sulfur dioxide, nitrogen and urban pollutant mixtures (Palmieri et al. 1997; Pesch and Schroeder 2006; Cowden et al. 2015; Pescott et al. 2015). Mosses obtain some nutrients through aerial sources (Brown and Bates 1990). This may be compared to that of vascular plants, which obtain their nutrients via roots (Winner and Atikinson 1987). As a result, mosses react more rapidly than vascular plants to changes in the environment and are particularly susceptible to changes in pH in peatlands (Hajkova and Hajek 2004; Vellak et al. 2014). Moss growth-form presence in bogs may be influenced by adjacent urban land-use or roadways. Mosses are also susceptible to nitrogen addition (Kleinebecker et al. 2010), temperature changes (Weltzin et al. 2000) and to water table alterations (Potvin et al. 2015; Udd et al. 2015). Sphagnum Moss, the most dominant moss in bog wetlands are susceptible to shading from taller graminoids, shrubs, and trees (Pellerin et al. 2008; Kettridge et al. 2013)

Generally, the competition for space in bogs between mosses and vascular plants is high, though mosses tend to dominate in such low nutrient environments (Wardle 1991). When mosses dominate, they contribute to the acidity of the bog, further inhibiting vascular plant growth (van Breeman 1995). However, changes to the bog ecosystem can have a great influence on mosses, allowing for shifts from moss to vascular plant dominance. For example, in nutrient addition studies, moss coverage decreased while vascular plant coverage increased (Jagerbrand et al. 2012; Nishimura and Tsuyuzaki 2015; Hedwall et al. 2017). Similarly, mosses decreased and vascular plants increased in studies of peatland warming (Arft et al. 1999; Bragazza et al. 2012; Dieleman et al. 2015; Hedwall et al. 2017). When vascular plants increase, there is a feedback where-by increased shading by vascular plants further suppress shade intolerant mosses, contributing to greater vascular plant dominance (Udd et al. 2015). Mosses decompose slower than any other bog growth-form (Dorrepaal et al. 2005; Dorrepaal 2007) and as a result, are essential for peat-building. When environmental stressors negatively impact moss growth in bogs, there are implications on the peat-forming functions of bogs.

2.3.1.2 Lichens

Lichens are composite symbiotic organisms comprised of both photosynthetic algae/cyanobacteria and fungi (Ahmadjia 1993). Measures of lichen (e.g., percent cover, composition, diversity, species presence/absence and vitality) have frequently been used as bio-indicators of air pollution (Llop et al. 2012; Degtjarenko et al. 2016; Koch et al.

2016). The physiology of lichen causes them to be particularly susceptible to airborne pollutants, because lichens absorb atmospheric constituents through their entire thallus and leaf surfaces (Britton and Fisher 2010; Stevens et al. 2012; Djekic et al. 2017), making them an effective bio-monitoring tool. In highly industrialized areas where air pollutants are high, lichen abundance and diversity can be low (Wannaza et al. 2012; Koch et al. 2016).

Non-industrialized urban areas often contain complex mixtures of pollutants and, as a result, several studies have used land cover as a representative of air pollution, to examine the effects of ambient urban air pollution on lichen measurements (Pinho et al. 2008; Llop et al. 2012; Stevens et al. 2012). Koch et al. (2016) found that lichen cover was negatively correlated with open fields, an overall decrease in cover with increasing urbanization, and a positive correlation with agricultural areas, at a scale of 3200m. Outside of pollution, the effects of urban heat islands can also influence lichen presence (Munzie et al. 2014; Adam 2017).

In a study of bog lichen, it was found that *Cladonia* (common in bogs) occurrence increased with increasing nitrogen deposition (Stevens et al. 2012). Other studies of nitrogen deposition on lichen have found contrasting results, where in some cases, increased nitrogen increased the growth of terricolous lichen, and in others increased nitrogen caused a decrease in growth (Britton and Fisher 2010). In Netherlands bogs, the almost complete disappearance of lichens in these ecosystems has been attributed partially to atmospheric pollution (Tomassen et al. 2004).

Other influences on lichen presence in bogs includes water level. Because of the poikilohydric nature of lichens, they are more likely to occur in dry bogs or dry portions of bogs as they can out-compete other vegetation in such situations (South 1983; Green and Lange 1995). Kleinebecker at al. (2010) found that the abundance of lichens in South Patagonian bogs was greater in the drier areas, where they outcompete bryophytes more common in moist conditions. Because of this, there is potential that alterations of a bog's water table influence on lichen presence/absence or lichen composition relative to other vegetation growth-forms.

2.3.1.3 Forbs and Ferns

Forbs include all non-graminoid, herbaceous vascular plants (Chapin et al. 1996). Ferns are vascular but are reproduce using spores. In ombrotrophic bogs, where there are low levels of soil nutrients and highly saturated conditions, mosses can out-compete forbs and ferns (Wardle 1991). Ferns and forbs obtain their nutrients using roots and because mosses obtain nutrients through direct contact with water and air. Additionally, mosses contribute to the highly acidic environment of bogs, acidity which is not conductive of forb and some fern growth (Wardle 1991). In conditions where there is greater ecosystem nutrient availability forb and fern functional types will have greater success at competing with moss dominance (National Wetlands Working Group 1997). Amongst the vascular plants present in bogs, herbaceous functional types like forbs, ferns, and graminoids are generally more tolerant overall to wet and flooded conditions than are woody growth-forms (Cronk and Fennessy 2001; Churchill et al. 2015).

Given this difference in acquiring nutrients, it has been hypothesized that forb and fern presence may be greater in bogs when a greater input of nutrients occurs (Jonasson et al. 1999). This has been tested in some nutrient addition experiments, where vascular plant coverage (including forbs) increases over moss (Jagerbrand et al. 2012; Nishimura and Tsuyuzaki 2015; Hedwall et al. 2017). Amongst vascular plants in bog's however, graminoids were found to outcompete forbs when nutrients were added, as a result of graminoids having a greater capacity to succeed in shoot competition, where forbs are more effective at root production (Nishimura and Tsuyuzaki 2015). Additionally, warming experiments in peatlands found that moss species presence tends to decrease while vascular (including forb) presence increases (Arft et al. 1999; Hedwall et al. 2017) although other evidence shows that the responses are not always consistent (Dorrepaal et al. 2005). Adjacent land-use development has the protentional to influence nitrogen (and other pollutant) deposition, warming, and water table alteration, and thus such has the potential to influence forb and fern presence in bogs, which have been shown above to be influenced by nutrient input, temperature, and wetness variation (Jagerbrand et al. 2012; Churchill et al. 2015; Nishimura and Tsuyuzaki 2015; Hedwall et al. 2017). Forbs have been associated with human activity, where they favored sample plots closest to and associated with disturbance in a peatland complex in Quebec in a study of impacts of human disturbance on peatland vegetation (Tousignant et al. 2010).

Graminoids are vascular and herbaceous that are generally narrow-leafed (Chapin et a. 1996) and are deeper rooted than woody vascular plants (Jassey et al. 2014). Graminoids are typically more prominent in fens, due to the minerotrophic status of fen wetlands (National Wetlands Working Group 1997). Minerotrophic wetlands have a greater input of nutrients including nitrogen, in which groups such as graminoids thrive and mosses and shrubs do not (Kleinebecker et al. 2010). Similarly, fens are typically wetter, due to contact with surface- and ground-waters. Graminoids are more likely to grow due to their greater tolerance to flooding than woody trees and shrubs (Cronk and Fennessy 2001; Churchill et al. 2015).

The herbaceous forbs, ferns and graminoids share similar reactions to various alterations in bog environments (Murphy et al. 2009; Jagerbrand et al. 2012) However, graminoids as a group react differently than do forbs to the alteration of bog environments. For example, graminoids were found to outcompete forbs in bogs when nutrients were added due to the ability of graminoids to outcompete forbs in shoot competition (Nishimura and Tsuyuzaki 2015). In saturated conditions, although both graminoids and forbs are more flood tolerant than woody trees and shrubs graminoid abundance was found to be greater than that of forb in a study of sustained flooding of a peatland (Cronk and Fennessy 2001; Churchill et al. 2015). Forbs are more likely to decompose faster than graminoids due to differences in leaf tensile strength (Dorrepaal et al. 2005).

Graminoids and woody plants show notable differences in their responses to changes to the bog environment (Ward et al. 2013; Kuiper et al. 2014; Robroek et al. 2015). Graminoid removal resulted in lowered potential methane production compared to shrub removal (Robroek et al. 2015). In another removal study (Ward et al. 2013), carbon dioxide sink strength was greatest under warming conditions when shrubs were present and weakest when graminoids were present, and graminoid presence contributed most to methane emission. Another study found that warmed bogs produced more shrubs and less graminoids (Weltzin et al. 2000). Under a constant water tables verses fluctuations, graminoid production was greater than that of shrub production (Weltzin et al. 2000; Riutta et al. 2007; Breeuwer et al. 2009).

2.3.1.4. Woody Plants (Shrubs and Trees)

Woody plants such as shrubs and trees become more common in bogs that have undergone some form of direct or indirect drying. For example, Pellerin et al. (2008), partially attributed increases in percent cover of trees in two Quebec bogs to the interacting effects of time and human activity, including ditch development, land cultivation, and presence of other disturbed surfaces inside and adjacent to the bogs. In another study of Quebec bogs, Pasquet et al. (2015) examined bogs isolated in agriculturally developed landscapes, which caused the drainage of the surrounding catchment. This drainage was attributed partially as the cause of an increase in trees. Similarly, Sensbusch (2015) found an increased tree and shrub presence in a bog with an adjacent roadway, concluding that the roadway diverted some water flow, allowing for the growth of the woody forms. The cause of drying of bogs has been demonstrated in water table manipulation experimental studies, where shrubs increased in presence over that of graminoids when a dry-down period was present or when the water table remained low (Breeuwer et al. 2009; Potvin et al. 2015).

Temperature increase in a bogs climate, which is expected in the SJMA in the future via a changing climate (Finnis and Daraio 2018), may result in the increased woodiness of bogs (Heijmans et al. 2013). Increased shrub and tree presence have the potential to influence litter quality and decomposition (Dorrepaal et al. 2005; Ward et al. 2015), and moss presence through shade competition (Udd et al. 2015). Nutrient addition in the form of nitrogen has also been attributed to increased shrub presence in peatlands, where moss competition is suppressed (Limpens et al. 2003; Hedwall et al. 2017). In the opposite case, when water tables in bogs are increased via flooding or diversion of water sources, resulting in a transition away from bog status, woody forms may decrease due to their inability to survive in saturated conditions and a transit (Breeuwer et al. 2009; Potvin et al. 2015). Loss of woody growth-forms in peatlands has been shown to increase potential methane oxidation, decrease carbon dioxide exchange (Kuiper et al. 2014) and decrease the ability of the bog ecosystem to act as a carbon sink (Ward et al. 2013).

Changes such as increased nitrogen loading due to atmospheric pollution or adjacent anthropogenic activity has been shown to cause shifts in moss growth-form dominance and increases in vascular plant growth-forms such as graminoids (Wu and Blodau 2015), which in turn can cause a lowering of the water table (Eppinga et al. 2009). Lowered water tables in bogs can then favor shrub or tree encroachment (Breeuwer et al. 2009; Pellerin et al. 2016), which has been shown to shift peatlands from carbon sinks to sources (Trettin et al. 2006). Specific adjacent anthropogenic land-use features including roads, industrial features, and agriculture have been shown to alter bog vegetative growth-form composition because of pollutants and ground-water level alteration (Bignal et al. 2007; Pasquet et al. 2015; Sengbusch 2015). Changes in the composition of bog growth-forms can then affect the carrying-out of various functions related to for example, nutrient cycling (Kuiper et al. 2014; Robroek et al. 2016) or water-retention (Robroek et al. 2010).

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Chapter 3: A PRELIMINARY ASSESSMENT OF THE INFLUENCE OF ADJACENT LAND-USE ON GROWTH-FORM COMPOSITION OF VEGETATION IN SMALL NEWFOUNDLAND BASIN BOGS.

3.1 Introduction

Research on the influence of surrounding land-use on ecosystems has been carried out extensively in the context of wetlands such as coastal (Margriter et al. 2014; Nestlerode et al. 2014) and freshwater wetlands (Weller et al. 2007; Fernandes et al. 2011; Miller et al. 2016), as anthropogenic activity in the surrounding landscape is known to have direct, secondary, and cumulative effects on wetland ecology (Brown and Vivas 2005). Much of the research that examines such landscape influences focuses on those wetlands that have direct relationships to the greater watershed through waterflow connectivity such as estuary, marsh or shallow water systems (Chipps et al. 2006; Hychka et al. 2007; Rooney and Bayley 2011; Martinez-Lopez et al. 2014). Comparatively less research has specifically considered the effects of landscape on the ecosystem and function of those wetlands that have less-direct connections to their surrounding uplands, such as bogs.

Though having historically been viewed as generally stable ecosystems (Zobel 1988), changes to the bog-environment including changes to water table depths, acidity, nutrient availability and light, can result in shifts in typical bog communities (Pellerin et al. 2008; Breeuwer et al. 2009; Jagerbrand et al. 2012). The sources of these environmental changes can be a result of anthropogenic disturbances both within the bog and in its surrounding landscapes (Bignal et al. 2007; Konvalinkova and Prach 2014; Pasquet et al. 2015). One of the many consequences of anthropogenic activities on bogs is the modification to their functions (Robroek et al. 2010; Ward et al. 2013; Evans et al. 2014; Robroek et al. 2015). Bog functions are natural processes carried out within the bog ecosystem such as nutrient cycling and water retention. These functions are the product of complex interactions amongst various biotic and abiotic bog features including hydrology, geomorphology, chemistry, and flora and fauna (Hanson et al. 2008; Hongjun et al. 2010) and are the source of important ecosystem services such as climate regulation, water regulation, and biodiversity habitat (Joosten and Clarke 2002; Hanson et al. 2008; Kimmel and Mander 2010). Anthropogenic activities have the potential to alter biotic and abiotic aspects of bogs, which in turn can alter normal bog functioning, impacting the quality of bog-derived services.

Growth-form composition within bogs, for example, has been shown to be altered by anthropogenic activity in a variety of ways (Bignal et al.2007; Pasquet et al. 2015; Sengbusch 2015) and has been used to examine and model the effects of various environmental effects such climate change, nitrogen deposition, and water table alteration on peatlands including bogs and associated bog functions (Bombonato et al. 2010; Jagerbrand et al. 2012; Nishimura and Tsuyuzaki 2015; Hedwall et al. 2017). Additionally, vegetation measurement at the scale of growth-form composition provides an alternative to a species approach marred by challenges such as a large number of species requiring identification, difficulties in identifying certain species particularly sedge and grass species, and the challenges of associating individual species with wetland functioning (Duckworth et al. 2000; U.S. EPA 2002; Houlahan et al. 2006).

Other benefits of measuring growth-form composition are the ability of moderate resolution multi-spectral remote sensing imagery to capture information at the scale of vegetative growth-form, verses more detailed classification such as species or genus (Ustin and Gamon 2010; Harris et al. 2015; Guo et al. 2017). Recently, remote sensing methods are increasingly being applied in studies of the estimation of abundance and changes of wetland vegetation including growth-forms (Dronova et al. 2012; Dronova 2015; Guo et al. 2017). Specifically, vegetation indices (VI) derived from remote sensing imagery spectral bands are an easy way to quantify changes in vegetation composition, when correlation between vegetation and the VI exist (Harris et al. 2015; Guo et al. 2017). The benefits of using remote sensing methods to track changes in bog vegetation includes the ability to more easily (verses field visits): assess bogs multiple times to track changes over time, identify bogs that are of greater concern in terms of changing condition, assess the effectiveness of bog management, assess bogs in hard-to-visit locations which are common in places such as NL, avoid the bias of different field-visitors, and avoid trampling and other negative effects of field visits (He et al. 2015). Remotely sensed imagery has the potential to aid in the measurement of changes in bog ecosystems as bog vegetative growth-forms have been proven to change predictably in response to various types of environmental changes (Wu and Blodau 2015), changes which are often a result of products of adjacent land-use (Bignal et al. 2007; Pasquet et al. 2015).

Despite the sensitivity of bogs to anthropogenic development (as demonstrated by shifts in growth-form composition), wetland conservation and managerial plans across Canada are often lacking in the context of peatlands specifically (Poulin et al. 2004). In the province of Newfoundland (NL) for example, the governmental policy managing these wetlands (and wetlands in general) restricts development within wetland boundaries that adversely impact functions, but has no restrictions on the lands outside, but within close proximity, of wetland boundaries (NLDME 2018). This is despite studies providing evidence that wetlands functions can be impacted in various ways through the presence of anthropogenic developments adjacent to, but not within, wetland boundaries (Burbridge 1994; Houlahan et al. 2006; Reiss et al. 2010; Miller et al. 2016).

The aim of this study was to examine the potential relationship between vegetation growth-form composition and adjacent anthropogenic land-use, taking small basin bogs in an urbanized area of north-eastern NL as a case study. Additionally, the potential of remotely sensed VI to estimate basin bog growth-form composition and changes of bog growth-form composition was examined in an exploratory context. Remotely sensed indices have the potential to be used in lieu of traditional field-work and can allow for more cost- and time-effective assessment of impacts of environmental changes such as adjacent anthropogenic land-use on bog wetlands. This is of interest for NL as there has been increasing interest and funding contributions towards developing methods for remote sensing of wetlands and wetland health in the province (Amani et al. 2017; Mahdavi et al. 2017) and given that population, house development and agricultural development within the study area is expected to increase in the future (City of St. Johns 2014a; Simms and Ward 2017; NLDFL 2019). Specifically, we asked: (1) Do basin bogs with similar adjacent land-use characteristics (natural, urban, or pasture) have similar growth-form compositions? (2) How do a) the quantity of anthropogenic land-use and b) the scale at which land-use is quantified influence the strength of associations between growth-form composition and adjacent land-use in bog wetlands? (3) Do common remotely sensed VI correlate with variability in bog growth-form composition? Given that some conservation and management policies, as is the case in NL, rarely place restrictions on development in adjacent land-use and given that many of these policies are not optimally designed to consider peatlands (Poulin et al. 2004; NLDME 2018), it is important to understand how this type of management may influence bog wetlands in the future. Because growth-forms are often used to understand and model wetland function and service provision, and if adjacent land-use does in fact impact bog growth-form composition, arguments can be made for building on the results of this preliminary study and further examining the influence of adjacent land-use on bog growth-forms in both field-based and remotely-sensed contexts, as well as justifying the consideration of alternative policies or management methods on bog adjacent land-use, which in-turn protects bog functioning.

3.2 Methods

3.2.1 Study Area

The St. John's Metropolitan Area (SJMA; Statistics Canada 2011) is located on the east coast of the island NL, Canada (Figure 3.1). SJMA is the most populated region in the province, with a population of 209, 995 and a population density of 255.9 people per square kilometer (Statistics Canada 2017). It is currently projected that the SJMA is the only region in NL that will have an increasing population over the next 15 years (Simms and Ward 2017). Arable land in the province is limited (NLDFL 2017), and much of the farmland in the SJMA is designated as pasture rather than crop agriculture. Recently however there have been recent initiatives to increase local food production by further developing land for agriculture and to ease the approval process for farmers purchasing land (NLDFL 2019).

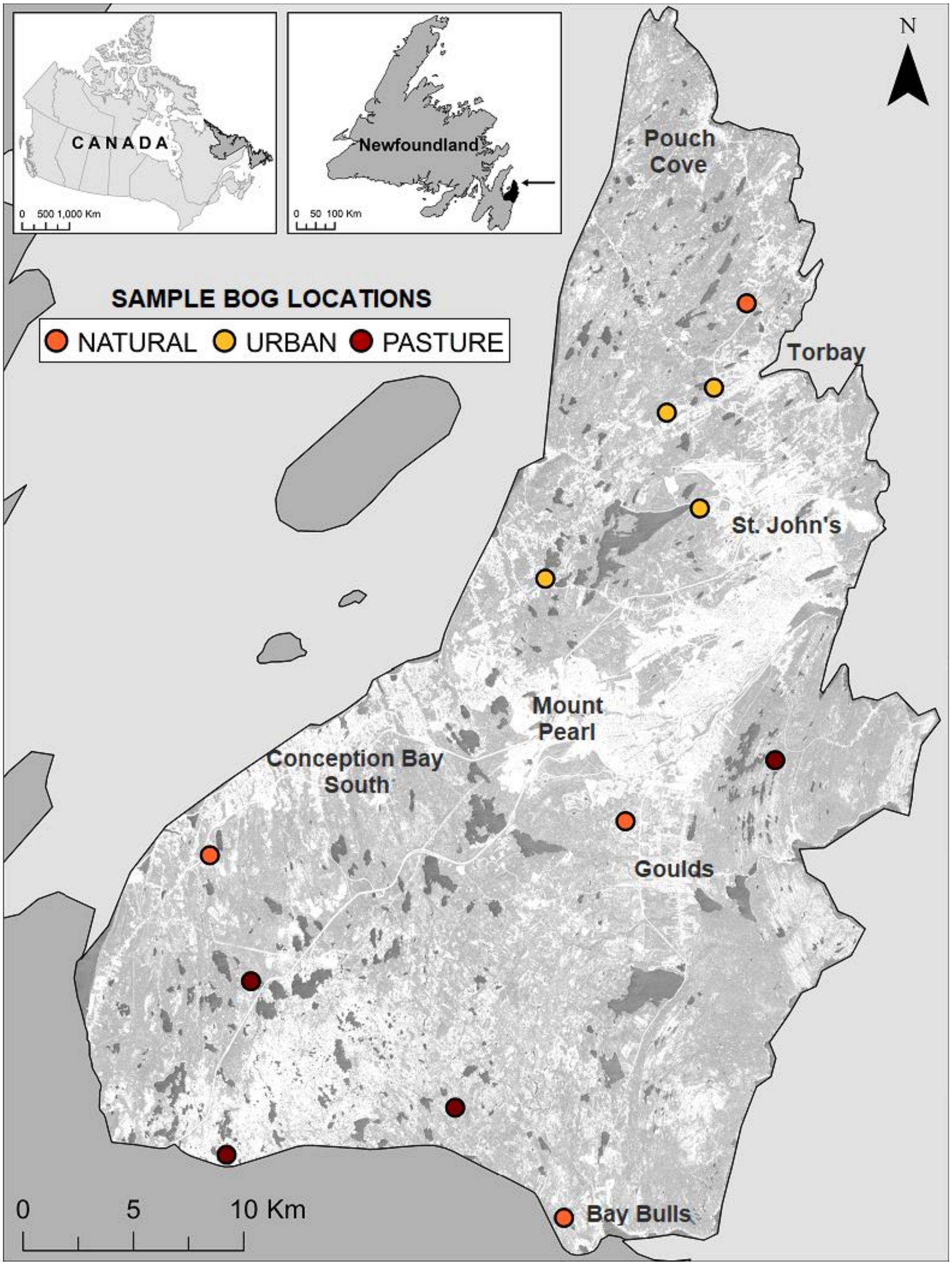

Figure 3.1. Sampled bog sites located across the St. John's Metropolitan Area of Newfoundland and Labrador.

The SJMA falls within the Boreal Forest Biome, and the Maritime Barrens Ecoregion (MBE; South 1983) which is characterized by an oceanic climate, where summers are cool and foggy, and winters are relatively mild (South 1983). In the SJMA area, the average summer and winter temperature is around 14° Celsius and -4° Celsius respectively (Government of Canada Environment and Natural Resources 2019) and the growing season of the area is within a range of 140-150 days (Government of Canada Agriculture and Agri-Food Canada 2019). Natural land-cover in the area is dominated balsam fir (*Abies balsamea*) stands and large swaths of nutrient poor ecosystems, including extensive heath barrens and peatlands (bog and fen) dominated by species of dwarf shrub, sphagnum mosses, and lichen (South 1983). The most dominant peatland in the area is bog (C-Core 2017), and the most common bog forms are domed, blanket, slope and basin (Wells 1981). Historic use of these peatlands include agriculture, peat harvest and recreation (Wells 1981).

Eastern NL bogs are dominated by species of sphagnum moss, mainly *Sphagnum fuscum*, shrubs including *Rhododendro groenlandicum*, *Kalmia angustifolia*, *Empetrum nigrum*, sedge including *Trichophorum cespitosus*, and lichen species particularly belong to the *Cladonia* genus such as Cladonia *rangiferina* (Wells 1981). Other common bog species include *Sphagnum tenellum, Vaccinium vitis-idaea, Cladonia boryi, Larix laricina*, and *Rhynchospora alba* (Wells 1981). Typical bog vegetation communities differ somewhat across different regions of NL (Pollett and Bridgewater 1973; Wells et al. 1981).

A recent remotely-sensed wetland inventory conducted within the SJMA (Amani et al. 2017; C-Core, 2017; Mahdavi et al. 2017) mapped and classified wetlands as one of five classes defined by the CWCS; National Wetlands Working Group 1997). The most recent inventory reports that the SJMA has an estimated wetland coverage of around 45%, of which 36% is bog, 21% is fen, 32% is swamp, 3% is marsh, and 8% is shallow water (C-Core 2017).

3.2.2 Study Site Selection

The bog sites that are the focus of this study were selected from a large pool of previously classified wetlands produced via the field-work and remote-sensing results of the recent NL wetland classification described above (Amani et al. 2017; C-Core 2017; Mahdavi et al. 2017). From the bogs classified by the inventory, a total of 12 (see Figure 3.1 and Table 3.1) were selected for vegetation sampling. While the sample size of study bogs is small, the intention was to provide a preliminary basis to justify further examination of anthropogenically-influenced bogs in the SJMA at the scale of growth-form (using remote sensing and/or field work). The selection of the 12 study sites was done so on the basis of the following criteria: (1) characteristics of the adjacent land-use within 200m of the bog boundary, (2) bog size less than or equal to 10,000 m^2, (3) at least 2 km away from another study bog site, (4) accessibility via roads or trails, and (5) consistent bog form.

Table 3.1. Characteristics of the study bog wetlands where U=urban dominated, N=natural dominate, and P=pasture dominated adjacent land-use.

Name	Land-use Class	Size (m^2)	Lat	Lon
N1	Natural	5000	47°20’18.50”N	53°02’20.30”W
N2	Natural	3610	47°30’17.79”N	52°42’55.01”W
N3	Natural	3931	47°24’34.44”N	53°01’37.34”W
N4	Natural	5000	47°21’36.32”N	52°54’09.46”W
P1	Pasture	1700	47°28’43.07”N	52°48’16.08”W
P2	Pasture	7649	47°18’37.44”N	52°50’24.35”W
P3	Pasture	7000	47°41’25.92”N	52°44’20.29”W
P4	Pasture	5638	47°27’38.03”N	53°03’13.76”W
U1	Urban	2109	47°39’21.49”N	52°45’25.73”W
U2	Urban	5683	47°36’23.85”N	52°45’50.88”W
U3	Urban	2791	47°38’42.95”N	52°47’07.39”W
U4	Urban	5405	47°34’35.43”N	52°51’22.27”W

With the purpose of assessing whether the type of adjacent land-use, specifically dominant urban, pasture, and natural, have similar influence on bog growth-form composition, bogs were selected based on dominant land-use in the adjacent landscape. Here, we define adjacent land-use as land-use that occurs within 200m of a bog’s boundary. Various studies investigating the influence of adjacent land-use of wetlands have addressed the issue of scale and the spatial extent of adjacent land-use effects, with the optimal landscape scale differing depending on wetland type, geography and geomorphology of the study area, and the specific response variables being investigated (Houlahan and Findlay 2004; Fernandes et al. 2011). For example, wetlands connected to hydrological systems often have landscape characterized optimally at the scale of catchment (Crosbie and Chow-Fraser 1999; Brooks et al. 2004; Nestlerode et al. 2014). No study was found that defined an optimal scale for the study of the influence of landscape and land-use on bog wetlands specifically. However, it has been suggested that a buffer of 100m around isolated depressional wetlands is effective to capture the influence of adjacent landscape on, among other variables, vegetation in wetlands located in the flat Florida terrain (Lane 2003). Like bogs (National Wetlands Working Group 1997), these isolated depressional wetlands receive water largely from atmospheric sources (Gala and Young 2015). Slightly larger scales of ~300m have been suggested in more irregular, hummocky terrain (Mita et al. 2007). Here, we chose to examine land-use within a 200m buffer surrounding the study bogs as the Avalon Peninsula is characterized by both irregular terrain and flat barrens (NLDME 2017). It is acknowledged that there are differences between the terrain in the Avalon Peninsula and the terrain in the studies from which the 200m buffer was based (Lane 2003; Mita et al. 2007), and differences between bogs and isolated depressional wetlands (Gale and Young 2015). As such, it is likely that the buffer of 200m does not capture certain characteristics unique to the land in the Avalon area.

Bogs of a size smaller than or equal to 10,000 m^2 were selected not only to keep size somewhat consistent across sites (see Table 3.1), but because most bogs within the anthropogenic landscape of the SJMA tended to be smaller than those bogs located in less urbanized locations in the study area (C-Core 2017). Additionally, adjacent land-use may impact the total area of smaller bogs differently than larger bogs (Bignal et al. 2007; Laroche et al. 2012; Feldmeyer-Christe and Küchler 2017).

Bogs in eastern NL may be classified into different bog forms based on surface formation, relief, and proximity to water bodies (Wells 1981; National Wetlands Working Group 1997). Bogs of different forms have different typical vegetation composition and are influenced differently by surrounding run-off from adjacent landscapes (Wells 1981;

National Wetlands Working Group 1997). Growth-form composition in different bog forms may not be impacted by adjacent land-use in the same ways and as a result, bogs selected in this study were basin bogs only.

To select bogs following the above criteria, classified bog data were obtained from the NL wetland inventory (Amani et al. 2017; C-Core 2017; Mahdavi et al. 2017) and were filtered to include those bogs with an area of 10000 m^2 or less. Some of the field data for the wetland inventory project included information on bog form and only those classified as basin bogs were selected. These bogs were then viewed within ArcMap 10.5.1 (ESRI 2017) in comparison with classified landcover data. These data were obtained from the remotely sensed Annual Crop Inventory 2016 (CI) (Government of Canada Agriculture and Agri-food Canada. 2016). CI landcover classification was obtained using Decision Tree methods and Landsat-8, Sentinel-2, Gaofen-1 and RADARSAT02 satellite imagery, resulting in a 30 m resolution classification. This data source was selected because it is one of the most up-to-date (2016) and free-to-obtain land-use data sets available for the province. The CI was downloaded from the Government of Canada Open Government portal (Government of Canada 2018). The land-use classes provided by the CI were reclassified as urban, pasture, or natural.

Landcover within 200 m of the small bogs was assessed to select twelve bogs in total, four “natural”, four “urban”, and four “pasture” bogs” (see Table 3.1). The four natural bogs (bogs surrounded by non-developed adjacent land-use) were selected to act as reference sites and were assumed to represent the range of variability in the growth-form composition of bogs relatively less impacted by anthropogenic land-use. The concept of reference ecosystems has been used previously in various studies of wetlands and adjacent land-use, and in the production of indices of biotic integrity (IBI; Fennessy et al. 2007; Lunde and Resh 2012; Moges et al. 2016). It should be noted that here, “natural” means the surrounding land has not undergone any major land conversion, though there is some anthropogenic activity present in the form of small paths, ATV use, and some wood cutting. Due to the distribution of bog wetlands in the SJMA, it was not possible as it was with the natural sites to select bogs surrounded almost purely by urban or pasture land-use. For example, many of the bogs in pasture landscapes also have certain amounts of undeveloped treed areas. This variability must be considered when interpreting the results.

3.2.3 Field Sampling

Vegetation sampling was conducted between July and early September 2017 using a quantitative quadrat random sampling method. Using ArcMap 10.5.1 (ESRI 2017), the boundaries of the bogs were delineated. Sampling points were randomly distributed within the delineated boundaries using the Create Random Points tool provided by ArcMap. The number of points per wetland was scaled based on bog area, with the largest bog (7000 m^2) containing 70 random points. The random points were converted to GPX format and imported into a GPS, which was used to identify points for sampling in the field.

At each random point, the abundance of vegetation growth-forms was visually estimated as areal percent coverage within a 0.25 m^2 quadrat. Smaller quadrat sizes are appropriate when collected cover data verses counting plant numbers (U.S. EPA 2002). Total percent values per quadrat exceeded 100% due to overlap. The percent cover recorded in each quadrat was later averaged within each site, resulting in each bog site having only a single percent associated with each growth-form. Growth-forms were defined following a modified version of Chapin et al. (1996). Although his growth-form classification method was specifically designed for low-nutrient, low-diversity tundra vegetation, it has frequently been applied in studies of low-nutrient, low diversity northern bogs to examine the influence of anthropogenic activities on bog vegetation and associated functions (Dorrepaal et al. 2005; Kleinebecker et al. 2010). For this study, growth-forms examined were as defined as follows: Mosses, Lichens, Shrubs, Trees, Graminoids, Forbs, and Ferns. These growth-forms have been shown to predictably change in response to various environmental changes (Jagerbrand et al. 2012; Nishimura and Tsuyuzaki 2015; Hedwall et al. 2017).

3.2.4. Remotely Sensed Vegetation Indices

Vegetation Indices (VI) were calculated from 5m resolution Rapid Eye Imagery of the SJMA taken in late June 2016. Rapid Eye Imagery consists of five spectral bands in the blue, red, blue, green, near-infrared (NIR), and the red-edge (RE) part of the electromagnetic spectrum. The red-edge band has been found to be a particularly important band in the context of wetlands as it is frequently used for wetland delineation, detection, and classification (Mahdavi et al. 2017). The VI are listed in Table 3.2 VI have previously been used to measure vegetation composition, vigor, richness, and physical structure in several ecosystems including wetlands (Loris and Damiano 2006; Rahilly et al. 2012; Comer and Harrower 2013). VI were calculated using the Band Equation function in ArcMap 10.5.1 (ESRI 2017) following the equations listed in Table 3.2. For each vegetation index, the zonal statistic tool in ArcMap 10.5.1 (ESRI 2017) was used to calculate the mean of each vegetation index value for each bog. This resulted in a total of five averaged values for each VI for each bog.

Table 3.2. Vegetation indices (Loris and Damiano 2006; He et al. 2009; Rahilly et al. 2012; Comer and Harrower 2013)

Index	Band Equation
Normalized Difference Vegetation Index (NDVI)	(NIR-RED)/(NIR+RED)
Green NDVI	(NIR-GREEN)/(NIR+GREEN)
Red-Edge NDVI	(NIR-RE)/(NIR+RE)
Simple Ratio	NIR/RED
Red-Edge Simple Ratio	NIR/RE

3.2.5 Statistical Analysis

All statistical analyses were conducted using R studio version 1.1.383 (R Core Team 2017), and associated packages including vegan (Oksanen et al. 2017) and car (Fox and Weisberg 2011).

One-way analysis of variance (ANOVA; R Core Team 2017) was used to assess whether the average cover of individual growth-forms differ amongst urban, pasture, and natural bogs by comparing the mean percent cover of each growth-form between adjacent land-use types. ANOVA results were considered significant at an α less than 0.05. This analysis was visually accompanied by the production of box-plots to view the statistical distribution of the growth-forms amongst bog land-use groups.

Non-metric multidimensional scaling (NMDS; Oksanen et al. 2017) ordination was used to visually assess if bogs with the same adjacent land-use have more similar growth-form composition with one another than to those having different adjacent land-use groups. Using the metaMDS() function, a Bray-Curtis dissimilarity matrix was calculated from the averaged percent-cover growth-form data for each bog site. Before calculating the dissimilarity matrix, metaMDS() performed a Wisconsin double standardization and a square-root transformation on the input data (Oksanen et al. 2017). metaMDS() was run with 9999 permutations at 2 dimensions.

Though NMDS allows for the visual assessment of bog similarity and dissimilarity in terms of growth-form composition, it does not permit statistical comparison of bogs in terms of land-use effects on growth-form composition (Oksanen et al. 2017). Analysis of Similarities (ANOSIM) and Permutational Multivariate Analysis of Variance (PERMANOVA) were used to test if there is a statistically significant difference in growth-form composition amongst pasture, natural, and urban bogs. ANOSIM and PERMANOVA were applied to the Bray-Curtis dissimilarity matrix calculated previously. ANOSIM, carried out using the anosim() function in the vegan package, was used to test the null hypothesis that average rank similarity of growth-form composition between bogs within one land-use group is the same as the rank similarity between bogs of other land-use groups (Oksanen et al. 2017). We used PERMANOVA, carried out using the adonis() function in the vegan package, to test the null hypothesis that the centroids and dispersion values of the growth-form composition of one land-use group are the same amongst all groups (Oksanen et al. 2017). The resulting R2 statistic represents the variation amongst sites that can be explained by the groupings of interest (Oksanen et al. 2017), in this case, adjacent land-use. Both anosim() and adonis() were run with 9999 permutations each. Statistical significance was defined at an $\alpha < 0.05$.

A Mantel test using the mantel () function in the vegan package (Oksanen et al. 2017) was used to test if bogs closer spatially have more similar growth-form composition. Here, we compared the Bray-Curtis dissimilarity amongst bog growth-form composition, previously calculated to produce the ordination, and the Euclidean distances between the geographical co-ordinates of the center of each of the 12 bogs. Spearman's correlation and 9999 permutations were applied.

General linear regressions were used to examine if different growth-forms are related with increases in the percent of urban, natural or pasture land-use present in bog-adjacent landscapes, and at what scales such relationships were present or strongest. As described above, buffers between 100 and 300 m have previously been used to examine the effects of adjacent land-use on isolated wetlands (Lane 2003; Mita et al. 2007). Frequently different types of vegetation are most influenced at different scales using only one scale may result in missing important relationships between growth-form and land-use. Such methods have been applied in other studies examining the scale at which the presence or abundance of some vegetation is most correlated with proportions of various landscape metrics (Hychka et al. 2007; Weller et al. 2007). Here, the percent of urban, pasture, and natural land-use were calculated from the CI land cover data at scales of 100m, 200m, 300m, 400m, and 500m from the bog edge using buffer analysis performed using ArcMap version 10.5.1 (ESRI 2017). Spearman's correlations of the land-use quantities were carried out with arcsine-transformed growth-form percentages (to improve the normality of the distribution of residuals and reduce heteroscedasticity) measured in each bog. Both the direction and value of the correlations were reported and were considered significant when $\alpha<0.05$.

A Mantel test using the mantel () function in the vegan package (Oksanen et al. 2017) was used to determine if dissimilarities of bog growth-form composition correlate with dissimilarities of bog vegetation index mean values, for each vegetation index listed in Table 3.2 To do this, a Euclidean distance matrix was calculated for each vegetation index using the mean values of each bog. For example, a Euclidean distance matrix for mean NDVI reports on the distance between the mean NDVI at each bog site from the mean NDVI at each other bog site. Mantel tests were performed using each Euclidean vegetation index distance matrix and the Bray-Curtis dissimilarity matrix calculated based on each bog's growth-form composition.

3.3 Results

Figure 3.2 shows the growth-form composition of each bog. All growth-forms were present in most wetlands. Lichen growth-forms that were sampled in all but two wetlands (U1 and U3). Tree growth-forms were present in all but two wetlands (N1 and P4), and fern growth-forms were present in five of the twelve wetlands (U1, U3, U4, N4, and P3; Figure 3.2). Table 3.3 lists all the species encountered in this research and their associated growth-form.

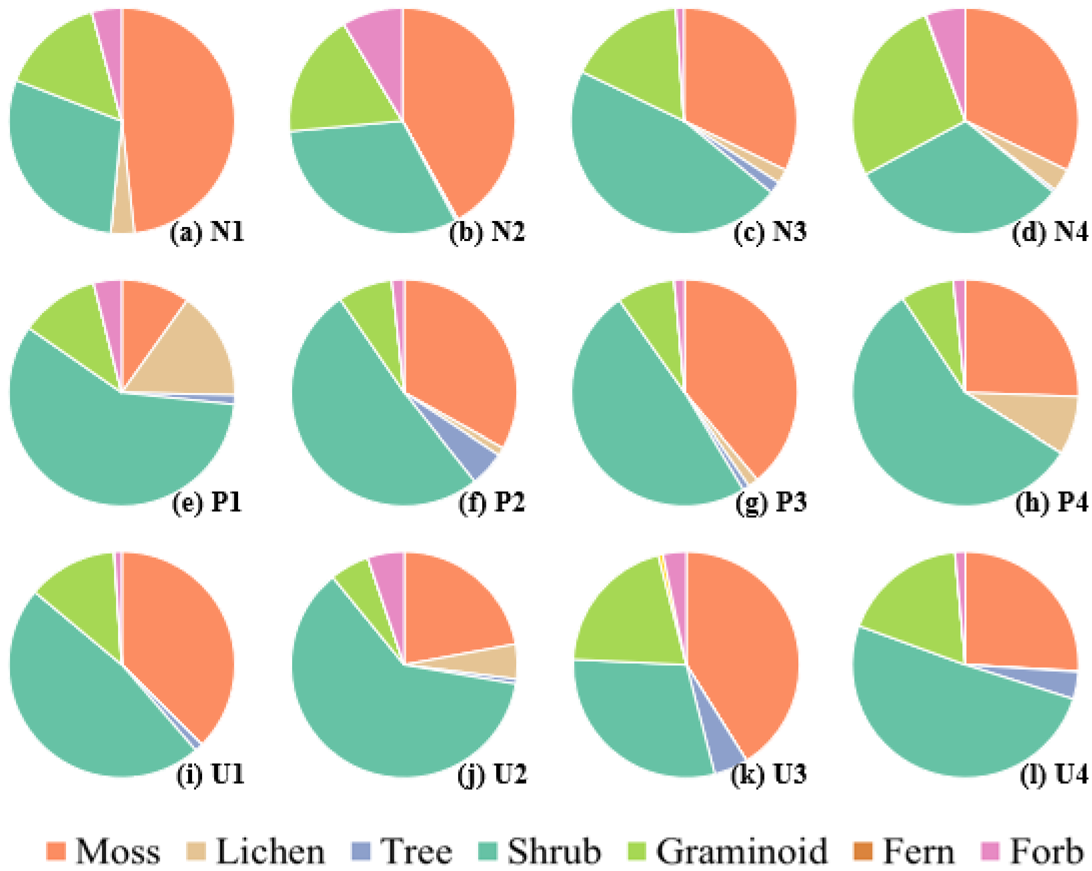

Figure 3.2. Percent cover of growth-forms in each of the 12 study site bogs, arranged by adjacent land-use dominance: natural bogs (a-d), pasture bogs (e-h), and urban bogs (i-l).

Table 3.3. Vegetation species identified across the 12 bogs and their associated growth-forms.

Growth-Form	Common Name	Scientific Name
Fern	Ferns	
Forb	Bake-Apple	*Rubus chamaemorus*
Forb	Bog Aster	*Oclemena nemoralis*
Forb	Bog Goldenrod	*Solidago uliginosa*
Forb	Bunchberry	*Cornus canadensis*
Forb	Canadian Burnet	*Sanguisorba canadensis*
Forb	Canadian Mayflower	*Maianthemum canadense*
Forb	Pitcher Plant	*Sarracenia Spp.*
Forb	Sundew	*Drosera Spp.*
Forb	White Bog Orchid	*Platanthera dilatata*
Forb	Other Forb	
Graminoid	*Carex* Sedges	*Carex Spp.*
Graminoid	Cotton Grass	*Eriophorum Spp.*
Graminoid	Other Grasses	Poaceae
Graminoid	Other Sedges	Cyperaceae
Lichen	Reindeer Lichen	*Cladonia Spp.*
Lichen	Other Lichen	
Moss	*Sphagnum* Moss	*Sphagnum Spp.*
Shrub	Black Crowberry	*Empetrum nigrum*
Shrub	Blueberry	*Vaccinium angustifolium*
Shrub	Bog Laurel	*Kalmia polifolia*
Shrub	Bog Rosemary	*Andromeda polifolia*
Shrub	Chokeberry	*Aronia Spp.*
Shrub	Chuckley Pear	*Amelanchier Spp.*
Shrub	Common Juniper	*Juniperus communis*
Shrub	Huckleberry	*Gaylussacia Spp.*
Shrub	Labrador Tea	*Rhododendron groenlandicum*
Shrub	Leatherleaf	*Chamaedaphne calyculata*
Shrub	Meadow Sweet	*Spiraea latifolia*
Shrub	Mountain Holly	*Ilex mucronata*
Shrub	Other Shrub	
Shrub	Partridge Berry	*Vaccinium vitis-idaea*
Shrub	Pincherry	*Prunus pensylvanica*
Shrub	Rose	*Rosa* Spp.
Shrub	Sheep Laurel	*Kalmia augustifolia*
Shrub	Small Cranberry	*Vaccinium oxycoccos*
Shrub	Sweet Gale	*Myrica gale*
Tree	Balsam Fir	*Abies balsamea*
Tree	Black Spruce	*Picea mariana*
Tree	Larch	*Larix laricina*

Visual assessment of the box-plots indicated notable overlap in the statistical distribution of the abundance of most growth-forms for the three land-use categories (Figure 3.3). This observation was supported by ANOVA results, showing that only the difference in mean abundance of the graminoid growth-form in natural and pasture wetlands was statistically significant at α of 0.05, with a p-value of 0.00916 (Table 3.4).

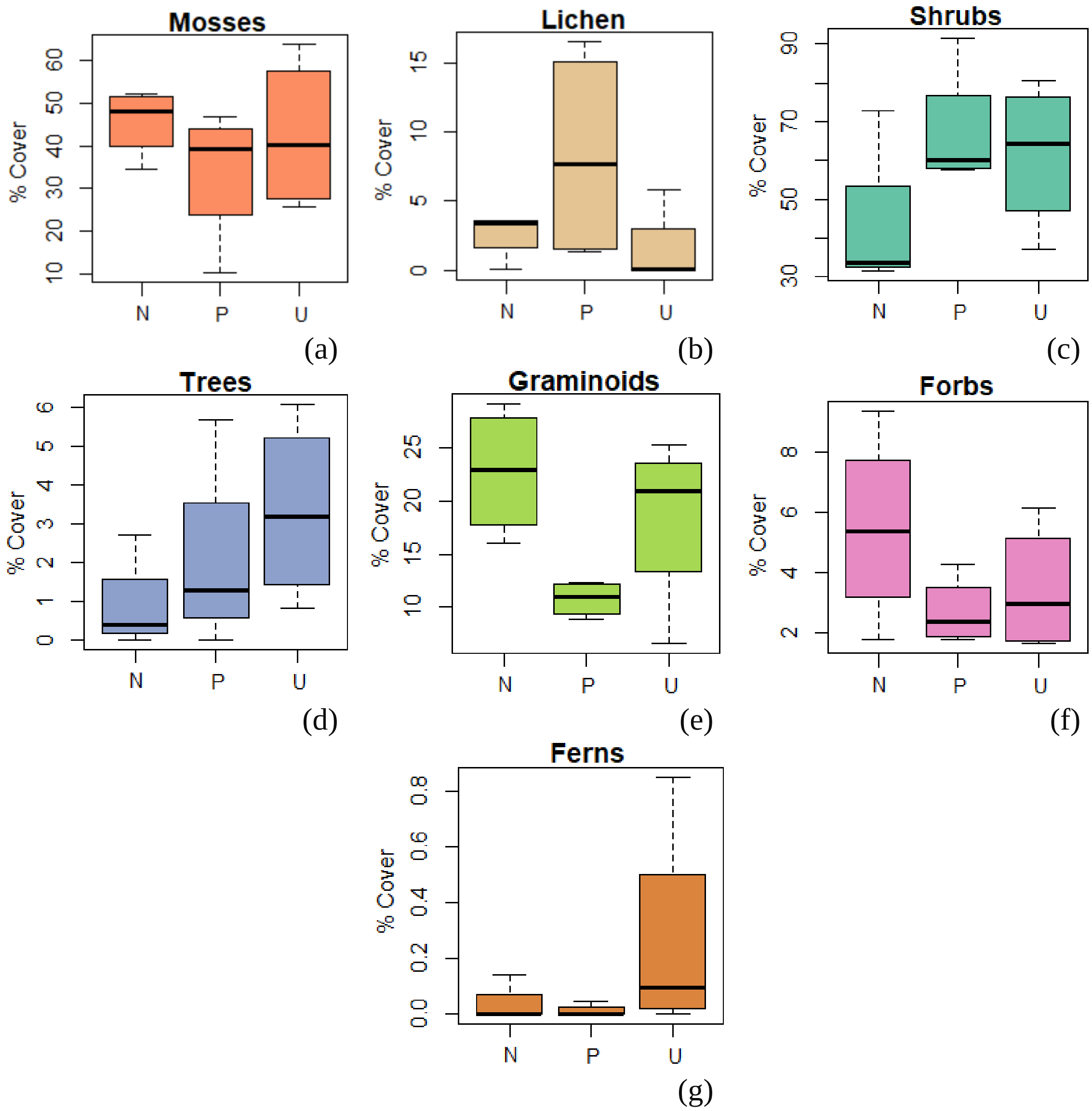

Figure 3.3. Boxplots comparing the average percent-coverage per land-use class for each growth-form: (a) mosses, (b) lichens, (c) shrubs, (d) trees, (e) graminoids, (f) forbs, and (g) ferns. The x-axis displays the land-use categories where N=Natural, P=Pasture, and U=Urban. The y-axis displays the average percent cover of growth-forms for each grouping of bog sites.

Table 3.4. Results of ANOVA statistical analyses comparing growth-abundance means across land-use types. * indicates p-value significant at an $\alpha < 0.05$.

Growth-Form	Urban vs. Natural	Urban vs. Pasture	Pasture vs. Natural
Moss	0.758	0.508	0.240
Lichen	0.533	0.157	0.200
Tree	0.117	0.494	0.424
Shrub	0.225	0.664	0.107
Graminoid	0.435	0.116	0.009*
Forb	0.327	0.569	0.150
Fern	0.311	0.26	0.535

A two-dimensional NMDS was produced with a final stress of 0.0839, represented in the Shepard plot in Figure 3.4. The Shepard's plot visualizes the disagreement (here described as stress) between the visualized NMDS ordination dissimilarities (Figure 3.5) and the original dissimilarities (Oksanen et al. 2017). 0.0839 is an acceptable stress level (Clarke 1993). The NMDS ordination plot (Figure 3.5) shows the sites loosely grouped into their *a-priori* land-use groups, though no grouping of sites is tightly concentrated and individual sites of some groups are closer to individual sites of other groups. Generally, the natural sites grouped near the top of the ordination, the urban sites grouped to the left and the pasture sites grouped to the lower right. Based on the general pattern of the visual assessment of Figure 3.5 the bogs in this study that have similar adjacent land-use are somewhat alike in terms of growth-form composition. Bogs in this study with different adjacent land-use are somewhat different in terms of growth-form composition.

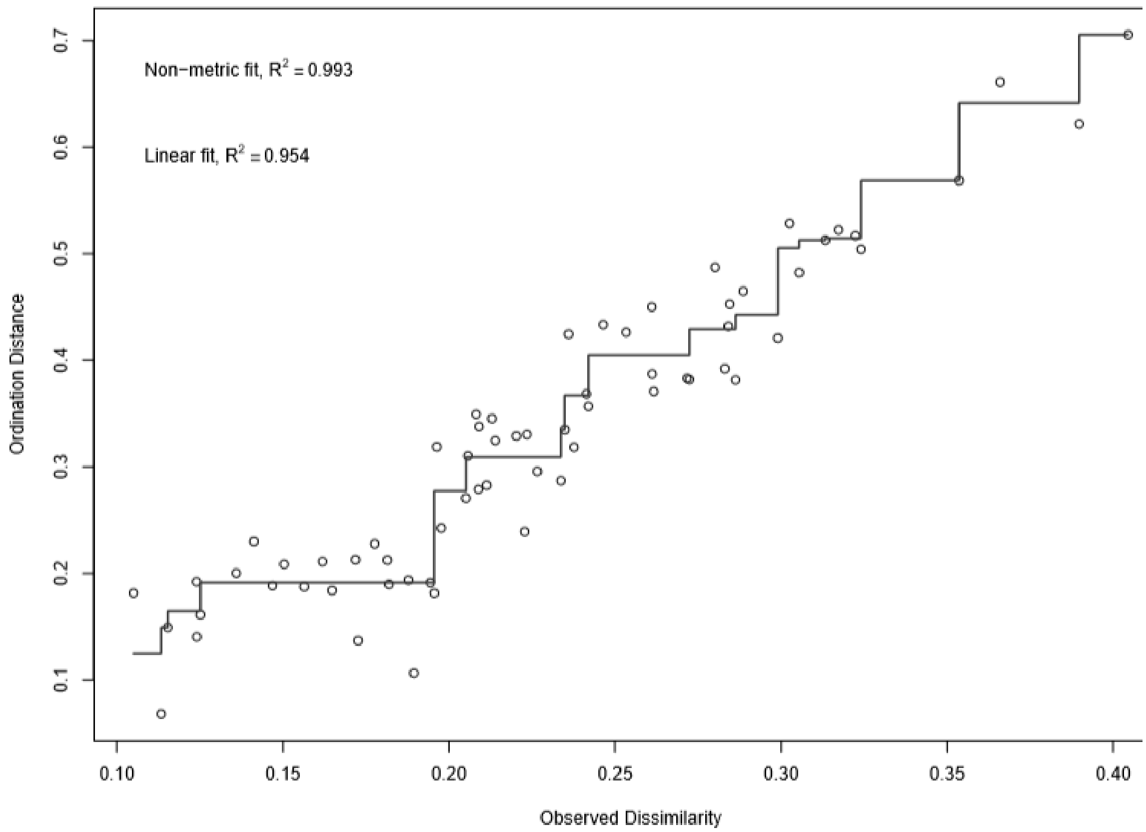

Figure 3.4. NMDS Shepard plot of bog growth-form composition showing the relationship between ordination dissimilarities and original dissimilarities.

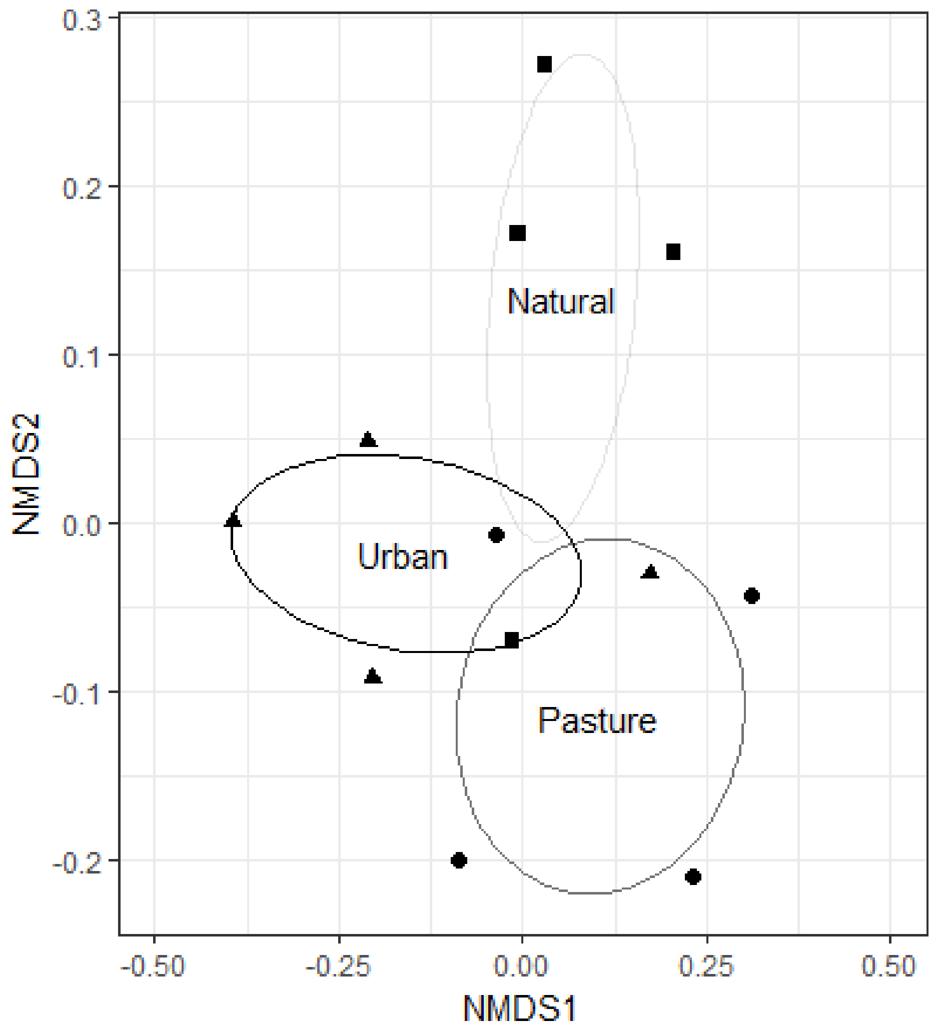

Figure 3.5. NMDS ordination plot for the sample sites growth-form composition, where squares represent natural wetlands, triangles represent urban wetlands and circles represent pasture wetlands. The ovals are shaded differently depending on land-use and represent 95% confidence intervals for each group of bogs based on land cover.

ANOSIM and PERMANOVA results are summarized in Table 3.5. Both statistical analyses were significant ($\alpha < 0.05$), with p-values of 0.027 and 0.034 respectively, rejecting the null hypotheses of both statistics. Specifically, the ANOSIM null hypothesis can be rejected, implying that rank similarities within the land-use groups are not statistically different to the rank similarities amongst land-use groups. Similarly, the PERMANOVA null hypothesis can be rejected, implying that the centroid and dispersion of groups in the ordination space are not statistically different. Additionally, the Mantel test was not significant at $\alpha<0.05$ level (p=0.6), implying an absence of relationship between the growth-form dissimilarity and geographical distance.

Table 4.5. Results of ANOSIM and PERMANOVA statistical analyses. * indicates p-value significant at an α of 0.05.

Name	R (R^2 for PERMANOVA)	p-value
ANOSIM	0.287	0.027*
PERMANOVA	0.3429	0.034*

ANOSIM compares the mean of the ranked dissimilarities between land-use groups to the mean of the ranked dissimilarities within land-use groups. Here, the ANOSIM R was 0.287, which indicates that dissimilarity is greater between land-use groups than within, although this dissimilarity is not strong. This reflects what is seen in Figure 3.5, where the sites of the same groups are loosely grouped together but form no concentrated clustered. For PERMANOVA, the resulting R^2 for the land-use groupings was 0.34. This can be interpreted as 34% of the variation in bog growth-form composition can be explained by the land-use groupings. This falls in line with both the ordination (Figure 3.5) and ANOSIM results.

In summary, the multivariate statistical tests indicate that different adjacent land-use groups (bog, pasture, and natural) have influence on the growth-form composition of bog wetlands. Based on visual ordination, the natural bogs sites formed the most distinctive grouping in terms of land-use class but were not tightly clustered. Additionally, the results of the Mantel test indicate that spatial autocorrelation is not an issue.

Of all the correlations tested, only six were significant (bolded values in Table 3.6). Based on these results, the percent cover of graminoids in a bog was negatively correlated with the percent of pasture land-use present at all scales. The variance in the percent graminoid cover explained by percent pasture ranged between -0.59 and -0.62, which represents a moderate to strong negative relationship. The percent of shrub present in a bog was negatively correlated with the percent of natural land-use present at scales of 500m. The variance in the percent shrub explained by the percent natural land-use in adjacent landscapes is -0.60.

Table 3.6. Univariate correlations of growth-forms with potential landscape correlates at five different scales. Bolded numbers are significant (α <0.05).

Scale		Land-Use	%Moss	%Lichen	%Tree	%Shrub	%Graminoid	%Forb	%Fern
Scale	*100m*	**% Urban**	-0.15	-0.2638	0.54	0.38	-0.34	-0.30	0.34
		% Pasture	-0.28	0.4168	-0.06	0.35	**-0.59**	-0.22	-0.21
		% Natural	0.16	0.2068	-0.47	-0.40	0.31	0.30	-0.38
	200m	**% Urban**	-0.12	-0.1411	0.34	0.44	-0.35	-0.32	0.24
		% Pasture	-0.29	0.3668	0.01	0.30	**-0.62**	-0.22	-0.21
		% Natural	0.11	0.0776	-0.25	-0.40	0.48	0.37	-0.33
	300m	**% Urban**	-0.22	-0.1509	0.35	0.37	-0.29	-0.34	0.35
		% Pasture	-0.29	0.3668	0.01	0.30	**-0.62**	-0.22	-0.21
		% Natural	0.24	-0.0807	-0.08	-0.46	0.51	0.33	-0.25
	400m	**% Urban**	-0.08	-0.18	0.31	0.41	-0.27	-0.30	0.45
		% Pasture	-0.29	0.37	0.01	0.30	**-0.62**	-0.22	-0.21
		% Natural	0.14	-0.08	-0.05	-0.53	0.48	0.26	-0.27
	500m	**% Urban**	-0.08	-0.05	0.20	0.54	-0.30	-0.30	0.35
		% Pasture	-0.18	0.367	0.12	0.39	**-0.62**	-0.34	-0.27
		% Natural	0.08	-0.07	-0.06	**-0.60**	0.40	0.33	-0.28

The Mantel test results of the relationship between the means of VI and bog growth-form composition showed a significant positive correlation between the growth-form dissimilarity matrix, the mean RE NDVI distance matrix, and the mean RE Simple Ratio, as seen in Table 3.7, both with a Mantel statistic r of ~0.52. Significant results were not found for the Green NDVI, the Simple Ratio, and the NDVI VI and as a result cannot be used to estimate growth-form composition in bogs.

Table 5.7. Mantel test of correlation between bog growth-form dissimilarity matrix and mean vegetation index distance matrices.

Band/Index	Statistic	Mantel Statistic r	Significance
Green NDVI	Mean	0.1726	0.151
Red Edge Simple Ratio	Mean	0.5244	0.002*
Red Edge NDVI	Mean	0.5172	0.001*
Simple Ratio	Mean	0.2691	0.056
NDVI	Mean	0.2105	0.101

3.4 Discussion

Multivariate analyses of growth-form composition provided some evidence to support our hypothesis that growth-form composition in bogs was influenced by the presence of urban, pasture, or natural land-use in adjacent landscapes. Though this research does not examine specific causes of change, land-use such as urban, pasture and natural act as a generalized representation, or surrogate qualification, of numerous environmental influences that have the potential to influence bog growth-form vegetation composition such as land clearing (Dean et al. 2015; Adane et al. 2018; NLDFL Agriculture and Lands Branch 2017), increased impervious cover (Ausseil et al. 2011; Patenaude et al. 2015), cattle grazing (and various air-borne pollutants ;Zaman et al. 2007; NLDFL 2018) which in turn can influence local water tables and bog hydrology and nutrient and other chemical deposition. It is unlikely that the bog vegetation composition in this study are influenced by only one type (hydrology alteration, air-borne pollutants) of land-use disturbance, but instead by multiple of these anthropogenic changes interacting, including some broader scale anthropogenic changes not considered here such as increasing temperatures and rates of precipitation (Finnis and Daraio 2018).

In the SJMA, much of the urban land-use is low density residential with some medium and high density containing assisted living complexes, apartment buildings, and duplexes (City of St. John's 2014b). The number of households in the SJMA has increased relatively steadily since 1986 and as of 2013 there were around 83, 00 homes (City of St.John's 2014a). Projections predict that there will be between 93, 530 to 103, 657 homes by 2036 (City of St.John's 2014a). Urban sprawl in the area contributes to high vehicle use in the area (CBC News 2012; The Telegram 2013). Pastures are used for hay, cattle, horse, and sheep and goat faming (Statistics Canada 2016; NLDFL Agriculture and Lands Branch 2017). As of 2016, an estimated 37 farms are used for the farming and ranching of farm animals (Statistics Canada 2016) and an estimated 13 farms are used for hay production. These numbers are down for 2011 (Statistics Canada 2016), however there are currently initiatives to increase agricultural production, with emphasis on making available more land to farmers, redeveloping historic pasture lands, and identify new areas for potential agricultural development (NLDFL 2019). There has also been an increase in sod farming in the area that correlates with increasing population and development, and as of 2010 there was an estimated 280 acres of sod farm in the Northeast Avalon (Northeast Avalon ACAP 2014).

Urban and pasture land-use require land-clearing and the removal of vegetated areas. Characteristics of densely vegetated areas allows for not only filter air-borne pollutants but the alteration of local ground-water recharge and water table levels via transpiration and water interception (Dean et al. 2015). When vegetated lands are cleared for pasture or crop-land, it has been shown that local water tables have the potential to rise as much more water is able to seep into the ground (Bekele et al. 2007). Urban land-use, on the other hand, is usually associated with impervious cover (Ausseil et al. 2011; Patenaude et al. 2015) which not only intercepts rainfall (and thus potentially lowering local water table), but re-directs water flow which in turn, can influence growth-form composition. Bog vegetation, which typically have adaptations to an ombrotrophic environment (National Wetlands Working Group 1997; Goud et al. 2018), can be altered when this hydrology is disrupted by increases or decreases in water table levels (Murphy et al. 2009; Potvin et al. 2015). Lowered water tables in bogs can result in the increased growth of woody vegetation, which in turn can lower water tables even further (Breeuwer et al. 2009; Potvin et al. 2015). Increased water tables have resulted in shifts of bog vegetation composition to greater moss presence, decreased shrub presence, and increased graminoid presence (Weltzin et al. 2000).

Pasture and urban land-use are not only associated with hydrological alterations but are also the source of various air-borne pollutants (Bignal et al. 2007; Ausseil et al. 2011). As discussed, previous, bogs are typically ombrotrophic and vegetation is adapted to low nutrients and air-borne pollutants and associated deposition have the potential to throw off the natural low-nutrient environment and create new competition amongst vegetation

(Limpens et al. 2003; Schrijver et al. 2011). Pasture and agriculture, for example, is usually a major source of nitrogen due to fertilization. In NL specifically, pastures are usually fertilized over the soil (NLDFL 2018). Urban land-use, on the other hand, is a source of not only nitrogen, but heavy metals and other contaminants from automobiles (Bignal et al. 2007; Pescott et al. 2015). Lichens have been shown to be sensitive to urban pollutants and may decrease within range of urban areas or highways (Munzi et al. 2014; Koch et al. 2016). Mosses have also been shown to be sensitive to urban-associated pollutants. It is likely that the differences seen in the vegetation composition amongst the natural, pasture, and urban bogs (see Figure 3.5) are a result, at least in part, of a the impacts of land clearing and pollutants typical of developed landscapes on growth-forms, and the interactions within the bog amongst the growth-forms themselves.

Analyses of the individual growth-forms grouped by land-use type supported the multivariate results, in that they indicated extensive overlap in the cover of most growth-forms across land-use types. Only graminoid growth-form cover in pasture bogs differed significantly from graminoid growth-form in natural bogs. Graminoid cover was lower both in mean cover and in range in pasture bogs, versus the greater range of graminoid cover in natural bogs and the overall greater mean (see Figure 3.3). Similarly, the linear correlations at various scales are only significant for graminoid growth-forms (see Table 3.6). All scales (100m, 200m, 300m, 400m, and 500m) the percent cover of graminoid growth-forms decreased with increased pasture land-cover in adjacent landscapes. Based on these results, we suggest that adjacent pasture land-use may have an impact on the composition of bog growth-form vegetation by decreasing the presence of graminoid growth-forms.

In the context of this study, graminoids are described as narrow-leafed vascular plants (Chapin et a. 1996). Generally, graminoids are typically more prominent in fens than in bogs due to the minerotrophic status of fen wetlands (National Wetlands Working Group 1997). Minerotrophic wetlands have a greater input of nutrients including nitrogen, in which growth-forms such as graminoids competitively thrive and mosses do not (Kleinebecker et al. 2010). In bogs, tussock species of sedge (such as *Carex* spp.) are most common (National Wetlands Working Group 1997). Various environmental changes have been shown to influence the presence of graminoid growth-forms bogs, some of which may be the result of adjacent pasture land-use including water table fluctuations (Weltzin et al. 2000; Riuttaetal et al. 2007; Breeuwer et al. 2009) nitrogen influx (Kool and Heijmans 2009; Kleinebecker et al. 2010) and animal-use (grazing and trampling) (deBlois and Bochard 1995; Cole 1995; Welch 1984; Cabezas et al. 2015).

Grazing animals using bog-adjacent pastures have the potential to enter the near-by bogs and modify the environment via grazing, trampling, and fecal deposition. In the SJMA, pastures are used for the farming of various grazing animals including cattle, sheep, goats and horses (Statistics Canada 2016; NLDFL Agriculture and Lands Branch 2017). Not only does some grazing have the potential to decrease overall graminoid cover but trampling by animals has been shown to modify the hydrology of habitats in such a way that can alter plant composition (Cole 1995; Poullot et al. 2011). When bog water tables remain relatively constant in elevation as would be the case in more natural bogs, greater graminoid cover is maintained (Breeuwer et al. 2009). However, when water tables in bogs fluctuate, as could be the result of trampling, graminoid cover in bogs has been shown to decrease (Weltzin et al. 2000; Weltzin et al. 2003). Similarly, when increased temperatures interact with altered bog water table, graminoid cover was found to decrease (Weltzin et al. 2000; Weltzin et al. 2003). Other aspects of pasture land-use such as land-clearing can impact bog water tables and alter graminoid presence.

Graminoid presence in bogs has also been shown to be influenced by changes in nitrogen levels, and it is likely that bogs near-by pasture land-use experience greater influx of nitrogen. However, increased nitrogen typically increases the growth of graminoid growth-forms and shrubs (Gerdol et al. 2007) in contrast to the decrease as seen in the results of this study. However, at higher nitrogen levels, other vascular plants such as shrubs tend to out-compete graminoids concluding that bogs under higher nitrogen availability are more likely to become dominated by shrubs over graminoids (Kool and Heijmans 2009). In studies of bog growth-forms, graminoids and shrubs often show opposite reactions to environmental change (Ward et al. 2013; Kuiper et al. 2014; Robroek et al. 2015). In the SJMA, it appears that the average cover of shrubs in pasture bogs is highest compared to that of urban and natural bogs, and the average cover of graminoids in pasture bogs is lowest compared to urban and natural bogs.

Graminoids contribute to various bog and peatland functions, including carbon dioxide sink and source potential, methane production potential and primary production (Malmer et al. 2005; Churchill et al. 2015; Strack 2017). These relationships have been demonstrated in numerous experimental studies, where graminoid growth-forms have been removed from peatlands. For example, graminoid removal has resulted in lowered potential methane production versus that of shrub removal (Robroek et al. 2015). In another removal study (Ward et al. 2013), peatland carbon dioxide sink strength was greatest under warming conditions when shrubs were present and weakest when graminoids were present, and graminoid presence contributed most to methane emission.

Though not statistically different, shrub presence appears to be lower in natural bogs in comparison to urban and pasture bogs. This can be seen in Figure 3.3. This is consistent with some of the literature on anthropogenically impacted bogs, where woody plant cover tends to be much higher. For example, Pasquet et al. (2015) examined bogs isolated in agriculturally developed landscapes which caused the drainage of the surrounding catchment. This drainage was attributed partially as the cause of an increase in trees in the bogs. Similarly, Sensbusch (2015) found an increased tree and shrub presence in a bog with an adjacent roadway, concluding that the roadway diverted some water flow, allowing for the growth of the woody forms. It seems that a higher presence of woody growth-forms in bogs may be indicative of anthropogenic influence.

The results of the VI analysis show a significant positive correlation between the bog growth-form dissimilarity matrix and the RENDVI and RE Simple Ratio distance matrices, suggesting their potential for the assessment of changes in bog vegetation composition as a result of adjacent land-use influence. Notably, only those VI that included the RE band were significant, which is unsurprising given the usefulness of this band in multiple wetland studies (Mahdavi et al. 2017). Additionally, applications of the RE has been found to discern between different types of vegetation including that of peatlands

(Bubier et al. 1997; Radoux et al. 2016; Arroyo-Mora 2018). This study applies the use of 5m resolution rapid eye data, which is not free and as a result perhaps not available for use in certain applications. However, Sentinal-2 data is freely available and provides access to the red-edge band, though at a coarser resolution of 20m. Recent Sentinal-2 data has the potential for assessment of vegetation in peatlands (Radoux et al. 2016; Arroyo-Mora 2018). Freely available access to the RE band will allow for better assessment of wetlands.

Some considerations should be made when considering the results. Bogs have been found in other studies to be relatively resilient to influences caused by anthropogenic change (Jasieniuk and Johnson 1982; Gunnarsson et al. 2000). Studies showing changes in bog growth-forms due to adjacent land-use tend to be conducted in areas where there primarily one type of land-use. For example, Pellerin et al. (2008) found that bog wetlands isolated in agricultural landscapes were encroached by woody vegetation. This landscape has an extremely high agricultural presence and the SJMA, though being the most populated and urbanized region in NL, does not have as dense an agricultural presence, nor the highly-urbanized city scape of other locations. The four natural landscape bogs were the only class of sites that were completely dominated by a single adjacent land-use group. The urban and pasture sites were bordered by mixed land-use compositions, commonly in conjunction with natural land-use. An additional constraint is that only four bog sites were examined in each of the three generalized land-use classes. Additionally, this study only considers basin bogs, though there are three other common bog formations in Eastern Avalon including domed, blanket and slope (Wells 1981), and the results may not be consistent with other bog forms given differences in surface formation and connection to the surrounding landscape (National Wetlands Group 1997). Finally, Newfoundland is highly diverse in terms of climate, containing nine ecoregions (South 1983) and three wetland regions (National Wetlands Working Group; 1986) and as a result, bogs in the SJMA are not representative of bogs at a provincial level.

These conditions should be considered when interpreting the results of this study, and as a result, the findings of this research cannot justify implementing restrictions on bog-adjacent landscapes at provincial scale. However, the findings do emphasize a need for wetland developmental studies to be locally distinct. For example, urban-related development is not a significant problem for most of the province outside of the SJMA. Additionally, the ways in which wetlands and wetland vegetation composition will be impacted by anthropogenic land-use will differ due to the variability of climate across the province (South 1983). Further research may obtain a clearer picture of land-use, growth-form relationships by increasing the number of sites examined selecting sites with more consistent adjacent anthropogenic land-use, examining bogs wetlands of different forms, and examining bogs in different ecoregions across the province. Additionally, further studies could define more specific adjacent land-use that may produce more accurate results. Several studies on the effects of adjacent land-scape on wetlands have employed the Land Development Intensity Index (LDI; Brown and Vivas 2005). The LDI, defines the intensity of land-use through calculations of energy use-per unit area where land-use such as an industrialized area will have a higher LDI coefficient than a natural grassland (Brown and Vivas 2005; Mack 2006; Reiss et al. 2010; Stapanian et al. 2016). Future studies in NL may use LDI as an estimate of anthropogenic activity, though the LDI would have to be modified to consider anthropogenic activity typical of NL.

By finding correlations between bog vegetation patterns and adjacent land-scape features, it may be possible to develop a suite of easily obtained and cost-effect remotely-sensed metrics that can be used to quantify at a large scale the overall condition of wetlands NL, identify wetlands under stress from adjacent land-use, and infer the impact of land-use on the quality of functions or services provided (Stuber et al. 2016). The Environmental Protection Agency (EPA) has a three-tier process for monitoring and assessing wetland condition (Nestlerode et al. 2014), which includes the use of land-scape metrics such as land-use as a primary assessment tool for identifying wetlands under threat. Adjacent land-use metrics are also used to develop Indices of Biotic Integrity (Karr and Chu 1999). IBIs have been used to assess the condition of many ecosystems around the world, including wetlands (Miller et al. 2006). It has been recommended that metrics such as those used by the EPA or by IBIs should be developed specifically for the region in which they will be applied. Currently, NL has yet to develop such metrics for wetlands of any type. Given the results of this research, there remains a possibility that growth-forms (particularly graminoid) may have the potential to act as an indicator of condition.

There also exists the possibility that adjacent land-use does not have a dramatic effect on bog growth-form composition. As mentioned previously, bogs have been shown to have significant resilience to anthropogenic changes (Jasieniuk and Johnson 1982; Gunnarsson et al. 2000). It may only be in extreme cases that adjacent land-use can cause major shifts in bog growth-form composition (Pellerin et al. 2016). However, under the effects anthropogenic climate change, and the growing evidence that a changing climate may alter plant responses in bog wetlands (Heijmans et al. 2008; Breeuwer et al. 2009; Buttler et al. 2015), it may be possible that the interacting effects of both anthropogenic global warming, and adjacent land-use will result in more dramatic changes in bog vegetation (Sengbusch 2015) in the future. In a place such as NL where the wetland policy is lenient and where there is potential for greater anthropogenic development in the form of a growing population, and increased housing and agricultural development, which all have the potential to result in altering wetlands and the services those wetlands provide.

3.5 References

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CHAPTER 4: CONCLUSION

Landscape-scale anthropogenic activity exerts influence over the typical bog ecosystems (growth-form composition) which can in turn result in the alteration of typical bog functioning and therefore the provision of bog wetland services. Protection and preservation of wetland functions and services is often the goal of provincial and federal wetland policies in Canada (Environment of Canada 1991; NLDME 2018), however, these polices rarely place restrictions on anthropogenic activity in wetland-adjacent landscapes. Additionally, these policies are often not optimally designed for bogs (Poulin et al. 2004). The lack of priority for the protection of bogs may be the result of several reasons including: their extent in the Canadian landscape (Poulin et al. 2004), their reputation as being stable ecosystems (Jasieniuk and Johnson 1982; Gunnarsson et al. 2000), the lack of faunal species of interest for conservation compared to other wetland types, the general public perception that bogs have limited value, and the lack of obvious pollutant inputs via water sources. However, this study has shown that bogs susceptible factors with the potential to impact important bog-related services such as carbon storage and ground-water management (Robroek et al. 2010; Ward et al. 2013; Robroek et al. 2015). As such, policies that do not implement restrictions on bog-adjacent landscapes will be less effective at protecting and preserving bog ecosystems and ecosystem services. Bog protection may not currently be viewed as pertinent by policy-makers under restrictive time and budgets. However, these issues may become exacerbated in a future of growing populations, expanding human-related land-use, and climate change.

The results of this study suggest that even in less densely population landscapes such as the SJMA, at least relative to other Canadian cities (for example the population density in the SJMA is 255.9 people per square kilometer verses the Greater Tornto area which has a population density of 1,003.8 people per square kilometer; Statistics Canada 2017), generalized landscapes such as urban and pasture have the potential to alter bog growth-form composition from that of bogs in less developed landscapes. It can be predicted that changes to bog form composition amongst bogs in the SJMA will continue given that: the PDW places no restrictions on development in wetland adjacent landscapes (NLDME 2018), the PDW is more lenient on allowing development of peatlands specifically (NLDME 2018), populations are predicted to increase over the next 15 years (Simms and Ward 2017), housing development is predicted to increase (City of St. Johns 2014), and initiatives are being made to increase agricultural activity across the province (NLDFL 2019).

Additionally, these changes in vegetation composition may be exacerbated given climate change (Evardsson et al. 2015). Future climate change projections predict that by mid-century in the SJMA, precipitation will increase seasonally by around 1mm (Finnis and Daraio 2018), the summer and winter temperatures will increase from 14° Celsius and -4° Celsius to 16.5°Celcisus and 0.2°Celcisus respectively (Finnis and Daraio 2018), and the growing season will increase by around 10 days from a current range of 140-150 days (Finnis and Daraio 2018; Government of Canada Agriculture and Agri-Food Canada 2019). For example, increased levels of nitrogen can interact with increases in temperature to impact bog vegetation and in turn, associated levels of decomposition (Breeuwer et al.

2008; Hedwall et al. 2017), which has implications on bog carbon storage capabilities. Water table alteration and increased temperatures may have similar effects (Buiber et al. 2003; Potvin et al. 2015).

4.1 Study Considerations

While the results of this research provide significant results for the influence of adjacent land-use on basin bog wetlands in the SJMA, there are several considerations that can be made with regards to: the inherent characteristics of bog wetlands (*bog stability*), temporal issues (*when were things built, lag in the impacts of different anthropogenic land-use*), limitations due to study area (*quality of adjacent land-scape and population density*), bog form, the diversity of NL regions, and study design (*site selection, vegetation sampling methods*, *total study sites*). The specifics of these considerations will be discussed below.

Bog Stability: Many older studies have found that bogs (and other *Sphagnum*-dominated ecosystems) exhibit stability and resistance to changes in vegetation, with some showing constant stability over of tens to thousands of years even in the context of anthropogenic and natural disturbance (Backe´us 1972; Gunnarsson et al. 2000; Rydin & Barber 2001). This stability is largely attributed to the low-nutrient content and associated low rate of production that is typical of these ecosystems. Other studies have shown that bog vegetation may be resistant to inputs of air pollution if all other bog features are held constant, such as water level (Hájkova and Hajek 2004), suggesting that bogs may require multiple environmental changes to display vegetation shifts, and that these changes may happen slowly. Similar studies of non-isolated (and non-ombrotrophic) wetlands and landscapes often show dramatic shifts in vegetation due to pollutants and nutrients being carried directly into the wetland from anthropogenic areas via surface and ground-water flow (Craft et al. 2007; Reiss et al. 2010). Such high influx of pollutants into bogs is unlikely due to the ombrotrophic nature of these ecosystems. Generally, studies that show substantial shifts in bog vegetation composition are under direct and perhaps unrealistic or uncommon levels of environmental change, as is the case in direct nutrient addition experiments (Limpens et al. 2003; Nishimura and Tsuyuzaki 2015) or bogs studied in the context of extremely degraded landscapes (Pasquet e al. 2015). Ambient and subtle changes in the environment may not be substantial enough to cause dramatic shifts in bog vegetation, particularly at the level of growth-form. Finally, the natural microtopography of bogs, including the characteristics of hummocks, hollows and lawns may help to support vegetation stability.

Temporal Issues: It is possible that there is time-lag in the response of bog vegetation to landscape change (Kapfer et al. 2011; Evans et al. 2014). In the context of this study, it was not considered when certain adjacent land-use developments were established, and it is likely that different areas were developed and built-upon at different times. This issue of time-lag may have played a role in the ordination results of this study. For example, one of the urban bogs is in a neighborhood (locally known as the Airport Heights area) is more recently developed compared to that of the other neighborhoods in the study. Of note, this bog is the one urban bog farthest from the other urban bogs (and closest to the pasture wetlands) in the ordination shown in Figure 3.5, and this may potentially be attributed to the differences in the timing of the development of the areas. These temporal differences in environmental change may contribute to in-group variation and the spread of the ordination results. Future studies may consider comparing topographic maps of the study area prior to selecting study sites. However, there are likely few bogs present in the oldest neighborhoods of the SJMA such as the downtown area, as these neighborhoods are more developed compared to that of newer neighborhoods.

Study Area Restrictions: As has been mentioned, many studies of the impact of landscape on bogs occur in areas where there is a very high level of anthropogenic activity, for example bogs isolated fully in a dense agricultural area (Pasquet e al. 2015), a level of which is not present within the SJMA study area. Other studies are carried out where there are highly polluting industries (Vellak et al. 2014), which again is not present in the study area. Though the SJMA is the most populated and developed urban centre in NL, it remains that compared to that of other provinces, NL has a low population density and still maintains many natural and undeveloped locations. While this is beneficial because many bogs remain in proximity to various types of land-use, it also means that most bogs still have much “untouched” or natural land-use in their proximity, and often maintain some natural lagg. Bog laggs have been shown to be important in protected bogs from adjacent land-use, filtering out pollutants and maintaining bog function (Howie and Meerveld 2011). In fact, most of the bogs selected for this research maintain at least some small lags along their boundaries. While the natural bogs were all surrounded by near 100% non-developed land-use, the urban and pasture bogs did not have the same benefit (often containing some mix of forest area) and as a result, the mixing of anthropogenic and more natural land-use in their boundaries may contribute to differences in their vegetation composition.

Bog Form: The 12 bogs that were selected for this study are classifiable as basin bogs, which are described as being confined to a small basin, flat with no raised surface, and fed mainly by the atmosphere with some inputs in the form of run-off from the surroundings (Wells 1981; National Wetlands Working Group 1997). Basin bogs differ in various ways from the other common eastern NL bog forms (domed, slope, blanket) with regards to both structure, vegetation composition, and the way in which surrounding run-off can influence bog ecology (Wells 1981). Thus, a study like this one, done on a non-basin bogs, may obtain different results. For example, domed bogs have a convex surface that can be upwards of 500m above the bog edge (National Wetlands Working Group 1997). Not only does this difference in structure have implications on the typical distribution of growth-forms within domed bogs, with different compositions along the low edges and raised centre, but there are implications as well on the strength of influence of landscape-scale changes on the lower part of the bog which likely receives some landscape run-off and the convex portion which does not. This means that the results of this study should not be extended to other bog forms.

Newfoundland Ecoregion Diversity: The island of Newfoundland contains a total of nine ecoregions (Ecological Stratification Working Group 1996), which are different on the basis of vegetation patterns and soil development as controlled by local climate and geography (South 1983; Ecological Stratification Working Group 1996). Figure 1.5 shows the ecoregions across the province. Even within ecoregions, there are sub regions that are different based on climatic and lithological variation (South 1983). For example, the SJMA occurs within the Maritime Barrens ecoregion, which is the ecoregion with the coldest summers, mild winters, and experiences frequent fog (South 1983; Ecological Stratification Working Group 1996). Compare this to the Central Newfoundland ecoregion, which has the highest summer temperatures, lowest winter temperatures, and high rates of evapotranspiration (South 1983; Ecological Stratification Working Group 1996). Given the dependence on bog development on climate, geography, and geology (Mitsch and Gosselink 2000), it is not possible to extend the results of this study to all basin bogs across the province. Thus, changes to NL wetland policy based on the results of this study at a provincial scale may not be an effective policy for non-SJMA wetlands. This emphasizes the need for more local NL wetland research, policies, and legislation.

Study Design: Aspects of the study methods could be altered or improved, with the potential of obtaining improved or different results. This includes the vegetation sampling method and growth-form classification. Vegetation sampling, carried out here as a random quadrat method, could instead be carried out using a larger quadrat (here the quadrat size was 0.25 m^2 quadrat). Smaller quadrat sizes may be appropriate when collected cover data verses counting plant numbers (U.S. EPA 2002) as was the case for this study. However, such a small quadrat size may have resulted in the missing of some vegetation patterns, particularly for some larger shrubs and so a 1 m^2 quadrat may improve results. Alternatively, the study could instead be designed using a transect method. A transect method would ensure that the microtopography of each sample was recorded, and the aspects of the microtopography were sampled equally. The microtopography of bogs consists of hummocks, lawns, and hollows which have characteristic local hydrology and vegetation (Potvin et al. 2015). Shrubs tend to dominate the drier hollows, and grasses and other herbaceous plants dominate the more wet lawns and hollows and mosses are present in all (Laine et al. 2009). It may be possible that certain environmental changes impact different microtopographic habitats differently or more strongly. Studies have found for example, that hummock vegetation is more resistant to persistent dry conditions when water table levels change (Robroek et al. 2007).

While the growth-forms used in this study are associated with bog functioning and importantly changes in bog functioning, alternative classifications of vegetation into broader or smaller groups may provide different information or be more sensitive to changes in the environment (Dorrepaal 2007). For example, distinguishing between evergreen and deciduous shrubs would be useful because in some cases they react differently to anthropogenic impacts and have different functional associations. For example, evergreen shrubs are good predictors of low nitrogen and potassium concentration in peatlands versus deciduous shrubs which was more predictive of intermediate concentrations of all nutrients (Bombonato et al. 2010) Additionally, evergreen litter takes longer to decompose (Dorrepaal et al. 2005). Because vegetation can react similarly and differently and various levels of classification, it may be helpful to group vegetation at multiple levels, ranging from the broad vascular/non-vascular dichotomy, down in some cases to species level. Species such a bog orchid can act as indication of bogs under anthropogenic distress and could be helpful when looking for changes in bog vegetation status due to environmental change (Laroche et al. 2012).

4.2 Potential for Future Research

The results and methods of this study may be further applied or enhanced in several potential ways including further studies on a more similar design and incorporation into potential future wetland assessment methods.

4.2.1 Further Studies

The design of this study was not extensive enough in terms of number of study sites, number of bog forms considered, and number of study areas to suggest that the evidence presented here is conclusive or that policy should be modified based on these results. However, it provides a basis for further studies of a similar kind on bog wetlands, considering that bogs are often not studied in such a context and considering that bogs are often poorly managed by governmental policy. A similar study should include more bog study sites, use more specific land-use categories, include additional bog forms, include more study areas representative of the regional diversity of the province, and consider distance from disturbance within the bogs.

Urban land-use, for example, includes a broad range of anthropogenic land-use including low-intensity sprawling neighborhoods, populated city centres, highways, single-lane roads, cabins, and parks. Each of these, though still considered urban, will produce different types and levels of air-borne pollutants and different amounts of impervious surface. Another way to improve the study design would be to consider the distance from disturbance in which a vegetation sample was taken within the bog. Though small bogs were chosen for this study with the purpose of lessening this potential problem, it has been shown not only in bogs but in many ecosystems that vegetation reaction to disturbance often depends on distance. Perhaps taking samples from within defined bog zones (ie: edge, centre) would provide more strong results, as a difference would be made between the areas closest to the disturbance (edge) and the area farthest from the disturbance (centre).

4.2.1.1 Wetland Assessment

The associations between wetland ecosystems and adjacent landscapes have been studied extensively in the US, and such knowledge has been incorporated into assessments of wetland condition. Wetland assessment works under the assumption that wetland features will reflect its environment in ways that will vary with the quantity or quality of natural or anthropogenic influences in the environment (Karr 1993; Brinson and Rheinhardt 1996; Reiss 2006). For example, the EPA has developed a three-tier approach to wetland conditional and functional assessment (Johnson 2005; Nestlerode et al. 2014). This approach involves intensive field assessment (level 3), rapid field assessment (level 2) and landscape scale assessment. Level 1 involves the application of landscape-scale data sets (such as landcover) to loosely infer wetland function and condition based on evidence that the surrounding landscape influences ecosystems integrity (Brown and Vivas 2005). Landscape metrics are only chosen only when they have been shown to impact wetland ecosystems (Johnson 2005). The results of this study would justify the application of generalized landscape scale information (urban, pasture) in the conditional assessment of bog wetlands.

Canada does not currently have a country-wide tool for wetlands (Dahl and Watmough 2007; Rubec and Hanson 2009) such as that of the US EPA-WA, however recently steps are being made towards province-based assessment tools such as the

NovaWET, developed initial in Nova Scotia, which being tested and altered for use in other provinces (Tiner 2018). This assessment tool does in-fact incorporate some landscape-scale information (Tiner 2018). Specifically, estimating the percent of a wetlands watershed that contains various types of landcover, such as pasture, urban, road, and forested. NovaWET justifies its application of landscape-scale land cover information in that developed watersheds will deliver more polluted waters to wetlands, degrading them. While this is certainly true for hydrologically connected wetlands such as marsh, the scale of watershed may not be as applicable to hydrologically isolated bogs, which do not receive water inputs from the greater watersheds and whose hydrology is likely altered more closely to the bog boundaries (Lane 2003). The results of this (and other) studies would support the use of a more limited landscape scale when assessing the condition of bog wetlands. Dedicated research to studying the impacts of landscape scale landcover on various aspects of bog ecology (not just vegetation) at various scales may provide a better estimate of an optimal scale for estimated bog condition.

Another type of wetland assessment method that has been highly studied in the US are IBIs. The first IBI was developed by Karr in 1981 to assess the health of water resources using fish, working under the assumption that fish communities will vary with the quality of the catchment of the associated stream. Since the time of Karrs' research, IBIs have been developed for many wetlands. Much like this study, wetland IBIs are developed by associating aspects of anthropogenic disturbance with changes to different aspects of ecosystems ecology. IBIS differ however in that they often test a suite of anthropogenic disturbance (not only landscape) and a suite of ecosystem characteristics (verses a single aspect) including vegetation at various levels of classification from species to genus to growth-form (Miller et al. 2006), fauna at various levels of classification (Veselka et al. 2010; Lunde and Resh 2012) and chemistry (King and Richardson 2003). Those aspects of the ecosystem that show the strongest relationship with anthropogenic activity are included in the final IBI with different values assigned to each ecosystem factors. The results of this study may be incorporated into a future bog IBI. For example, the % cover of graminoids, which has been shown to be influenced by adjacent pasture, may be included along with a suite of other bog-related metrics that may be studied while developing a bog IBI.

4.3 Conclusion

Bogs are often ignored in studies of landscapes and impacts on wetlands. Currently, in the SJMA, there has already been an extensive loss of wetlands and wetland functions (Slaney 2006; Ren 2014). This is unfortunate as bogs are societally and biologically important ecosystems. Though their extensive nature in the province (C-Core 2017) may make them a low priority in terms of conservation effort, this will certainly change as populations (Simms and Ward 2017) and temperatures (Finnis and Daraio 2018) increase in the future. If bogs and the impacts of landscape on bog ecology are not effectively incorporated into future conservation and policy efforts, the result will not only be the loss of a variety recreational actives, but the loss of important services such as ground-water regulation, carbon storage, and nutrient transformation. Though there are now rules being implemented around the globe to restore and replace wetlands impacted or lost by anthropogenic activity (see no net loss and wetland restoration wetland policies), there is variable evidence to support the effectiveness of these methods for returning wetlands to highly functional states (Moreno-Mateos et al. 2012; Meli et al. 2014). Additionally, wetland restoration efforts have been shown to be more successful in tropical climates verses colder, northerly climates (Moreno-Mateos et al. 2012). As such, it is ideally preferred to protect and conserve wetlands without the need to restore or replace. This means not only preventing direct loss of wetland area but managing anthropogenic impacts at the scale of the adjacent-landscape, as the results of this study shows. Further studies such as this will hopefully contribute to improved management of one of NLs most common, yet often ignored, wetland.

4.4 References

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