Macrophyte community dynamics in Lake Simcoe's fringe wetlands: Potential use as biological indicators of water quality

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Abstract

Indices have been developed using macrophytes and water quality parameters to detect the impact of anthropogenic disturbance on coastal wetlands of the Great Lakes, but such an index does not currently exist for Lake Simcoe. As wetland macrophytes are influenced by water quality, any impairment in wetland quality should be reflected by taxonomic composition, biomass and dynamics of the macrophyte community. This study investigates the potential use of macrophytes as a tool for monitoring water quality by examining the dynamics (species richness, density, diversity, and above-ground biomass) in emergent macrophyte communities in fringe wetlands around Lake Simcoe exposed to contrasting degrees of disturbance.

Macrophytes and limnologic data were collected from six wetlands over four seasons from 2013 – 2014. The macrophytes were identified to species and limnologic data was quantified to reflect the water quality, which was measured as a proxy for site disturbance. Wetlands in this study correspond to a wide range of environmental conditions, ranging from very clear and nutrient poor oligotrophic conditions (e.g., TP =11.25 µg/L, TN = 345.25 µg/L, CHL *a* =1.37 mg·m^{-3}) to turbid and eutrophic wetlands (e.g., TP = 47.25 µg/L, TN = 2285 µg/L, CHL *a* = 3.26 mg·m^{-3}. Overall, the limnologic parameters indicated that water quality reflected the degree of anthropogenic degradation influencing the wetland. Taxonomic composition and population dynamics reflected the water quality. The least disturbed site was dominated by native *Scirpus acutus* and *Scirpus pungens*, which are mostly intolerant of environmental degradation, as well as the cosmopolitan *Sparganium eurycarpum*, found in all sites and tolerant of many different conditions. The moderately disturbed sites were dominated by both intolerant and tolerant species, including *Scirpus acutus*, *Leersia oryzoides*, *Eleocharis smallii*, *Typha* x *glauca*, *Typha angustifolia*, and *Sparganium eurycarpum*. The highly disturbed sites were also dominated by *Sparganium eurycarpum*, and species that are considered to be invasive, aggressive, and/or very tolerant of degradation, including *Typha* x *glauca*, *Typha angustifolia*, *Phragmites australis*, and *Calamagrostis canadensis*. These species are indicator species of wetland integrity and their relationship with the limnologic parameters as determined by ordination demonstrated responses that were consistent with the literature. Thus this study validates that macrophytes could be used as an indicator of water quality changes in this study area.

Lay Summary

The mission statement of Lakehead University's Department of Biology is "Faculty and students in the Department of Biology are bound together by a common interest in explaining the diversity of life, the fit between form and function, and the distribution and abundance of organisms." The current study focuses on the dynamics of wetland macrophytes. This study contributes to one of the central research themes outlined in the mission statement, which is the relationship between life forms and their environmental functions. The study advances our understanding of various biotic and abiotic factors influencing macrophyte growth in fringing wetland environments. Understanding the dynamics of wetland macrophyte communities and their environmental niches is a valuable tool in assessing water quality as macrophytes possess the ability to integrate the temporal, spatial, chemical, biological, and physical dynamics acting concurrently on the wetland system. Three major research questions were investigated. 1. What are the temporal and spatial effects of macrophyte dynamics? 2. How do macrophyte composition and dynamics relate to water quality? 3. Can the macrophyte composition function as a bio-indicator of water quality? Results showed that the species composition varied considerably between sites and sampling periods and limnologic parameters of water quality influenced the presence or absence of certain species. This study provides a baseline dataset for macrophyte and water quality measurements in Lake Simcoe's fringe wetlands. Furthermore, it may be useful to the development of a macrophyte-based water quality index for Lake Simcoe which will assist scientists and policy makers in their efforts towards more efficient water resources management.

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Chapter 1 Introduction

1.1 Overview

Lake Simcoe has a long history of a wide range of environmental challenges. Research and monitoring have demonstrated how anthropogenic activities have impaired the heath of the Lake Simcoe ecosystem through direct changes including intensive agriculture, urbanization, and effluent release (North et al. 2013; Palmer, Winter, Young, Dillon, & Guildford 2011; Winter et al. 2007). These activities have caused the loss, fragmentation, and/or degradation of many wetlands and natural areas (Ministry of the Environment 2009). Furthermore, due to the influence of excessive nutrient input and sedimentation originating from these activities, water quality has been degraded in both the lake as a whole and in its fringe or coastal wetlands. "Fringe" or "coastal" (used interchangeably throughout this thesis) wetlands are those adjacent to lakes where the water elevation of the lake maintains the water table in the wetland (U.S. Department of Agriculture, Natural Resources Conservation Service 2008). Any impairment of wetland water quality is reflected in the aquatic plant community, as wetland macrophytes are directly influenced by it (Croft & Chow-Fraser 2007). In this sense, macrophytes can be used as biological indicators of the health or integrity of the fringe wetlands, which ultimately reflects the ecological health of the lake. Therefore, the purpose of the research is to examine the response of Lake Simcoe's fringe wetland emergent macrophyte community to degraded water quality caused by the anthropogenic activities. Responses of macrophytes to water quality may include changes in community composition, species richness, density, diversity, and above-ground macrophyte biomass.

A review of the literature indicates that there is no published research on the Lake Simcoe watershed that attempts to relate fringe wetland macrophyte dynamics, degraded water quality, and anthropogenic stressors. As Lake Simcoe's fringe wetlands are the focus of this research, lacustrine wetland site types are discussed. Details of the Lake Simcoe watershed and its anthropogenic activities are referenced in order to provide the context of degradation in water quality. Responses of fringe wetland vegetation to water quality degradation and the use of macrophytes as biological indicators are also described.

Since no research could be found that specifically involves Lake Simcoe's coastal wetlands, a majority of the review focuses on studies completed on coastal wetlands of the Great Lakes. Although Lake Simcoe is not a part of the Great Lakes system (Environment Canada 2013), it is located within the Great Lakes Drainage Basin, and therefore shares similar underlying physical and geographic attributes that influence the wetland hydrology.

1.2 Wetlands, Definitions, Classification, Types and Site Types

Wetlands are lands where water collects on the land surface or within the root zone long enough to promote unique soil development that differs from adjacent uplands and supports plant and animal communities that are adapted to saturated conditions and intermittent anoxia (Cronk & Fennesy 2001; Grady 2007; National Wetlands Working Group [NWWG] 1997). They are large or small expansive areas of the landscape where the water table is near or at the surface, or where the land is covered by shallow water for a majority of the growing season. Wetlands are discrete entities and lie between unsaturated terrestrial upland and aquatic deep water in the landscape mosaic (Mitsch 1995; Mitsch & Gosselink 2007; NWWG 1997). The importance of wetlands is evident through their limnologic and water quality functions, including shoreline stabilization, nutrient cycling, carbon storage, sedimentation and heavy metal movement; as well as contributions to biodiversity through providing significant habitats, and native plant species richness (Levine & Willard n.d.; Newmaster, Harris & Kershaw 1997; Beacon Environmental & the Lake Simcoe Region Conservation Authority [LSRCA] 2007).

Wetlands show several distinguishing features. Their limnologic characteristics and tendency to form an intermediate zone along the margins (and consequent significant interaction) of aquatic and terrestrial ecosystems make them difficult to delineate (van Dam, Camilleri, & Finlayson 1998; Mitsch & Gosselink 2007). Nevertheless, definitions are important both for the scientific understanding of these systems and for their appropriate management (Mitsch & Gosselink 2007). Formal definitions of wetlands have been developed in Canada both nationally and provincially, as well as internationally through a treaty known as the Ramsar Convention. The International Union for the Conservation of Nature and Natural Resources (IUCN) at the Ramsar Convention adopted the following definition of wetlands:

> They are the areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six metres (van Dam et al. 1998, p. 298).

Absent from this definition is vegetation or soils, as well as the extension of wetlands to water depths of 6 meters or more, well beyond the depth usually considered in Canada (usually less than 2 meters deep) (NWWG 1997; Mitsch & Gosselink 2007). The official definition of wetlands in Canada is provided by The Canadian Wetland Classification System (Mitsch & Gosselink 2007). The System represents the development of a national wetland classification system in Canada. Its focus includes identification of the three basic hierological levels of wetland classification: class, form, and type (NWWG 1997). The Canadian Wetland Classification System defines a wetland 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" (NWWG 1997, p. 1).

At the provincial level, the Ontario Ministry of Natural Resources (MNR) has mandated the sustainable management and protection of Ontario's natural heritage features, including wetlands (MNR 2013). To aid in the identification of wetlands that are having a value at the provincial scale, the MNR has developed the Ontario Wetland Evaluation System (OWES). This evaluation system is the only way of evaluating wetlands in Ontario to establish whether they are provincially significant. The Ontario Wetland Evaluation System defines wetlands as:

> Lands that are seasonally or permanently flooded by shallow water as well as lands where the water table is close to the surface; in either case the presence of abundant water has caused the formation of hydric soils and has favoured the dominance of either hydrophytic or water tolerant plants (MNR 2013, p. 7).

In this definition in contrast to the Ramsar definition, soil and vegetation characteristics are included as well as limnologic attributes.

Whether for scientific or management purposes, it is imperative that there is both precision in the definition and consistency with which it is used. For ecological studies and inventories, the Canadian Wetland Classification System may be better applicable than the Ontario Wetland Evaluation System, which is probably more appropriate for recognizing the values of wetlands for management purposes.

Since wetlands are a consequence of the interaction of numerous environmental factors, they develop varying characteristics that can be utilized to group them into classes (NWWG 1997). These classifications are contingent on a well-comprehended general definition of wetlands, although a classification contains definitions of individual wetland types (Mitsch & Gosselink 2007). Various wetland classification systems are found throughout the literature, such as the Circular 39 Classification, The United States Classification of Wetlands and Deepwater Habitats, International Wetland Classification System, etc. (Mitsch & Gosselink 2007). The Canadian Wetlands Classification System is hierarchical in design. In this classification system, the greatest importance is attached to the numerous conditions that affect wetland development, including wetland morphology (elevation relative to the surrounding terrain, surface pattern and form), water source, basin depth and shape, water chemistry (nutrient levels, base saturation, pH), phytosociology and physiognomy (plant communities and their structure), and peat and sediment characteristics (chemical and physical properties). Therefore, a combination of particular geomorphologic, hydrologic, chemical and biological factors characterizes wetland systems at a general level of classification of wetland regimes (NWWG 1997).

The major features of the Canadian Wetland Classification System include wetland classes, forms, and subforms. Wetlands at the class level are recognized on the basis of properties of the wetland that mirror the overall genetic origin of the wetland ecosystem and the nature of the wetland environment. Wetland forms are subdivisions of each wetland class based on surface pattern, surface morphology, water type and morphology characteristics of underlying mineral soil. Some forms can be additionally subdivided into subforms. The forms and subforms vary from each other in surface relief, basin topography, surface pattern, and proximity to water bodies (NWWG 1997). Currently, the system recognizes 5 wetland classes (bog, fen, swamp, marsh, and shallow water marsh), 49 wetland forms, and 75 subforms.

Between the Canadian Wetland Classification System and the Ontario Wetland Evaluation there is a lack of uniformity in the terminology for classification. The Canadian

Wetland Classification System uses the term "wetland class" to identify bogs, fens, swamps, and marshes, and shallow water marshes as wetland classes. In the Ontario Wetland Evaluation System, the term "type" is used for these same ecosystems. They are based on the definitions of Jeglum et al. (1974), Zoltai et al. (1975), Riley (1983), Damman & French (1987), and Environment Canada (1987). As noted above, the Canadian Wetland Classification System recognizes five wetland types; however, in the Ontario Wetland Evaluation System, "marsh" and (shallow) "open water marsh" are regarded as two categories of the marsh wetland type, since the descriptive terms for shallow water wetlands in the Canadian Wetland Classification System are analogous to the descriptions for marsh wetland (MNR 2013). In this study, the separation of wetland ecosystems as "types" will be utilized.

Within the Lake Simcoe watershed, 4 types of freshwater wetlands are present: marshes, fens, bogs, and swamps (LSRCA 2007). The following is a general review of the information found in the literature in regards to the characteristics of each type. Because bogs and fens are uncommon within the watershed, they will be discussed in less detail compared to the swamp and marsh classes.

Fens

Fens are peatlands with a fluctuating water table rich in dissolved minerals, therefore characterizing them as minerotrophic. The common characteristics of fens include groundwater and surface water movement, which can be directed through pools, channels, and other open water bodies (NWWG 1997). In the Lake Simcoe Watershed, almost all known fens are peat-based, with approximately 450 hectares present within the watershed. These peat-based systems can be referred to as "poor" fens, as compared to the "rich" fens that can develop on mineral-rich limestone systems (which are commonly found, for example, on the Bruce Peninsula) (LSRCA

2007). Important attributes that allow identification of this wetland type are fen indicator species, including moss species with narrow pH tolerances (MNR 2013). Fen vegetation is closely associated with the depth of the water table, the chemistry of the water, and regional geographic variations (NWWG 1997). *Sphagnum* species can spread as a mat in nutrient-poor fens, and can be found in nutrient-rich fens dominated by sedges and grasses. Low shrubs, such as *Myrica gale* (sweet gale) or ericaceous species associated with mycorrhizal fungi such as *Kalmia angustifolia* (sheep laurel) can occur along with sedges and grasses in rich fens. There can be a tall shrub layer exceeding 25% cover that often includes *Larix laricina* (stunted tamarack) and *Thuja occidentalis* (eastern white cedar). In other cases a sparse layer of trees, typically tamarack or eastern white cedar, and in poor fens *Picea mariana* (black spruce) may also be present (MNR 2013). Fens have a higher plant diversity compared to bogs, which typically possess less than 14 species of vascular plants (LSRCA 2007).

<u>Bogs</u>

The bog is a very scarce wetland type in the Lake Simcoe watershed with only 25 ha identified so far (LSRCA 2007). A bog is a peat landform of variable extent (NWWG 1997). Often with a raised surface, it is essentially secluded from the mineral soil waters. Consequently, bog water is strongly acidic due to organic acids produced in the decaying organic matter layer (L. Lovett-Doust, personal communication, April 22 2016) and upper peat layers are extremely deficient in mineral nutrients. Peat is usually formed in-situ under conditions of closed drainage and low oxygen levels (MNR 2013). All bogs are considered as ombrogenous because the source of water is only from precipitation, fog and snowmelt. Since precipitation is slightly acidic and lacks dissolved minerals, the surface waters of bogs are low in dissolved minerals (NWWG 1997). The acidity in bog is also due to the organic acids formed during the decomposition of peat and the acids present within *Sphagnum* leaves (NWWG 1997). A bog can be either treed or treeless, and when treed it consists largely of *Picea mariana* and small number of *Larix laricina* (LSRCA 2007). Bogs are routinely identified by a layer of ericaceous shrubs, including *Chameadaphne calyculata* (leatherleaf) (MNR 2013). Despite bogs often being covered with *Sphagnum*, they can also support sedges such as *Carex oligosperma* (few-flowered sedge). Many bogs begin as fens, and therefore peat may be found in the deepest layers of bog (NWWG 1997).

Swamps

In Ontario the term swamp denotes a wooded area of standing water or saturated soil, which may or may not dry up over the summer (Grady 2007). A swamp can be defined as a wetland regulated by minerotrophic groundwater with either mineral or organic soils that is prevalently covered by trees or tall shrubs (also called thicket) (MNR 2013; NWWG 1997). The water table is found below the major portion of the ground surface. Pools and channels are abundant, which provides an indication of subsurface water flow (MNR 2013; NWWG 1997). The available aerated or partly aerated zone of substrates, in or above the water, allows for a suitable place for root growth of trees and/or tall shrubs. Not as wet as marshes, fens and the open bogs, swamps have a similar moisture content to treed bogs. Drier treed swamps grade into upland forest on mineral soil, and the wettest treed swamps grade into treed fen, which is wetter with less tree canopy cover. Tall shrub swamps grade into treed fens, which are wetter with less tree canopy cover, or into wetter marshes (NWWG 1997). Mineral swamp soils have a versatile consistency, frequently they are gleysols, but can range from clays to sands. The variable nutrient regime in swamps allows for basic conditions having pH above 7.0, to acidic conditions where pH can be lower than 4.5. Swamp subtypes may be characterized by this pH gradient, being either calcareous rich (eutrophic), intermediate (mesotrophic), or poor (oligotrophic) (NWWG 1997). Typically there are three physiognomic types of swamps; each is separated by the predominance of either "tree" or "shrub" (MNR 2013; NWWG 1997). There are shrub (thicket) swamps, coniferous swamps, and hardwood (deciduous) swamps, and mixed swamps with conifers and deciduous trees. In shaded understory swamps only forest species that can tolerate shade are seen. The deciduous swamp is commonly found in dry locations with silver maple, hybrid soft maple, white elm, black ash and yellow birch (MNR 2013), while the shrub swamp is seen in somewhat wetter locations. A coniferous swamp can appear across a range of trophic levels with white cedar, eastern hemlock, tamarack and black spruce as indicator species (Grady 2007; MNR 2013; NWWG 1997). Thicket swamps, characterized by thick growths of tall shrubs such as willow species, speckled alder, red-osier dogwood, and buttonbush, possess similar water levels and chemistry of forest swamps. Regardless of the type of woody vegetation each swamp must contain at least 25% cover or more of trees or tall shrubs (Beacon Environmental & LSRCA 2007; MNR 2013). Up to 86% of all remaining wetlands in Lake Simcoe's watershed are swamps. They have developed or persisted in seasonally wet areas that have not been farmed, though some are used for livestock grazing and shade in the summer and fall. In the Lake Simcoe watershed there is some difficulty distinguishing between treed swamps and moist upland forest (Beacon Environmental & LSRCA 2007).

Marshes

A marsh is a minerotrophic and usually eutrophic wetland that has shallow water which usually fluctuates daily, seasonally or annually due to tides, evapotranspiration, flooding, groundwater recharge, or seepage losses. Marshes occur in many geomorphological sites, associated with a pond, stream, or shallow edge of a lake (Grady 2007). They comprise approximately 10% of all wetlands in southern Ontario and 12.6% of the wetlands within the Lake Simcoe Watershed (Beacon Environmental & LSRCA 2007). In the case of the water level drawdowns that may occur, marshes can dry up exposing areas of sediment. Depending on the location of the site, water is drawn in from the catchment as precipitation, surface runoff, stream inflow, storm surges, and groundwater discharge. Marshes reliant upon surface runoff usually retain less permanent water than those supplied by groundwater. Except in years of extreme drought, the water table generally remains at or below the soil surface; however, throughout the growing season the soil remains waterlogged within the rooting zone (NWWG 1997).

Due to regular saturation or semi-permanently flooding, the shifts in the hydrology of marshes can be more drastic seasonally, compared to other classes of wetlands. In response to rapid fluctuations in surface water, marshes go through cycles of drawdown and regeneration, degeneration and open water stages over the course of a few years (NWWG 1997). The characteristic high productivity of vascular plants and high decomposition rates of the plant material at the culmination of the growing season is due to the high level of nutrients derived from the substrate as a function of periodic aeration. Due to the presence of dissolved minerals such as calcium, potassium carbonate, or potassium bicarbonate freshwater marshes are generally circumneutral to highly alkaline (NWWG 1997). Marsh soils and substrates generally range from mineral soils including Humic and Rego Gleysols to organic soils such as Mesisols and Humisols. The marsh sediment mixture is commonly made up of organic and inorganic material (NWWG 1997). Clumps, tussocks, or hummocks of dead and live herbaceous vegetation may exist in standing water. In lakeshore marshes, these hydrologically stable and permanently saturated wetlands develop Humisols. This organic material can accumulate but rarely is more than 40 to 50 cm deep. In comparison to the permanently saturated and more hydrologically stable marshes, those that are seasonally dry or exposed to high energy currents or tides usually accumulate little organic matter.

Establishing from sedges and aquatic mosses, floating mats of vegetation form in response to persistent conditions of stable water (NWWG 1997). Marsh vegetation is typified by emergent aquatic macrophytes, mainly reeds, rushes, grasses, sedges, and shrubs as well as other herbaceous species such as broad-leaved emergent macrophytes. To a lesser extent marsh vegetation includes anchored floating plants, submergents and non-vascular plants such as liverworts, brown *Sphagnum* mosses, and algae (MNR 2013; NWWG 1997). Characteristic of marshes are zones or mosaics of vegetation, often intermingled with pools or channels of deep or shallow open water, such as ponds, oxbows, shallow lakes, reaches or impoundments. Along with the aforementioned non-woody emergent plants, a border of peripheral bands of trees and shrubs may also be found. Low shrubs such as red-osier dogwood, sweet gale, waterwillow, and winterberry can also occur. In the open water areas, a variety of submerged or floating plants flourish, such as water-milfoils, waterweeds, stonewort (*Chara*), pondweeds, bladderworts, coontails, tape-grass, duckweeds, watermeals, and water lilies (MNR 2013). In low-lying areas adjacent to rivers or lakes marshes exist in many geomorphological sites (MNR 2013; NWWG 1997).

<u>Wetland site types</u>

A wetland’s site type is defined by its physiographic position in the landscape. In the Ontario Wetland Evaluation System, four fundamentally different site types are defined, including isolated, palustrine, riverine and lacustrine. Riverine and lacustrine are further subdivided because the location of a wetland on a lake or river shore influences the nutrient concentrations of the water and therefore the productivity of the site (MNR 2013). In studies of the Great Lakes coastal wetlands, these “site types” are instead referred to as a hydrogeomorphic classification scheme that divides coastal wetlands into specific hydromorphic systems of lacustrine, riverine, and barrier-protected wetlands. This division is based on the geomorphic position, major hydrologic source, and current hydrologic connectivity of the wetland to the lake. Each hydrogeomorphic wetland type has characteristic associated floral and faunal communities and specific physical attributes related to sediment type, water quality, hydrology, and wave energy (Albert, Wilcox, Ingram, & Thompson 2005). Since Lake Simcoe’s lacustrine fringe wetlands are the focus of the present research, details of lacustrine fringe wetlands are discussed below.

The Ontario Wetland Evaluation System defines lacustrine (lake-associated) wetlands as “Areas of open water that are greater than 8 ha in size and at some location are greater than 2 m depth from the normal low water mark” (MNR 2013, p. 56). Lacustrine wetlands include areas normally covered by the seasonally high water level (MNR 2013). They are confined to the high and low shore zone and littoral zone of freshwater lakes (Cronk & Fennessy 2001). Water is fresh and originates from precipitation, groundwater discharge, surface runoff, and rivers and streams. Here, the water levels may fluctuate dramatically (NWWG 1997) where the vegetation is influenced by changes in lake level (MNR 2013). Lacustrine wetlands in the Great Lakes are controlled directly by waters of the lakes and are directly affected by lake-level fluctuations, seiches, nearshore currents, and ice scour. Geomorphic features along the shoreline offer a varying extent of protection from coastal nearshore processes (Albert et al. 2005). Cronk and Fennessy (2001) contend that lacustrine wetlands are likely to occur along shallow lake basins, often formed by glaciation. The landscape of the Lake Simcoe basin is the product of four major glacial advances and retreats of the Pleistocene Epoch (Johnson 1997), and this explains the scattered coastal wetlands along the Lake Simcoe littoral and limnetic zones (County of Simcoe Interactive Map 2013).

Within these hydrologically based systems, differing geomorphic features and shoreline process permit for further classification of Great Lakes coastal lacustrine wetlands, as open lacustrine, which includes open shoreline and open embayment types, and protected lacustrine, which includes protected embayment and sand-spit embayment. Open lacustrine wetlands develop in beach or strand areas or the zone of wave action, and include the high shore, low shore and littoral zones (NWWG 1997). Open lacustrine wetlands are directly exposed to nearshore processes, with little or no physical protection by offshore sandspits and sandbars. Consequently, vegetation development is restricted to relatively narrow nearshore bands and little organic sediment is accumulated. Exposure to nearshore processes also causes a variable bathymetry, ranging from relatively steep profiles to more shallow sloping beaches (Albert et al. 2005). Within the open lacustrine division, open shoreline wetlands are characterized by erosion-resistant substrate of either clay or rock, with infrequent patches of mobile substrate. No detrital sediment is present and almost no organic sediment is accumulated. The resulting expanse of shallow water functions to dampen waves, and if littoral sediment is available sand-bars may form. In lacustrine wetlands narrow fringes of emergent vegetation extending offshore to a point where wave action is tolerated, is all that is able to develop. Some smaller embayments also fit into this class due to exposure to prevailing winds; most of these have relatively narrow littoral vegetation zones of 100 meters or less.

The open embayment wetlands are typically large and consequently exposed to storm-generated waves and surges. Substrates present may be gravel, sand, and clay (fine), and organic sediments may accumulate only near the shoreline edge. This stable sediment along with shallow waters reduces the harsh effects of wave action on the emergent plant communities. Most bays larger than 3 or 4 km in diameter fit into this class, allowing wetlands that are 100 to 500 m wide to develop over broad expanses of shoreline. The high wave energy results in little organic sediment accumulation and relatively low plant diversity, as most emergent and submergent aquatic plants cannot tolerate such vigorous conditions. Higher diversity can be found locally in shallow, nearshore areas. In the shallowest open embayments, a strong chemical gradient develops between the outer marsh and the protected inner marsh, resulting in distinctly different invertebrate and fish populations for these marsh zones (Cardinale, Brady, & Burton 1998; Burton, Stricker, & Uzarski 2002). Despite the generally low productivity of open embayments, the overall area of the wetlands can be large, rendering them significant as wildlife and fish habitat (Albert et al. 2005).

Protected lacustrine wetland type is characterized by a sand-spit, offshore bar, or till- or bedrock-enclosed bay that offers increased protection. Consequently, shallower off-shore profiles, increased mineral sediment accumulation, and more extensive aquatic vegetation development compared to open lacustrine systems is a result of this protection. Organic sediment development is also evident. Protected embayments are stretches of bedrock or till-derived shorelines that form small protected bays, usually less than 3 or 4 km wide. These bays can be completely vegetated with a diversity of submergent or emergent vegetation. Wave action is strong enough to limit organic material accumulation to beneath wet meadow vegetation at the wetland boundary. Protected embayment wetlands are one of the most biologically stable wetland types in the Great Lakes, as major water-level fluctuations of the Great Lakes do not typically result in major changes in vegetation. Basin morphology is diverse in this wetland type and determines the range of plants found in a specific wetland (Albert et al. 2005).

Sand-spit embayment wetlands are typically shallow and created by sand spits projecting along the coast protecting shallow embayments on their landward side. Spits frequently occur along gently sloping and curving sections of shoreline. They are characterized by moderate levels of organic soils, similar to those found in other protected embayments. Sand-spit embayments possess broad zones of wet meadow, emergent and submergent vegetation, but are subject to increased erosion during Great Lakes high water conditions (Albert 2005, personal observation). Small sand-spit embayments are typically shallow (less than 2 m depth of water), but in larger spits water is deeper and wave action is stronger, resulting in plant communities that are similar to open embayments. Year-to-year fluctuations in water level cause dramatic vegetation change in the Great Lakes; and the high diversity of the viable seed bank in the organic sediment results in major changes in plant composition, coverage, and structure, intermittently are occurring on an almost annual basis (Albert et al. 2005).

The greatest physical and biological differences between coastal wetlands are typically seen at the hydrologic system level, resulting from differences in water-flow characteristics and residence time (Albert et al. 2005). With reference to Lake Simcoe, no published studies discuss the differences of its fringing wetlands at the hydrologic system level, but applying the above classifications from the Great Lakes to Lake Simcoe would be a valuable step in understanding macrophyte composition, before the anthropogenic stressors on vegetation can be evaluated.

1.3 Lake Simcoe Watershed, Anthropogenic Activities, and Water Quality Impacts

Following the identification of wetland types within the Lake Simcoe watershed, an examination the watershed, anthropogenic activities occurring within it, and evidence of these activities degrading the water quality can be discussed. Chapter 2 provides a detailed physical description of the lake. Briefly, Lake Simcoe (44°25’N, 79°24’W) is a large lake with a surface area of 722 km^2, maximum depth is 42 m; mean depth 15 m, and a shoreline length of 240 km located in southern Ontario (Ginn 2011; Johnson 1997; Stantec Consulting Ltd. 2007). It is the sixth largest inland lake in Ontario (Johnson 1997), with only the five Great Lakes being larger (Stantec Consulting Ltd. 2007).

The Lake Simcoe Watershed has been influenced by human activities for over 200 years (Stantec Consulting Ltd. 2007) and consequently has undergone environmental changes (Ginn 2011). Development within the Lake Simcoe watershed was primarily agricultural until the 1960s–70s when urbanization rapidly took place (Ginn 2011). Agricultural use still dominates the watershed, accounting for 47% of the catchment area. Urban areas and roadways occupy approximately 12%. The remainder is highly fragmented natural cover (forests, wetlands, and “cultural greenlands” (maintained, non-agricultural vegetative areas)) (Ginn 2011; LSRCA 2008; Palmer et al. 2011). Agricultural land use is concentrated along the Holland River and includes the largest area of cultivated marsh in Ontario that empties into Cook’s Bay at the south end of the lake. While only 12% of Lake Simcoe’s watershed is urban, the population has doubled in the past two decades and projections suggest further growth (642,000 people by 2031) (Palmer et al. 2011; Stantec Consulting Ltd. 2007). The land use pattern in the basin closely mirrors the underlying physical characteristics of the landscape. For example, the wetland areas have been largely maintained in natural vegetation cover due to their high water tables and poor drainage. The exceptions are areas along portions of the Black River and the Holland River which have been drained and converted to cropland farms (Johnson 1997).

Anthropogenic activities, particularity agriculture and urban land use, have caused a significant proportion of the watershed to be changed from its natural state, resulting in dramatic ecological changes that caused extensive degradation of the lake's ecological health (Johnson

1997; LSRCA 2013b; Palmer et al. 2011). Water quality issues first became obvious in the 1970s with the recruitment failure of popular coldwater sport fishes such as *Coregonus clupeaformis* (lake whitefish) and *Salvelinus namaycush* (lake trout) and excessive algal and shoreline macrophyte growth (Winter et al. 2007). Early investigations indicated these changes were caused by anthropogenic phosphorus inputs that promoted algal production, where the subsequent algal decomposition consumed hypolimnetic oxygen and constrained the availability of suitable coldwater fish habitat (Palmer et al. 2011).

The development of land for urban, industrial, and institutional uses constitutes the most drastic changes to a natural system. Evidence seen through the watershed includes changes to flows in urban watercourses to accommodate development activities such as removal of sediment, filling in, and/or reduction of forest, wetland, and grassland vegetation. Agriculture also causes alterations to the natural system, including channelized watercourses and the removal of riparian vegetation, which leaves little habitat value for aquatic communities and removes the opportunity for filtration of contaminants, as runoff passes quickly as surface runoff through the riparian zone. Additionally, the presence of tile drains impact the local hydrology by lowering the water table thereby making less water available for wetlands (LSRCA 2013b).

Common to both agriculture and urbanization and perhaps of greatest significance for watershed health is the impact these activities have on water quality. The chemical, physical and microbiological characteristics of natural water comprise an integrated index defined as “water quality” (NWWG 1997). Water quality is a function of both natural processes and anthropogenic impacts. Natural processes such as mineral weathering and various kinds of erosion can affect the quality of groundwater and surface water (LSRCA 2013b). There are several different types of anthropogenic influences on water quality (both point and non-point pollution), which are discussed below.

Urbanization leads to degraded water quality due to contaminants including sediment, nutrients, metals, and chloride from winter salt being transported by overland flow. Also evident due to urbanization are changes in sediment transport, including streambank erosion due to high flows, and sediment deposition following a drop out of suspension due to slow moving water. Agriculture leads to degraded water quality due to the contribution of contaminants such as nutrients, sediment, pesticides, and, potentially bacteria, to the water body. This can lead to eutrophication of the water body and the deterioration of aquatic habitat, as well as decreased dissolved oxygen concentrations (Johnson 1997; LSRCA 2013b).

Water quality degradation of Lake Simcoe due to nutrient enrichment, specifically phosphorus levels, has been thoroughly reported on for decades. Phosphorus (P) loading has increased from pre-settlement (pre-1800) period of 32 t (P)/y (Nicholls 1997) to a peak 100 t (P)/y around 1990 (Winter et al. 2007) resulting in cultural eutrophication (hypoxic/anoxic bottom waters, algal blooms, high macrophyte biomass, etc.) and simultaneous recruitment failures of several coldwater fish species (Ginn 2011). High-intensity agricultural practices in the Holland Marsh are known to significantly contribute phosphorus to Lake Simcoe. Furthermore, four sewage treatment facilities are also introducing effluent into the Holland River and Cook's Bay directly as a point source of phosphorus contribution (Johnson 1997; Stantec Consulting Ltd. 2007). The urbanization of the Holland River watershed with the large urban centers of Newmarket, Aurora and Bradford make additional contributions to phosphorus loading. The combination of phosphorus contributors historically has impacted water quality in Cook's Bay (Stantec Consulting Ltd. 2007).

1.4 Wetland Macrophyte Responses to Water Quality and Their Utility as Biological Indicators

Generally, biological indicators are used to monitor the environmental changes occurring in an ecosystem (Simon, Stewart, & Rothrock 2001). Predictable changes in community composition, species abundance, productivity, and other ecosystem properties have been used as indicators of environmental changes (Cronk & Fennessy 2001). Many studies have developed biotic indicators of ecosystem health (Albert & Minc 2004). Early studies focused on fish and invertebrates as biologic indicators in Great Lakes coastal wetlands (Burton et al. 1999; Kashian & Burton 2000). Biological assessments using primary producers have focused on algal and periphyton communities (Bahls 1993; Kentucky Division of Water 1993; Oklahoma Conservation Commission 1993; Rosen 1995; Stewart 1995; as cited by Simon et al. 2001). Only recently has the utilization of plants as indicators of the heath of aquatic ecosystems in both inland and Great Lakes coastal wetlands been explored (Simon et al. 2001; Albert & Minc 2004; Croft & Crow-Fraser 2007). Such studies are notably absent from Lake Simcoe's wetlands. The composition of a wetland's plant community has been shown to serve as an indicator of ecological stress (Albert & Minc 2004). They suggest that wetland plants can be used to track four dimensions of anthropogenic stress on wetlands such as water-level regulation, nutrient loading, sedimentation, and physical degradation/disturbance. There are numerous biological attributes that can be measured, but only a few provide useful indications of the impact of human activities (Cronk & Fennessy 2001).

Croft and Chow-Fraser (2007) developed a Wetland Macrophyte Index (WMI) with plant presence/absence data for 127 coastal wetlands over 154 years from all five Great Lakes. They used results of a Canonical Correspondence Analysis (CCA) to ordinate plant species along a water quality gradient. The WMI that they developed was specifically for coastal systems that have a hydrological linkage to a large lake or bay. Results of their study included a valuation or ranking of the tolerance of particular macrophyte species of water degradation. Emergent species and their specific tolerances are elaborated upon in Chapter 4, section 4.1. The results of the study by Croft and Chow-Fraser suggested that the presence or absence of tolerant or intolerant species is a useful indicator water quality, and therefore a WMI index could be usefully applied in Lake Simcoe. Simon et al. (2001) examined riverine and palustrine wetland plant communities in order to propose a multimetric plant index of biotic integrity. The objectives were to determine the structural and functional attributes of these wetland plant communities, calibrate reference conditions in assessing aquatic plant communities, and provide approaches for further development and testing of the index. More than 20 characteristics of aquatic plant communities were evaluated and 12 metrics in 5 categories were developed. While the structural metrics focused on community composition, key indicator species such as number of *Carex* and *Potamogeton* species, and guild type (whether the guild is composed of obligate wetland species or invasive emergents like *Typha* or *Phragmites* species). The functional metrics included sensitivity and tolerance measures, percent emergent, pioneer, and obligate wetland species, and the number of weed species as a substitute metric.

Since wetland macrophytes are directly influenced by water quality, any impairment in wetland water quality should be reflected by taxonomic composition of the aquatic plant community (Croft & Chow-Fraser 2007). Lougheed, Crosbie, & Chow-Fraser (2001) carried out a comparison of the macrophyte community composition of 62 wetlands on the Canadian side of the Great Lakes basin to examine how water quality and sediment quality affect the taxonomic composition and community structure of macrophytes and relate these to land use in their watershed. Their results confirmed that submergent macrophyte biodiversity in Great Lakes wetlands declines with deterioration in water quality. This is consistent with a study by Findlay & Houlahan (1997), who also found reduced species richness of both aquatic and terrestrial organisms with increasing development in wetland watersheds of southeastern Ontario. In the study by Lougheed et al. (2001), the important predictors of macrophyte distribution were total suspended solids, total phosphorus, chlorophyll *a*, total nitrogen, and conductivity.

Many studies have shown that nutrient enrichment can cause substantial changes in the species richness, composition, and density of aquatic vegetation in lakes (e.g., Lougheed et al. 2001; Magee et al. 1999; Toivonen & Huttunen 1995). Albert & Minc (2004) reported that several species of both submergent and emergent macrophytes in Great Lakes responded with increased growth when organic nutrients are added to wetlands. In their study of macrophyte community structure Lougheed et al. (2001) found that certain macrophyte taxa were identified as intolerant of turbid, nutrient-rich conditions, while others were identified as tolerant of a wide range of conditions, including *Typha* spp., occurring in both degraded and pristine wetlands. Overall, the research generally suggested that some emergent and submergent plant species indicate a turbid, nutrient-rich wetland, while high quality wetlands with clearer water and lower nutrient levels contain a mix of emergent and floating-leaved taxa with a diverse and dense submergent plant community (Lougheed et al. 2001).

The impact of sedimentation is expected to be evident within both emergent and shoreline herbaceous vegetation. Responses include the severe loss of plant diversity within the submergent zone, the relative dominance of submergent species that are more tolerant of low light levels, and a reduced presence of species intolerant of turbidity. Changes in species composition are also expected in the emergent and wet-meadow zones (Albert & Minc 2004). Sediment phosphorus may be a source of wetland degradation (Albert & Minc 2004) and inputs vary considerably across Lake Simcoe, with high values being recorded near Barrie, as well as the Black, Beaver, and Talbot River subwatersheds (LSRCA 2013b). The effect of increased sedimentation from agricultural land use can be reflected by a loss of plant diversity within the submergent zone (Albert & Minc 2004), as deposition of sediment can prevent seed germination for both emergent and submergent aquatic plants (Barko, Gunnison, & Carpenter 1991). Furthermore, deposition of thick sediments in the shoreline herbaceous zone favours a collection of aggressively colonizing species, including native annuals and several exotic species.

In contrast to the viewpoint of Lougheed et al. (2001) that the most accurate indicator of wetland quality is the type of community present, rather than the presence of certain indicator species, several studies assert that specific plants also function as indicators of wetland health. For example, it has been suggested that the number of exotic plant species present at a wetland site is a good indicator of the level of site degradation (Simon et al. 2001; Stewart et al. 1999).

There is a need to clarify and comprehend the distinction between native, introduced/exotic, and other associated terminology. In general, introduced, exotic, non-native, nonindigenous, alien, nuisance, and invasive species refer to plants, animals, or microscopic organisms living outside their native range. Among these terms, the major difference is between introduced and invasive. An “introduced” species is an organism (plant, animal, microbe) found living beyond its historic native range, which is typically understood as the area where it evolved to its present form. “Alien” species are any species, including its seeds, spores, or other biological material capable of propagating that species, that is not native to the particular ecosystem in which it is found. Thus, alien can be used interchangeably with non-native. The terms exotic and nonindigenous are both synonyms for non-native. Therefore, introduced, non-native, alien, exotic, and nonindigenous share the same meaning.

"Nuisance" species are a non-native species that threaten the abundance or diversity of native species or the ecological stability of the particular ecosystem, or commercial, agricultural, aquacultural or recreational activities dependent on such systems. Likewise, invasive species are alien species whose introduction is likely to or does cause environmental, economic, or human health harm. Thus, invasive and nuisance species are synonymous and can be used interchangeably. An invasive species is also, by definition, nonnative, but not all non-native species are invasive (Great Lakes Environmental Research Laboratory [GLERL] 2002). It should also be noted that native species can also be aggressively dominant, or "weedy" (L. Lovett-Doust, personal communication, April 22, 2016). These weeds are r-strategists that opportunistically colonize disturbed sites (Clewell & Aronson 2012). The understanding between native and introduced species is essential because many non-native species are invasive and alter the local ecosystem. Invaders frequently alter productivity, biomass, litter dynamics, and nutrient cycling (Ehrenfeld 2003; as cited by Tuchman et al. 2009). Invasive wetland plants may also act as drivers of ecological change (MacDougall & Turkington 2005).

Introduced plants establish in disturbed wetlands by positively responding to the increased nutrient levels or increased sedimentation (Albert & Minc 2004). These exotic plants can form dense monotypic stands that often replace the native flora of wetlands, and provide only a few benefits to the fauna that the native flora would have (Cronk & Fennessy 2001).

In their study, Albert & Minc (2004) explained that the number of exotic species co-varies with wetland size and therefore the number alone cannot indicate the health of the system clearly. Instead, based on their study from Great Lakes coastal marshes, they suggested that the total coverage of exotic plants appears to more accurately represent the condition of the wetland than the number of exotic species. This perspective is valuable in assessing the response of wetland macrophytes in the Lake Simcoe watershed, as several macrophyte studies reported the presence of various invasive species (Ginn 2011; LSRCA & Bradford 2011; Stantec Consulting 2007).

Vegetation can integrate the temporal, spatial, chemical, physical, and biological dynamics of the system (Cronk & Fennessy 2001). Anthropogenic stress is not the only factor that can influence the vegetation dynamics, but the natural variability in ecological factors can influence the wetland plant communities and hence a thorough knowledge on the local environmental conditions is important to understanding the species distribution (Albert & Minc 2004). Therefore, an initial consideration for utilizing wetland plants as indicators of water quality or wetland integrity involves identifying the local and regional variability in wetlands that reflects the distinctive plant associations and major physical factors (Minc 1997). A discussion of the natural variability creating distinctive wetland types appears in Chapter 3, section 3.1. Once the understanding of plant response to the physical environment is achieved, the next action in developing plant-based indicators of wetland health is to identify the major types of environmental degradation within the framework of regional wetland types (Albert & Minc 2004; Johnston et al. 2007).

Much research has been accomplished in the Great Lakes region that has studied the responses of coastal wetland macrophyte communities to degraded water quality caused by anthropogenic stressors. Even though the relationship between water quality and aquatic vegetation in coastal wetlands of the Great Lakes has been well-studied (Lougheed et al. 2001, McNair & Chow-Fraser 2003), no existing research completed in the Lake Simcoe watershed attempts to relate fringe wetland macrophyte dynamics, degraded water quality, and anthropogenic stressors in the landscape, and consequently, no basin-wide biotic index of anthropogenic disturbance based on aquatic wetland plants for Lake Simcoe has yet been developed. Some macrophyte studies have focused on submergent species in the pelagic zone of Lake Simcoe (Ginn 2011; Stantec Consulting 2007; Winter et al. 2007), while the near shore zone, where fringe wetlands are located, have been relatively ignored. Yet it is the near shore zone that exhibits the most visible effects of water quality degradation, such as increased nutrient flows and higher nutrient concentration, as the anthropogenic activities that contribute to the degradation often occur near the shoreline (LSRCA 2011). The near shore habitat is influenced greatly through landscape changes from agriculture and urbanization, effluent discharges, and other human activities (fishing, boating, and hunting) (LSRCA 2013b). Since fringe wetland macrophytes grow in the near shore zone, they are the communities exposed to these anthropogenic activities directly, and therefore are the ideal community to observe water quality changes. The LSRCA (2011) has recommended that limited surveys in target locations be completed to capture seasonal changes in vegetation species composition and biomass, therefore studying emergent fringe wetlands addresses these 'limited' and 'targeted' research gaps. The data generated will help to design a macrophyte based water quality index for fringe wetlands in Lake Simcoe. The study will emphasize on emergent species of the macrophyte community. This study will therefore help to design a more cost effective management strategy for Lake Simcoe through the examination of community composition and dynamics and the representative water quality it provides.

The remainder of this thesis is divided into 4 chapters. Chapter 2 describes the general methodology employed in this study. Chapter 3 describes the spatial and temporal variation of limnologic and macrophyte data collected from six different fringe wetlands around Lake Simcoe exposed to varying degrees of exposure to anthropogenic stressors. Water quality was measured as a proxy of site disturbance. The survey was carried out four times to represent different seasons of the year. Several objectives and hypotheses are tested in this chapter. The limnologic objectives were to analyze the variance of water chemistry at all sites and seasons to determine if water quality reflects the degree of anthropogenic disturbance impacting the wetland, and it is hypothesized that water quality will reflect the degree of anthropogenic degradation influencing the wetland. The objective of the macrophyte survey is to determine dominant species at each site; and analyze the variation in species richness, diversity, stem density, and above-ground biomass between sites and seasons. The hypothesis tested was the emergent macrophyte community and metrics including species richness, density, diversity, and above-ground biomass will change with the quality of the wetland water which reflects the wetland disturbance. Chapter 4 describes macrophyte community dynamics as an index of water quality changes caused by site disturbance. The objective of this chapter is to explore the relationship between macrophyte species and specific limnologic parameters. Investigation of this relationship will reveal the responses of different species to various water quality parameters. The hypothesis tested was that particular indicator species show distinctive relationships with the limnologic parameters measured in this study. The responses and tolerance of indicator species to water quality will be compared with accounts in the literature, and are expected to support the potential use of emergent macrophytes in Lake Simcoe's fringe wetlands as indicators of water quality, and wetland integrity. Finally, Chapter 5 provides an overall summary, conclusion and suggestions related to future research.

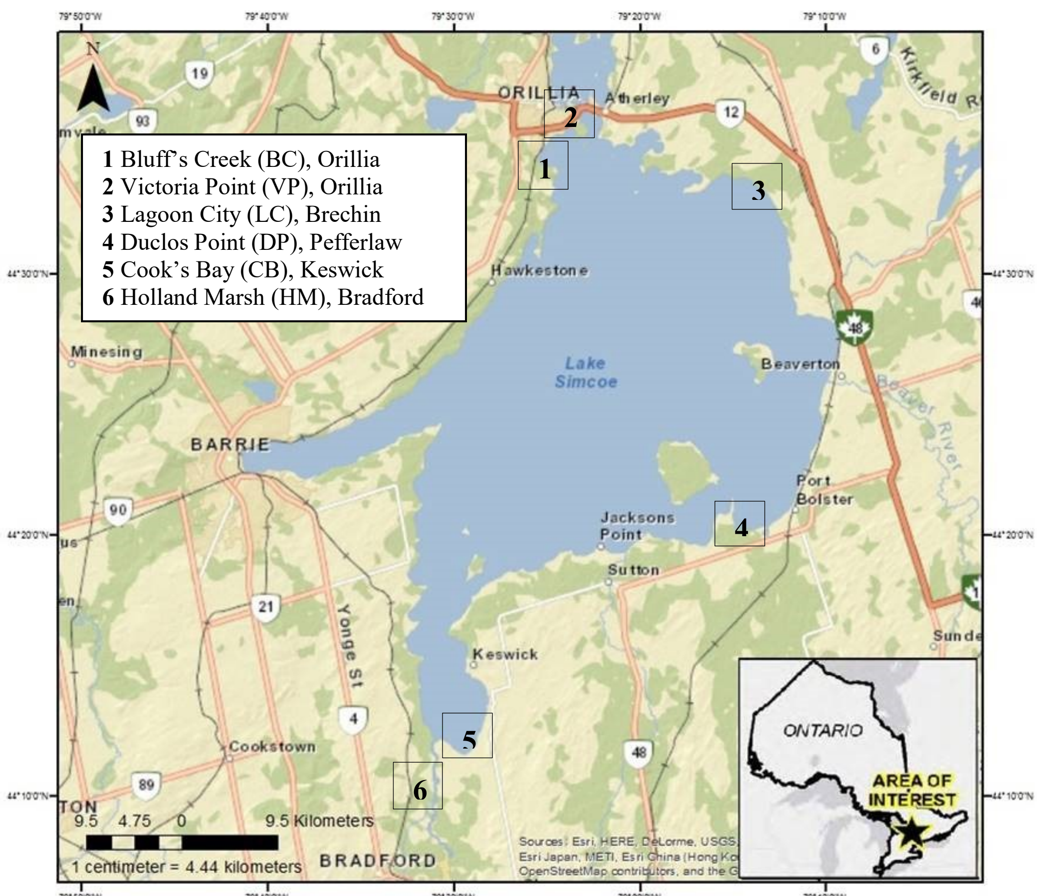
Chapter 2 Materials and Methods

2.1 Study location

The study was conducted at six sampling sites (Figure 1) located on northern, eastern, and southern Lake Simcoe, with one site being offshore of the lake. The wetlands sampled were Victoria Point (VP), Orillia, ON (44°35'38.18", -079°23'11.73"); Bluff's Creek (BC), Orillia, ON (44°34'31.9200", -079°25'32.8200"); Lagoon City (LC), Brechin, ON (44°33'06.0000", -079°13'18.2400"); Duclos Point (DP), Pefferlaw, ON (44°20'20.4600", -079°14'52.7400"); Cook's Bay (CB), Keswick, ON (44°11'38.4600", -079°29'05.7600"); and Holland Marsh (HM), Bradford, ON (44°09'46.9200", -079°31'15.7800"). Lake Simcoe is a hard water, dimictic lake with a surface area of 722 km^2 (Stantec Consulting Ltd. 2007). It is currently classified as meso-eutrophic, as the offshore water had a mean total phosphorus (TP) concentration of approximately 14 μg/L in 2008 (Ginn 2011). However, in some areas, oligotrophic and eutrophic conditions occur (Johnson 1997; Stantec Consulting Ltd. 2007). Lake Simcoe is the sixth largest inland lake in southern Ontario, after the Great Lakes (Johnson 1997), and is part of the Trent-Severn Waterway (TSW) that connects Lake Ontario to Georgian Bay. The lake itself drains northward into Lake Couchiching and from there into the Severn River. Lake Simcoe is divided by two large bays, Kempenfelt Bay near Barrie, Ontario (mean depth 14 m, maximum depth 42 m, surface area 34 km^2), and Cook's Bay, comprising the southern tip (mean depth 13 m, maximum depth 15 m, surface area 44 km^2), and the large shallow main basin which covers the northeastern portion of the lake (mean depth 14 m, maximum depth 33 m, surface area 643 km^2) (North et al. 2013; Winter et al. 2007; Young et al. 2010). The lake's water retention time is roughly 11 years and drains through the Atherley Narrows into Lake Couchiching (North et al. 2013; Young et al. 2010). The total Lake Simcoe watershed area is 2899 km^2 (North et al. 2013;

Palmer et al. 2011). Management of Lake Simcoe's water level is complex as it involves consideration of the entire TSW. Typically, in any given year, water levels vary by about 0.4-0.5 m. The highest levels usually occur between April and June, following snow melt, and will begin to drop as temperatures rise due to increased evaporation and evapotranspiration and reduced inflows. The lowest levels occur in late fall and winter. Drawdown is the deliberate lowering of water levels and is necessary to make room for high inflows and precipitation during the fall, winter and spring. It is important to note that Lake Simcoe water levels do not actually fluctuate all that much; historic lows were recorded at 218.4 m and historic highs at 219.5 m, and actual fluctuations over the course of any one year are even smaller (LSRCA 2013a).

Figure 1. Map of Lake Simcoe and the six sampling sites. Base map from ESRI 2015.



2.2 Sampling site description

Key dimensions of wetland stress as identified by Albert and Minc (2004) include hydrologic flow modification (through water regulation and dyking), water quality degradation (through nutrient loading, sedimentation, and chemical pollution), and ecological structural breakdown or physical degradation (through land-use alterations, filling, etc.). The sampling sites were selected based on their exposure to of one or more of these disturbances, and on the basis of a disturbance gradient they were assigned a degree of anthropogenic stress level as low, moderate, and high for studying the status of water quality and emergent macrophyte community dynamics. The wetlands were chosen to represent the range of human disturbance that are likely to be encountered across the various sections of the Lake Simcoe shoreline.

All of the sampling wetlands are designated as "Provincially Significant Wetlands" (PSWs). PSWs are those areas identified by the province as being the most valuable. They are determined by the science-based OWES described in section 1.2 (MNR 2013). Duclos Point wetland is part of the "Areas of Natural and Scientific Interest" (ANSI), which are areas of land and water containing natural landscapes or features that have been identified as having life science or earth science values related to protection, scientific study, or education. ANSIs are identified by the MNR to represent the full spectrum of Ontario's biodiversity, natural landforms and environments (MNR 1999).

Site 1: Bluff's Creek (BC), Orillia

Sampling in Bluff's Creek occurred in the fringing wetland area located on the shoreline of Shingle Bay, within the boundaries of urban Orillia (Figure 1). The sampling site, Bluff's Creek East Wetland, is identified as a PSW and is considered to have local significance for conservation and protection (along with the adjacent Bluff's Creek West wetland) (LSRCA 2013c).

Bluff's Creek is located in the Oro Creeks North subwatershed, which drains into Shingle Bay. The majority of the subwatershed (78.1%) is located within the Township of Oro-Medonte, with the remaining 21.9% in the northern portion of the subwatershed occupied by the City of Orillia (LSRCA 2013c). The most significant land use activities in this subwatershed are natural heritage (such as forests and wetlands) and agriculture, occupying close to 80% of its area. Ten percent of the subwatershed area is occupied by urban land use, the majority of this is found within the City of Orillia. There are approximately 1018 hectares of wetland in the Oro Creeks North subwatershed, which is approximately 13.5% of the landscape (LSRCA 2013Ac). Despite the large natural cover of the subwatershed, Site 1 itself is located within the urban boundaries of Orillia.

Bluff's Creek originates in an agricultural area and flows through urban and residential areas (LSRCA 2013c). The quality of water in Bluff's Creek has been compromised by agricultural practices, development, and pollutants such as salt and other contaminants from roads. This has resulted in an increase in the amount of phosphorous in the creek and in Lake Simcoe. Barriers to fish movement are also present within the creek. These barriers include perched culverts, small dams and online ponds (Ontario Streams 2009). Monitored sites within Bluff's Creek support successfully reproducing populations of brook trout, indicating areas of extremely healthy habitat in the vicinity of these sites, and both tributaries support other coldwater species such as mottled sculpin. Several branches of Bluff's Creek flowing through the Bluff's Creek East wetland are designated as municipal drains, and have the potential to impact this provincially significant wetland (LSRCA 2013c). The most significant instance of a municipal drain potentially impacting a natural heritage feature is in the lower reaches of Bluff's Creek, in proximity to the sampling site. The inputs into the drain consist of both overland flow and tile outlets and can carry contaminants, sediment, and debris into the drain (LSRCA 2013c). This site is considered to be a moderately stressed area. It is representative of a degraded water quality disturbance type.

Site 2: Victoria Point (VP), Orillia

Site 2 (VP) is located in the Victoria Point fringe wetland area on the shoreline of Smith's Bay just south of the Atherley Narrows within the City of Orillia (Figure 1). The entire Victoria Point wetland area is situated at the outlet of Lake Simcoe into Lake Couchiching. It is a PSW (LSRCA 2013c). The sampling site itself receives waters only from Lake Simcoe (O'Connor 2014).

As with Site 1, Site 2 is located in the Oro Creeks North subwatershed within the boundaries of urban Orillia, and is subject to various urban stressors. Site 2 is surrounded by commercial and residential buildings. The Narrows is home to many marinas thus attracting boaters (OMOA 2010). Further boat traffic is attracted by the presence of a public boat launch in Smith's Bay. Also in the vicinity of Smith's Bay is high traffic tourist public park/beach, Mara Provincial Park and Tudhope Park (Google Earth 2015c). This site is considered to be a moderately disturbed area. It undergoes hydrologic flow modification and water quality degradation.

Site 3: Lagoon City (LC), Brechin

Site 3 (LC) is located in the north western area of Lake Simcoe, in Lagoon City, Brechin, Ontario (Figure 1). The fringe wetland is a PSW that is located near a system of lagoons and canals that includes residential and commercial land use (County of Simcoe 2015; Ontario

Council of University Libraries [OCUL] 2015). Lagoon City is a popular tourist location for boaters and cottagers as a consequence of its easy access to Lake Simcoe and the Trent Severn waterway via its canal system. Prior to 1970s, the area of Lagoon City was a wetland (Lagoon City Community Association [LCCA] 2014). The sampling site is located adjacent to Lagoon City beach and on the shores of Lake Simcoe and is approximately 1.3 kilometers away the sewage treatment plant (Google Earth 2015b). This plant was constructed in the 1990s and treats wastewater for the 3,000 residents of Lagoon City (South Georgian Bay-Lake Simcoe Source Protection Committee [SGBLS] 2015). As a result of anthropogenic stressors such as a high intensity of boating, a direct release of “grey water” to the water, nutrient enrichment and degradation of natural terrain, Site 3 is considered as a moderately disturbed one. The treated effluent released from the sewage treatment plant represents a potential point source of nutrients. Overall Site 3 is representative of hydrologic flow modification, water quality degradation, and ecological structural breakdown disturbance types.

Site 4: Duclos Point (DP), Pefferlaw

Sampling in Duclos Point (DP) was carried out in a fringe wetland located within the boundary of the Morning Glory Provincial Nature Reserve (formerly known as the Duclos Point Provincial Nature Reserve) (Town of Georgina 2009), on the southeastern shoreline of Lake Simcoe (Figure 1). It is an ANSI and PSW (Ontario Parks 2007), situated in the Pefferlaw River subwatershed, which is considered to be a rural watershed, with just 5.5% of land being used as urban development. The rural/agricultural category accounts for 48% of the subwatershed, and natural cover accounts for 43%. The Black River subwatershed is one of the healthiest subwatersheds in the basin, especially in regards to natural cover. The health is attributed to the fact that in the 1980s this area was one of the first regions to have its wetland cover evaluated and classified as provincially significant. It contains a large valley that spans the central portion of the watershed, the Oak Ridges Moraine, which makes up 20% of the watershed and is protected under the Oak Ridges Moraine Conservation Plan (LSRCA 2012).

The Nature Reserve includes areas of open water, shallow marsh, deep marsh, thicket swamp and deciduous and coniferous swamp forest (Ontario Parks 2007). It protects wetland habitats, including open marsh, swamp and two small streams, and represents a large portion of a PSW marsh-swamp complex in the watershed. The park contains a natural shoreline, sandbar and associated backshore marsh and is one of the few remaining undisturbed lake front wetland complexes on Lake Simcoe (Hanna 1984; as cited by Ontario Parks 2007). Furthermore, the backshore marsh complex is specifically unique to Lake Simcoe (Lake Simcoe Environmental Management Strategy [LSEMS] 2008).

The emphasis of nature reserves is to provide representation of and protection for Ontario's geological, ecological and species diversity. Low intensity day-use activities are encouraged. The park does not have any parking or staging areas, and can only be accessed by pedestrians from the nearby road. The 1km long, unopened access road that enters the park is often flooded and unusable; furthermore, no future road development is permitted. An entrance gate restricts vehicular access. Tourists do frequently access the park and adjacent wet beach by boat via Lake Simcoe. In summer months there can be large numbers of boats intermittently moored off the park shoreline for recreation activities. However, care is taken to ensure that these activities do not adversely impact the natural values of the reserve. The lands adjacent to the park to the south, east and west are private and are rural residential, residential or vacant. There are also no dams, water control structures, or diversions within the park boundaries that might disrupt hydrology patterns (Ontario Parks 2007).

In order to understand patterns of regional ecological conditions it has become necessary to establish expectations based on least-impacted areas. These benchmarks of biological expectations provide best estimates of acceptable or desirable ecological conditions and should represent 'natural' conditions of a region (Simon et al. 2001). A crucial component of a biological assessment program is the careful selection of least-impacted reference sites (US EPA 2002). Anthropogenic influences are considered at a lesser degree at this site compared to the five other sites, justified by the reasons above. Therefore Site 4 is considered as the least disturbed site in this study. It is representative of minimal to low anthropogenic disturbance type.

Site 5: Cook's Bay (CB), Keswick

Site 5 (CB) is a fringe wetland located on the shoreline of Cook's Bay on the southern tip of the bay, in the East Holland River subwatershed (Figure 1). It is part of the Holland Marsh Wetland Complex and is characterized as a PSW (OCUL 2105).

The East Holland River subwatershed is one of the most urbanized, populated subwatersheds in the Lake Simcoe basin, with17.3% of the land use being urban area. The largest land use is designated as natural heritage features including forests and wetlands at 32.6%, and secondly intensive and non-intensive agriculture at 30.7%. The cumulative effects of land conversion for agriculture, urbanization and other uses that involve increases in impervious surfaces, plus other additional stressors have caused this subwatershed to be one of the most stressed, and to be one of the largest contributors to Lake Simcoe's phosphorus loads (LSRCA 2010a). Cook's Bay itself is shallow (average depth of 3.4 m) and receives input from several tributaries including the Holland River. The Holland River drains the Holland Marsh, and the high-intensity agricultural activities in the Marsh can contribute significantly the contribution of phosphorus to Lake Simcoe, as well the Holland River watershed which includes the large urban centers of Newmarket, Aurora and Bradford all contributing to phosphorus loading (Stantec 2007).

Land uses in the watershed have had substantial impacts. The wide-ranging marsh network occupies a broad valley originating from Cook’s Bay and extends southwest towards Schomberg. Since 1925, large tracts of this marsh area have been dyked and drained for agricultural purposes (Ontario Department of Energy and Resources Management 1966; as cited by LSRCA 2010a). In the East Holland’s agricultural areas, stressors include the input of nutrient-laden sediment eroding from unstable banks and fields, the use of large volumes of water for irrigation, the elimination of streambank vegetation, and the transfer of storm water directly to watercourses via tile drainage (LSRCA 2010a). The Holland River system, a cultivated marsh or polder, feeds into Cook’s Bay. The water level within the polder is maintained through a series of canals and pumping stations which discharge excess water to Cook’s Bay (Eimers, Winter, Scheider, Watmough, & Nicholls 2005).

Water quality and quantity have declined as a result of the introduction of harmful elements from both urban and rural areas and because of the increasing amount of impervious surfaces in this rapidly urbanizing watershed. Urban areas make up 14% of the East Holland subwatershed and approximately 20% is impervious, exceeding Environment Canada’s Area of Concern (AOC) proposed 10% impervious guideline. There are numerous consequences resulting from the high level of impervious surfaces in the East Holland’s urban areas, including reduced infiltration of rain and melt water, which can result in low groundwater levels in the watercourse areas. There is an effect on water quality as contaminants are carried with storm water runoff; streambank erosion and instability increase; and there are adverse effects on stream habitats due to sediment deposition or disruption of natural riffle-pool sequences, resulting in alterations to the composition of aquatic communities and biodiversity (LSRCA 2010a).

In addition, there are issues associated with other activities in the subwatershed such as recreation and industrial uses, including water consumption, the introduction of invasive species, and the input of nutrients and other contaminants. The summative effects of these activities have caused the East Holland to be one of the most stressed subwatersheds in the Lake Simcoe watershed, and one of the largest contributors to Lake Simcoe's phosphorus loads (LSRCA 2010a).

Adjacent land to the south and west of the sampling site is intensely farmed land (polders), and to the east are located residential buildings in the town of Keswick. There are also four Water Pollution Controls Plants that are introducing effluent into Cook's Bay directly as point sources of phosphorus contribution, (Stantec 2007; LSRCA 2013b). The closest point source is 4.5 km from Site 5 (Google Earth 2015a). For these reasons, and the description provided above, Site 5 is considered a highly disturbed area. It is representative of a site experiencing hydrologic flow modification, historic water quality degradation, and severe breakdown of ecological functioning.

Site 6: Holland Marsh (HM), Bradford

Sampling in the Holland Marsh occurred in a fringe wetland area located at the end of Line 10 in Bradford, east of Yonge St., where the road meets the West Holland River (Figure 1). It is classified as a PSW (OCUL 2015).

Site 6 is located in the West Holland River subwatershed, where the cumulative effects of anthropogenic activities have caused the West Holland to become one of the most stressed subwatersheds in the Lake Simcoe watershed, and one of the largest contributors to Lake

Simcoe's phosphorus loads (LSRCA 2013Ac). Land use in the subwatershed is primarily agricultural, occupying 58% of the subwatershed area. Natural areas contribute 31%, and 2.6% is occupied by urban areas, though this component is quickly expanding. Agricultural areas are found throughout the subwatershed, with natural areas interspersed throughout. The Holland Marsh is a significant feature, running northeast from Highway 9 to Cook's Bay (LSRCA 2013Ac).

This large wetland complex was drained for agricultural purposes during the 1920s and 1930s (O'Connor 2014). Conversion to agricultural activities has caused natural heritage areas to decline in quality and integrity, resulting in increased fragmentation, a decline in species richness, increased predation, and increased edge effects. As farming continues to expand, increasing levels of contaminants, including phosphorus are being released into Lake Simcoe. Consequently, many of the areas categorized as natural heritage sites, including wetlands, have degraded ecological integrity and habitat functionality. Impacts from the agricultural areas include loss of riparian vegetation; sediment-laden water; the consumption of enormous volumes of water for irrigation, and changes to the hydrology through artificial polder system implementation, channelization, and rapid diversion of storm water directly to the lake by tile drainage (LSRCA 2010b).

Urban areas in the subwatershed are expected to increase in the near future by roughly 4% (an additional 1,557 ha) with the majority of this increase attributable to high intensity urban development, with wetlands projected to decrease by approximately 27 ha (Louis Berger Group 2006). Impacts from the high level of impervious surfaces include reduced infiltration of rain and melt water, which can result in decreased baseflow and low groundwater levels and in the watercourse areas. This has an effect on water quality as contaminants are carried with storm water runoff; there is increased streambank erosion and instability causing negative effects on stream habitats due to sediment deposition or disruption of natural riffle-pool sequences, resulting in alterations to the composition of aquatic communities and biodiversity.

In addition to these stresses, there are issues associated with other activities in the subwatershed such as recreation and industrial uses, including water consumption, the introduction of invasive species, and the input of nutrients and other contaminants into area watercourses (LSRCA 2010b).

Specific sources of stress in proximity to Site 6 include a point source Water Pollution Control Plant located at 225 Disette St. in Bradford, discharging into the West Holland River (LSRCA 2013b) just over 5 km upstream from the sampling site (Google Earth 2015b). There is also large on-line pond, where the volume of water in the pond is controlled by a berm or other form of control structure. On-line ponds restrict the natural streamflow as a large volume of water becomes contained in the pond (LSRCA 2010b). Immediate stressors include non-point sources of nutrient enrichment from surrounding agricultural land, and recreational use of the site by hunters and anglers. This site is considered to be a highly anthropogenically disturbed area. It is representative of a site experiencing hydrologic flow modification, water quality degradation, and ecological structural breakdown. While the sampling site is located on the shore of the Holland River and not actually on Lake Simcoe, it was considered necessary to sample this location as a tributary of Lake Simcoe.

2.2.1 Macrophyte Survey

2.2.1.1 Field Work

Before the actual sample collection took place each wetland was subjected to pre-field investigations. Pre-field investigations involved the assembly of maps, aerial photographs and past reports to determine the suitability of each wetland site to be included in the study. Maps and aerial photographs helped to plan access routes, assess wetland size, plant community distribution, habitat characteristics that may influence plant distribution, and identify sampling obstacles or hazards. A field reconnaissance followed, to visit each wetland site, to verify observations made from maps and aerial photographs, to choose the sampling location, and to develop a first impression of the plant communities present and their distribution. Pre-field investigations were also necessary to ensure that the wetland sampling sites would represent a disturbance gradient of increasing anthropogenic disturbance from low to high.

At each sampling location, 7 quadrats were randomly placed to collect emergent macrophytes in wet meadow and emergent marsh zones. Macrophytes and water quality parameters were collected during 4 sampling periods: Sampling Period (SP) 1 (Fall 2013): October 16, 2013 - November 7, 2013; SP 2 (Spring 2014): June 10 - 17, 2014; SP3 (Summer 2014): August 19 – 28, 2014, and SP4 (Fall 2014): October 9 - 28, 2014. Quadrat locations were determined based on accessibility (water depth and density of emergent cover) and wetland size. Consequently, quadrats for each season and site were randomly selected based on these factors. The 7 quadrats ($0.0625 m^2$ each) were identified as 1-1, 1-2, 1-3, 2, 3, 4, and 5. Quadrats 1-1, 1-2, and 1-3 are replicate quadrats that were collected in close sequence or proximity (within 5-10 meters of one another) and that provide a rough estimate of the overall precision associated with the field technique. Quadrats 2 to 5 were generally 15-30 m apart, but with the exception of HM SP3 and SP4, where these quadrats were only 3-10 meters apart due to limited accessibility (density of plant cover limited access). The specific distances for each were recorded. A metre stick was used to measure a $0.25 m^2$ length and width of the quadrat and 4 small wooden poles were used to mark the quadrat area. The coordinate location was recorded using a GPS and the depth of the water (if applicable) was measured in metres using a metre stick. If no water was present then the type of substrate was noted. Within each quadrat all standing crop (above-ground) biomass was collected, including emergent, submergent, and floating vegetation. Water depth and flow allowed specimens to be collected by hand by wading through the wetland. This permitted a detailed and comprehensive evaluation of the macrophyte community. Manual samplers – cutting shears – were used to cut macrophytes because shears can be manipulated in deeper or shallow water, they are relatively inexpensive, and were easily purchased from a commercial store. Cutting shears were used to obtain estimates of above-ground biomass (standing crop) from quadrats (Madsen 1993). Above-ground biomass is used because of the challenges of collecting underground plant parts, such as rhizome and roots. It should be noted however that without the underground parts, the data do not represent total primary production (Eaton, Clesceri, Rice, & Greenburg 2005). Macrophytes were placed in garbage bags and stored in the freezer until identification could take place. Other data collected to characterize vegetation included measures of abundance in terms of density, frequency, cover, and above-ground biomass. Quadrat data were used for estimates of species frequency, stem density, species diversity, calculation of species richness and total biomass (Eaton et al. 2005). It should be noted that because of accessibility and safety concerns, the entire expanse of the wetland could not be sampled; therefore some species that were present may have been missing from the quadrat samples.

2.2.1.2 Species Identification

Macrophytes were removed from freezer and allowed to thaw prior to identification. The identification of unknown plant material was accomplished using dichotomous keys, published plant descriptions, illustrations and photographs. Specimens were also compared with properly identified herbarium specimens from Simcoe County donated to Lakehead University. Table 1 lists all resources referenced during identification. All identifications were accomplished with the assistance of a wetland macrophyte expert (Robert (Bob) Bowles, who has served as an environmental consultant for various provincial ministries and is a naturalist who has a long-established relationship with Lakehead University). Macrophyte identification included macroscopic/organoleptic techniques and microscopic analysis (Health Canada 2015). Macroscopic techniques involved the unaided senses of sight, smell or taste, as well as the use of a hand lens with 4-20x magnification for visual identification. Analysis was based on both defined morphological and/or anatomical characteristics of the whole plant or individual plant parts (e.g., leaf, flower, fruit, seed, bark) and characteristic colour, fracture, smell, or taste (U.S. Food and Drug Administration 1999). Identification was done by comparing the morphological characteristics with herbarium reference plants and other sources of references (Table 1). Microscopic identification involved the use of a dissecting microscope to check histological characteristics of plant parts (e.g. stems, bark, leaves, flowers, seeds, wood); vegetative and floral structures, as well as cell and tissue structures (Radford, Massey, Dickison, & Bell 1974). A dissecting microscope was used for many plants, particularly the aquatic grasses and sedges. Nearly all classifications and keys are based on the sexual parts of the plant (the flowers and the fruit). A major reason for this is that floral parts tend to remain much more stable over time and under different environmental conditions than do the vegetative parts and they better reflect the truly distinctive characteristics of particular plant species. The classification of the angiosperms is mostly based upon comparative studies of the structure of their flowers and fruits (Wondafrash 2008). While wetlands were visited once during each season, certain species are difficult to identify accurately without flowering parts (Lougheed et al. 2001), consequently some samples were identified to genus only. However, other features of their ecology and physiology, as well as vegetative characters can also be extremely useful in identification. Basic features used to identify specimens included: habitat, habit (tree or shrub, or herb), morphology (leaves, reproductive characters (inflorescence)); and perianth (B. Bowles, personal communication, January 8 2015; Wondafrash 2008).

Table 1. Identification keys and manuals used for macrophyte species identification.

Title	Author
Manual of Vascular Plants of Northeastern United States and Adjacent Canada	Henry A. Gleason and Arthur Cronquist.
Shrubs of Ontario	James H. Soper and Margaret L. Heimburger
Manual of Aquatic Plants	Norman C. Fassett
Aquatic and Wetland Plants of Northeastern North America: A Revised and Enlarged Edition of Norman C. Fassett's A Manual of Aquatic Plants	Garrett E. Crow and C. Barre Hellquist
Sphagnum Mosses of Ontario: Identification by Macroscopic Features	Haavisto, V. F.
Illustrated companion to Gleason and Cronquist's Manual: Illustrations of the Vascular Plants of Northeastern United States and Adjacent Canada	Noel H. Holmgren
Mosses of Michigan	Henry T. Darlington
A Graphic Guide to Mosses	Robert Muma
Woodlot Biodiversity	Dr. Steven Newmaster, Chris Earley, Dr, Aron Fazekas, Carole Ann Lacroix, Troy McMullin, Brain Lacey, Jose Maloles, Thomas Henry, and Peter Williams
Mosses, Liverworts and Lichens of Elgin County Ontario	William G. Stewart
Mosses	Howard Crum
Mosses With Hand-Lens and Microscope	A.J. Grout

2.2.2 Biomass Measurements

To measure the fresh weight of the macrophytes, they were washed to remove silt and debris and excess moisture was removed with absorbent tissue paper. Mass was determined to the nearest 0.1 g. To remove all water, and assess dry mass macrophytes were oven-dried at 50°C - 75°C for 4 days, or until constant mass was achieved. In each instance all macrophytes from each individual quadrat were sorted by species (or genus) and weighed separately.

2.3 Water Parameters

On each field sampling occasion, water samples were collected to assess parameters such as dissolved oxygen (DO), conductivity (COND), pH and temperature (TEMP). Water pH, dissolved oxygen and conductivity were measured in-situ using a hydro lab sampler (VWR Symphony SB9 0M5), ambient water temperature was measured using a calibrated Fisher Scientific thermometer. Surface water samples from each sampling area were also collected for nutrient analysis and chlorophyll *a* using clean 1L polyethylene bottles. Unfiltered water samples for nutrient analysis were stored in a freezer at -18°C until analyses were carried out at Lakehead University Analytical Laboratory (LUEL) at the Lakehead University, Thunder Bay campus.

Chlorophyll *a* (CHL *a*) measurements provide a surrogate for actual measurements of algal biomass, and can be used to estimate of the level of primary productivity in the water column, which may contribute to water turbidity, along with other suspended solids (Bruckner 2016). In this study, chlorophyll *a* from the water column was measured according to the APHA standard protocol (APHA 2005). This was done by filtering 1 litre of the water sample through a 0.045mm Whatman GF/B 47mm glass fibre filter and extracting chlorophyll *a* in 12 ml of 90% acetone at 4°C for 16-24hr. Upon extraction, the samples were centrifuged for 15 minutes at 4200 rpm, the supernatant was extracted using a Pasteur pipette and transferred into a 10ml acid washed, glass test tube which was inserted directly into a Beckton Dickinson Spectrophotometer (DU700) where the corrected absorbance measurements (750, 664, 647, and 630 nm) were recorded and the chlorophyll *a* concentration was calculated and expressed as mg·m^{-3} (APHA2005).

Total Phosphorus (TP) (uv digestible) and Total Nitrogen (TN) (uv digestible) analysis was completed at the Environmental Laboratory at the Lakehead University Centre for Analytical Services in Thunder Bay, ON. For TP, polyphosphate and some organophosphorus compounds are determined by conversion to ortho-phosphorus using acid hydrolysis. This was performed online with the Skalar autoanalyzer system by adding sulphuric acid to the sample stream and heating at 97°C. Following hydrolyzation, the sample underwent further digestion with peroxodisulfate under uv radiation generating the orthophosphate ion. The ortho-phosphate ions in a sample reacted online in an acidic solution containing molybdate and antimony ions to form a phospho-molybdic acid, which was reduced by ascorbic acid to an intensely blue complex which is measured at 880 nm (SKALAR Methods for Total Phosphorus in Water Catrn# 503-010) (J. Joncas, personal communication, June 30, 2015). TN was analyzed using the Skalar autoanalyzer system; the sample was mixed with a potassium peroxodisulfate/sodium hydroxide solution and heated to 90°C. The solution was then mixed with a borax buffer and all nitrogen species were converted by uv radiation to nitrate. Colourimetric determination followed WNOX. This method accounted for nitrogen in the form of azide, azine, azo, hydrazone, nitrate, nitrite, nitro, nitroso, oxime and semi-carbazone, as opposed to the Total Kjeldahl Nitrogen method (SKALAR Methods for Total Nitrogen in Water Catrn# 475-426) (J. Joncas, personal communication, June 30, 2015).

2.4 Statistical Analysis

Data related to description of the vegetation included measures of abundance including frequency, stem density, species above-ground biomass, species richness, and species diversity. Percent composition of native, introduced, and invasive species, Importance Values and similarity coefficients of species were also determined. Methods of determination are presented in detail in Chapter 3. Frequency was expressed as the number of times a species appeared. Density was expressed as the number of stems of a species observed per unit area. Stem density was determined due to the large number of clonal species that spread via rhizomes and stolons, which makes collection of individuals difficult. Density was also used to determine percent composition, which was expressed as the proportions (%) of various plant species in relation to the total on a given area. Species richness was assessed as the number of species per unit area. Diversity was described using Simpson's Reciprocal Index of Diversity. Dominant species were determined based on the total stem density of each species present. In this study dominance is defined as the species having highest stem density at each site during each sampling period.

One-way ANOVA was conducted to determine whether there were any statistically significant differences between the means of two or more independent groups. Water quality parameters and macrophyte measures of richness, density, diversity, and above-ground biomass within study period and between sampling sites were compared using one-way ANOVA (IBM SPSS Statistics, Ver20, SPSS Inc, Armonk, New York, USA). If species dynamics were different for different seasons and different locations, *post hoc* tests were carried out using a Bonferroni Correction to test pair-wise comparisons. In several cases, the data failed to meet the assumptions of the one-way ANOVA tests. When the Shapiro-Wilk test for normality showed the data did not meet parametric assumptions, the one-way ANOVA was carried out regardless of non-normality because the sample sizes (numbers in each group) were equal, a condition which contributes to the robustness of the one-way ANOVA to deviations from normality (Lund & Lund 2013). If inspections of boxplots revealed outliers in the data, they were not removed. The outliers were included in the analysis for a number of reasons. One reason was because there were a limited number of data points. The outliers were also not removed because a one-way ANOVA was run with and without the outliers being included in the analysis. The results were then compared and the conclusions both resulted in a statistically significant result (the conclusions were essentially the same). As a result the outliers were retained in the data set. If the assumption of homogeneity of variances was violated ($p > 0.05$), as assessed by Levene's test for equality of variances, then the Welch ANOVA was applied (Lund & Lund 2013).

Two-way ANOVA analysis was conducted to examine the combined effects of sampling period and sampling site on limnologic parameters and macrophyte dynamics (IBM SPSS Statistics, Ver20, SPSS Inc, Armonk, New York, USA). Residual analysis was performed to test for the assumptions of the two-way ANOVA. Outliers were assessed by inspection of a boxplot and were removed if present, normality was assessed using Shapiro-Wilk's normality test for each cell of the design and homogeneity of variances was assessed by Levene's test. If a statistically significant interaction between sampling period and site was revealed an analysis of simple main effects was performed with statistical significance receiving a Bonferroni adjustment and being accepted at the $p < 0.05$ level. All pairwise comparisons were run for each simple main effect with reported 95% confidence intervals and p-values Bonferroni-adjusted within each simple main effect. If the interaction effect was not statistically significant it was followed up by an analysis focusing on the main effects.

Multiple regression was run to see if macrophyte richness and diversity could be predicted by any of the measured limnologic parameters (IBM SPSS Statistics, Ver20, SPSS Inc., Armonk, New York, USA). In all analyses the assumptions of linearity, independence of errors, homoscedasticity, unusual points and normality of residuals were met.

Measures of density and above-ground biomass (grouped into sites of their disturbance levels) were ordinated using Canonical Correspondence Analysis (CCA) and Redundancy Analysis (RDA) using CANOCO 4.5 software to examine the relationship between density and above-ground biomass and limnologic parameters. Detrended correspondence analysis (DCA) was initially used to verify that the species data had strong or weak unimodal distributions across the environmental (water quality) gradient. If DCA analysis revealed the data had only a weak unimodal or more linear distribution (lengths of gradients were less than 4 for axes 1 to 4, respectively), RDA, a constrained linear canonical ordination technique, was used (Leps & Smilauer 2003). If DCA showed the data to have a unimodal distribution (at least one gradient length was 4 standard deviations or greater for axes 1 to 4), CCA was appropriate. CCA and RDA were used to ordinate the species along the environmental gradient, where the ordination is constrained by the environmental variables. Environmental variables were standardized to a mean of 0 and a standard deviation of 1. The forward selection option was implemented to both rank the importance of limnologic parameters to species distribution patterns and to exclude less relevant environmental variables from the model. The procedure was as follows: the environmental variable best fitting the species data was selected first, followed by the inclusion of the next best fitting variable, and so forth, until additional variables no longer significantly contributed to explaining the observed variation. The significance of the explanatory effect of an environmental variable was determined by using the Monte Carlo permutation test (449 permutations, α= 0.05) prior to the addition of the next best fitting variable (Reyes & Kneeshaw 2008). Variables entered into the CCA and RDA included the presence of macrophyte taxa except mosses, as well as the corresponding environmental variables (i.e., SITE, SP, TEMP, pH, DO, COND, TP, TN, CHL *a*, DEPTH). The analysis was re-run without those variables not significant to the model (α >0.05) without forward selection. Consequently, all included variables had variance inflation factors <20, indicating that they contributed uniquely to the analysis. Since CCA has the tendency to over-emphasize rare species, these species were downweighted (ter Braak & Smilauer 2002); RDA does not give rare species high weighting. Bi-plot scaling was used and the scaling was focused on interspecies distances. Points in the ordination plot were based on linear combination scores (biological data are described in relation to the environmental variables), which is the standard method in CANOCO 4.5 (Lougheed et al. 2001).

Chapter 3 Macrophyte Survey and Water Quality Parameters in Lake Simcoe's Fringe Wetlands

3.1 Introduction

It has been well recognized for over two decades that different amounts and types of nutrients exported from agricultural and urbanized landscapes have a direct impact on the quality of the Lake Simcoe water system (Ginn 2011; LSRCA & Bradford 2011; Ministry of the Environment 2007; Stantec Consulting Inc. 2007). By comparison, the impact of these different land uses on the water quality of wetlands specifically in the watershed has seldom been documented, and even less so for Lake Simcoe's fringe wetlands. However, studies have been done within the Great Lakes coastal wetlands linking anthropogenic land use activities to water quality of the wetland. Crosbie & Chow-Fraser (1999) created a multidimensional index of trophic quality for 22 marshes to test the relationship between that index and the relative distribution of agricultural, urban, and forested land in each wetland's respective watershed. A significant relationship was determined: Principal Component scores in relation to water turbidity, total suspend solids, total organic suspended solids, total inorganic suspend solids, total phosphorus, chlorophyll, and total NH_4-N varied directly with the proportion of agricultural land, ultimately revealing that concentrations of phosphorus, nitrogen, and inorganic suspended solids in wetlands increased predictably as agriculture became the dominant land use in the respective watersheds. Accordingly, these wetlands with watersheds that were dominated by agricultural areas tended to be turbid and nutrient rich, an indication of degraded water quality. In a study of the primary determinants of macrophyte community structure of 62 fringe and inland marshes, Lougheed et al. (2001) collected water quality, land use and aquatic macrophyte information and found that the proportion of agricultural and urban land in wetland watersheds was a highly significant predictor of water quality, which coincides with the results of Crosbie & Chow-Fraser (1999). Their study also described differences in the sources of water quality degradation between inland and coastal wetlands. For inland marshes of the Great Lakes, the primary source of water quality degradation was excess inputs of nutrients and sediment associated with agricultural development in the watershed. For coastal wetlands, however, water quality may have also been influenced by mixing with water in the lake itself.

As aquatic plants are tightly linked to the functional capacity of wetlands and with water quality parameters, the impact of anthropogenic activities, and resulting degradation of water quality is evident in the species density, richness, and dynamics of wetland macrophyte vegetation (Lougheed et al. 2001). As anthropogenic disturbances increase over time, the ecological integrity of the wetland is lessened due to changes in processes such as nutrient cycling, photosynthesis, hydrology, predation or competition (Karr 1993). At the community level, anthropogenic disturbances tend to alter community composition and species richness (Cronk & Fennessy 2001). Odum (1985) also discussed plant community composition and predicted general effects of anthropogenic disturbances in wetlands, including an increase in the proportion of r-strategists, decrease in the mean size of plant species, decrease in the mean life span of plants or plant parts (leaves), shortened food chain, decrease in species diversity, and increasing dominance by fewer species. Within the Great Lakes, agricultural and urban land use in wetland catchments of the lower lakes has been shown to affect nutrient enrichment, water clarity and sediment quality (Crosbie & Chow-Fraser 1999) and will therefore likely have profound effects on macrophyte growth (Crosbie & Chow-Fraser 1999; Magee et al. 1999) and distribution (Crosbie & Chow-Fraser 1999; Lougheed, Crosbie & Chow-Fraser 1998). The same outcome is expected in Lake Simcoe fringe wetlands as well.

The quality of the water in a wetland is related to the health or integrity of the wetland ecosystem. Degradation of water quality of a main lake basin potentially indicates that the health or biological integrity of both the entire lake ecosystem and localized fringe wetlands is degraded. The degraded health of the wetland caused by water quality degradation by anthropogenic stressors is reflected by the macrophytes present in the wetlands (Simon et al. 2001).

There is a need to describe Lake Simcoe's coastal wetland water quality across the full range of anthropogenic impacts, particularly agriculture and urbanization. It is also important to note that water chemistry is not only influenced by the type and amount of anthropogenic land use but it is also dependent on natural factors, such as climate, hydrology, and geography – factors that should also be taken into consideration when monitoring Lake Simcoe's fringe wetland water quality, and relating it to anthropogenic stressors (Kurtz et al. 2006). Albert & Minc (2004) have stated that a description of the natural variability within coastal wetlands is necessary to understand the temporal and spatial variation in limnologic data and species distributions. These authors explored the potential for developing plant-based water quality indicators for Great Lakes wetlands exposed to varying degrees of stresses. They began their study by identifying the major ecological factors that generate distinct wetland types. Anthropogenic stress is not the only deciding factor for vegetation dynamics, as the natural variability within coastal wetland plant communities and major environmental factors are also influential (Albert & Minc 2004). Therefore, any application of the use of wetland plants as indicators of water quality or wetland integrity requires identification of the local and regional variability of wetlands (Minc 1997; Albert and Minc 2004). As in the Great Lakes, local and regional variability in the aquatic system, bedrock, geomorphology, land use, as well as temporal variability in Lake Simcoe's water levels will create a series of distinctive wetland types (Albert and Minc 2004). The morphology of the wetland can influence the type of aquatic vegetation through differences in exposure, wave action, sedimentation, and organic soil deposition (Trebitz et al. 2009; Cvetkovic, Wei & Chow-Fraser 2010).

Regional and site-specific information on Lake Simcoe's fringe wetlands is well-documented in Johnson (1997), LSRCA (2013b), and LSEMS (2003). Johnson (1997) described the landscape ecology of the Lake Simcoe basin. The largest portion of the drainage basin lies to the south of Lake Simcoe in an area where streams flow north into the lake through a landscape that is highly modified by agricultural and urban activities. The entire basin lies in a transitional portion of the southern Ontario landscape. To the south, deep fertile soils were deposited on top of limestone and shale bedrock by glacial activities. In the north, a thin layer of soil is sparsely spread over the limestone, shale, and granite bedrock that have been eroded in many places by glacial activities. The transitional nature of the basin, a varied geological past and over 200 years of intensive exploitation by European immigrants have created the complex ecological tapestry of the Lake Simcoe basin (Johnson 1997).

In contrast to the Great Lakes, Lake Simcoe lies within one ecoregion, the Lake Simcoe - Rideau Ecoregion (Crins, Gray, Uhlig, & Wester 2009). However, local heterogeneity in terms of topography and bedrock geology is very apparent. Table 2 summarizes the physical characteristics of the Lake Simcoe wetlands sampled during this study (LSEMS 2003; LSRCA 2010a; LSRCA 2013c).

Table 2. Classification of the studied fringe wetlands in Lake Simcoe.

Location	Unit Type	Bedrock	Geomorphic Context (Site Type)	Substrate	Significant Human Impacts
Southern (Keswick Marsh, CB)	Marsh	Limestone (sedimentary rock)	Lacustrine Open Embayment	Organic	Nutrient enrichment, urbanization, degradation of natural terrain
Southern (Holland Marsh, HM)	Swamp	Limestone	Riverine	Organic	Nutrient enrichment, hydrologic flow modification, urbanization
Southern (Duclos Point, DP)	Swamp	Limestone	Lacustrine Open Shoreline	Organic (peat and muck)	None or localized; minimal.
Northern (Victoria Point, VP)	Marsh	Shale (sedimentary), limestone	Lacustrine Open Embayment	Organic	Nutrient enrichment, urbanization, hydrologic flow modification
Northern (Lagoon City, LC)	Swamp	Shale, limestone	Lacustrine Open Shoreline	Organic, sandy loam	Nutrient enrichment, degradation of natural terrain
Northern (Bluff's Creek, BC)	Swamp	Shale, limestone	Lacustrine Open Shoreline	Organic, clay loam	Nutrient enrichment, chemical contamination, physical degradation

In this study, limnologic parameters were measured at the surface (at a depth of approximately 10 cm). Surface water is generally faster-moving, with higher dissolved oxygen concentrations and exposed to a greater risk of point source pollution (Credit Valley

Conservation 2010). Water depth was still measured as emergent cover tends to decrease as water depth increases (Tulbure, Johnston, & Auger 2007).

Measurements of pH can distinguish acidic (pH<5.5), circumneutral (pH 5.5 to 7.4) and alkaline (pH>7.4) waters (Tiner 1999). Provincial Water Quality Objective guidelines for flowing water for pH range from 6.5-8.5. Any deviation of pH from this recommended range can be lethal for fish and invertebrates. In addition, the toxicity of other water components (e.g. trace metals and carbon dioxide) may change with changes in pH (Ministry of the Environment 1979). These guidelines, however, may not be applicable in wetlands, as they were not designed for wetland systems (Credit Valley Conservation 2010).

Dissolved oxygen (DO) is the most fundamental parameter in water; oxygen is essential to the metabolism of all aerobic aquatic organisms (Canadian Council of Ministers of the Environment [CCMOE] 1999). In surface waters, DO may range from non-detectable to 18.4 $mg \cdot L^{-1}$ (NAQUADAT 1985; as cited by CCMOE 1999). Wetland systems tend to have naturally low DO concentrations (U.S. EPA 2009). In wetlands bacteria, protists (unicellular eukaryotes), plants and animals have adapted to deal with these naturally anoxic conditions (Mitsch & Gosselink 2007). In addition, in areas with aquatic macrophytes, oxygen levels are also affected by photosynthesis, with higher DO levels during the day when photosynthesis exceeds respiration, and lower levels at night when only respiration is occurring. Macrophytes can also affect DO by generating higher oxygen levels during the growing season and low levels later in the season during vegetation dieback and decomposition (CCMOE 1999). Measurement of DO also provides evidence of nutrient enrichment and eutrophication (Minc & Albert 2004).

Conductivity (COND) indicates the ionic strength of the water and therefore is a good indicator of urban run-off (Lougheed et al. 2001; Minc & Albert 2004). It is affected by the concentrations of inorganic dissolved solids such as chloride, nitrate, sulphate, and phosphate anions and sodium, magnesium, calcium, iron, and aluminum cations in the water. Conductivity increases with increasing water temperature and is therefore reported as conductivity at 25°C (U.S. EPA 1997). The implication of high conductivity levels is that water has become saline, which is stressful for plants with low to medium salt tolerance (Credit Valley Conservation 2010).

Chlorophyll is the major green photosynthetic pigment in cyanobacteria and algae. Chlorophyll *a* (CHL *a)* is a measure of the portion of the pigment of algae and cyanobacteria that is still active; that is, the portion that was still actively photosynthesizing at the time of sampling. The Lake Simcoe Monitoring Report (2014) demonstrated the connection between chlorophyll *a*, DO, and nutrients: high phosphorus loading increases phosphorus concentration, which increases chlorophyll *a* concentrations from algal and cyanobacterial growth, and ultimately affects dissolved oxygen concentration (Young & Jarjanazi 2014).Thus, high nutrient loads can lead to excessive algal growth and may subsequently contribute to high water turbidity (Chow-Fraser 1998). The turbidity of a body of water is related to the cleanliness of the water. Turbidity can be caused by high concentrations of biota such as phytoplankton, which is made up of green algae and cyanobacteria, or by loading of abiotic matter such as sediments (Bruckner 2016). Therefore, determining the concentration of CHL *a* in the water reflects the concentrations of suspended biota in the water, which, along with other sediments and particulate matter contributes to the turbidity of the water.

Phosphorus (P) is a key macronutrient required by biota for biological metabolism. It tends to be the least abundant macronutrient, and therefore, is most often the limiting factor for biological productivity. Water with a low P concentration tends to support relatively diverse aquatic life. However, high P concentrations can lead to adverse effects on aquatic ecosystems, such as anoxic conditions, increased turbidity, increased floral and faunal biomass, a decline in ecologically sensitive species and an increase in nutrient tolerant species, and decreased biodiversity (CCMOE 2004). Excessive P concentrations can also stimulate the production of algae and aquatic macrophytes (Eimers et al. 2005). Many other nutrients besides phosphorus are essential for aquatic plant growth and can affect ecosystem functioning, including nitrogen (N), which is available in many forms in aquatic ecosystems. Total N (TN) includes nitrate plus nitrite, ammonium and organic N (Young & Jarjanazi 2015), and provides an indication of nutrient enrichment from sewage, fertilizers, and manure, and the presence of industries; primarily wastewater treatment plants (Minc & Albert 2004).This chapter describes the spatial and temporal variation of water quality and macrophyte data collected from six different fringe wetlands around Lake Simcoe exposed to varying degrees of anthropogenic stressors. The study was repeated four times to represent different seasons of a year. Therefore this describes variations in limnologic parameters and macrophyte communities observed (a) among the six wetlands, and (b) with the four seasons.

Several objectives and hypothesis were tested in this study. The first objective was to analyze the variance of limnologic parameters at all sites and seasons to determine if water quality reflects the degree of anthropogenic disturbance impacting the wetland. It was hypothesized that water quality will reflect the degree of anthropogenic activities influencing the wetland, as water quality was measured as a proxy of site disturbance. The least anthropogenically disturbed wetland (DP) will have the healthiest water quality; the high anthropogenically disturbed sites (CB and HM) will have the poorest water quality, while moderately disturbed sites (BC, VP, and LC) will fall somewhere between the two extremes. The second objective was to identify the dominant species at each site, and analyze the variation in species richness, diversity, density, and above-ground biomass between sites and seasons. The hypothesis tested was emergent macrophyte community/composition and metrics including richness, density, diversity, and above-ground biomass, as well as indicator characteristics, will reflect the quality of the wetland water, a proxy of site disturbance. Specifically, the site with the highest water quality (and thus the least anthropogenically disturbed, DP), is expected to have higher species richness and diversity, but lower plant densities and above-ground biomass. Conversely, the sites with the poorest water quality, and thus highly degraded (CB and HM) will have the lowest species richness and diversity, but plants will be present at higher densities and will have greater above-ground biomass per unit area.

3.2 Methods

3.2.1 Study Sites

Detailed general methodology is described in Chapter 2. Briefly, the study was conducted at six sampling sites located in northern, eastern, and southern Lake Simcoe, and in its proximal watershed, namely Victoria Point (VP), Orillia, ON (44°35'38.18", -079°23'11.73"); Bluff's Creek (BC), Orillia, ON (44°34'31.9200", -079°25'32.8200"); Lagoon City (LC), Brechin, ON (44°33'06.0000", -079°13'18.2400"); Duclos Point (DP), Pefferlaw, ON (44°20'20.4600", -079°14'52.7400"); Cook's Bay (CB), Keswick, ON (44°11'38.4600", -079°29'05.7600"); and Holland Marsh (HM), Bradford, ON (44°09'46.9200", -079°31'15.7800") (Figure 1).

3.2.2 Macrophyte Sampling

Macrophyte samples from each wetland were collected from seven 0.0625 m^2 quadrats spread across the shoreline, therefore, in total, 28 quadrats were sampled at each wetland site over the four study seasons. Emergent macrophytes were quantified and identified to the species level using various identification keys and manuals (Table 1). Parameters collected to describe the vegetation included frequency, density, above-ground biomass, richness, and diversity. Composition of native, introduced, and invasive species, an Importance Value Index and similarity coefficients of species were also determined.

Several metrics were used to characterize the macrophyte community composition and dynamics. Frequency (F) refers to the degree of dispersion of individual species in an area and is usually expressed in terms of percentage occurrence. Density (D) is an expression of the number of stems of a species in a unit area. Density was used to determine the percent composition of native, introduced, and invasive species in the study, each being expressed as the percent (%) of total community at each site (University of Idaho 2009a). Relative frequency (RF) and relative density (RD) refer to the ratio of individual species to the total number of species in an area and the ratio of the number of individuals of a species to the total number of individuals of all the species in an area, respectively (Krebs 1999). In this study, RF and RD values were calculated for all species at each site for the entire study period. These two metrics were used to calculate the Importance Value Index (IVI) of each species at each wetland. This index is used to determine the overall importance of each species in the community. The IVI for each species was calculated as an average of its relative frequency and its relative density (Krebs 1999; Williams-Linera, Palacios-Rios, & Hernandez-Gomez 2005).

The macrophyte community analysis of each wetland allows assessment of the ecological similarities among wetlands in the study (Jongman, Ter Braak & Van Tongeren 1995). The Sørensen coefficient (S_S) was calculated to determine the similarity index between the reference site (DP), and the five other study sites. The Sørensen coefficient is a simple index that uses presence/absence data, and the similarity coefficient ranges from 0 (no similarity) to 1.0 (complete correspondence) (Jongman et al. 1995).

Stem species data were also grouped together to determine richness, density, above-ground biomass and diversity of each site and of each sampling period. Species richness is a measure of the number of species found in a sample. Species diversity differs from species richness in that it takes into account both the numbers of species present and the dominance or evenness of species in relation to one another (Krebs 1999). Above-ground biomass is used as an indicator of ecological processes in the vegetation. Plants that dominate a site, in terms of biomass, are likely to be the species that are controlling the nutrient, water, and solar energy resources. Consequently, biomass is often measured to assess the ecological status of a site. Measures of standing crop also reflect the amount of energy stored in the vegetation, which can indicate the potential productivity at the site (University of Idaho 2009b).

Briefly, the methods of determination of these metrics presented below:

(a) Frequency and Relative Frequency

Frequency (%) = (Number of quadrats in which a species occurred at each site / 28 (Total number of quadrats studied) X 100

Relative frequency (%) = (Number of occurrence of the species at each site / Number of occurrence of all the species at each site) X 100

(b) Stem Density, Relative Density and Dominance

Density = Total number of stems of a species in all quadrats at each site / 1.75m^2 (Total area studied; 28 quadrats x 0.0625m^2)

Relative density = (Number of stems of the species / Number of individual of all the species) X 100

(c) Percent Composition

Composition (%) = (Number of stems of each species at each site / Total number of stems at each site) X 100

Species were organized as native, introduced or invasive and percent composition of the species in each group was totaled.

(d) Importance Value Index

Importance Value Index = (Relative Density of each species at each site + Relative Frequency of each species at each site) / 2

(e) Index of Similarity

$S_S = 2a/(2a + b + c)$, where

S_S = Sørensen similarity coefficient,

a = number of species common to both quadrats,

b = number of species unique to the first quadrat, and

c = number of species unique to the second quadrat

S_S usually is multiplied by 100% (i.e., S_S = 67%) (Jongman et al. 1995)

(f) Richness

Richness = Number of different species in each studied quadrat

(g) Above-ground Biomass

Above-ground Biomass = Dry Mass (of above ground tissues) of each species / $0.0625m^2$ (Quadrat Area)

(h) Diversity

Species diversity was calculated using Simpson's Reciprocal Index.

$D = \Sigma(n/N)^2$, where

D= Simpson's Index

n= the total number of stems of a particular species

N= the total number of stems of all species

Simpson's Reciprocal Index was then calculated by 1/D.

3.2.3 Water sampling and limnologic parameters monitored

Water samples from each of the study sites were collected in cleaned 1L polyethylene bottles. Limnologic parameters monitored during this study included DO, COND, pH and TEMP. Ambient water temperature was measured using a calibrated Fisher Scientific thermometer. The water samples collected were used to monitor the nutrients (TP and TN) and chlorophyll *a*. Water samples for nutrient analysis were stored in a freezer until the analyses were performed at the Environmental Laboratory at the Lakehead University Centre for Analytical Services in Thunder Bay. The data were analyzed using several statistical tests. In order to understand the variation in data between sites of sample collection (spatial) and sampling period (temporal or seasonal), one-way ANOVA and two-way ANOVA was carried out (IBM SPSS Statistics, Ver20, SPSS Inc., Armonk, New York, USA). The details of the analysis are described in detail in Chapter 2.

3.3 Results

3.3.1 Comparisons of Limnologic Parameters

Wetlands in this study are experiencing a wide range of environmental conditions, ranging from oligotrophic (very clear and nutrient poor, e.g., TP =11.25 μg/L, TN = 345.25 μg/L, CHL *a* =1.37 $mg \cdot m^{-3}$) to turbid and eutrophic (e.g., TP = 47.25 μg/L, TN = 2285 μg/L, CHL a = 3.26 mg·m^{-3} (Table 3). The pH of water samples ranged from a minimum of 6.48 in Fall 2013 in HM, to 8.04 in Spring 2014 also at HM (Table 3). There were no statistically significant differences in pH between seasons ($F_{3,20}$ = 0.546, p=0.656), or sampling sites, ($F_{5,18}$ =1.085, p=0.402). The maximum water temperature was recorded during sampling period 2 at LC and DP (25°C). The lowest temperature value was recorded during SP1 and SP4 (both Fall) at HM and LC (5°C) (Table 3). There were no significant differences in temperature between sites ($F_{5,18}$ = 0.356, p=0.872). However temperatures did differ significantly between sampling periods, ranging from 5°C - 25°C ($F_{3,20}$=10.409 , p<0.05). The dissolved oxygen concentration did not differ significantly either between sites ($F_{5,18}$ = 0.496, p=0.775) or seasons ($F_{3,20}$ = 2.053, p= 0.139). Conductivity values ranged between 143-849 µS/cm (Table 3). The highest conductivity value recorded was at CB, located near high agricultural areas with significant non-point source pollution and urbanization. Conductivity differed significantly between sites ($F_{5,18}$=12.506, p<0.05), but did not differ significantly between sampling periods ($F_{3,20}$=0.501, p= 0.686).

Over the entire study period the water TN concentrations ranged from 289 to 6694 µg/L (Table 3). One-way ANOVA results showed a significant variation between sites ($F_{5,7}$= 20.916, p< 0.05), but not sampling periods of ($F_{3,10}$= 0.731, p= 0.556). More specifically CB, VP, and HM, situated in the vicinity of agricultural effluent release and urban areas had consistently higher TN concentrations than the other three sites. Table 3 shows highest mean TN concentrations were recorded in Fall 2013 (1891.83 µg/L) and the lowest in Summer 2014 (586.67 µg/L).

During the study period water TP concentrations ranged between 5 - 81 µg/L (Table 3). One-way ANOVA results showed a significant variation between sites ($F_{5,7}$= 3.889, p<0.05), and sampling periods ($F_{3,9}$= 4.047, p< 0.05). Again, VP, CB and HM, had consistently higher TP concentrations than the other three sites (mean values of 45.25, 47.25, and 43 µg/L, respectively) (Table 3). In terms of compliance with provincial environmental standards, 15 of the 24 TP samples collected exceeded the Provincial guideline of 20 µg/L to avoid nuisance concentrations of algae in lakes (Environment Canada 2004a). Table 3 reveals the highest mean TP concentration was recorded in the Fall 2014 sampling period (48 µg/L) and lowest in the Fall 2013 (13.83 µg/L) sampling period.

Chlorophyll *a* concentrations in the water column during the study period ranged from 0.29 to 7.17 mg/m^{-3}; the minimum concentration was during SP4 (Fall 2014) at HM and the maximum also occurred at HM during SP2 (Table 3). There were no significant differences between sites (one-way ANOVA $F_{5,7}$=1.015, p=0.472) or sampling periods (one way ANOVA $F_{3,20}$= 0.923, p= 0.448).

Table 3. Raw data, mean values and standard error (in parenthesis) of limnologic parameters at all sites and sampling periods.

	Fall 2013 (SP1)	**Spring 2014 (SP2)**	**Summer 2014 (SP3)**	**Fall 2014 (SP4)**	**Mean Value by Site (± S.E.)**
pH					
BC	7.42	7.07	7.15	6.96	7.15 (0.07)
VP	6.39	6.98	6.67	6.65	6.67 (0.12)
LC	7.62	7.01	7	6.9	7.13 (0.16)
DP	7.34	6.73	8.25	7.21	7.38 (0.32)
CB	7.58	6.5	6.8	6.72	6.9 (0.24)
HM	6.48	8.04	7.28	6.7	7.125 (0.35)
Total Mean Per SP	7.14 (0.23)	7.06 (0.22)	7.19 (0.23)	6.86 (0.09)	
DO (mg/l)					
BC	7.07	3.49	14.76	4.99	7.58 (1.89)
VP	4.65	0.12	9.99	1.58	4.085 (2.19)
LC	3.14	5.18	9.04	4.83	5.55 (1.25)
DP	3.46	3.98	9.05	8.61	6.28 (1.48)
CB	9.91	0.47	1.68	5.14	4.3 (2.12)
HM	6.87	6.07	2.85	8.13	5.98 (1.13)

Total Mean Per SP	5.85 (1.06)	3.22 (1.00)	7.895 (1.99)	5.55 (1.05)	
COND (µs/cm)					
BC	450	614	716	489	567.25 (45.87)
VP	161.6	190.4	211.1	142.8	176.48 (15.14)
LC	330	354	461	271	354 (39.70)
DP	362	330.5	299	330.5	330.5 (12.86)
CB	849	612.5	612.5	376	612.5 (96.55)
HM	623	690	617	538	617 (31.10)
Total Mean Per SP	462.6 (98.91)	465.23 (81.73)	486.1 (81.08)	357.88 (59.03)	
TEMP (°C)					
BC	5.4	21.5	12.3	10	12.3 (2.56)
VP	16.7	19.5	15.33	9.8	15.33 (2.04)
LC	6.2	25	22	5	14.55 (5.21)
DP	15.1	25	23	10.8	18.48 (3.33)
CB	7.21	19.5	20	19.3	16.50 (3.10)
HM	5	22.8	24.1	18	17.46 (4.36)
Total Mean Per Site	9.27 (2.13)	22.22 (1.02)	19.46 (1.91)	12.15 (2.22)	
TP(µg/L)					
BC	13	24	15	33	21.25 (3.47)
VP	28	29	56	68	45.25 (9.98)
LC	14	24	25	21	21 (2.48)
DP	4.5	13	14.33	13.33	11.25 (2.28)
CB	11	73	24	81	47.25 (17.46)
HM	12	33	55	72	43 (13.06)
Total Mean Per SP	13.83 (3.16)	32.67 (8.52)	31.5 (7.78)	48 (11.86)	
TN(µg/L)					
BC	313	726	289	370	424.5 (77.05)
VP	1274	1254	1350	1645	1380.75 (90.48)
LC	445	426	431	553	463.75 (30.02)
DP	365.33	365.33	344.67	306	345.25 (13.99)
CB	6694	641	544	1261	2285 (1478.22)
HM	2260	685	561	1049	1138.75 (387.83)
Total Mean Per SP	1891.83 (1008.21)	682.83 (128.66)	586.67 (158.80)	864 (220.15)	
DEPTH (cm)					
BC	18.48	25.86	11	18.57	18.48 (2.29)
VP	6.45	16.57	0.21	2.57	6.45 (3.61)
LC	14.04	26.71	14.04	1.37	14.04 (5.17)
DP	43.65	57.29	42.37	31.29	43.65 (5.33)
CB	8.40	16.16	1.8	7.23	8.40 (2.96)

HM	6.11	0.1	4.23	14	6.11 (2.91)
Total Mean Per SP	16.19 (5.83)	23.78 (7.76)	12.28 (6.40)	12.50 (4.63)	
Chl *a* (mg·m^{-3})					
BC	3.30	1.67	0.88	1.15	1.75 (0.41)
VP	0.84	0.99	2.57	0.78	1.30 (0.3)
LC	3.27	1.44	1.97	6.52	3.30 (1.14)
DP	1.09	1.61	1.47	1.33	1.37 (0.11)
CB	2.86	5.72	0.77	2.72	3.02 (1.02)
HM	5.20	7.17	0.39	0.29	3.26 (1.73)
Total Mean Per SP	2.76 (0.66)	3.10 (1.08)	1.34 (0.3)	1.13 (0.94)	

3.3.2 Macrophyte Species Composition and Dynamics in Wetlands

3.3.2.1 Species Composition

Table 4 lists all species identified during the study at each site. A total of 72 different emergent macrophyte species were collected. 58 species were identified, 7 could only be identified to genus level, and 7 species were unidentifiable. Those identified only to genus level included *Salix* spp., *Acer* spp., *Ribes* spp., *Sparganium* spp., *Typha* spp., *Prunus* spp., and *Cornus* spp. Four species of moss were present, *Brachythecium populeum* (matted feather moss), *Brachythecium salebrosum* (golden ragged moss), *Brachythecium reflexum* (cedar moss), and *Brachythecium velutinum* (velvet ragged moss). Only one species was present in all six sites, *Sparganium eurycarpum* (broadfruit bur-reed). *Typha* x *glauca* (hybrid cattail) was the only species to be present in five of the six sites (except DP), and *B. populeum* (matted feather moss), *Scirpus acutus* (hardstem bulush), *Typha angustifolia* (narrow-leaved cattail), and *Typha latifolia* (broad-leaved cattail) were present in four sites.

Table 4. List of all plant species present in the study sites throughout all four sampling periods. "X" indicates presence of the species in a wetland. Empty cells indicate the absence of a species.

Taxa	Common Name	Site					
		BC	VP	LC	DP	CB	HM
Acer negundo	Manitoba maple						X
Acer spp.	Maple						
Alnus incana	Speckled alder						
Betula papyrifera	White birch						
Bidens cernua	Nodding bur-marigold	X				X	
Brachythecium populeum	Matted Feather moss	X	X			X	X
Brachythecium salebrosum	Golden ragged moss			X			
Brachythecium reflexum	Cedar moss						X
Brachythecium velutinum	Velvet ragged moss		X				
Calamagrostis canadensis	Blue-joint grass	X				X	X
Calystegia sepium	Hedge bindweed						X
Carex hystericina	Bottlebrush sedge					X	X
Carex lacustris	Lake sedge	X					
Cornus obliqua	Silky dogwood		X				
Cornus spp.	Dogwood		X				X
Cornus stolonifera	Red osier dogwood	X	X				X
Eleocharis smallii	Creeping/common spike rush	X					
Epilobium ciliatum ssp. *glandulosum*	Willowherb					X	
Equisetum arvense	Field horsetail						X
Equisetum fluviatile	Water horsetail						X
Euthamia graminifolia	Grass-leaved goldenrod						X
Galium trifidum	Three-petaled bedstraw	X				X	
Glyceria striata	Fowl mannagrass/Ridged glyceria						X
Ilex verticillata	Winterberry		X				
Impatiens capensis	Jewelweed					X	X
Leersia oryzoides	Rice cutgrass	X				X	
Lysimachia terrestris	Swamp candles						X
Lythrum salicaria	Purple loostrife		X				X
Melilotus albus	Sweet clover					X	
Mimulus ringens	Allegheny monkeyflower/Square-stemmed monkeyflower	X					X
Myrica gale	Sweet gale	X	X				X

Onoclea sensibilis	Sensitive fern		X				X
Persicaria lapathifolia	Pale persicaria					X	
Phalaris arundinacea	Reed canary grass	X				X	X
Phragmites australis	European common reed						X
Pontederia cordata	Pickerelweed		X		X		
Prunus spp.	Cherry		X				
Rhamnus frangula	Alder buckthorn						X
Ribes spp.	Currant						X
Rosa palustris	Swamp rose		X				
Rubus idaeus L. var. strigosus	American red raspberry						X
Rumex orbiculatus	Water dock					X	X
Sagittaria graminea	Grassy arrowhead			X			
Sagittaria latifolia	Broadleaf arrowhead			X			
Salix spp.	Willow						
Salix x *rubens*	Hybrid crack-willow					X	X
Sambucus canadensis	American elderberry						X
Scirpus acutus	Hardstem bulrush	X		X	X		X
Scirpus pungens	Common three-square bulrush				X	X	
Scirpus cyperinus	Woolgrass	X					
Scutellaria galericulata	Common skullcap					X	X
Scutellaria lateriflora	Mad-dog skullcap					X	
Sium suave	Water parsnip			X			
Solanum dulcamara	Bittersweet nightshade					X	X
Solidago canadensis	Canada goldenrod						X
Sparganium americanum	American bur-reed	X					
Sparganium eurycarpum	Broadfruit bur-reed	X	X	X	X	X	X
Sparganium spp.	Bur-reed				X		X
Thelypteris palustris	Marsh fern	X					X
Typha angustifolia	Narrow-leaved cattail	X	X			X	X
Typha latifolia	Broad-leaved cattail	X	X			X	X
Typha spp.	Cattail	X				X	X
Typha x *glauca*	Hybrid cattail	X	X	X		X	X
UnID twigs		X					
UnID vines						X	
Unidentifiable emergent		X					
Unidentifiable Wood Stems		X					
Unidentified plant							X
Unidentified plant							X
Unidentified plant							X

Vitis riparia	Riverbank grape	X					X
Zizania palustris	Northern wild rice				X	X	X

Table 5 lists all species present in the quadrats in each of the four sampling periods. Sixteen species were observed across all four sampling periods, including *Brachythecium populeum*, *Brachythecium velutinum*, *Calamagrostis canadensis* (blue-joint grass), *Carex hystericina* (bottlebrush sedge), *Carex lacustris* (lake sedge), *Cornus stolonifera* (red osier dogwood), *Eleocharis smallii* (creeping/common spike rush), *Leersia oryzoides* (rice cutgrass), *Myrica gale* (Sweet gale), *Scirpus acutus*, *Scirpus pungens* (common three-square bulrush), *Scutellaria galericulata* (common skullcap), *Sparganium eurycarpum*, *Typha angustifolia*, *Typha latifolia*, and *Typha* x *glauca*.

Table 5. List of all plant species present in each sampling period in at least one of the six sampling sites. "X" indicates presence of a species. Empty cells indicate the absence of a species.

Taxa	Common Name	Sampling Period			
		SP1	SP2	SP3	SP4
Acer negundo	Manitoba maple		X		
Acer spp.	Maple				
Alnus incana	Speckled alder				
Betula papyrifera	White birch				
Bidens cernua	Nodding bur-marigold	X	X	X	
Brachythecium populeum	Matted feather moss	X	X	X	X
Brachythecium salebrosum	Golden ragged moss			X	X
Brachythecium reflexum	Cedar moss	X	X	X	
Brachythecium velutinum	Velvet ragged moss	X	X	X	X
Calamagrostis canadensis	Blue-joint grass	X	X	X	X
Calystegia sepium	Hedge bindweed		X		
Carex hystericina	Bottlebrush sedge	X	X	X	X
Carex lacustris	Lake sedge	X	X	X	X
Cornus obliqua	Silky dogwood	X	X		
Cornus spp.	Dogwood	X	X		
Cornus stolonifera	Red osier dogwood	X	X	X	X
Eleocharis smallii	Creeping/common spike rush	X	X	X	X
Epilobium ciliatum spp. *glandulosum*	Willowherb			X	X

Equisetum arvense	Field horsetail	X	X	X	
Equisetum fluviatile	Water horsetail	X	X	X	
Euthamia graminifolia	Grass-leaved goldenrod		X	X	
Galium trifidum	Three-petaled bedstraw	X		X	
Glyceria striata	Fowl mannagrass/Ridged glyceria		X		
Ilex verticillata	Winterberry		X		
Impatiens capensis	Jewelweed	X	X	X	
Leersia oryzoides	Rice cutgrass	X	X	X	X
Lysimachia terrestris	Swamp candles		X		X
Lythrum salicaria	Purple loostrife	X		X	X
Melilotus albus	Sweet clover			X	
Mimulus ringens	Allegheny monkeyflower/Square-stemmed monkeyflower	X		X	
Myrica gale	Sweet gale	X	X	X	X
Onoclea sensibilis	Sensitive fern		X	X	
Persicaria lapathifolia	Pale persicaria		X		
Phalaris arundinacea	Reed canary grass	X	X		
Phragmites australis	European common reed	X	X	X	
Pontederia cordata	Pickerelweed		X	X	X
Prunus spp.	Cherry		X	X	
Rhamnus frangula	Alder buckthorn			X	
Ribes spp.	Currant			X	
Rosa palustris	Swamp rose	X	X	X	
Rubus idaeus L. var. strigosus	American red raspberry	X	X	X	
Rumex orbiculatus	Water dock		X	X	X
Sagittaria graminea	Grassy arrowhead		X	X	X
Sagittaria latifolia	Broadleaf arrowhead			X	
Salix spp.	Willow				
Salix x *rubens*	Hybrid crack-willow	X	X		X
Sambucus canadensis	American elderberry	X	X		
Scirpus acutus	Hardstem bulrush	X	X	X	X
Scirpus pungens	Common three-square bulrush	X	X	X	X
Scirpus cyperinus	Woolgrass	X		X	
Scutellaria galericulata	Common skullcap	X	X	X	X
Scutellaria lateriflora	Mad-dog skullcap			X	
Sium suave	Water parsnip			X	
Solanum dulcamara	Bittersweet nightshade	X		X	
Solidago canadensis	Canada goldenrod			X	
Sparganium americanum	American bur-reed	X		X	X

Sparganium eurycarpum	Broadfruit bur-reed	X	X	X	X
Sparganium spp.	Bur-reed	X	X		
Thelypteris palustris	Marsh fern		X	X	
Typha angustifolia	Narrow-leaved cattail	X	X	X	X
Typha latifolia	Broad-leaved cattail	X	X	X	X
Typha spp.	Cattail	X	X		X
Typha x *glauca*	Hybrid cattail	X	X	X	X
UnID twigs			X		
UnID vines		X			
Unidentifiable emergent				X	
Unidentifiable Wood Stems			X		
Unidentified plant		X			
Unidentified plant			X		
Unidentified plant				X	
Vitis riparia	Riverbank grape	X	X	X	
Zizania palustris	Northern wild rice		X	X	

Table 6 shows the distinction of vascular species in this study as native, introduced, or invasive. All introduced species in this study were invasive, so this category alone was not included. The table is based on the number of different native and invasive species at a site, and therefore is a measure of the richness of these groups. The richness of invasive species in all sites was 12; and the richness of native species was 41.

Table 6. List of native and invasive plant species in this study. "X" indicates whether a species was native or invasive.

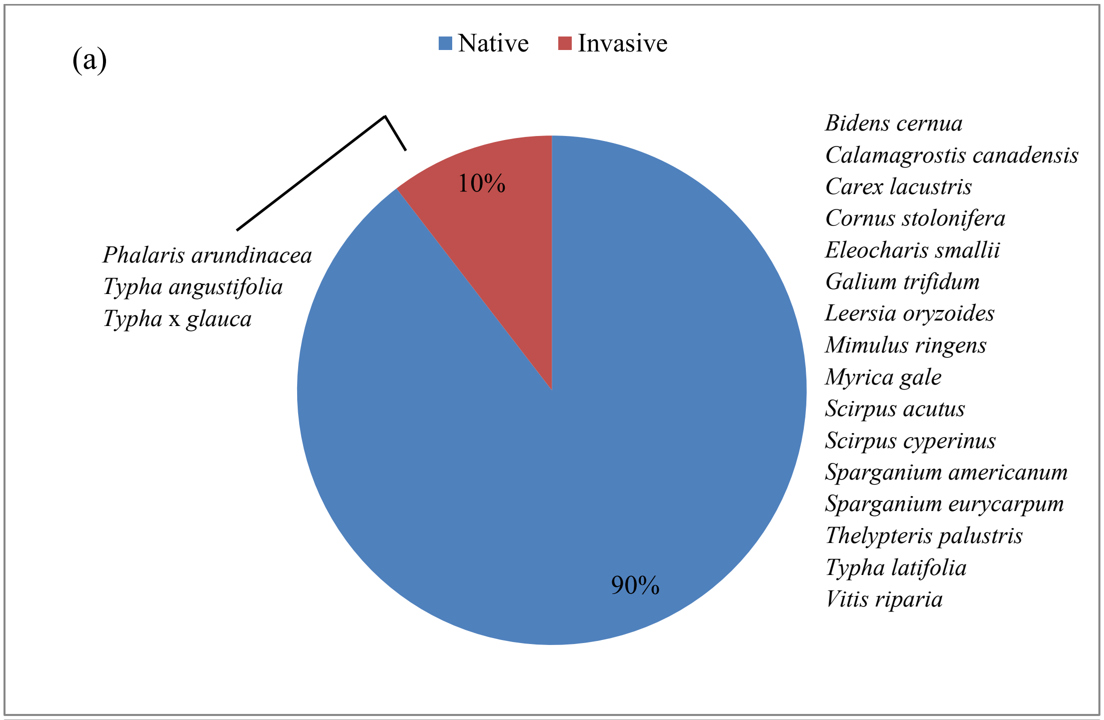
Taxa	Common Name	Native	Invasive
Acer negundo	Manitoba maple		X
Alnus incana	Speckled alder	X	
Betula papyrifera	White birch	X	
Bidens cernua	Nodding bur-marigold	X	
Calamagrostis canadensis	Blue-joint grass	X	
Calystegia sepium	Hedge bindweed		X
Carex hystericina	Bottlebrush sedge	X	
Carex lacustris	Lake sedge	X	
Cornus obliqua	Silky dogwood	X	
Cornus stolonifera	Red osier dogwood	X	

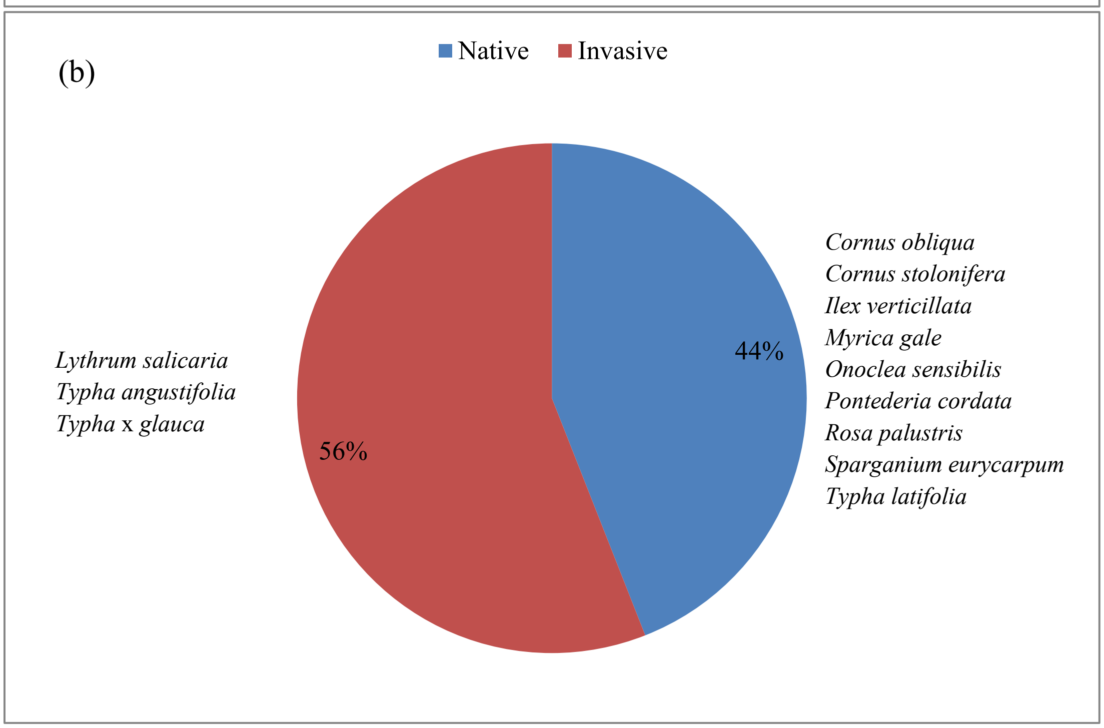
Eleocharis smallii	Creeping/common spike rush	X	
Epilobium ciliatum spp. *glandulosum*	Willowherb	X	
Equisetum arvense	Field horsetail	X	
Equisetum fluviatile	Water horsetail	X	
Euthamia graminifolia	Grass-leaved goldenrod	X	
Galium trifidum	Three-petaled bedstraw	X	
Glyceria striata	Fowl mannagrass/Ridged glyceria	X	
Ilex verticillata	Winterberry	X	
Impatiens capensis	Jewelweed	X	
Leersia oryzoides	Rice cutgrass	X	
Lysimachia terrestris	Swamp candles	X	
Lythrum salicaria	Purple loostrife		X
Melilotus albus	Sweet clover		X
Mimulus ringens	Allegheny monkeyflower	X	
Myrica gale	Sweet gale	X	
Onoclea sensibilis	Sensitive fern	X	
Persicaria lapathifolia	Pale persicaria		X
Phalaris arundinacea	Reed canary grass		X
Phragmites australis	European common reed		X
Pontederia cordata	Pickerelweed	X	
Rhamnus frangula	Alder buckthorn		X
Rosa palustris	Swamp rose	X	
Rubus idaeus L. var. strigosus	American red raspberry	X	
Rumex orbiculatus	Water dock	X	
Sagittaria graminea	Grassy arrowhead	X	
Sagittaria latifolia	Broadleaf arrowhead	X	
Salix x *rubens*	Hybrid crack-willow		X
Sambucus canadensis	American elderberry	X	
Scirpus acutus	Hardstem bulrush	X	
Scirpus pungens	Common three-square bulrush	X	
Scirpus cyperinus	Woolgrass	X	
Scutellaria galericulata	Common skullcap	X	
Scutellaria lateriflora	Mad-dog skullcap	X	
Sium suave	Water parsnip	X	
Solanum dulcamara	Bittersweet nightshade		X
Solidago canadensis	Canada goldenrod	X	
Sparganium americanum	American bur-reed	X	
Sparganium eurycarpum	Broadfruit bur-reed	X	
Thelypteris palustris	Marsh fern	X	
Typha angustifolia	Narrow-leaved cattail		X

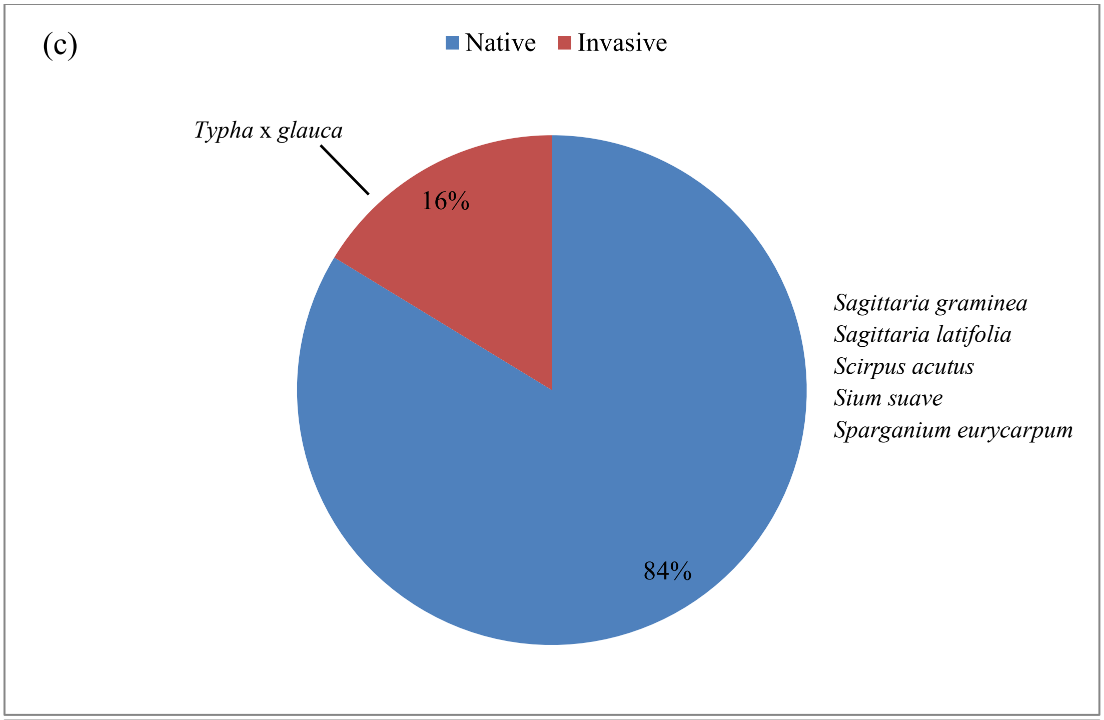
Typha latifolia	Broad-leaved cattail	X	
Typha x *glauca*	Hybrid cattail		X
Vitis riparia	Riverbank grape	X	
Zizania palustris	Northern wild rice	X	

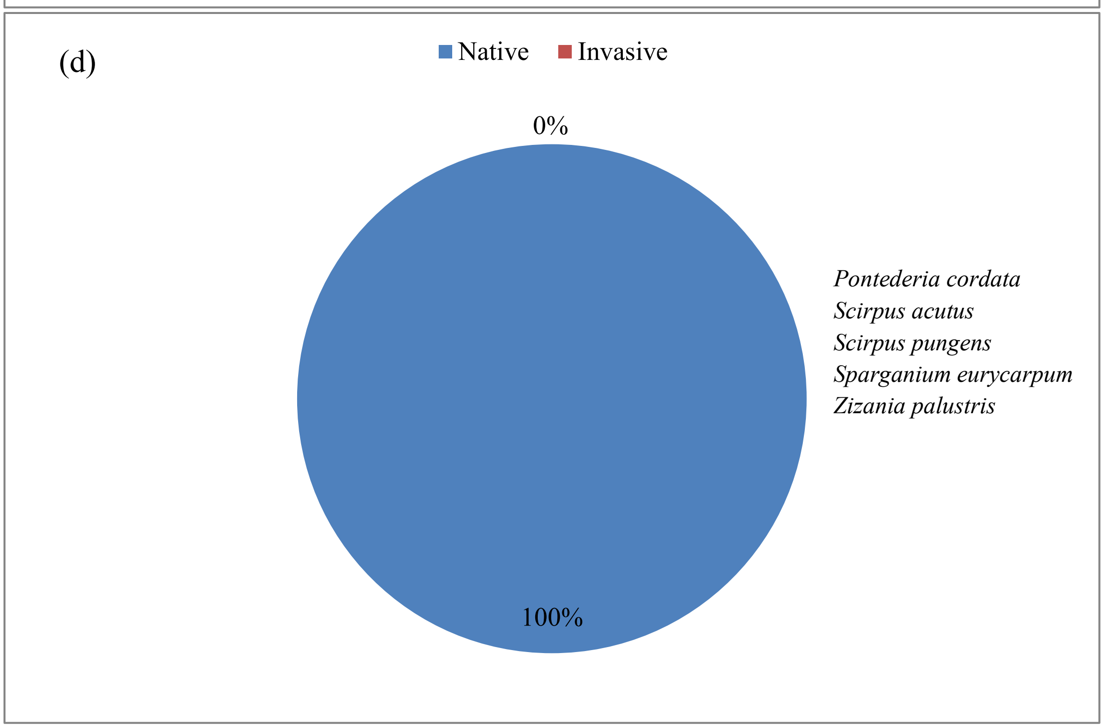
The percent composition of native and invasive species in each site is shown in Figure 2. A list of species comprising the native and invasive composition of each site is provided in the charts to illustrate the species richness of each group. DP was composed of 100% native species and 0% invasive species; CB had the highest percent composition, followed by VP with 56% (Figure 2). All sites had a greater richness of native species than invasive species, but VP and CB had greater percent compositions of invasives. The percent composition of native and invasive species was the same at HM (50%) despite having a lower richness of invasive species (Figure 2). Figure 3 shows a comparison of the percent composition of only the invasive species among each site. The percent composition of invasive species ranged from 0% to 64% composition among the sampling sites.

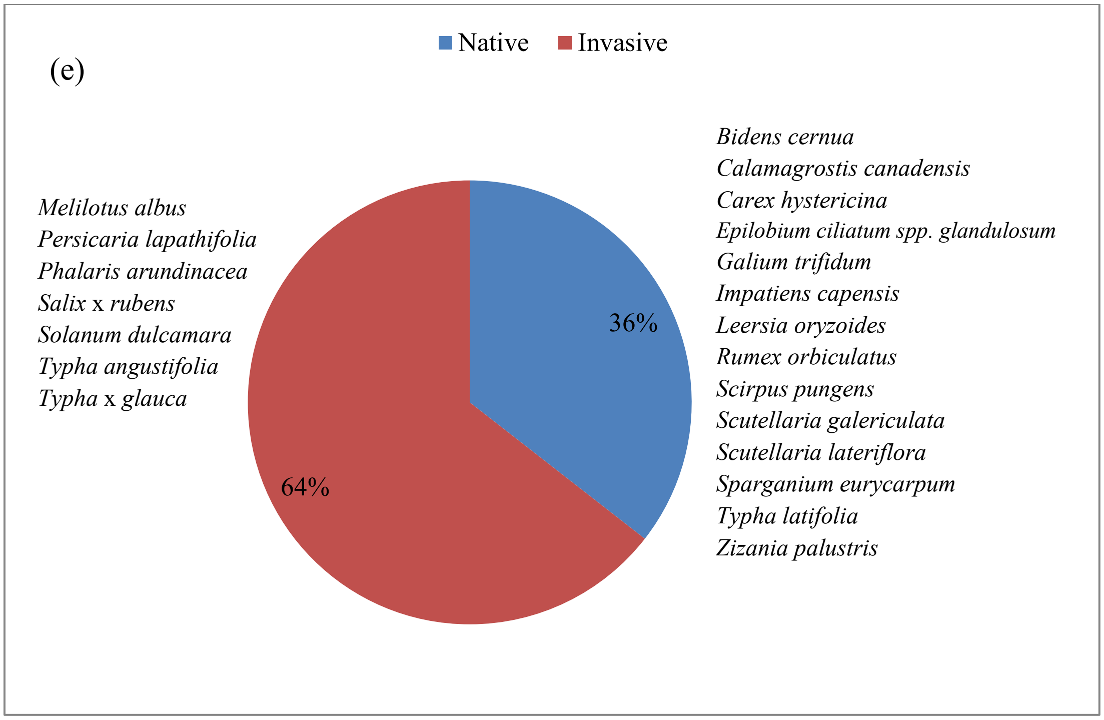
Figure 2. Percent composition of native and invasive species observed during the study period in a) BC b) VP c) LC d) DP e) CB f) HM. Species contributing to the percent composition are provided.






(c)
Native
Invasive
Typha x glauca
16%
84%
Sagittaria graminea
Sagittaria latifolia
Scirpus acutus
Sium suave
Sparganium eurycarpum


(d)
Native
Invasive
0%
100%
Pontederia cordata
Scirpus acutus
Scirpus pungens
Sparganium eurycarpum
Zizania palustris


(e)
Native
Invasive
Melilotus albus
Persicaria lapathifolia
Phalaris arundinacea
Salix x rubens
Solanum dulcamara
Typha angustifolia
Typha x glauca
64%
36%
Bidens cernua
Calamagrostis canadensis
Carex hystericina
Epilobium ciliatum spp. glandulosum
Galium trifidum
Impatiens capensis
Leersia oryzoides
Rumex orbiculatus
Scirpus pungens
Scutellaria galericulata
Scutellaria lateriflora
Sparganium eurycarpum
Typha latifolia
Zizania palustris

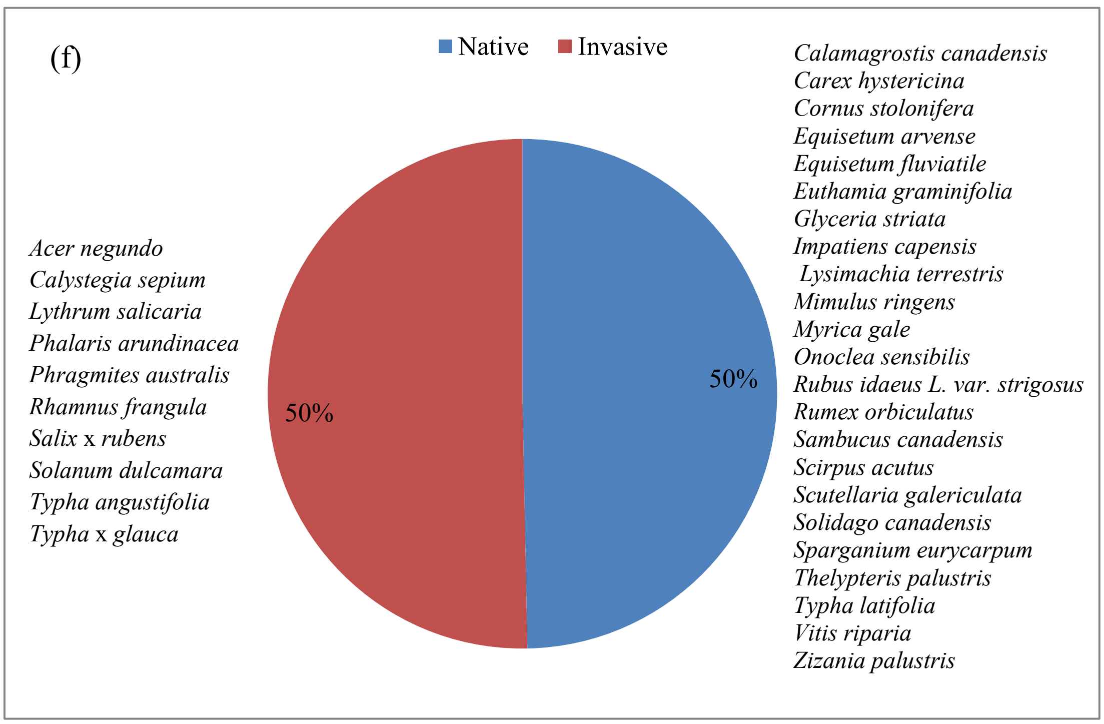
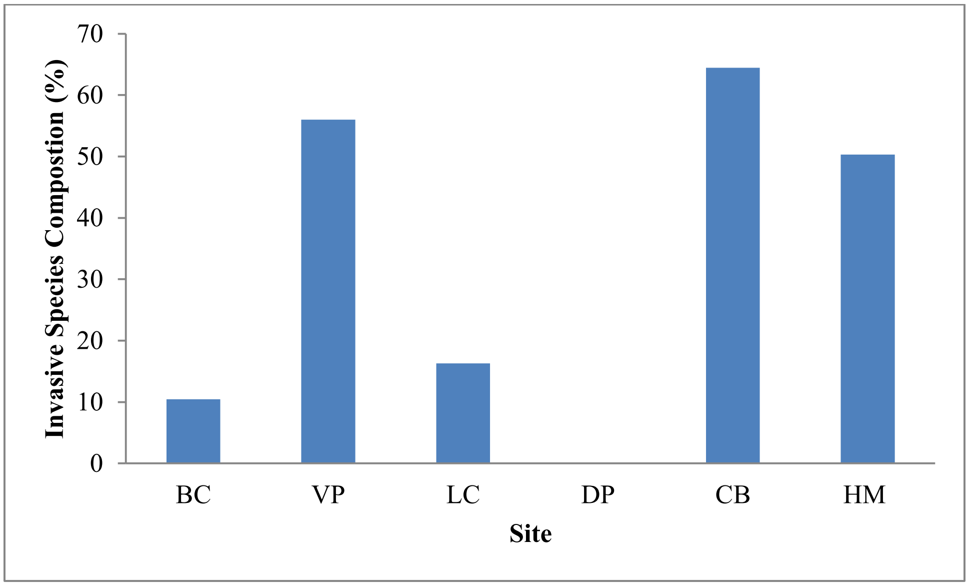

(f)
Native
Invasive
Acer negundo
Calystegia sepium
Lythrum salicaria
Phalaris arundinacea
Phragmites australis
Rhamnus frangula
Salix x rubens
Solanum dulcamara
Typha angustifolia
Typha x glauca
50%
50%
Calamagrostis canadensis
Carex hystericina
Cornus stolonifera
Equisetum arvense
Equisetum fluviatile
Euthamia graminifolia
Glyceria striata
Impatiens capensis
Lysimachia terrestris
Mimulus ringens
Myrica gale
Onoclea sensibilis
Rubus idaeus L. var. strigosus
Rumex orbiculatus
Sambucus canadensis
Scirpus acutus
Scutellaria galericulata
Solidago canadensis
Sparganium eurycarpum
Thelypteris palustris
Typha latifolia
Vitis riparia
Zizania palustris

Figure 3. Percent composition of invasive species at each site in the study.



3.3.2.2 Frequency

The relative frequencies of all macrophyte species are summarized in Table 7. In BC, *Typha* x *glauca*, *Leersia oryzoides*, and *Eleocharis smallii* were the most frequent species (20.34%, 16.95%, and 11.02%, respectively) with frequencies greater than 10%, indicating greater abundance of these species in BC relative to other species (Table 7). Most of the other species present were at frequencies <10%. At VP, *Myrica gale*, *Typha* x *glauca*, and *Brachythecium velutinum* were the most frequent species, with frequencies of 24.72%, 20.23% and 10.11%, respectively (Table 7). Again this indicated that relative to all species present in VP, these species occurred equal to or more than 10% of the time. At LC, only two of the seven species present had a frequency of less than 10%. Frequent species at this site included *Sparganium eurycarpum* (29.79%), *Scirpus acutus* (25.53%), *Typha* x *glauca* (19.15%), *Sagittaria graminea* (grassy arrowhead) (10.64%), and *Brachythecium salebrosum* (10.64%) (Table 7). At DP, the most frequent species were *Scirpus acutus* (54.72%) *Scirpus pungens* (16.67%), and *Sparganium eurycarpum* (16.6%) (Table 7). Of the 23 species present in CB, only three species had frequencies equal to or greater than 10%. *Typha* x *glauca* was most frequent at this site (25.23%), followed by *Typha angustifolia* (14.56%) and *Scutellaria galericulata* (10.68%) (Table 7). *Typha* x *glauca* was also the most frequent species at HM (13.04%), and was the only species with a frequency >10%. *Phragmites australis* (European common reed) had the second greatest frequency (9.32%), followed by *Equisetum fluviatile* (water horsetail) and *Vitis riparia* (riverbank grape) both with a frequency of 6.83% (Table 7). In five the six sites *Typha* x *glauca* was at a frequency >10%, and is consistently one of the most frequent species. The only site it is absent from (i.e. where it has 0% occurrence), is DP, the least disturbed site.

Table 7. Relative frequency (RF) and relative density (RD) of each species present at each site (BC, VP, LC, DP, CB, and HM).

Site	Species	RF (%)	RD (%)
BC	*Bidens cernua*	5.09	0.96
	Brachythecium populeum	4.24	0.51
	Calamagrostis canadensis	1.7	1.25
	Carex lacustris	8.48	5.44
	Cornus stolonifera	0.85	0.07
	Eleocharis smallii	11.02	24.61
	Galium trifidum	0.85	0.07
	Leersia oryzoides	16.95	48.79
	Mimulus ringens	1.7	0.15
	Myrica gale	0.85	0.07
	Phalaris arundinacea	1.7	0.29
	Scirpus acutus	1.7	0.37
	Scirpus cyperinus	1.7	0.44
	Sparganium americanum	4.24	1.54
	Sparganium eurycarpum	0.85	0.37
	Thelypteris palustris	0.85	3.31
	Typha angustifolia	6.78	1.91
	Typha latifolia	5.09	7.57
	Typha spp.	3.39	0.81
	Typha x *glauca*	20.34	1.03
	Vitis riparia	0.85	0.07
VP	*Brachythecium populeum*	2.25	0.66
	Brachythecium velutinum	10.11	2.62

	Cornus obliqua	4.49	1.31
	Cornus spp.	1.12	0.33
	Cornus stolonifera	8.99	2.62
	Ilex verticillata	1.12	0.33
	Lythrum salicaria	1.12	1.31
	Myrica gale	24.72	7.54
	Onoclea sensibilis	1.12	2.62
	Pontederia cordata	5.62	4.59
	Prunus spp.	1.12	0.33
	Rosa palustris	5.62	2.62
	Sparganium eurycarpum	1.12	14.10
	Typha angustifolia	8.99	18.69
	Typha latifolia	2.25	6.56
	Typha x *glauca*	20.23	33.77
LC	*Brachythecium salebrosum*	10.64	1.24
	Sagittaria graminea	10.64	2.72
	Sagittaria latifolia	2.13	0.99
	Scirpus acutus	25.53	31.44
	Sium suave	2.13	0.50
	Sparganium eurycarpum	29.79	47.03
	Typha x *glauca*	19.15	16.09
DP	*Pontederia cordata*	7.14	1.61
	Scirpus acutus	54.76	61.01
	Scirpus pungens	16.67	13.07
	Sparganium eurycarpum	16.67	23.39
	Sparganium spp.	2.38	0.46
	Zizania palustris	2.38	0.46
CB	*Bidens cernua*	1.94	0.47
	Brachythecium populeum	4.85	1.17
	Calamagrostis canadensis	6.8	3.75
	Carex hystericina	4.85	1.64
	Epilobium ciliatum spp. *glandulosum*	2.91	0.70
	Galium trifidum	1.94	0.47
	Impatiens capensis	0.97	2.58
	Leersia oryzoides	0.97	0.47
	Melilotus albus	0.97	0.23
	Persicaria lapathifolia	0.97	0.23
	Phalaris arundinacea	2.91	1.17
	Rumex orbiculatus	1.94	0.47
	Salix x *rubens*	1.94	0.47
	Scirpus pungens	0.97	0.23

	Scutellaria galericulata	10.68	3.75
	Scutellaria lateriflora	1.94	0.70
	Solanum dulcamara	0.97	0.23
	Sparganium eurycarpum	6.8	15.46
	Typha angustifolia	14.56	13.58
	Typha latifolia	1.94	0.47
	Typha spp.	0.97	0.70
	Typha x *glauca*	25.24	47.07
	Zizania palustris	0.97	3.75
HM	*Acer negundo*	0.62	0.20
	Brachythecium populeum	4.97	1.57
	Brachythecium reflexum	2.48	0.79
	Calamagrostis canadensis	5.59	17.52
	Calystegia sepium	0.62	0.20
	Carex hystericina	2.48	1.38
	Cornus spp.	1.24	0.39
	Cornus stolonifera	1.24	0.59
	Equisetum arvense	3.11	1.97
	Equisetum fluviatile	6.83	4.53
	Euthamia graminifolia	2.48	1.18
	Glyceria striata	0.62	0.59
	Impatiens capensis	2.48	0.98
	Lysimachia terrestris	1.24	0.39
	Lythrum salicaria	2.48	2.95
	Mimulus ringens	0.62	0.20
	Myrica gale	1.86	0.59
	Onoclea sensibilis	1.86	1.77
	Phalaris arundinacea	3.73	7.09
	Phragmites australis	9.32	20.08
	Rhamnus frangula	0.62	0.20
	Ribes spp.	0.62	0.20
	Rubus idaeus L. var. *strigosus*	3.73	2.36
	Rumex orbiculatus	1.24	0.79
	Salix x *rubens*	1.24	0.39
	Sambucus canadensis	1.24	0.39
	Scirpus acutus	0.62	2.76
	Scutellaria galericulata	2.48	0.79
	Solanum dulcamara	1.86	0.59
	Solidago canadensis	0.62	0.20
	Sparganium eurycarpum	1.24	2.56
	Sparganium spp.	0.62	0.59

	Thelypteris palustris	1.86	1.97
	Typha angustifolia	1.86	2.17
	Typha latifolia	0.62	0.39
	Typha spp.	1.24	0.39
	Typha x *glauca*	13.04	14.57
	Vitis riparia	6.83	2.95
	Zizania palustris	0.62	0.20

3.3.2.3 Species Relative Density, Stem Density, and Dominance

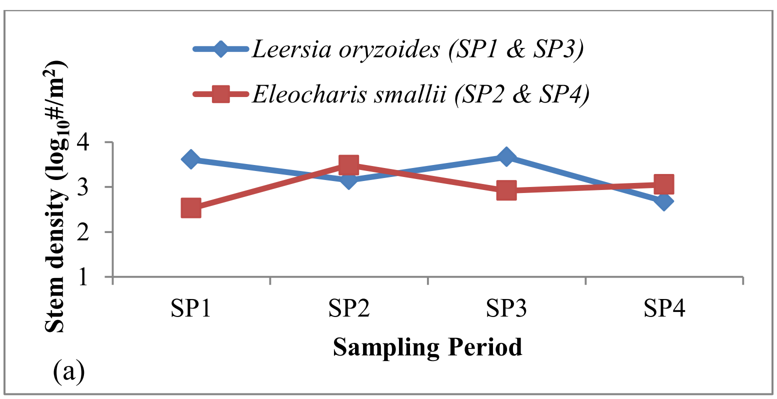
The species density values based on stem counts are presented in Table 7, and are expressed as the percent relative density. In BC, the species with the greatest relative densities were *Leersia oryzoides* (48.79%), *Eleocharis smallii* (24.61%), and *Typha* x *glauca* (7.57%) (Table 7). In VP, *Typha* x *glauca* and *Typha angustifolia* had the greatest densities (33.77% and 18.69%, respectively), followed by *Sparganium eurycarpum* with a relative density of 14.10%. *Sparganium eurycarpum* was also the most dense species at LC (47.03%), along with *Scirpus acutus* (31.44%) and *Typha* x *glauca* (16.09%) (Table 7). Other species present at this site contributed less than 3% of the relative density. At DP, *Scirpus acutus*, had a density value of 61.01% (the highest among all sites), followed by *Sparganium eurycarpum* (23.39%) and *Scirpus pungens* (13.07%). *Typha* x *glauca* had the highest density value at CB (47.07%), followed by *Sparganium eurycarpum* (15.46%) and *Typha angustifolia* (13.58%) (Table 7). All of the other 23 species present at CB had relative density values <5%. At HM, *Phragmites australis*, *Calamagrostis canadensis*, and *Typha* x *glauca* were the most dense species (20.08%, 17.52%, and 14.57%, respectively) (Table 7). Similar to the relative frequency values, *Typha* x *glauca* showed the highest relative density values in five out of six sites.

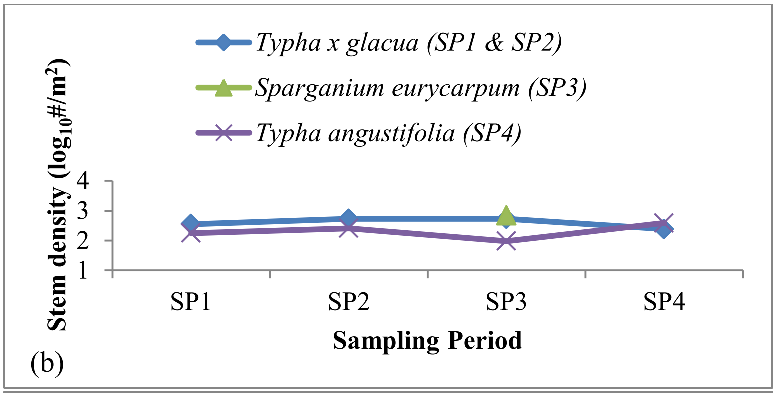
In this study, the dominant species were apparent in terms of their absolute stem density (that is, not relative density as discussed above); those species with the highest densities in each sampling period were dominant, and are shown in Figure 4. At DP, the least disturbed site, *Scirpus acutus* was the dominant species in Fall 2013, Summer 2014, and Fall 2014 (SP1, SP3, and SP4), while *Sparganium eurycarpum* was dominant in Spring 2014 (SP2). Figure 7 provides photographs of these dominant species at each site.

At the moderately disturbed site BC, *Leersia oryzoides* was dominant in SP1 and SP3, while *Eleocharis smallii* was dominant during SP2 and SP4 (Figure 4). At VP, dominance shifted to three different species, *Typha* x *glauca* in SP1 and SP2, *Sparganium eurycarpum* in SP3, and *Typha angustifolia* in SP4 (Figure 4). At LC, *Scirpus acutus* was dominant for one season only (SP1), while *Sparganium eurycarpum* dominated for the remainder of the study (SP2 to SP4) (Figure 4).

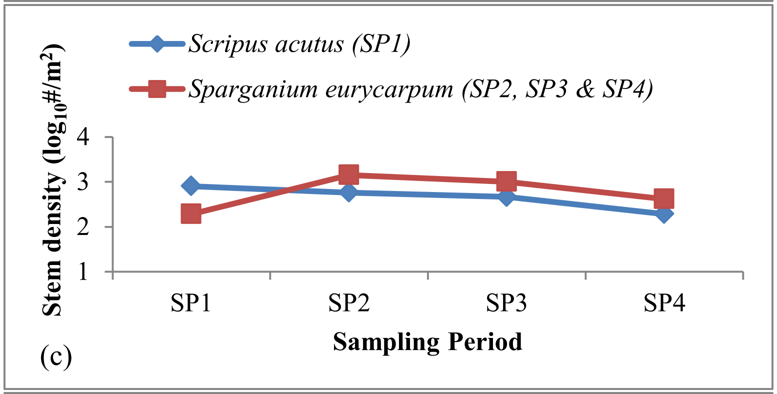
At CB, a highly disturbed site, *Typha* x *glauca* was the dominant species from SP2 to SP4, while *Sparganium eurycarpum* dominated in Fall 2013 (SP1) (Figure 4). At HM, three species were dominant, *Calamagrostis canadensis* during SP1 and SP2, *Phragmites australis* in the following sampling period (SP3), and *Typha* x *glauca* in SP4 (Figure 4).

Figure 4. Variation of dominant macrophyte species based on stem density at each site observed during the study period. a) BC b) VP c) LC d) DP e) CB f) HM

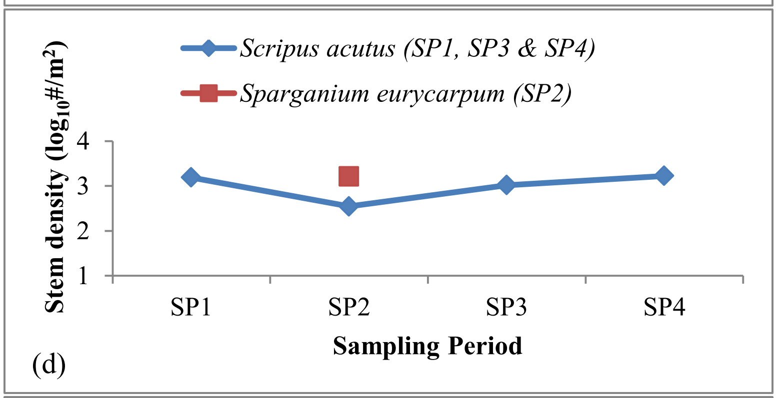




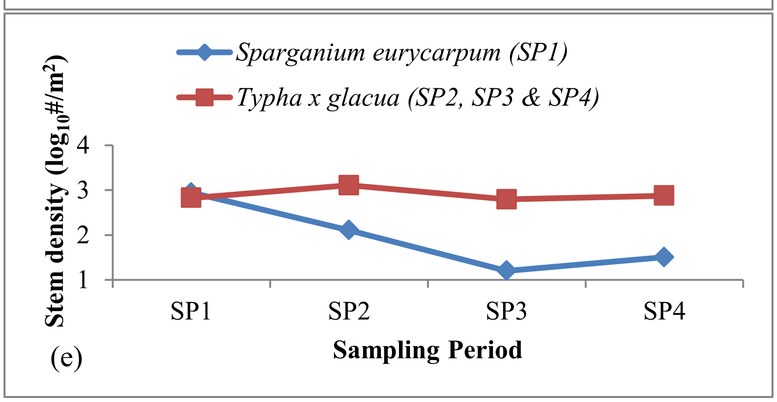
Typha x glacua (SP1 & SP2)
Sparganium eurycarpum (SP3)
Typha angustifolia (SP4)
Stem density (log10#/m2)
4
3
2
1
SP1
SP2
SP3
SP4
Sampling Period
(b)



Scripus acutus (SP1)
Sparganium eurycarpum (SP2, SP3 & SP4)
Stem density (log10#/m2)
4
3
2
1
SP1
SP2
SP3
SP4
Sampling Period
(c)



Scripus acutus (SP1, SP3 & SP4)
Sparganium eurycarpum (SP2)
Stem density (log10#/m2)
4
3
2
1
SP1
SP2
SP3
SP4
Sampling Period
(d)



Sparganium eurycarpum (SP1)
Typha x glacua (SP2, SP3 & SP4)
Stem density (log10#/m2)
4
3
2
1
SP1
SP2
SP3
SP4
Sampling Period
(e)

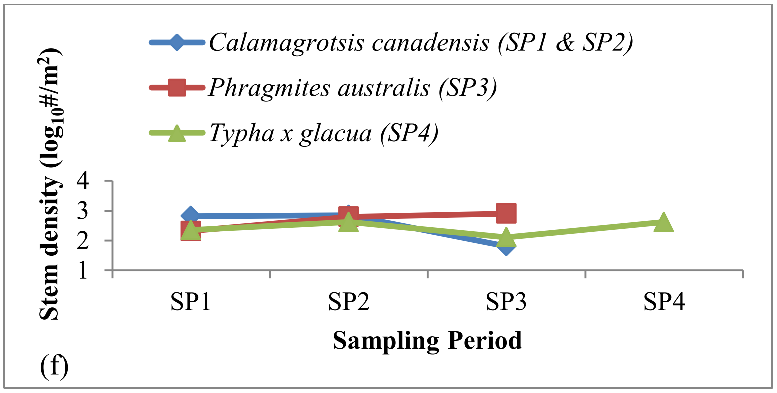
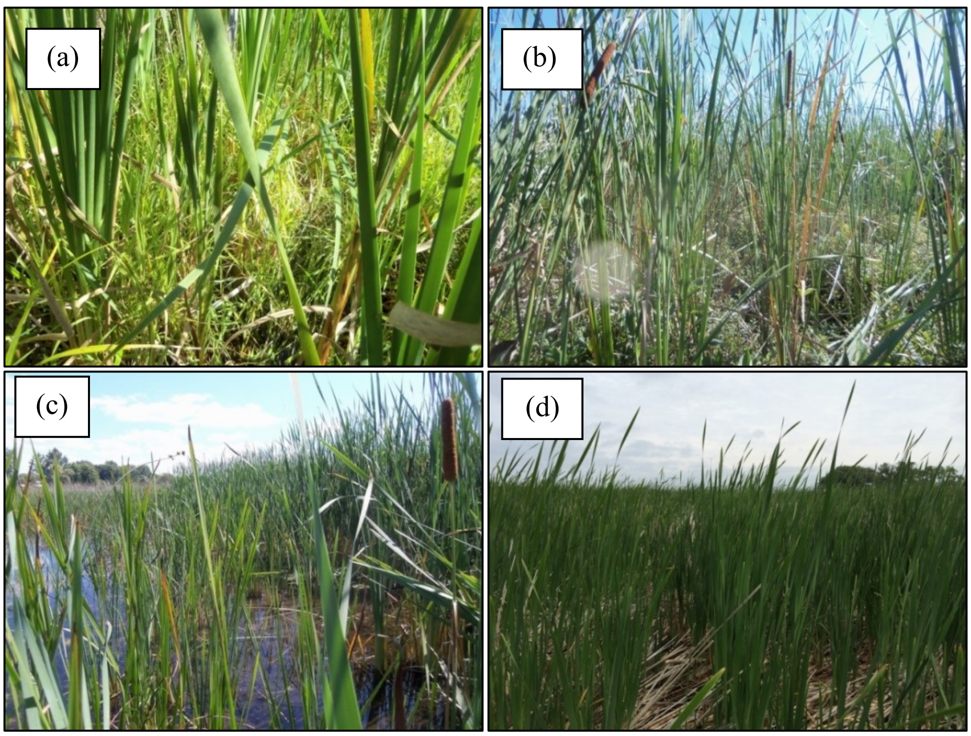
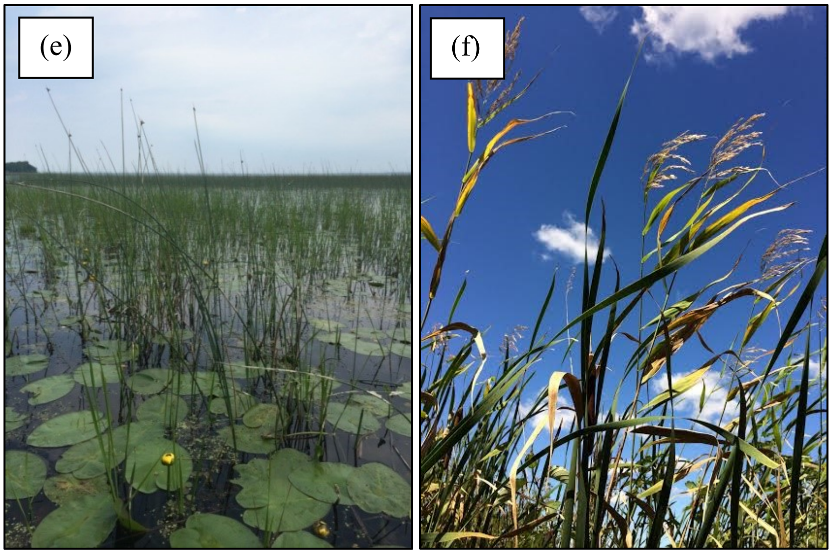


Figure 5. Some photos of the dominant macrophytes at a) BC – *Typha* spp. and *Leersia oryzoides*; b) VP – *Typha* spp. and *Sparganium eurycarpum*; c) LC – *Typha* spp., *Scirpus acutus*, and *Sparganium eurycarpum*; d) CB – *Typha* spp. e) DP – *Scirpus acutus* f) HM – *Typha* spp. and *Phragmites australis*





3.3.2.4 Importance Value Index

The overall importance of each species in the community was determined for each site using relative frequency and relative density data. In BC, *Leersia oryzoides, Eleocharis smallii, Typha* x *glauca,* and *Carex lacustris,* were the most important species. In VP, *Typha* x *glauca*, *Myrica gale*, *Typha angustifolia*, and *Sparganium eurycarpum* had the greatest importance. *Sparganium eurycarpum* had the greatest overall importance in the community structure at LC, followed by *Scirpus acutus*, *Typha* x *glauca*, and *Sagittaria graminea*. Similar to LC, at DP, *Sparganium eurycarpum* had the greatest overall importance, followed by *Scirpus acutus* and *Scirpus pungens*, as well as *Pontederia cordata* (pickerelweed). At CB, *Typha* x *glauca*, *Typha angustifolia*, *Sparganium eurycarpum*, and *Scutellaria galericulata* were the most important species in the community. Similar to CB, *Typha* x *glauca* also important in HM, as was *Calamagrostis canadensis*, *Phragmites australis*, *Equisetum fluviatile*, and *Phalaris arundinacea.*

Species with an IVI of 0.025 or greater in at least one sampling site are organized in Table 8. The species with greatest overall importance value was *Scirpus acutus* (IVI=0.579), followed by *Sparganium eurycarpum* (IVI=0.384), *Typha* x *glauca* (IVI=0.362), *Leersia oryzoides* (IVI=0.329), and *Scirpus acutus* (IVI= 0.285).

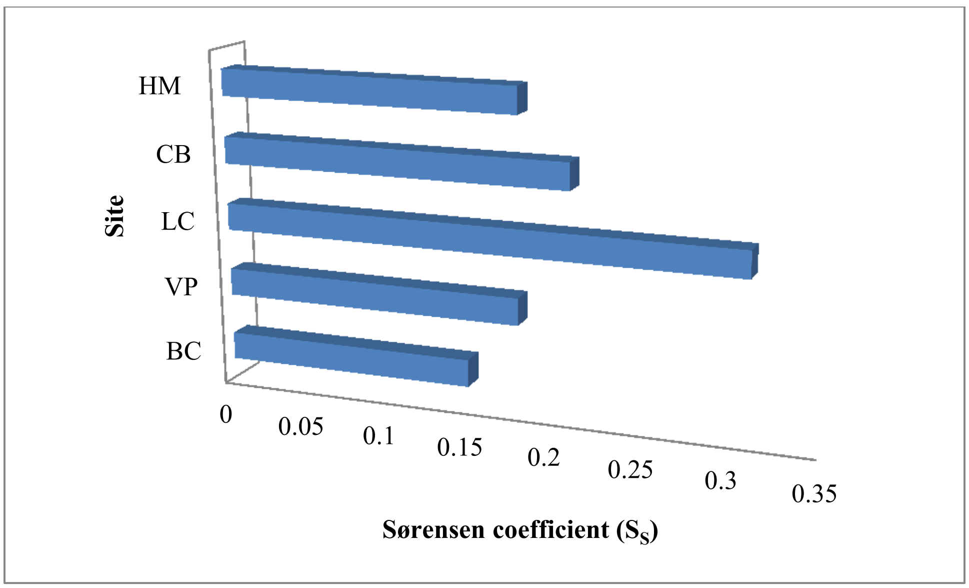
Table 8. IVI for species with the highest importance value indexes (0.025 and greater) in at least one sampling site. – indicates the absence of the species.

Species	Study Sites					
	BC	**VP**	**LC**	**DP**	**CB**	**HM**
Scirpus acutus	0.011	-	0.285	0.579	-	0.017
Sparganium eurycarpum	0.006	0.076	0.384	0.200	0.111	0.019
Typha x *glauca*	0.140	0.270	0.176	-	0.362	0.138
Leersia oryzoides	0.329	-	-	-	0.007	-
Eleocharis smallii	0.178	-	-	-	-	-
Myrica gale	0.005	0.161	-	-	-	0.012
Scirpus pungens	-	-	-	0.149	0.006	-
Phragmites australis	-	-	-	-	-	0.147
Typha angustifolia	0.043	0.138	-	-	0.141	0.020
Calamagrostis canadensis	0.015	-	-	-	0.053	0.116
Scutellaria galericulata	-	-	-	-	0.072	0.016
Carex lacustris	0.070	-	-	-	-	-
Sagittaria graminea graminea	-	-	0.067	-	-	-
Brachythecium velutinum	-	0.064	-	-	-	-
Brachythecium salebrosum	-	-	0.059	-	-	-
Cornus stolonifera	0.005	0.058	-	-	-	0.009
Equisetum fluviatile	-	-	-	-	-	0.057
Phalaris arundinacea	0.010	-	-	-	0.020	0.054
Pontederia cordata	-	0.051	-	0.044	-	-
Rosa palustris	-	0.041	-	-	-	-
Brachythecium populeum	0.024	0.015	-	-	0.030	0.033
Carex hystericina	-	-	-	-	0.032	0.019
Rubus idaeus L. var. strigosus	-	-	-	-	-	0.030
Bidens cernua	0.030	-	-	-	0.012	-
Typha latifolia	0.029	0.044	-	-	0.012	0.005
Sparganium americanum	0.029	-	-	-	-	-
Cornus obliqua	-	0.029	-	-	-	-
Lythrum salicaria	-	0.012	-	-	-	0.027
Equisetum arvense	-	-	-	-	-	0.025

3.3.2.5 Similarity Index

Sørensen coefficient (S_S) was calculated to determine the similarity index between DP, the least disturbed site, and the other sites in the study. The coefficient of similarity between DP and BC was 0.15 (15%); between DP and VP was 0.18 (18%); between DP and LC was 0.31 (31%); between DP and CB was 0.21 (21%); and between DP and HM was 0.18 (18%) (Figure 6). A comparison of the Sørensen coefficients shows that species composition in LC was most similar to DP, while BC was the least similar.

Figure 6. Visual representation of similarity between DP and the other sampling sites. The range of similarity coefficients is 0 (no similarity) to 1.0 (complete similarity).



3.3.3 Species Richness

3.3.3.1 Variation in species richness between sampling sites

The mean species richness was highest at HM (6.29, SE= ±0.81) and lowest at LC (1, SE= ±0.22) during Fall 2013 (SP1). The reference site, DP, had the second lowest species richness 1.43, SE= ±0.22) (Table 9). During this sampling period, species richness was significantly different between sites, one-way ANOVA $F_{5, 36} = 12.435$, $p < 0.05$. Tukey's *post* *hoc* analysis revealed that species richness was significantly different between all pairs of sites. BC and LC (p =0.007), LC and CB (p =0.032), LC and HM (p =0.000), VP and HM (p =0.032), DP and HM (p =0.000), DP and BC (p =0.020), DP and VP (p =.0032), LC and VP (p =0.012), BC and HM (p =0.050), and CB and HM (p =0.012).

The following spring (SP2) the mean species richness was again highest at HM (9, SE= ±1.00), and lowest at LC (1.43, SE= ±0.20). The control site, DP, had the second lowest species richness value (2.43, SE= ±0.20). There was a mean difference of 2.52 between HM, the site with the greatest richness, and the second greatest richness, BC (Table 9). During this sampling period, species richness was significantly different between sites, one-way ANOVA $F_{5, 36}$ = 14.657, $p < 0.001$. Tukey's *post hoc* tests showed that species richness was significantly different between DP and HM, VP and HM, LC and HM, CB and HM, and BC and HM (p =0.000 for all).

During SP3 the mean species richness was highest at one of the highly disturbed sites, HM (6.43, SE= ±1.09), and lowest at DP, the least disturbed site (1.43, SE= ±0.30). During this sampling period, richness was significantly different between sites, one-way ANOVA $F_{5, 16}$= 6.594, $p < 0.05$. The *post hoc* analysis revealed that the richness values were significantly different between DP and HM (p = 0.028), as well as between DP and VP (p = 0.033).

During Fall 2014 (SP4), the mean species richness was highest at BC (5.29, SE= ±0.71), a moderately disturbed site, and lowest at DP (1.29, SE= ±0.29). HM, which had the greatest richness for all other seasons, had the second lowest richness (1.43, SE= ±0.20) (Table 9). During this sampling period, the richness was significantly different between sites, one-way ANOVA $F_{5,36}$ = 10.589, $p < 0.001$. Tukey's *post hoc* tests showed that the richness was significantly different between DP and CB (p =0.011) and DP and BC (p =0.000), BC and LC

(p =0.000), BC and VP (p =0.032), BC and HM (p =0.000), and between CB and HM (p =0.019).

The variation in mean richness between sampling periods is shown in Figure 9, while Figure 10 shows the variation in richness between sites. The site with the greatest mean species richness over all sampling periods was one the most disturbed, HM (5.79, SE= ±1.58), and the lowest was the least disturbed site, DP (1.64, SE= ±0.26) (Table 9). Thus, the mean species richness results revealed a trend that the site with higher disturbance had higher species richness than sites exposed to lower levels of anthropogenic disturbance.

3.3.3.2 Variation in species richness between sampling periods

The season with the greatest mean richness over all sites was SP2 (3.86, SE= ±1.08), and the lowest was Fall 2014 (SP4) (2.81, SE= ±0.63) (Table 9). Sites that showed no significant variation between sampling periods included CB, BC and VP.

At HM, species richness differed significantly between sampling periods, one-way ANOVA $F_{3, 24} = 13.322$, $p < 0.001$ Tukey's *post hoc* tests showed a significant difference between SP1 and SP4, SP2 and SP4, and SP3 and SP4 ($p = 0.003$, $p = 0.002$, $p = 0.000$, respectively). At DP, the species richness was significantly different with different sampling periods, one-way ANOVA $F_{3, 24} =3.727$, $p < 0.05$. Tukey's *post hoc* tests showed a significant difference between SP2 and SP4 (p =0.033). At LC, a moderately disturbed site, the species richness again varied significantly between sampling periods, $F_{3, 24} =3.727$, $p < 0.05$. Tukey's *post hoc* tests showed that richness was significantly different between SP2 and SP4 ($p = 0.023$).

3.3.3.3 Influence of sampling periods and sampling locations – Interaction outcome

The results of the two-way ANOVA showed that species richness varied significantly between sampling periods ($F_{3, 130}= 3.408$, p <0.05) and between sites ($F_{5, 130}= 40.554$, p <0.01)

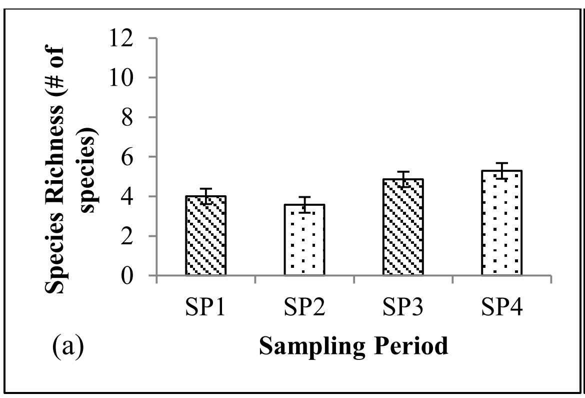
(Figure 9). The interaction results of the two-way ANOVA between sites and sampling periods also resulted in significant variation ($F_{15, 130}$= 9.516, p <0.01). *Post hoc* tests of species richness was performed with statistical significance receiving a Bonferroni adjustment and being accepted at the p <0.025 level showed that there was significant differences between sites in all sampling periods (p<0.01). Furthermore, *post hoc* tests showed significant variation between seasons in BC and HM only (p<0.05).

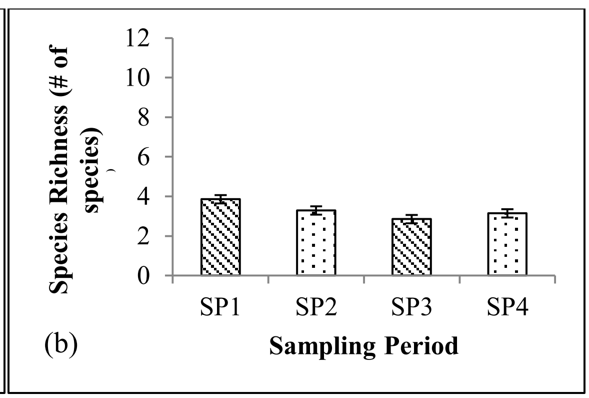
Table 9. Mean values and standard errors in parenthesis of macrophyte species richness, density, diversity and biomass observed at all sites and sampling periods.

	Fall 2013 (SP1)	**Spring 2014 (SP2)**	**Summer 2014 (SP3)**	**Fall 2014 (SP4)**	Mean Value by Site (± SE)
Richness					
BC	4.00 (0.69)	3.57 (0.81)	4.86 (0.96)	5.29 (0.71)	4.43 (0.39)
VP	3.86 (0.51)	3.29 (0.57)	2.86 (0.26)	3.14 (0.34)	3.29 (0.21)
LC	1.00 (0.22)	1.43 (0.20)	2.43 (0.30)	2.00 (0.44)	1.71 (0.31)
DP	1.43 (0.22)	2.43 (0.20)	1.43 (0.30)	1.29 (0.29)	1.64 (0.26)
CB	3.57 (0.48)	3.43 (0.90)	4.14 (0.74)	3.71 (0.64)	3.71 (0.15)
HM	6.29 (0.81)	9.00 (1.00)	6.43 (1.09)	1.43 (0.20)	5.79 (1.58)
Mean Value by Season (± SE)	3.36 (0.79)	3.86 (1.08)	3.69 (0.74)	2.81 (0.63)	
Density					
BC	834.29 (157.64)	774.86 (215.73)	1062.86 (595.58)	438.86 (96.69)	777.71 (128.89)
VP	123.43 (24.86)	185.14 (36.26)	240.00 (98.07)	144.00 (26.59)	173.14 (25.71)
LC	141.71 (37.84)	345.14 (82.17)	242.29 (49.68)	148.57 (33.98)	219.43 (47.77)
DP	226.29 (43.67)	320.00 (45.92)	235.43 (52.69)	267.43 (80.05)	262.29 (21.16)
CB	322.29 (60.11)	297.14 (59.24)	194.29 (32.83)	178.29 (26.92)	248.00 (36.15)
HM	374.86 (120.49)	425.14 (61.06)	292.57 (39.30)	68.57 (10.31)	290.29 (78.79)
Mean Value by Season (± SE)	337.14 (107.23)	391.24 (83.03)	377.90 (137.58)	207.62 (53.16)	

Diversity					
BC	2.00 (0.28)	1.50 (0.14)	2.81 (0.35)	2.66 (0.24)	2.24 (0.30)
VP	2.49 (0.43)	2.00 (0.76)	1.71 (0.16)	1.56 (0.07)	1.94 (0.20)
LC	1.00 (0.00)	1.54 (0.18)	1.85 (0.18)	1.39 (0.30)	1.44 (0.18)
DP	1.05 (0.05)	1.52 (0.16)	1.43 (0.33)	1.05 (0.05)	1.26 (0.12)
CB	2.18 (0.22)	1.74 (0.27)	2.47 (0.49)	2.23 (0.25)	2.16 (0.15)
HM	4.03 (0.78)	4.77 (1.01)	3.28 (0.64)	1.34 (0.18)	3.35 (0.74)
Mean Value by Season (± SE)	2.13 (0.45)	2.18 (0.52)	2.26 (0.29)	1.70 (0.25)	
Biomass					
BC	2050.17 (433.34)	548.11 (145.99)	1612.91 (520.46)	2077.83 (256.53)	1572.26 (357.60)
VP	4731.66 (593.48)	3645.26 (843.45)	5060.34 (528.03)	3266.06 (461.08)	4175.83 (428.22)
LC	193.83 (69.36)	1076.57 (479.52)	2073.83 (590.66)	1984.46 (499.62)	1332.17 (441.28)
DP	369.49 (96.87)	214.17 (65.44)	498.29 (111.56)	498.51 (136.46)	395.11 (67.54)
CB	3207.54 (528.48)	2812.00 (464.27)	3647.20 (637.50)	2473.37 (322.36)	3035.03 (253.26)
HM	2472.23 (471.31)	1747.09 (271.73)	1969.03 (377.19)	2258.29 (380.71)	2111.66 (159.37)
Mean Value by Season (± SE)	2170.82 (704.54)	1673.87 (545.95)	2476.93 (661.45)	2093.09 (369.80)	

Figure 7. Variation in species richness across all sampling periods at a) BC b) VP c) LC d) DP e) CB and f) HM. Error bars represent the standard error.





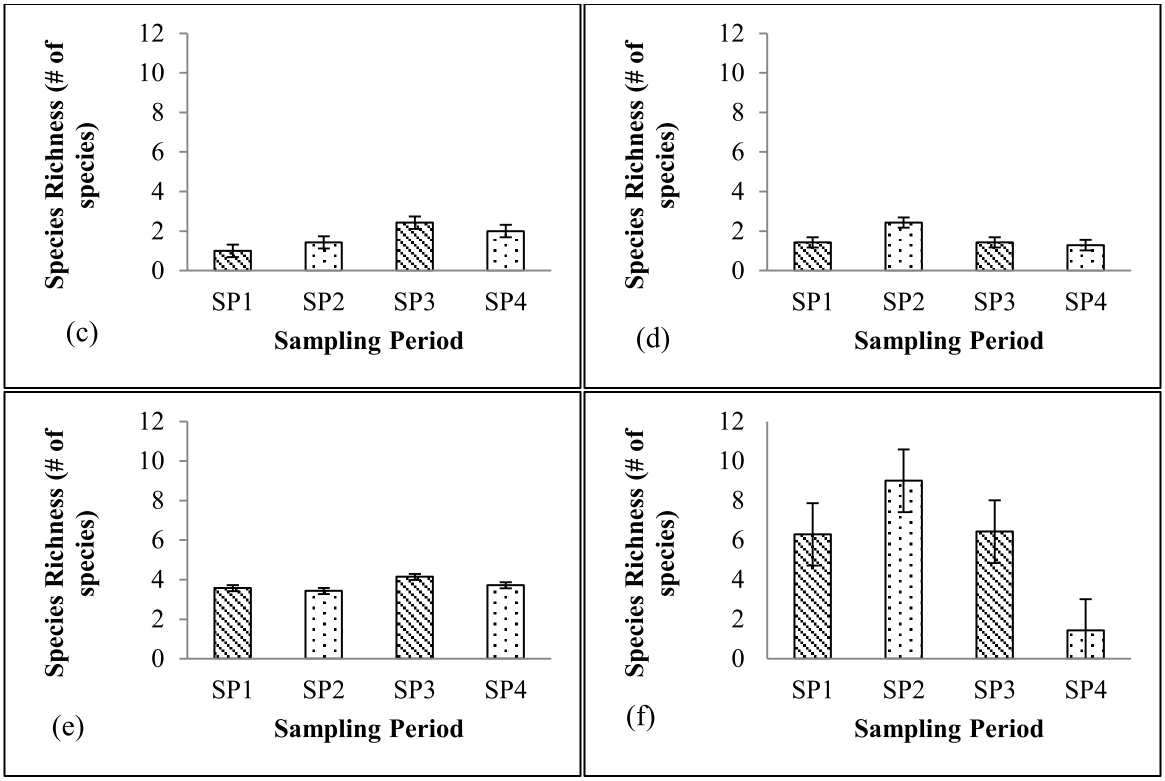


Figure 8. Variation in species richness observed across sampling sites during a) SP1 b) SP2 c) SP3 and d) SP4. Error bars represent the standard error.

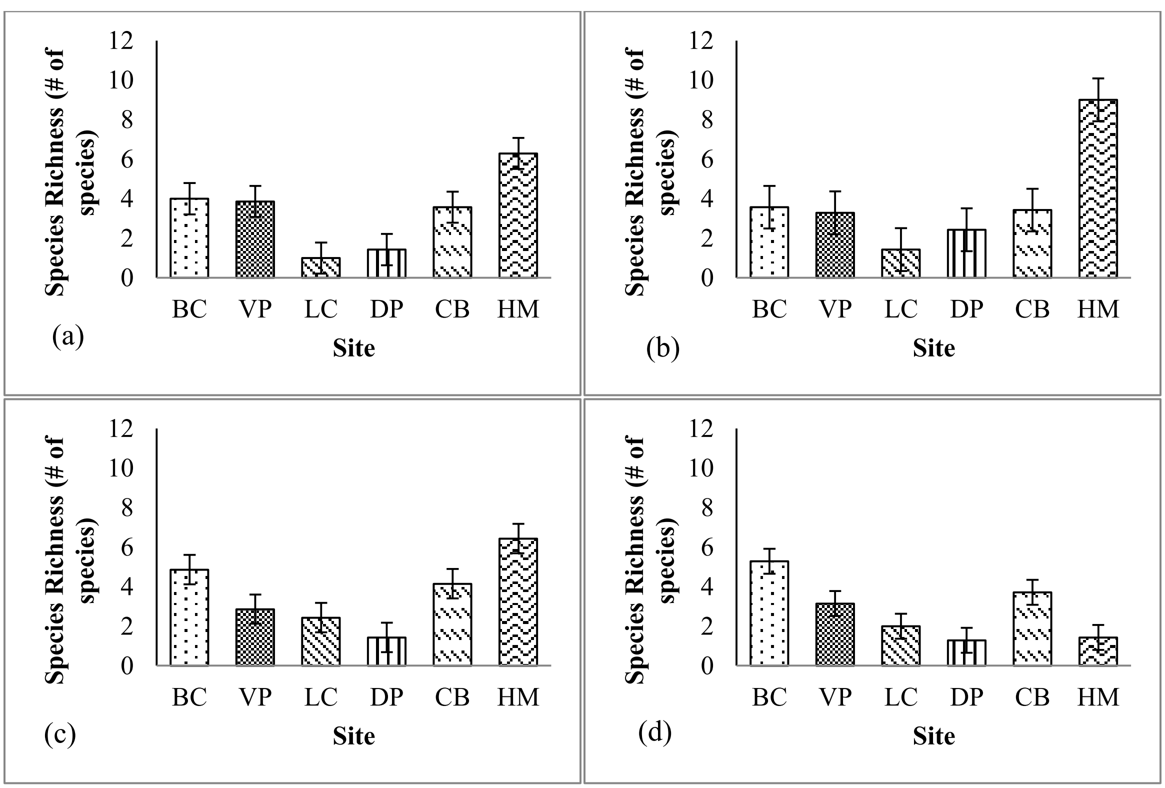
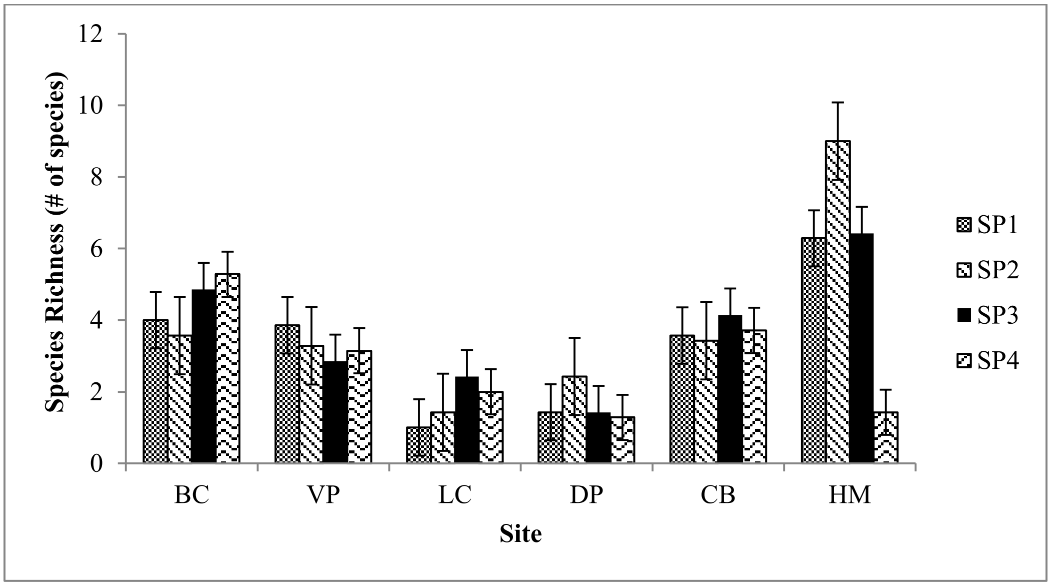


Figure 9. Mean species richness observed across all sites and sampling periods. Error bars represent the standard error.



3.3.4 Stem Density

3.3.4.1 Variation in density between sampling sites

Among the sites, BC had the greatest mean density (777.71 #/m^2, SE= ±128.89) while VP had the lowest (173.14 #/m^2, SE= ±25.71) over all seasons (Table 9). Changes in mean density between sampling periods is shown in Figure 10, while the mean density differences between sites is represented in Figure 11. A one-way ANOVA showed density differed significantly among sites in Fall 2013 (one-way ANOVA $F_{5, 16} = 6.570$, $p < 0.05$) and Fall 2014 ($F_{5, 36} = 6.964$, $p < 0.001$).

During Fall 2013 (SP1), the highest mean density was observed at BC (834.29 #/m^2, SE= ±157.64) while the lowest was at VP (123.43 #/m^2, SE= ±24.86); both are moderately disturbed sites. Both HM (374.86 #/m^2, SE= ±120.49) and CB (322.29 #/m^2, SE= ±60.11) had slightly higher density than the least disturbed site, DP (226.29 #/m^2, SE= ±43.67) (Table 9). During this sampling period, density was significantly different for different sampling sites (one-way

ANOVA $F_{5,16} = 6.570$, $p < 0.05$). Games-Howell *post hoc* test revealed that the density between DP and BC and between VP and BC were significantly different (p =0.012 and p=0.001, respectively).

In the following sampling period (SP2), the mean density was highest at BC (774.86 #/m^2, SE= ±215.73) and lowest at VP (185.14 #/m^2, SE= ±36.26) (Table 9). However, there were no significant differences in species densities between different sampling sites (one-way ANOVA $F_{5,16} = 2.528$, $p = 0.070$). In SP3, the density between sampling sites was not significantly different (one-way ANOVA $F_{5,16}$= 1.527, p =0.236). In this sampling period, the mean density was highest at BC (1062.86 #/m^2, SE= ±595.58) and lowest at CB (194.29 #/m^2, SE= ±32.83) (Table 9).

During Fall 2014 (SP4), the mean density was highest at BC (438.86 #/m^2, SE= ±96.69) and lowest at HM (68.57#/m^2, SE= ±10.31), one of the most stressed sites (Table 9). During this sampling period, density was significantly different between sites (one-way ANOVA $F_{5,36}$ = 6.964, $p < 0.001$). Tukey's *post hoc* test revealed that density was significantly different between DP and HM (p =0.009), BC and HM (p =0.000), BC and LC (p =0.016) CB and HM (p =0.047).

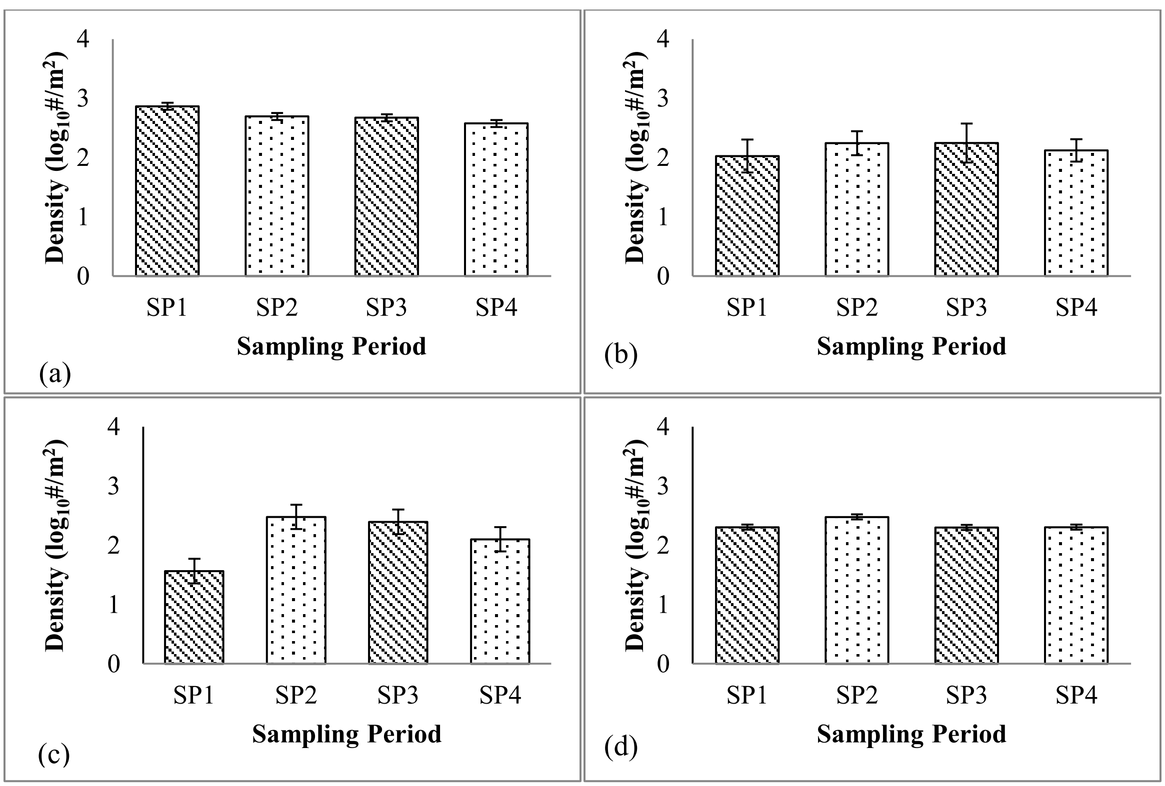
3.3.4.2 Variation in stem density between sampling periods

The season with the highest recorded mean density over all sites was Spring 2014 (SP2) (391.24 #/m^2, SE= ±83.03) and with lowest in Fall 2014 (SP4) (207.62 #/m^2, SE= ±53.16) (Table 9). One-way ANOVA results revealed that HM was the only site that showed significant variation between sampling periods ($F_{3,27} = 18.404$, $p < 0.001$). Tukey's *post hoc* test showed that the difference in density between SP1 and SP4, SP2 and SP4, and SP3 to SP4 was statistically significant (p =0.000 for all comparisons).

3.3.4.3 Influence of sampling periods and sampling locations – Interaction outcome

According to the results of two-way ANOVA, density showed significant variation between sampling periods ($F_{3, 138}$= 2.777, p<0.05) and sampling sites ($F_{5, 138}$= 5.690, p<0.05) (Figure 12). The interaction results of the two-way ANOVA between sites and sampling periods also resulted in significant variations ($F_{15, 138}$= 1.739, p=0.05). The results of the *post hoc* test with Bonferroni adjustment showed that there was significant variation at LC between SP1 and SP2, and between SP1 and SP3; and also at HM between SP2 and SP4. Furthermore, significant variation was observed in SP1, between LC and BC, LC and DP, LC and CB and LC and HM.

Figure 10. Variation in the stem density across sampling periods at a) BC b) VP c) LC d) DP e) CB and f) HM**.** Error bars represent the standard error.



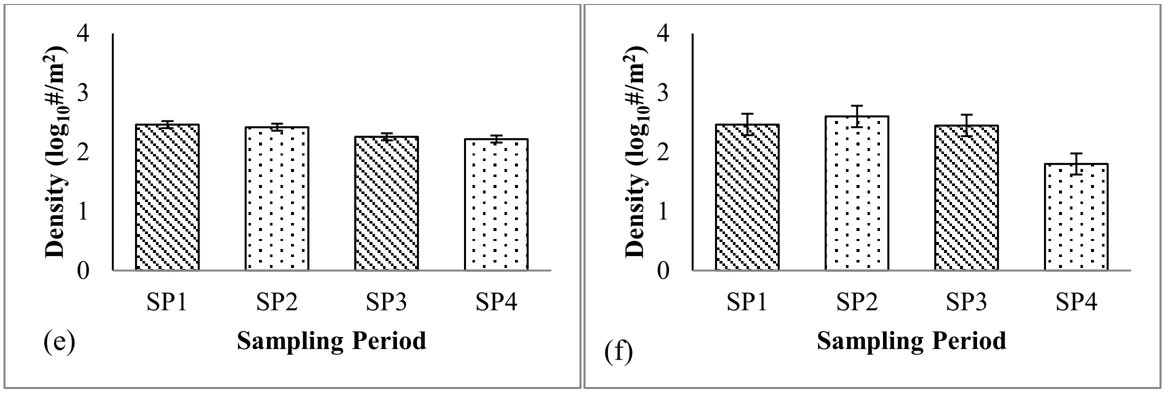


Figure 11. Variation in the stem density observed across sampling sites during a) SP1 b) SP2 c) SP3 and d) SP4. Error bars represent the standard error.

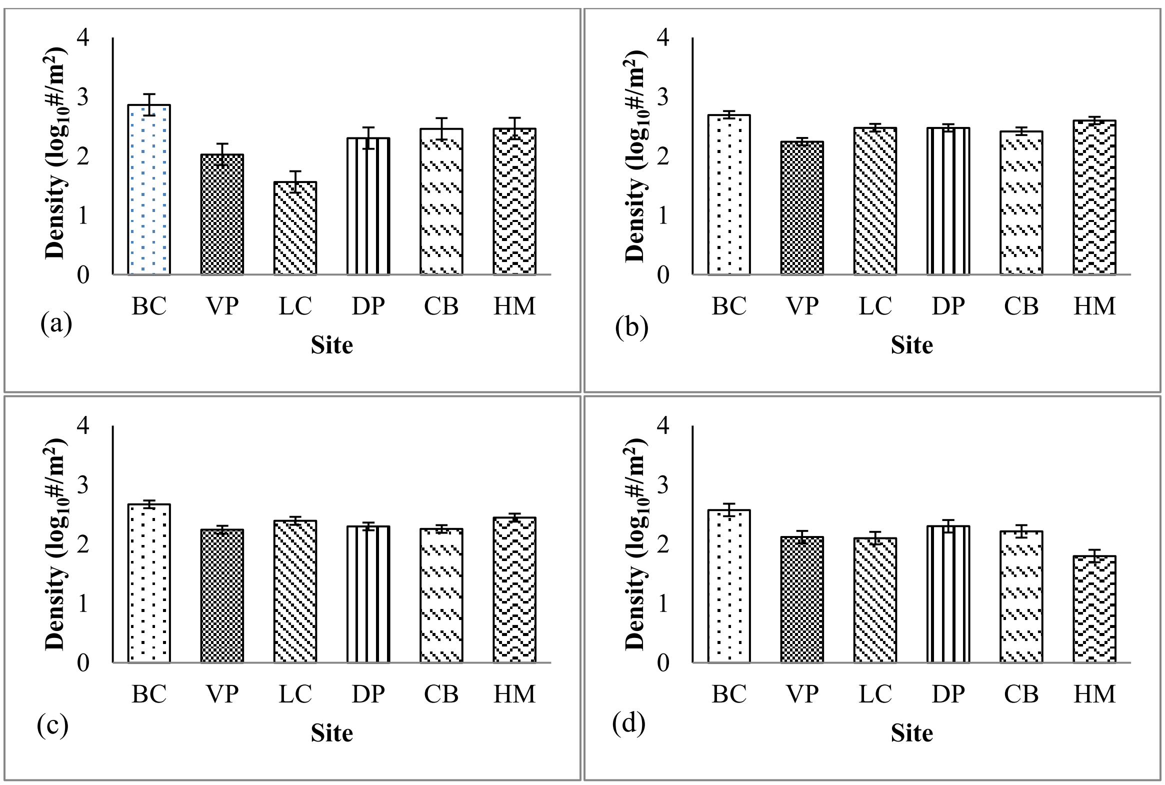
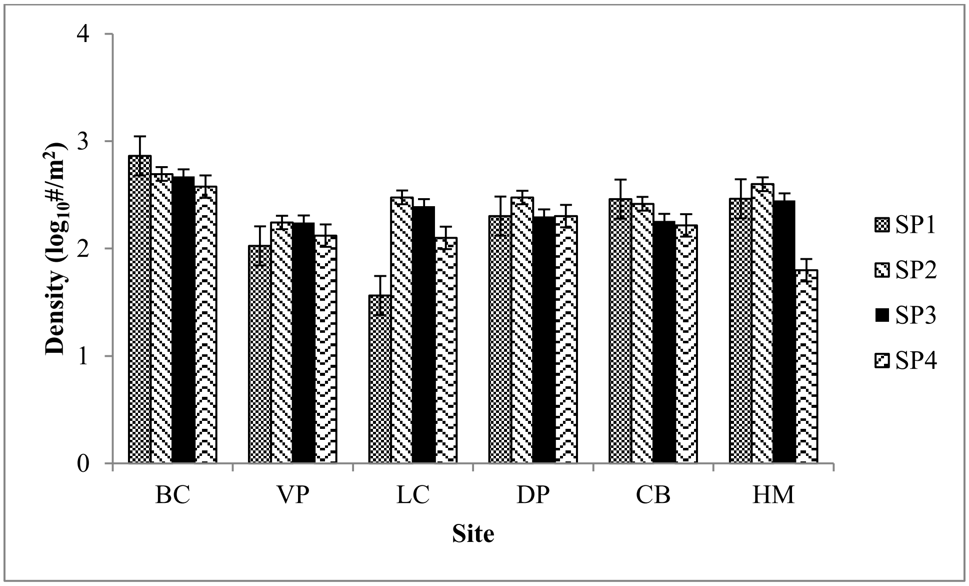


Figure 12. Variation in density observed across all sampling sites and sampling periods. Error bars represent the standard error.



3.3.5 Above-ground Biomass

3.3.5.1 Variation in above-ground biomass between sampling sites

During Fall 2013 (SP1), mean above-ground biomass was highest at VP (4731.66 g/m^2, SE= ±593.48), and lowest at LC (193.83 g/m^2, SE= ±69.36), both moderately disturbed sites. VP had the highest above-ground biomass for the entire study period (among all seasons) (Table 9). During SP1, above-ground biomass was significantly different between sites (one-way ANOVA $F_{5, 16} = 14.821$, $p < 0.001$). Games-Howell *post hoc* test showed that the increase in the mean biomass was significantly different between DP and CB, DP and HM, DP and BC and DP and VP (p =0.002, p =0.020, p =0.013, p =0.001, p = 0.049 respectively).

In the following Spring (SP2) the mean above-ground biomass was again highest at VP (3645.26 g/m^2, SE= ±843.45), and lowest at DP, the least disturbed site (214.17 g/m^2, SE= ±65.44) (Table 9). During this sampling period, above-ground biomass was significantly different between sites (one-way ANOVA $F_{3, 16} = 14.002$, p <0.001). Games-Howell *post hoc* test showed that biomass varied significantly between BC and CB (p =0.003), BC and HM (p =0.023), BC and VP (p =0.004) DP and HM (p=0.002), DP and CB (p =0.001) and DP and VP (p= 0.000).

As in the previous season, the mean above-ground biomass was again highest at VP (5060.34 g/m^2, SE= ±528.03) and lowest at DP (498.29 g/m^2, SE= ±111.56) (Table 9). Above-ground biomass varied significantly between sites (one-way ANOVA $F_{5, 41}$ = 8.316, p < 0.001). Tukey's *post hoc* test showed that biomass was significantly different between DP and BC, VP, LC, CB, and HM (p =0.046, p =0.000, p =0.012, p =0.000, p =0.003, respectively). The above-ground biomass was also significantly different between VP and BC (p =0.011) and VP and LC (p =0.045).

During Fall 2014 (SP4), the mean above-ground biomass was again highest at VP (3266.06 g/m^2, SE= ±461.08) and lowest at DP (498.51 g/m^2, SE= ±136.46) (Table 9). According to the one-way ANOVA, the biomass varied significantly between sites ($F_{5, 16}$ = 4.162, p < 0.05). Games-Howell *post hoc* test showed that biomass was significantly different between DP and VP and DP and CB (p =0.041, p =0.022, respectively).

Over the study period, the site with the greatest mean above-ground biomass was VP (4175.83 g/m^2, SE= ±428.22), and the control site DP had the lowest mean (395.11 g/m^2, SE= ±67.54) (Table 9). The variation in mean above-ground biomass between sampling periods is shown in Figure 13, while the mean species density variation between sites is represented in Figure 14.

3.3.5.2 Variation in above-ground biomass between sampling periods

The season with the greatest mean above-ground biomass for all sites was Summer 2014 (2476.93 g/m^2, SE= ±661.45), and the lowest was Spring 2014 (1673.87 g/m^2, SE= ±545.95)

(Table 9). Significant variation in biomass between seasons was observed only at BC and LC. At BC, the above-ground biomass varied significantly with sampling period (one-way ANOVA $F_{3, 27} = 7.561$, $p < .05$). Tukey's *post hoc* test showed a significant variation between SP2 and SP3 (p =.036), SP2 and SP4 (p =.001) and SP1 and SP2 (p =.003). Above-ground biomass was highest in Fall 2014 (SP4) (2077.83 g/m^2, SE= ±256.53) and lowest at Spring 2014 (548.11, SE= ±145.99) (Table 9). The variation in biomass showed a decrease in biomass from Fall 2013 (2050.17, SE=433.34) to Spring 2014, and then increased towards Fall 2014. At LC, the above-ground biomass was significantly different for different sampling periods ($F_{3, 27} = 3.729$, $p < 0.05$). Tukey's *post hoc* test showed that above-ground biomass was significantly different between SP1 and SP3 (p =0.037) and between SP1 to SP4 (p=0.038). The above-ground biomass was highest in Summer (2073.83 g/m^2, SE= ±590.66) and lowest in Fall 2013 (193.83 g/m^2, SE= ±69.36). From Fall 2013, to Spring 2014, biomass decreased, and then increased by Summer 2014, but dropped slightly by Fall 2014 (Table 9).

3.3.5.3 Influence of sampling periods and sampling locations – Interaction outcome

The results of the two-way ANOVA showed that above-ground biomass varied significantly between sampling periods ($F_{3, 139}$= 4.705, p<0.05) and between sites ($F_{5, 139}$=22.864, p<0.01) (Figure 15). Interaction results of two way ANOVA between sites and sampling periods resulted in significant variation ($F_{15, 139}$=3.998, p<0.01). The *post hoc* test showed that LC was the only site where the biomass between all sampling periods varied significantly (p<0.05). Furthermore, the *post hoc* test also showed significant variation between all sites in all sampling periods (p<0.05).

Figure 13. Variation of above-ground biomass observed across sampling periods at a) BC b) VP c) LC d) DP e) CB and f) HM. Error bars represent the standard error.

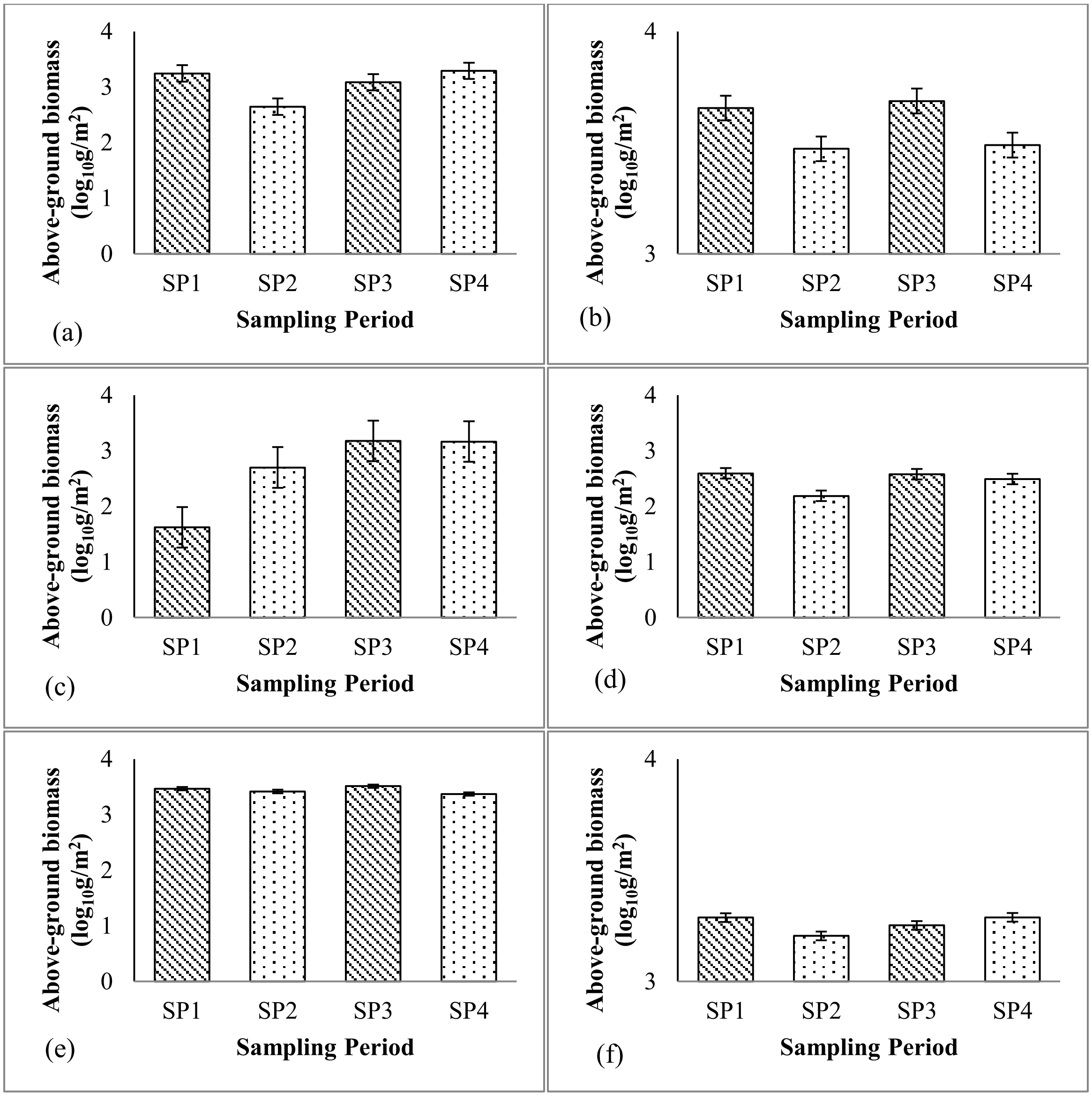


Figure 14. Variation in above-ground biomass observed across sampling sites during a) SP1 b) SP2 c) SP3 and d) SP4. Error bars represent the standard error.

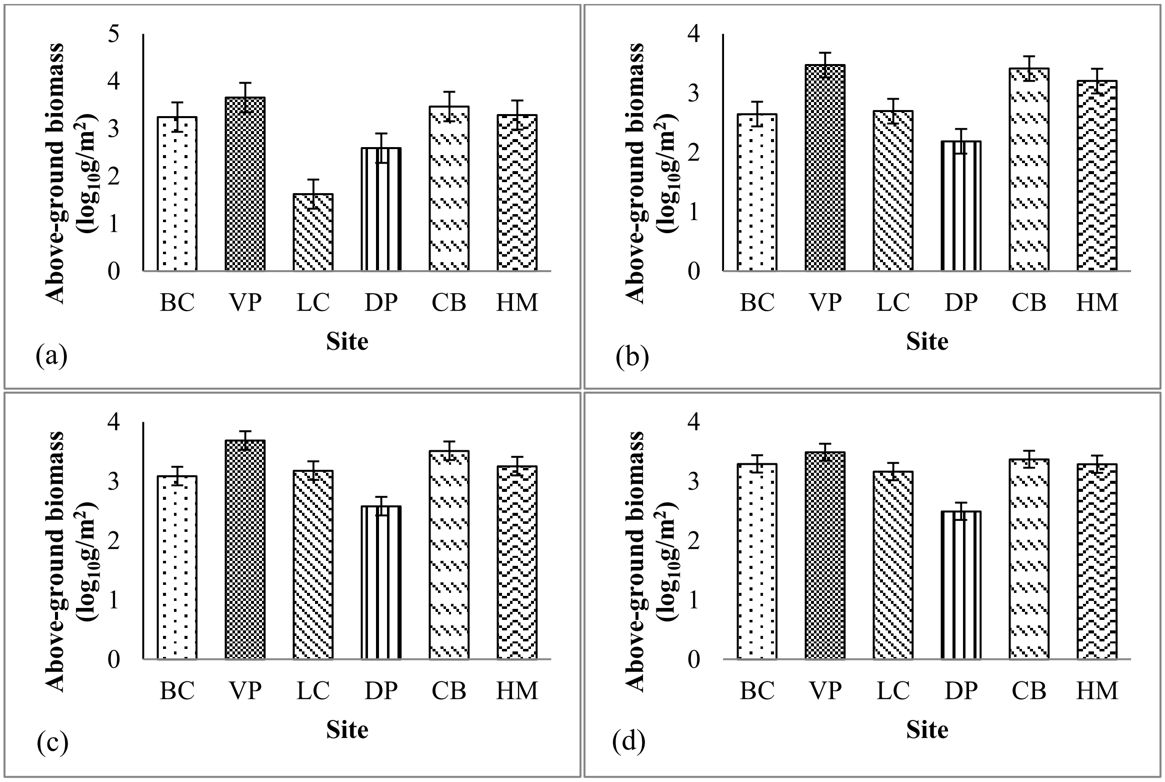
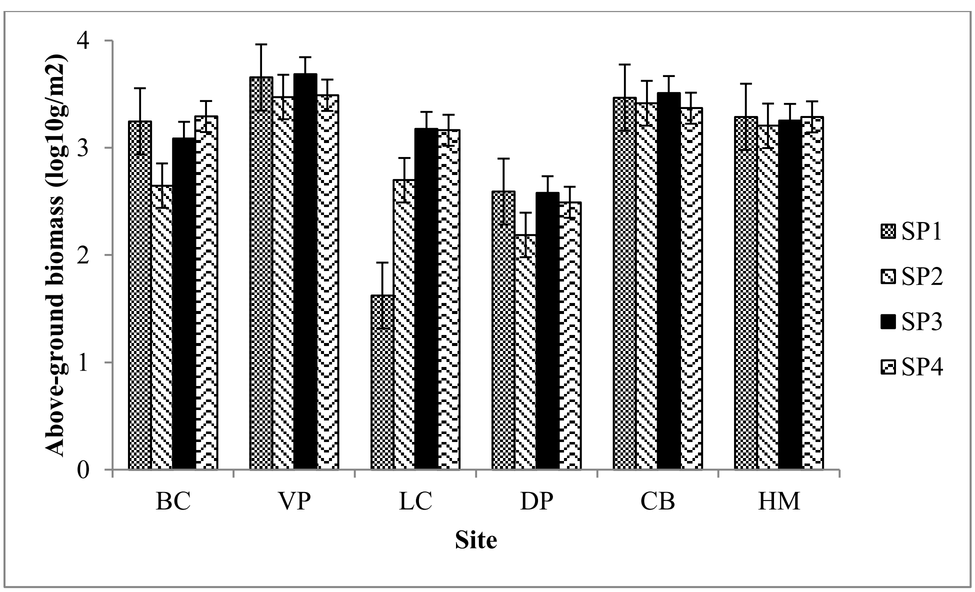


Figure 15. Variation in above-ground biomass across all sampling sites and sampling periods. Error bars represent the standard error.



3.3.6 Species Diversity

3.3.6.1 Variation in diversity between sampling sites

The mean Simpson's diversity was highest at HM (4.03, SE= ±0.78) and lowest at LC (1.00, SE= ±0.00) during Fall 2013 (SP1). All sites except LC exhibited a higher species diversity than the reference site DP (1.05, SE= ±0.05) (Table 9). Simpson's diversity was significantly different between sampling sites (one-way ANOVA $F_{3, 27}$= 8.316, $p < .001$). Tukey's *post hoc* test showed that diversity was significantly different between BC and HM (p =0.018), CB and HM (p =0.033), LC and HM (p =0.000), and DP and HM (p =0.000)

During SP2 the mean diversity was highest at HM (4.77, SE= ±1.01 and lowest at BC (1.50, SE= ±0.14) (Table 9). During this sampling period the diversity did not differ significantly between sites (one-way ANOVA $F_{5, 16}$ = 2.838, p= 0.05).

The mean species diversity was highest at HM (3.28 1.68, SE= ±0.64) and lowest at DP (1.43, SE= ±0.33) during Summer 2014 (SP3), indicating that all sites had a higher species diversity than that at the least disturbed site (Table 9). According to the one-way ANOVA, the Simpson's diversity varied significantly between sites ($F_{5, 15}$ = 8.295, $p < 0.05$). A Games-Howell *post hoc* analysis revealed that diversity was significantly different between the least disturbed site and the three moderately disturbed sites, DP and BC (p =0.037), DP and VP (p =0.027), and DP and LC (p =0.042).

During Fall 2014 (SP4), unlike the three previous sampling periods, the diversity was highest at BC (2.66, SE= ±0.24), but still lowest at DP (1.05, SE= ±0.05) (Table 9). During this sampling period, the diversity was significantly different between sites (one-way ANOVA $F_{5, 36}$ = 8.765, $p < 0.001$). Tukey's *post hoc* analysis revealed that the mean diversity varied significantly between VP and BC (*p* =0.023), LC and BC (*p* =0.002), DP and BC (*p* =0.000), DP and CB (*p*=0.003), BC and CB (*p*=0.001), BC and HM (*p* =0.001), and CB and HM (*p* =0.034).

The variation in mean diversity between sampling periods is shown in Figure 16, while the mean diversity variation between sites is represented in Figure 17.The highest diversity measure was at HM in Spring 2014 (4.77, SE= ±1.01) and the lowest was at VP (1.00, SE= ±0.00) and DP (1.05, SE= ±0.05) during SP1. HM also had the highest mean diversity over all sampling periods and sampling sites (3.35, SE=1.74) while the reference site had the lowest mean diversity (1.26, SE= ±0.12). The season with the highest mean diversity was Summer 2014 (2.26, SE= ±0.29) while the lowest was in Fall 2014 (1.70, SE= ±0.25) (Table 9).

3.3.6.2 Variation in species diversity between sampling periods

The mean species diversity was highest in Summer 2014 (2.26, SE= ±0.29) and lowest in Fall 2014 (1.70, SE= ±0.25) (Table 9). VP and CB were the only sites that did not differ significantly between seasons (p>0.05). This could be due to the dominant perennial species, *Typha*, persisting across the seasons.

At BC, the diversity varied significantly between sampling periods (one-way ANOVA $F_{3, 12}$= 6.438, p < 0.05). Games-Howell *post hoc* analysis revealed that the variation between SP2 and SP4 was significant (p = 0.014). During SP2 the mean diversity was 1.50, SE= ±0.14, while in SP4 it was 2.66, SE= ±0.24, illustrating a significant increase in diversity from SP2 to SP4 (Table 9). At LC, one-way ANOVA results revealed significant differences in diversity ($F_{3, 24}$ = 4.531, p < 0.05). Tukey's *post hoc* analysis revealed that variation between SP1 and SP3 was significant (p = 0.007). Figure 16 shows an increase in mean diversity from SP1 (1.00, SE= ±0.00) to SP3 (1.85, SE= ±0.18).

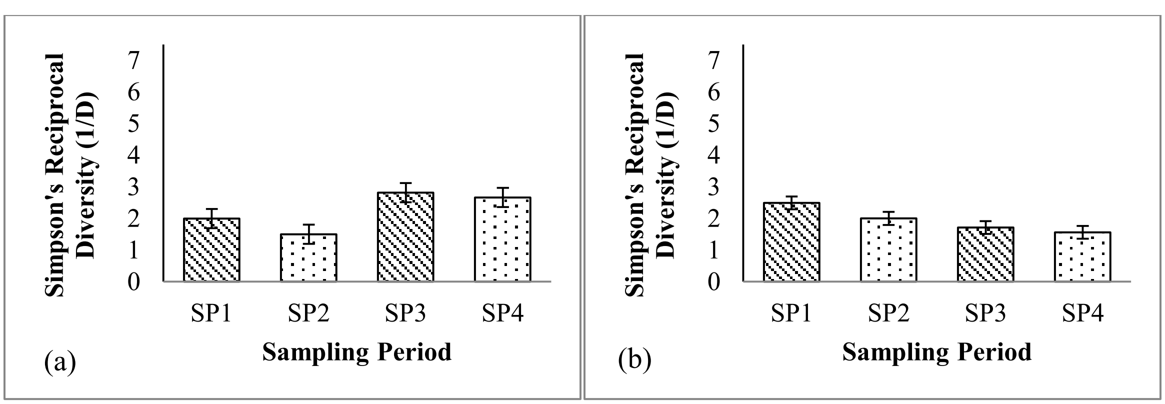
At DP, Simpson's diversity was significantly different between sampling periods (one-way ANOVA $F_{3, 24} = 4.554$, $p < 0.05$). Tukey's *post hoc* analysis revealed that the mean difference in diversity between SP1 and SP2 showed significant variation ($p = 0.010$).

At the highly disturbed site HM, the diversity varied significantly between seasons (one-way ANOVA $F_{3, 24} = 3.929$, p < 0.05). Tukey's *post hoc* analysis revealed that the variation between SP2 and SP4 was significant (p =0.018). During SP2 the mean diversity value was 4.77, SE= ±1.01, while during SP4 at HM the diversity had a mean value of 1.34, SE= ±0.18, illustrating a significant decrease between these sampling periods (Table 9).

3.3.6.3 Influence of sampling periods and sampling locations – Interaction outcome

The results of the two-way ANOVA showed that the Simpson's diversity index varied significantly between sites ($F_{5, 131}$=27.771, p <0.01) but not between sampling periods ($F_{3, 131}$=2.336, p=0.07) (Figure 18). Interaction results of the two-way ANOVA between sites and sampling periods resulted in a significant variation ($F_{15, 131} = 5.619$, $p = 0.000$). An analysis of simple main effects (the *post hoc*) after Bonferroni adjustment showed that there was significant variation between sampling periods in BC, LC, and HM (p <0.05). Furthermore, the *post hoc* analysis showed significant variation between all sites and all seasons (p <0.01).

Figure 16. Variation in **s**pecies diversity across sampling periods at a) BC b) VP c) LC d) DP e) CB and f) HM. Error bars represent the standard error.



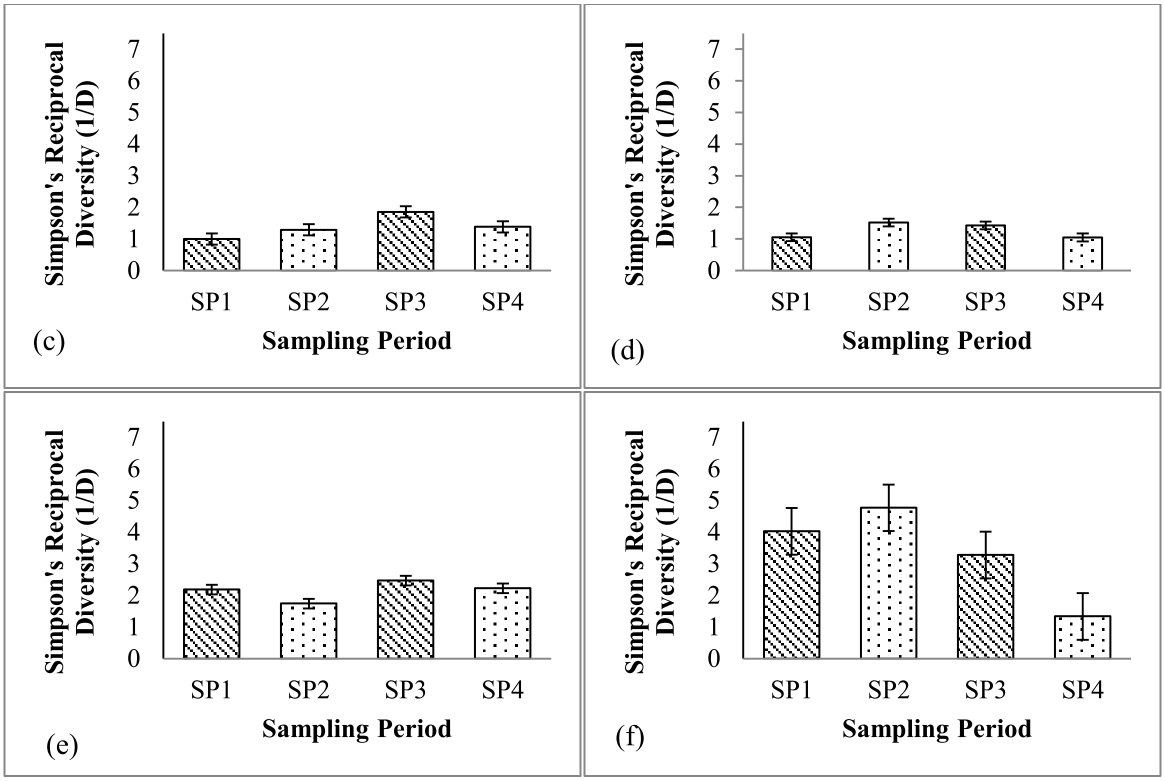


Figure 17. Variation in species diversity across sampling sites during a) SP1 b) SP2 c) SP3 and d) SP4. Error bars represent the standard error.

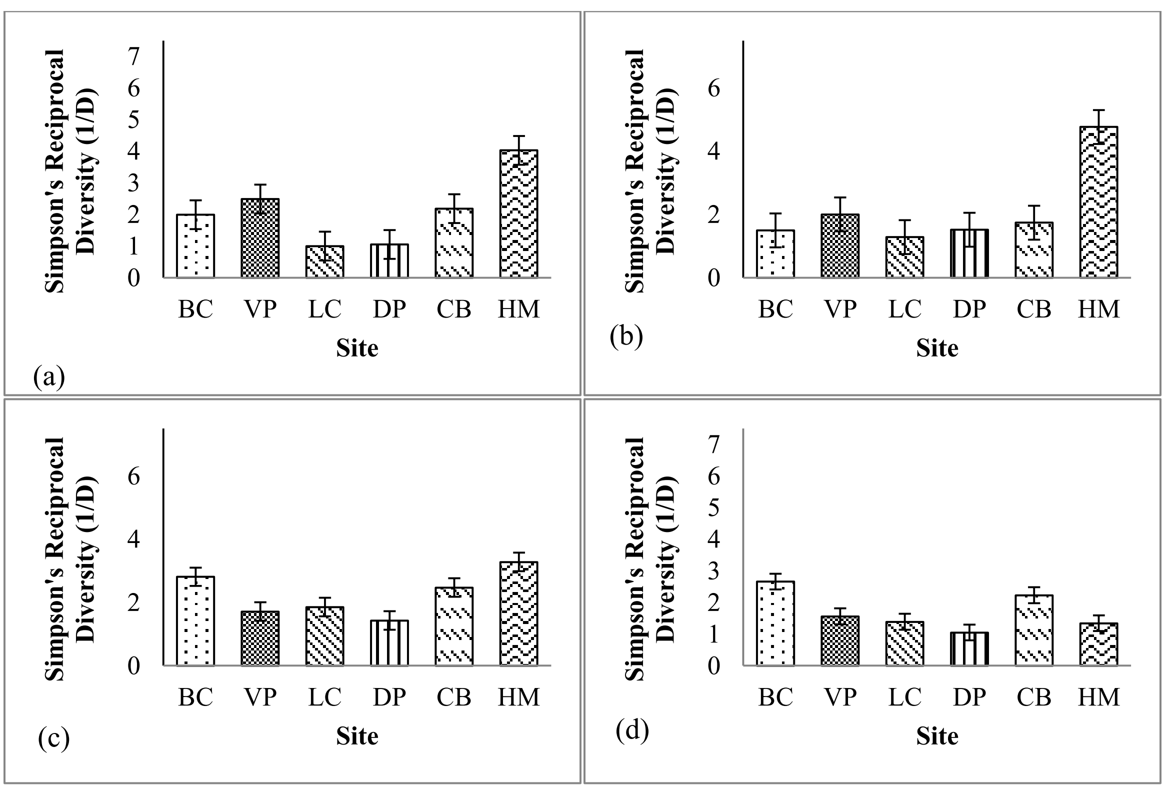
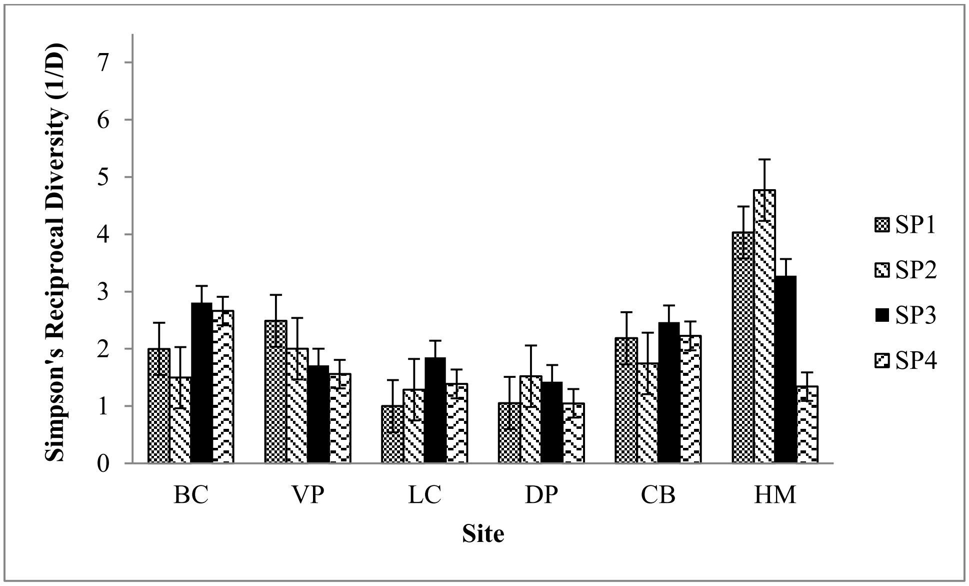


Figure 18. Variation in species diversity across all sampling sites and sampling periods. Error bars represent the standard error..



3.4 Discussion

3.4.1 Natural Variation

De Catanzaro, Cvetkovic, & Chow-Fraser (2009) examined the effect of watershed features on coastal water quality and showed that bedrock type did not have a significant influence on nutrient concentration and other water quality parameters, such as suspended solids and conductivity in Georgian Bay watersheds. Instead, factors such as wetland cover, watershed area, and road density were significantly correlated with water chemistry and overall water quality condition in these low-impact marshes. Similarly, McNair & Chow-Fraser (2003) showed regional variation in climate and geology did not significantly affect water quality in Great Lakes coastal wetlands. Hence, the underlying geology of the Lake Simcoe wetland sampling sites should not significantly impact measured limnologic parameters (Cvetkovic & Chow-Fraser 2011).

3.4.2 Variation in Limnologic Parameters

The temperature of the water column appeared to follow the natural seasonal thermal cycle, which explains the significant variation in temperature between seasons. The results support the general finding that temperature is changes significantly over the course of the day (diel variation) and between seasons (Credit Valley Conservation Authority 2010). Dissolved oxygen (DO) did not differ between sites and seasons, an unexpected and conflicting result, as levels of DO typically fall under conditions of nutrient enrichment and eutrophication at the levels evident in the moderately and highly disturbed sites. According to the Canadian Council of Ministers of the Environment (CCMOE) (1999) macrophytes are expected to have an effect on seasonal DO concentrations, with higher oxygen levels during the growing season and low levels in late fall as vegetation dies back and decomposition proceeds. It would also be expected then that variation in DO due to macrophytes would have also been evident, as different sites demonstrated variation in macrophyte density. However such differences were not reflected in the data. Still, as indicated in the literature as a general quality of wetlands (CCMOE 1999; Mitsh & Gosselink 2007) the mean DO concentrations of all sites were generally low. A concentration of 7 - 11 mg/L is required for most stream fish (Cary Institute of Ecosystem Studies 2015); in the present study all mean concentrations ranged from about 3 to 7 mg/L (Table 3). The anoxic conditions measured in VP and CB reflected the nonpoint source loading of organic materials and level of organic material observed at these sites. DO in surface waters may range from nondetectable to 18.4 mg/L (CCMOE 1999). DO concentrations across all sites and sampling periods are consistent with this, ranging from 0.47 mg/L to 14.76 mg/L (Table 3).

Measure of electroconductivity in the water differed significantly between sites. It is not surprising that CB and HM had the highest mean values, as these wetlands are located in urbanized and agricultural areas that deliver large volumes of highway run-off and fertilizer runoff that contain elevated concentrations of inorganic dissolved solids. Unexpectedly, rather than DP having the lowest conductivity values, VP did. This is an indication of pollution caused by organic compounds, such as oil or other organic contaminants, which decrease conductivity as these elements do not break down into ions (Kemker 2014).

An absence of variation of pH values across all six wetlands suggests that pH was generally circumneutral (pH 5.5 to 7.4), though some values were alkaline (pH>7.4) (Tiner 1999). When pollution results in higher productivity from increased temperature or excess nutrients, pH levels increase as allowed by the buffering capacity of the water (Washington State Department of Ecology 2016) Therefore, due to the known differences in anthropogenic disturbances among sites, some variation in pH would have been expected. Although these small changes in pH are not likely to have a direct impact on aquatic life, they can greatly influence the solubility of all chemicals and exacerbate nutrient levels, by, for example, increasing the solubility of phosphorus and thereby increasing its availability for plant growth (Kemker 2013). Minc & Albert (2004) collected water chemistry data in Great Lakes coastal wetlands in Northern Lake Michigan and Lake Huron, where typical pH values ranged between 7 - 9, well above the point at which acid pH is known to adversely affect plant growth. They suggest that pH may be spurious and not provide a reliable metric.

The chlorophyll *a* concentration did not differ significantly between seasons or sites. This was unexpected, but this finding may explain or correspond to the lack of variation in DO and pH. Algae, which produces chlorophyll *a*, can cause changes in pH through photosynthesis and respiration. Furthermore, when algae die and decompose, oxygen is consumed by decomposer aerobic bacteria, resulting in a reduction in DO level in the water (Washington State Department of Ecology 2016). Therefore, these relationships provide an insight to the lack of variation among all three parameters. It would have been expected that sites with higher nutrient inputs (CB and HM) would have had significantly greater chlorophyll *a* concentrations than those found in the least disturbed site DP. Though no significant variation was reported, the mean chlorophyll *a* concentrations at CB and HM were more than twice those found at DP. Mean chlorophyll *a* concentration reflected the degree of disturbance impacting each wetland site; the least disturbed site DP had one of the lowest concentrations, moderately disturbed sites reflected mid-level concentrations, and the most disturbed sites had the highest concentrations. This is consistent with McNair & Chow-Fraser's (2003) report that periphyton and phytoplankton chlorophyll *a* can be used as indicators of wetland degradation. They found that in undisturbed wetlands, both periphyton and phytoplankton biomass are limited by low nutrient availability, whereas in highly disturbed wetlands, excessive nutrient loading resulted in high periphyton and phytoplankton, generating elevated readings of chlorophyll.

Variation of total phosphorus (TP) between sites reflected the anthropogenic disturbances in those wetlands. Concentrations of TP indicate that the least disturbed site DP had oligotrophic conditions; BC and LC, which are moderately disturbed sites had mesotrophic conditions, and VP, though considered to be moderately disturbed site had eutrophic conditions, as did the highly disturbed sites CB and HM. In 2008, the LSRCA reported that mean annual TP values in nearshore areas of Cook's Bay were about 21.9 μg/L (range 12.0 – 48 μg/L), however in this study the mean annual TP was almost three times this value. Higher TP concentrations in CB, HM and VP reflect the high to moderate degree of anthropogenic disturbances at these sites due to agricultural and non-point sources of pollution, urbanization, etc. High TP (and TN) concentrations are interpreted as nutrient enrichment resulting from sewage, manure, fertilizer, and input (Minc & Albert 2004). The annual TP concentrations in the wetlands during this study were compared to those collected from the open lake closest to each wetland in 2009 – 2012 (LSRCA 2013b). The open water station distances from each wetland were: for BC and VP, less than 5 km; for LC and DP just over 8 km; and for CB and HM each station was approximately 1 km away. Open water measurements were similar to the corresponding nearby sites except in the case of VP (open lake monitoring site ranges of TP were 9-10 µg/L by VP; 11 – 20 µg/L by BC, LC, and DP; 31 – 50 µg/L by CB, and 51- 100 µg/L by HM). The open lake monitoring stations near VP showed that annual TP concentrations in the open water were 9-10 µg/L compared to average values of 45.25 µg/L (±9.98 SE) observed in the wetland (Table 3). The finding that open water TP at Atherley Narrows did not reflect the range of VP could be due to the fast dilution of the water occurring in the Narrows as part of the water flow to Lake Couchiching from Lake Simcoe (Young & Jarjanazi 2014). Nonetheless, all other wetland sites had recorded annual TP concentrations that were close to, or within the range measured by the closest open water monitoring stations. This underscores the value of TP monitoring at the fringe wetlands as a preliminary indicator of the local and overall trophic status of the lake. This result is consistent with Trebitz et al. (2007), who collected water quality data from 58 wetlands across the U.S. shoreline of the Laurentian Great Lakes and determined that the coastal wetlands broadly reflected the status of the adjacent lake.

Concentrations of TN at the sites showed a similar pattern to TP, with the lowest concentration at DP and highest at CB. However, TN and TP were not in agreement between sampling periods. These differences between seasons reflect natural seasonal fluctuations in nutrient supply to freshwater bodies driven by external factors (Heathwaite 1993). Higher TN concentrations in the moderately and highly disturbed sites compared to the reference site are an indication of both nutrient enrichment and industrial activity, in the form of wastewater treatment plants (Minc & Albert 2004) which, as mentioned in Chapter 2 are located in the vicinity of the macrophytes, especially at CB and HM.

A potential weakness and/or limitation to the limnologic data are that they vary in time, over both the short- and long-term. The set of individual measurements made in this study may not adequately reflect the prevailing chemical environment within which macrophytes grow. These temporal as well as spatial issues combined with an inadequate documentation in the literature of the relationships between specific macrophytes and chemical stressors make it difficult to define robust plant-based indicators of wetland health (Minc & Albert 2004). For example, repeated turbidity measurements in 15 Lake Ontario coastal wetlands showed that even within the same land-use environment, turbidity is highly variable through time (Environment Canada 2004b). Daily rainfall, wind-speed, and carp and waterfowl activity changed water clarity from week to week, with 6-fold increases in turbidity observed in a one wetland subsequent to a major rain storm. In this study, although repeated measurements were taken for water quality variables, it is clear that a single sampling of water quality may not reflect the range of conditions to which aquatic vegetation is responding to (Minc & Albert 2004). Another consideration was the collection of the limnologic data close to the shoreline. Chow-Fraser (1999) stated that in Great Lakes coastal wetlands, the flow of water between the marsh and its adjacent Great Lake can be reversed depending on wind direction, watershed inputs, and water level. Consequently, mixing with lake water may ameliorate the effects of upstream pollution, while wind and wave action in exposed coastal marshes may cause organic matter to be exported from the wetland to the lake (Lougheed et al. 2001).

In addition to collecting the limnologic parameters, it would have been advantageous to collect sediment quality parameters also, such as TP, TN and other trace elements. Sediment-water interactions occur in wetlands and biogeochemical processes regulate the exchange of solutes and other chemical constituents between the soil and water. The soil-water interface separates the mineral and organic matter of soils and sediments and pore waters from the overlying water column (Mitsch & Gosselink 2007: Reddy & DeLaune 2008). Large gradients of various dissolved substances can occur across the soil-water interface due to biogeochemical process. Such gradients can also develop as a consequence of anthropogenic inputs of nutrients and contaminants to the water column. These solutes are then subject to transport across the soil-water interface into the water column. The zone of soil-water interface is distinguished by residence time of reactive solutes and biological activities. In the water column of wetlands there is a biologically active layer, including bethnic periphyton mats and vegetation in the water column, such as a bed of submerged, floating, and emergent aquatic vegetation. Thus, the exchange of solutes across the soil-water interface is affected by not only the physicochemical processes at the interface but also by the activities of biotic communities such as aquatic vegetation (Reddy & DeLaune2008). Uptake up nutrients and trace chemicals is by the roots which are typically located in the soil (Kadlec & Wallace 2008). Although sediment would have been ideal to include in this study, the interactions of sediment with the water column validate the limnologic parameters measured and demonstrate the indirect connection of emergent macrophytes and water quality. Furthermore, emergents macrophytes may actually be exposed directly to the water column through growth of adventitious roots (Kadlec & Wallace 2008).

3.4.3 Limnologic Data Reflects Intensity of Anthropogenic Influences

The water chemistry data reflected the degree of anthropogenic disturbance influencing each site. The least impacted wetland, DP, had low nutrient levels, low conductivity, low CHL *a,* and higher DO concentrations. The most degraded sites, CB and HM, had high nutrient concentrations, high conductivity, high CHL *a,* and lower DO levels. The sites exposed to moderate disturbances, BC, VP and LC, had nutrient levels, conductivity, CHL *a* and DO concentrations that were intermediate between the least and most disturbed sites. In some cases a moderately disturbed site would have a value higher or lower than a value reported for the least or most disturbed site. Nonetheless, the results were consistent with other findings in the literature associating anthropogenic impacts and water quality. McNair & Chow-Fraser (2003) also found that in the Great Lakes basin, degraded wetlands had high nutrient concentrations, poor water clarity (measured in terms of high light extinction coefficient, suspended solids, turbidity), and high conductivity. At the other end of the gradient, pristine wetlands had low nutrient levels and good water clarity. McNair & Chow-Fraser (2003) found that wetlands with the lowest nutrient levels and clearest water were in the Lake Superior basin, in areas characterized by relatively minimal human influence, whereas those with the highest nutrient levels and most turbid water were located in one of the most heavily impacted areas of Lake Erie, in the western basin. In developing the WMI, Croft & Chow-Fraser (2007) found that pristine wetlands in Georgian Bay were associated with the lowest concentrations of nutrients and suspended solids, at levels that were significantly lower than the corresponding values for wetlands associated with urbanization and receiving large volumes of highway runoff. They determined that mean CHL *a* in the least degraded wetlands were ten times lower than those in the more degraded Lakes Erie and Ontario (2.28 versus 24.82 and 16.37 mg/m^3, respectively). In this study, CHL *a* levels in the least degraded wetland DP were just over two times lower than HP which had the highest mean CHL *a* concentration. Similarly, the mean conductivity ranged from 126 in Georgian Bay to a high of 388 μS/cm and 470 μS/cm for Lakes Erie and Ontario, and in Lake Simcoe a wide range of conductivity levels was evident (176 to 617 μS/cm); the higher conductivity levels were associated with CB and HM, the most degraded wetlands (Table 3). Croft & Chow-Fraser (2007) also found that pH values for all five Great Lakes were generally circumneutral, which agreed with the data from the Lake Simcoe wetlands.

According to McNair and Chow-Fraser (2003), in healthy systems aquatic vegetation is abundant, which reduces turbidity by trapping sediment and provides substrata for epiphyton and habitat for benthic invertebrates. Algae and vegetation absorb nutrients entering the watershed, but planktonic and benthic algal biomass is kept low by zooplankton and benthic grazers. Grazer populations in turn support a diverse planktivore and benthivore fish community controlled by piscivorous fishes. In contrast, degraded systems have little aquatic vegetation and are turbid and nutrient-rich, producing high planktonic and benthic algal biomass that further decreases light for aquatic plant germination. In DP, not only was there a high density of emergent macrophytes, but a high density of submergent vegetation was also observed during sampling. Nutrient levels were low and this site had one of the lowest CHL *a* measurements.

Lougheed et al. (2001) found that the proportion of agricultural and urban land in wetland watersheds was a highly significant predictor of water quality, and other studies also related water quality to land use in the watershed (Johnson, Richards, Host & Arthur 1997; Crosbie & Chow-Fraser 1999).

3.4.4 Macrophyte Community Analysis and Dynamics

The highest Sørensen coefficient between the reference site DP and the other sites impacted by increasing degrees of anthropogenic stress was 0.31 or 31% for LC, and the lowest coefficient was 0.15, or 15% for BC. This coefficient revealed only a 31% similarity between LC and DP, and this being the highest coefficient indicates that LC was the site most similar to DP. This and the other low similarity coefficients (Figure 6) signify great dissimilarity between the reference site and the other 5 study sites. This supports the contention that each sampling site is exposed to varying degrees of anthropogenic stresses, and as a result, differences between wetland sites in terms of community composition are apparent. It should be noted that binary similarity coefficients including Sørensen's can be crude measures available for judging similarity between communities, as commonness and scarcity are not into consideration. Binary coefficients thus weigh rare species the same as common species, and should be used whenever one wishes to weigh all species on an equal footing. More commonly, binary similarity measures are used because only lists of species names are available for particular communities and comparisons are possible only at this lower level of resolution (Krebs 1999).

It was initially hypothesized that the most degraded sites (CB and HM) would have the lowest species richness and diversity and the greatest densities and above-ground biomass. Along the disturbance gradient, the least disturbed site was expected to have the greatest species richness and diversity and lowest density and above-ground biomass. However, the data showed that species richness was highest in the most disturbed sites and lowest in the least disturbed sites. Density was greatest in the moderately disturbed sites, and lowest in the least disturbed site. Diversity was greatest in the highly disturbed sites and lowest in the least disturbed site. As had been expected, above-ground biomass was greatest in the highly disturbed sites and lowest in the least disturbed site.

3.4.5 Taxonomic Composition

This section of the discussion will broadly describe the wetland communities present that spanned across the gradient of anthropogenic stressors impacting Lake Simcoe. The discussion focuses on the species that were described as dominant having the greatest densities at each site, but other species are also mentioned. The discussion explains that the community structure was a function of the water quality, and thus degree of anthropogenic disturbance.

Simon et al. (2001) and Stewart et al. (1999) have asserted that the number of exotic plant species present at a wetland site is a good indicator of the level of site degradation. In this study, all sites had a greater richness of native species, but VP and CB, moderately and highly disturbed sites respectively, had a greater percent composition of invasive to native species (Figure 2). This result suggests that measures that incorporate both abundance and richness of invasive species, and not simply the number of species as suggested by Simon et al. (2001) and Stewart et al. (1999) are more accurate when assessing community composition. The highly and moderately disturbed sites HM, CB, LC, BC, and VP had greater percent composition of invasive species than DP, the least disturbed site (Figure 3). Disturbance can facilitate exotic invasions (Wienhold & van der Valk 1989; Ehrenfeld & Schneider 1991; as cited by Simon et al. 2001)), which is reflected by the percent composition of invasive species at each wetland site.

Across all sites, the dominant and important species were: *Leersia oryzoides, Eleocharis smallii, Typha* x *glauca, Typha angustifolia, Sparganium eurycarpum, Scirpus acutus, Scirpus pungens, Calamagrostis canadensis, and Phragmites australis.* Of these nine dominant species, six are native to Ontario (*Leersia oryzoides, Eleocharis smallii, Calamagrostis canadensis,* *Sparganium eurycarpum, Scirpus pungens, and Scirpus acutus*) while only three are exotic/invasive (*Typha* x *glauca, Typha angustifolia, and Phragmites australis*). Lougheed et al. (2001) collected water quality, land use, and aquatic macrophyte information from 62 coastal and inland wetlands in the Great Lakes basin and comparably found that *Typha* and *Scirpus* were the most dominant species, and *Lythrum salicaria was* also encountered in those wetlands. *Lythrum salicaria*, though not dominant was still present in this study.

At DP, *Sparganium eurycarpum Scirpus acutus,* and *Scirpus pungens* were dominant. *Sparganium eurycarpum* is a robust, emergent, perennial herb growing up to 1.2 m tall distinguished by erect strap-like leaves and burr-like spiky flower heads (Newmaster et al. 1997). It can be found in a variety of habitats including marshes, fens, swamps, borders of ponds and slow-moving rivers, and sloughs. It has been reported that *Sparganium eurycarpum* is found in less disturbed wetlands with fertile soil or mineral-rich water It functions to control shoreline erosion, by acting as an effective natural filter for water clarification, and is a food source for waterfowl and muskrats (Favourite 2003). These services support its presence in the least disturbed wetland, however, the literature more frequently describes the cosmopolitan nature of this species (Croft & Chow-Fraser 2007; US EPA 2000), which was evident in this study from its presence in all sampling sites. Sulsan (2014) studied 75 *Sparganium* habitat localities in Wisconsin and found that this genus was present in mesotrophic to hypertrophic conditions. Sulsan (2014) also mentioned species frequently found with *Sparganium eurycarpum*. Those associations with this species observed in Wisconsin and in Lake Simcoe included *Sagittaria graminea latifolia* (broadleaf arrowhead)*, Typha latifolia, Phalaris arundinacea, Carex lacustris, Sparganium americanum* (American burreed)*, Zizania, Bidens* spp.*, Leersia oryzoides, Sium suave* (water parsnip)*, Calamagrostis canadensis,* and *Eleocharis* spp.

The other species dominating DP were *Scirpus acutus*, and *Scirpus pungens*. While *Scirpus acutus* and *Scirpus pungens* were also present in the moderately to highly disturbed sites, the dominance of *Scirpus pungens* was exclusive to DP, while *Scirpus acutus* was only also dominant in LC. *Scirpus acutus* and *Scirpus pungens* are perennial, rhizomatous, wetland obligate species growing up to 3 m high (Newmaster et al. 1997; Stevens, Hoag, Tilley, & St. John 2012; Tilley 2012). *Scirpus acutus* is generally found in areas of standing water with depths varying from 10 cm to more than 1.5 m; while *Scirpus pungens* is found only up to 30 to 45 cm deep water. *Scirpus acutus* can grow on peat to coarse soils, while *Scirpus pungens* prefers fine silty clay loam to sandy loam soil. Both can grow and spread in alkaline, saline, and brackish sites, and can tolerate periods of drought and total inundation (Stevens et al. 2012; Tilley 2012). *Scirpus acutus* has a dense root mass to stabilize soil, and its above ground biomass provides protection from wave action and stream currents that erode shorelines or stream banks. The rhizomatous root system also acts a substrate for endophytic bacteria able to degrade toxins and transform nutrients to remove them from a water body, therefore acting as an agent of water treatment through bioremediation (Anderson & Coats1994; Jorgenson, 2013). *Scirpus acutus* and *Scirpus pungens* form large, often monospecific stands, but without posing any environmental concern to native plant communities (Stevens et al. 2012; Tilley 2012). The dominance of this native genus at DP reflects a less degraded water quality and least amount of anthropogenic stress.

BC, VP, and LC sites are dominated by both native and invasive species, signifying the intermediate or moderate anthropogenic activities impacting them. While the dominant native species varied according to site, all moderately disturbed sites had *Typha* x *glauca* as a dominant species, and at VP *Typha angustifolia* was also dominant. The dominant native species observed at BC were *Leersia oryzoides* and *Eleocharis smallii*; at VP was *Sparganium eurycarpum*; and those observed at LC were *Scirpus acutus* and *Sparganium eurycarpum*. *Leersia oryzoides* is a perennial obligate wetland grass most common near streams, ponds, ditches, canals, and freshwater marshes. This species thrives best in nutrient rich mud and slow moving or stagnant water, and is tolerant of highly acidic conditions (pH 3.0). *Leersia oryzoides* has creeping rhizomes that are effective in sediment stabilization and erosion control along the immediate shorelines of streams and lakes. Its seeds and rhizomes are an important food source for shorebirds, waterfowl, and small mammals. It forms dense colonies that provide cover and act as habitat for amphibians, fishes, and reptiles, however these colonies can have a negative impact by potentially excluding other native marsh grasses and herbs. However, under undisturbed conditions it is regularly replaced by other species (Darris & Bartow 2004).

Eleocharis smallii is a heavily rhizomatous perennial wetland obligate sedge that grows between 10 to 100 cm tall. It is found in most marshes, lakeshores and riverbanks with permanent water up to 1 m deep, but can survive in areas where the water drops to 30 cm below the surface (Newmaster et al. 1997; Ogle, Tilley, & St. John 2012). It grows on fine texture soils in neutral to alkaline or saline conditions, developing a thick root mass that is resistant to compaction and erosion. *Eleocharis smallii* is a pioneer species rapidly populating in mud flats as the water draws down. Its dense root mass provides the benefits of soil stabilization and erosion control. The rhizomes also form a matrix for endophytic bacteria able to degrade toxins and transform nutrients, making *Eleocharis smallii* effective in cleaning polluted water. Under favourable conditions it can spread, but does not pose any environmental concern to native plant communities (Ogle et al. 2012). The dominance of this pioneer species provides insight on the disturbances impacting at BC. Pioneer species are typically more abundant only when newly disturbed habitats are colonized, and as succession proceeds, they usually decline (Simon et al. 2001). Consequently, BC is likely a newly or only recently disturbed wetland that has experienced lowering of the water level, allowing this species to increase in abundance.

In all moderately and highly disturbed sites, *Typha* x *glauca* was a dominant invasive species. *Typha angustifolia* was dominant at VP and CB. Notably, these invasive species and others discussed below were not present at the reference site. *Typha* spp. are perennial, rhizomatous, emergent wetland macrophytes that generally grow in organic soil substrates (Hall 2008; Newmaster et al. 1997; Tulbure et al. 2007). *Typha latifolia* and *Typha angustifolia* hybridize to form *Typha* x *glauca*, a highly productive hybrid that spreads prolifically by rhizomes after seedlings establish in disturbed vegetation, frequently forming monotypes that reduce wetland plant and animal diversity (Hall 2008). In North America, *Typha* x *glauca* has expanded westward from the eastern coast along with *Typha angustifolia*, while *Typha latifolia* has been historically widespread (Hotchkiss & Dozier 1949; as cited by Hall 2008). Tulbure et al. (2007) combined the non-native cattail *Typha angustifolia* and its hybrid *Typha* x *glauca* into a single class "invasive *Typha*" when documenting the rapid change in wetland vegetation of a Green Bay lagoon. *Typha angustifolia* is competitive in deeper (greater than 15 cm) water, while *Typha* x *glauca* appears to tolerate widely variable hydroperiods (Frieswyk & Zedler 2006).

The widespread expansion of each *Typha* species is evident in this study. In all sites except LC, all three *Typha* species were present. In LC, only *Typha* x *glauca* was present. This species' widespread presence in moderately to highly degraded sites is also reported in the literature; it invades communities that have eutrophic conditions and altered hydroperiods, and therefore is prevalent in these areas (Hill 2008). *Typha* spp. formed dense stands at CB and HM.

The extensive growth of *Typha* x *glauca* accelerates further degradation in addition to the anthropogenic influences that degrade wetlands in the first place. *Typha* x *glauca* appears to dramatically increase primary productivity and organic matter accumulation relative to the native species it replaces (Angeloni, Jankowski, Tuchman, & Kelly 2006). This may explain the high CHL *a* concentration measured at CB and HM, with *Typha* x *glauca* having the second greatest relative density at this site. *Typha* x *glauca* is also reported to alter soil microbial community structure, resulting in higher N and P concentrations (Angeloni et al. 2006). Visual inspection of organic matter accumulation during sampling revealed greater organic matter accumulation at all sites, in contrast to DP, where *Typha* x *glauca* was absent. It dominates the seed bank which reduces diversity, and also reduces invertebrate density and waterfowl habitat (Frieswyk & Zedler 2006; Linz et al. 1999; as cited by Hall 2008).

Typha angustifolia is also prevalent in disturbed wetlands and its expansion appears to be enhanced by anthropogenic disturbance, road construction, and the application of road salts (Grace & Harrison 1986). In Chapter 2, the extent of urbanization is described and this phenomenon helps explain the prevalence of *Typha angustifolia* in the moderately and highly disturbed sites.

Typha latifolia is the only *Typha* species traditionally associated with undisturbed wetlands (Grace & Harrison 1986). In undisturbed North American wetlands, *Typha latifolia* often grows sparsely with rush species such as *Juncus* and *Eleocharis*, sedges such as *Carex* and *Scirpus* species, and shrub species like *Cornus* and *Salix* (Curtis 1959; as cited by Hall 2008). In DP, the interim management strategy reports that the west and east inlet marshes support *Typha latifolia, Scirpus pungens*, *Zizania aquatica* and *Decodon verticillatus* (water willow) (Ontario Parks 2007), however *Typha latifolia* was not found in samples collected in this study, and it is worth noting that these associations were observed back in 1988.

In the highly disturbed sites CB and HM, *Typha* x *glauca, Typha angustifolia*, *Sparganium eurycarpum*, *Phragmites australis* and *Calamagrostis canadensis* were dominant. *Calamagrostis canadensis* is a native perennial grass growing up to 1 m tall. This species thrives in nutrient rich, saturated soils, peat, or deep, fine textured substrates that are moist all summer, in beaver meadows, ditches and shores, swamps, and fens (Newmaster et al. 1997). *Calamagrostis canadensis* withstands seasonal inundation and temporary spring flooding up to 15 cm deep, and has adapted a wide pH tolerance from very acidic to slightly alkaline soils (pH 3.5 to 8) (Darris 2005). Its rhizomes help to bind soil and stabilize shorelines and streambanks, and it provides shelter and nesting areas for waterfowl and shore birds (Newmaster et al. 1997). Following a disturbance it will rapidly expand, forming a complete stand that restricts establishment of other species. It may also become weedy or invasive and exclude desirable vegetation (Darris 2005). Its relatively high density in CB and HM and presence in BC supports this.

Phragmites australis is a very large perennial grass growing 2 to 4 m tall in deep fresh, brackish, or saline water. It is found along the banks of watercourses, artificial channel systems of irrigated and drained agricultural areas, and roadside ditches (Newmaster et al. 1997; OFAH/OMNR 2012; Tulbure et al. 2007). It prefers rich muddy substrates but has a high degree of plasticity that allows it to adapt to wide ranging substrates and water conditions that are usually (but not exclusively) stationary or slow-moving (lentic). It is present in oligotrophic to eutrophic conditions (Haslam 1972; Hocking et al. 1983; as cited by Invasive Species Compendium 2015).

The recent expansion of *Phragmites* in the Great Lakes region has been reported by Tulbure et al. (2007). They described *Phragmites* as a cryptic invasive species with both native and non-native genotypes that differ in their aggressiveness. Similar to *Typha*, *Phragmites* is an invasive taxon that is a clonal dominant. Establishment of *Phragmites* typically occurs on bare un-vegetated moist soils after water levels recede (Chambers, Meyerson, & Saltonstall 1999). Invasion begins from the upland area of the wetland through stolons and rhizomes. It could also start on a high point within the wetland, such as a dike remnant, and then expand into the wetland plain (Tulbure et al. 2007). The invasive form of *Phragmites australis* was identified in this study, and its presence among other macrophyte species suggests that expansion of this species is just beginning and that soon the species richness of these sites will decrease as other species will be out-competed by *Phragmites australis*.

3.4.6 Richness

Species richness is dependent on many factors such as nutrient concentration, temperature, duration of irradiance, etc. Richness differed among sampling periods and was greatest during SP2, when the highest mean seasonal water temperatures were recorded. Differences between sites suggest that the different degrees of anthropogenic degradation are important to species richness. It was initially hypothesized that species richness would be greatest in DP and lowest in CB and HM. However, the mean species richness was greatest at the highly disturbed sites, and lowest at the least disturbed site. This result is readily explained by applying the intermediate disturbance hypothesis, the observation that in many communities species richness is greatest at an intermediate frequency and/or intensity of disturbance; and that pristine conditions will sometimes have fewer and often different species than those found under disturbed conditions. This is because very frequent disturbance eliminates sensitive species, whereas very infrequent disturbance allows time for superior competitors to eliminate species that are less successful competitors (McGinley 2014; Simon et al. 2001). Yet with this hypothesis, richness should have been greatest in the moderately disturbed sites, rather than, as was found, in the highly disturbed sites. *Phragmites australis* and *Typha* spp. were dominant at CB and HM and under the predictions of the hump-backed model, (Grime 2000; Moore & Keddy 1989) this large standing crop of clonal dominant species would be expected to reduce species richness, as the greatest species richness is usually reached at moderate standing crops. However, Catford et al. (2012) suggested that human disturbance, coupled with plant introductions (like invasive *Phragmites australis* and *Typha* spp.), extends the diversity–disturbance curve and shifts peak diversity (richness) towards higher disturbance levels, explaining why richness was greatest in the highly disturbed sites. Although they only monitored one wetland before and after rapid expansion of non-native *Phragmites* and *Typha* spp., Tulbure et al. (2007) also observed an increase in species richness before and after disturbance expansion, that is, the site transitioned from minimally to highly disturbed. Those authors reported that although there was an increase in species richness, they believe that it was transient based on data collected at the site, and expect to see a decrease in species richness with the continued expansion and increase in density of *Phragmites*. The same could be predicted for species richness at CB and HM. However, according to Simon et al. (2001) this intermediate hypothesis trend has not been confirmed for aquatic plant communities.

3.4.7 Density

Variation in the density of stems between sites may be explained by the different species present at each site. Mean density was greatest in BC; this large density can be attributed to the dominance of *Leersia oryzoides* and *Eleocharis smallii*. These species have smaller stem diameters than some other dominant species such as *Typha* spp., and they were observed to form dense mats at this site. *Eleocharis smallii* was identified as a pioneer species, which are early successional plants that first colonize disturbed habitats and show dramatic year-to-year differences in population and species density (Whittaker 1993). They differ from tolerant species since they are more abundant only when newly disturbed habitats are colonized and as succession progresses they usually decline over time (Simon et al. 2001). The mean density at BC was almost three times that of the HM, the site with the second highest mean density. As previously mentioned this suggests that BC is a newly disturbed habitat and the density of *Eleocharis smallii* will decline over time. Still, community composition can change dramatically without a large change in density (Coulloudon 1999). Taxonomic composition and Sorsenson's coefficient showed that LC and CB were most different than DP, yet their total mean densities are somewhat similar (DP's mean density was 262.29 #/m^2, LC's was 219.43 #/m^2, and CB's mean density was 248 #/m^2) (Table 9) .

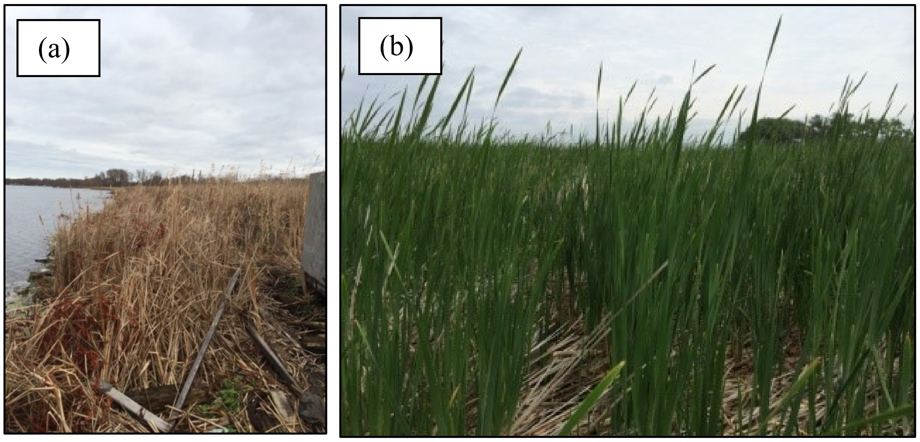
Plant density and species richness tend to be lower in oligotrophic systems compared to others (Lougheed et al. 2001). When compared in terms of their trophic status, DP, the only oligotrophic site, had a lower density than the mesotrophic and eutrophic sites, corroborating the finding of Lougheed et al. (2001). HM was the only site that showed variation in density seasonally, which was most likely a result of the large number of different perennial and annual species with various phenologies in terms of growth and dieback.

3.4.8 Diversity

Variation between sites in Simpson's diversity mirrors the same variation in species richness. Both take into account the number of species present, but diversity also incorporates the dominance or evenness of species in relation to one another (Krebs 1999). Most sites differed in diversity between sampling periods. The variation may be attributed to species tolerances of environmental conditions including parameters such as temperature, dissolved oxygen and availability of nutrients from the water column, which differ according to season, as well differences in phenologies in terms of growth and dieback of different species. The greatest mean diversity was during SP3, in the summer. During this sampling period, DO levels and water temperature were also high. It was hypothesized that diversity would be greatest in DP and lowest in CB and HM. However, the mean species diversity was highest at the highly disturbed sites, and lowest at the least disturbed site. This result is explained by the intermediate disturbance hypothesis, explored above in section 3.4.6, and see Catford et al. (2012).

The decrease of species diversity with increasing anthropogenic stress is well documented (Stuckey 1975, 1989; Van der Valk 1981; Keddy 1990; Wilcox et al. 1993; as cited by Albert & Minc 2004). Reduced natural water fluctuations, through manipulation of lake levels, can lead to an overall loss of both species diversity and richness (Albert & Minc 2004). Significant management and alterations of water levels of CB and HM are described in Chapter 2. Disruption of the natural cycle favours species intolerant of water-depth change and associated stresses, and/or excludes species requiring periodic exposure of fertile substrates. The result is frequently a monoculture of highly competitive species, particularly *Typha* spp., at the expense of diverse shoreline flora (Keddy 1989; as cited by Albert & Minc 2004). Figure 19 illustrates the dense stands of *Typha* observed at CB, yet mean species diversity at this site was high.

Figure 19(a) and (b). (a) CB during Fall 2013, demonstrating the lack of diversity of the shoreline and a dense stand of *Typha* spp. (b) CB in Spring 2014 showing the dense stand of *Typha* spp.



The dominant species at CB and HM included invasive *Phragmites australis*, *Typha* x *glauca* and *Typha angustifolia*. *Typha* x *glauca* can reduce diversity of not only plants, but also insects and birds by forming dense, monotypic stands in natural wetland communities. Plant species richness declined swiftly as *Typha* x *glauca* cover increased in wetlands of the Midwestern United States, including Great Lakes estuaries, constructed wetlands, sedge meadows, and marshes (Frieswyk & Zedler 2006; Hall 2008). Furthermore, its ability to produce allelopathic chemicals, along with increased competition for light and nutrients, should have resulted in a widespread pattern of reduced wetland plant diversity (Boers, Veltman, & Zedler 2007; Frieswyk & Zedler 2007; Hall 2008).

In BC, CB, and HM, the density of native *Scirpus* spp. was lower than *Typha* x *glauca*. In LC, however, the density of *Scirpus acutus* was actually greater than *Typha* x *glauca*, suggesting that a complete *Typha* invasion may be just beginning at this site, as the native species still dominate. Yet diversity was still greater in all of these sites compared to DP where not only did *Scirpus* spp. have the greatest density, but *Typha* x *glauca* was absent from this site. Therefore, in terms of *Typha* x *glauca* density and its effect on diversity, no clear relationship could be observed in this study, except for the result that DP, with the lowest species diversity, had no *Typha* x *glauca* present. *Phragmites* also forms monospecific stands with a consequent reduction in species diversity (Marks, Lapin, & Randall 1994). Tulbure et al. (2007) compared vegetation structure and community composition before and after *Phragmites* invasion and found in plots where *Phragmites* had a 100% cover, a reduction in diversity was evident. However, in plots where *Phragmites* was not present or had a low cover, species diversity had actually increased.

Overall species diversity is viewed by many as a good indicator of wetland quality or integrity (Minc & Albert 2004). In this study, species diversity alone did not point out degradation of the highly disturbed sites. However, in these sites the relative dominance of *Typha* reflected the degree of disturbance. Relative dominance of *Typha* spp., along with the measures of species diversity, potentially provides an index of stress resulting from water level regulation (Albert & Minc 2004).

3.4.9 Above-ground Biomass

Variation observed in the above-ground biomass between sampling periods could be due to the period when different species reach their peak standing crop. Biomass followed the same trend as richness, density, and diversity, where the highly disturbed sites together had a greater mean biomass than the moderately disturbed sites and least disturbed site. Biomass is a measure of productivity, and the productivity of a site can depend on water quality. Along with density, biomass is considered a good measure of plant dominance on a site because it reflects the amount of sunlight, water and nutrients a plant is able to capture and turn into plant mass (University of Idaho 2009b). Above-ground biomass was greatest in the highly disturbed sites with eutrophic conditions and the poorest water quality (e.g., TP = 47.25 μg/L, TN = 2285 μg/L, CHL *a* = 3.26

$mg \cdot m^{-3}$) and lowest in the least disturbed sites with the greatest water quality (e.g., TP =11.25 μg/L, TN = 345.25 μg/L, CHL *a* =1.37 $mg \cdot m^{-3}$) (Table 3), corroborating the relationship of productivity and water quality. The moderately and highly disturbed sites had greater TP and TN concentrations therefore allowing species like *Typha* to increase in biomass (Croft & Chow-Fraser 2007). Increased biomass thus could indicate increased anthropogenic stresses in the form of higher nutrient levels.

3.5 Conclusions

Wetlands in this study represented a wide range of environmental conditions, ranging from very clear and nutrient poor (e.g., TP =11.25 μg/L, TN = 345.25 μg/L, CHL *a* =1.37 $mg \cdot m^{-3}$) to turbid and eutrophic (e.g., TP = 47.25 μg/L, TN = 2285 μg/L, CHL *a* = 3.26 $mg \cdot m^{-3}$). The limnologic parameters showed temporal and spatial variation and overall, indicating that water quality reflected the degree of anthropogenic degradation of the wetland. The least anthropogenically disturbed wetland DP had the highest water quality, the most anthropogenically disturbed sites, CB and HM, had the poorest water quality, and moderately disturbed sites BC, VP, and LC reflected an intermediate level of water quality and disturbance. Species richness, density, diversity, and above ground biomass varied among sites and between seasons, and the dominant vegetation reflected the water quality of each wetland. The least disturbed site DP was dominated by the native species *Sparganium eurycarpum*, *Scirpus acutus* and *Scirpus pungens*. The moderately disturbed sites were dominated by both native species with varying aggressiveness and invasive species, including *Leersia oryzoides*, *Eleocharis smallii*, *Scirpus acutus*, *Sparganium eurycarpum*, *Typha* x *glauca*, and *Typha angustifolia*. Highly disturbed sites were dominated by only one native species, *Calamagrostis canadensis*, as well as *Typha* x *glauca*, *Typha angustifolia*, and *Phragmites australis*; all of the last three are considered either to be invasive, aggressive, tolerant of degradation, and/or able to compete with native species. *Sparganium eurycarpum* was the only species that was present in all sites, supporting the literature that this species is cosmopolitan and tolerant of a wide range of environmental conditions. Thus, the results from this study provide important and useful information to ecosystem managers to consider assessment of macrophyte community dynamics as a relatively simple method of assessment of water quality in this area.

Chapter 4 An Exploration of Wetland Macrophytes as Bio-Indicators of Water Quality

4.1 Introduction

The goal of biological assessment of an ecosystem is to identify biological attributes that provide reliable information about it. There are numerous biological attributes that can be measured, but only a few provide useful indications on the impact of human activities. Standard terms that have been identified to describe ecosystem integrity include ‘attribute,’ ‘metric,’ ‘multimetric index,’ and ‘biological assessment’ (Cronk & Fennessy 2001). Structural metrics can include community composition, guild type, and key indicator species, while functional metrics incorporate sensitivity and tolerance measures, percent emergent, obligate, and pioneer wetland species, and the number of weed species. As discussed in Chapter 1, numerous biological indicators have been incorporated into multimetric indices for assessing the condition of water quality (Albert & Minc 2004; Croft & Crow-Fraser 2007; Johnson et al. 2007; Simon et al. 2001). Many macrophyte taxa have been used as indicators due to their sensitivity to a large suite of physical and chemical variables (Lougheed et al. 2001). Studies have focused on broadly describing Great Lakes plant communities (e.g., Minc & Albert 1998; Smith, Glooschenko, & Hagen 1991) or evaluating vegetation responses to anthropogenic disturbance (e.g., Albert & Minc 2004; Lougheed et al. 2001; Rothrock & Simon 2007; Wilcox et al. 2002). As described in Chapter 2, key dimensions of wetland stress include hydrologic flow modification and control (through water regulation and dyking), water quality degradation (through nutrient loading and sedimentation), and ecological structural breakdown or physical degradation (through land-use alterations, filling, etc.) (Albert & Minc 2004), which can affect macrophyte growth (Lougheed et al. 2001). In developing macrophyte metrics or indices to assess these stresses, studies typically use one or several of the structural or functional metrics described above, including an assessment of the response or tolerance of different species to one or several water quality parameters, providing an appropriate metric that quantifies the amount of degradation.

In the Great Lakes, Croft & Chow-Fraser (2007) used macrophyte species' tolerance as a metric to understand anthropogenic stress. Using data from 176 wetlands along the Great Lakes they established a highly significant relationship between Wetland Macrophyte Index (WMI) and Water Quality Index (WQI) scores that supported the position that plants are good indicators of water quality conditions in wetlands. The general formula used to generate the WMI score has been used by other authors (Lougheed & Chow-Fraser 2002; Seilheimer & Chow-Fraser 2006). The "U" value and "T" value of a species used to generate the WMI and WQI can be applied and utilized in this study to evaluate species' responses. The U-value indicated the tolerance of a species to degradation, 1 being very tolerant and 5 being very intolerant, and the T-value indicated the niche breadth, 1 specifying a broad niche, 3 specifying a narrow niche. Croft & Chow-Fraser (2007) included emergent, floating and submergent taxa in their study, and many of the emergent taxa found in the Great Lakes were present in Lake Simcoe. Table 10 is a modified version of U and T values the authors determined for emergent taxa of species common to both Great Lakes and Lake Simcoe wetlands.

Table 10. Summary of U and T values for emergent taxa in Croft and Chow-Fraser's (2007) study that also appeared in the Lake Simcoe study. Common names and species codes are also included for convenience. U value indicates the tolerance of a species to degradation (1 = very tolerant, 5 = very intolerant) and T value indicates the niche breadth (1 = broad niche, 3 = narrow niche).*denotes that the species is not native to North America.

Emergent	Common name	U value	T value
Lythrum salicaria	Purple loosestrife*	1	1
Typha angustifolia	Narrow-leaf cattail*	1	1
Typha sp.	Cattail	1	1
Typha x *glauca*	Hybrid cattail*	1	2
Sagittaria graminea latifolia	Broad arrowhead	2	1
Sagittaria graminea sp.	Arrowhead species	2	1
Sparganium sp.	Burreed	2	2
Pontederia cordata	Pickerelweed	3	2
Sparganium eurycarpum	Giant burred	3	2
Typha latifolia	Broadleaf cattail	3	2
Eleocharis smallii	Marsh spike rush	4	2
Equisetum fluviatile	Water horsetail	4	2
Scirpus acutus	Hardstem bulrush	4	2
Scirpus sp.	Bulrush	4	1

Croft and Chow-Fraser (2007) identified three species indicative of excellent conditions (U-value of 5): *Eriocaulon aquaticum* (pipewort), *Scirpus americanus* (three-square bulrush) and *Utricularia cornuta* (horned bladderwort) none of which were present in this study. This suggests the possibility that no wetlands in Lake Simcoe, including the "clean" reference site DP, are representative of pristine or excellent conditions. Indicators of good conditions (U-value of 4) included *Sparganium androcladum* (branched burred), *Scirpus validus* (softstem bulrush), *Scirpus acutus*, *Equisetum fluviatile,* and *Eleocharis acicularis* and *Eleocharis smallii*, both spikerushes. *Sparganium androcladum, Scirpus validus,* and *Elecharis acicularis* were not present at any of the studied sites. *Scirpus acutus* was present in BC, LC, DP, and HM; *Equisetum fluviatile* was present at HM only; and *Eleocharis smallii* was found in BC only. Species the authors found to be indicative of degraded water quality (U-value of 1) were dominant in polluted sites in the Great Lakes and in Lake Simcoe and included *Lythrum* *salicaria*, *Typha angustifolia*, and *Typha* x *glauca*. Other species in the Great Lakes not present in Lake Simcoe were *Sparganium emersum* (unbranched burred) and *Polygonum amphibium* (smartweed). However, the present study is consistent with these findings in that *Typha angustifolia, Typha* x *glauca* and *Lythrum salicaria* were only found at the moderately and highly impacted sites. Croft & Chow-Fraser (2007) considered several species as "neutral" in that they were cosmopolitan and seemed to be tolerant of many different conditions. These species were *Pontederia cordata*, *Sagittaria graminea cuneate* (small arrowhead), *Sparganium eurycarpum* and native *Typha latifolia*, all of which were, in the Great Lakes, among the most common species of emergent plants encountered. The presence of *Sparganium eurycarpum* in Lake Simcoe in all six wetlands sites strongly supports the neutrality of this species described by Croft & Chow-Fraser (2007). *Pontederia cordata* was present in only the least and moderately disturbed sites in Lake Simcoe, while *Typha latifolia* was present in the moderately to highly degraded sites, also supporting the finding of their tolerance of many different conditions.

Albert and Minc (2004) carried out a review of Great Lakes coastal wetlands and identified species and species groups that function as metrics of individual dimensions of anthropogenic stress based on their responses to different sources of wetland degradation, including hydrologic flow modification, water quality degradation, and ecological structural breakdown or physical degradation. They discussed specific stresses, responsive species, and the proposed metric for evaluating the stress, summarized in Table 11. Species responses included intolerance of stress, tolerant of stress, and positive response to stress. In the Great Lakes, *Typha* spp. demonstrated a positive response to lack of water level fluctuations. In regards to nutrient enrichment, *Typha* spp., and *Phragmites australis* responded positively. *Lythrum salicaria*, *Phragmites australis*, *Phalaris arundinacea*, and *Persicaria lapathifolia* had a positive response to sedimentation and increased turbidity, while *Calamagrostis canadensis* was intolerant of this stress. In response to physical degradation, *Lythrum salicaria*, *Phragmites australis*, and *Phalaris arundinacea* all responded positively. While all of the aforementioned species were present in Lake Simcoe fringe wetlands, other species and their responses monitored by Albert & Minc (2004) included *Carex stricta* (upright sedge) and *Carex aquatilis* (water sedge) both of which were intolerant of sedimentation and increased turbidity.

Table 11. Wetland species responses to anthropogenic stress as determined by Albert & Minc (2004). Species responses are coded as: - Intolerant to stress; + Tolerant of Stress; ++ Positive response to stress.

	Responsive Species	
Stress	**Emergent/Wet Meadow Zone**	**Proposed Metrics**
Dampening of Water-Level Fluctuation	*Typha* sp.(++)	a. Total coverage value of *Typha* in emergent and wet meadow zones. b. Width of *Typha* zone. c. Algal coverage.
Nutrient Enrichment	Typha sp.(++) Phragmites australis (++)	a. Total coverage of *Typha* spp. and *Phragmites australis* in emergent and/or wet meadow zones. b. Algal coverage.
Sedimentation and Increased Turbidity	*Carex stricta* (−) *Carex aquatilis* (−) *Calamagrostis canadensis* (−) *Lythrum salicaria* (++) *Phragmites australis* (++) *Phalaris arundinacea* (++) *Polygonum lapathifolium* (++)	a. Absence of turbidity intolerant species. b. Relative dominance of perennials vs. annuals in wet meadow zone.
Physical Degradation	*Lythrum salicaria* (++) *Phragmites australis* (++) *Phalaris arundinacea* (++) *Polygonum lapathifolium* (++)	a. Major loss of species diversity. b. Elimination of natural zonation. c. Relative dominance of exotics and aggressive native species in wet meadow zone.

Another approach to summarize plant responses to anthropogenic stressors was undertaken by the United States EPA (2000) to develop the *Sensitivities to Enrichment and Hydrologic Alteration Database*, which collected information from published, peer-reviewed sources on the responses of plants to anthropogenic stress. The database compiled 222 studies and contains references to over 2300 taxa describing species responses to changes in nutrient inputs and water regime. Species responses to these stresses were recorded as either "Increased" or "Decreased" in dominance, or along a 6-point scale of tolerance, ranging from unaffected, intolerant, somewhat tolerant, moderately tolerant, tolerant, and very tolerant. The database provides a general guideline for assessing wetland quality, in that "sites with a large component of reputedly tolerant species but with only few intolerant species might be considered in many instances to be ecologically degraded" (US EPA 2000).

Even though the relationship between water quality and aquatic vegetation in coastal wetlands of the Great Lakes has been well-studied (Lougheed et al. 2001; McNair & Chow-Fraser 2003), a basin-wide biotic index of anthropogenic disturbance based on aquatic wetland plants for Lake Simcoe has yet to be developed, despite the advantages of using plants as biotic indicators. Some advantages outlined in the literature include their non-motility (except for a few free-floating species), which permits for their distribution to be georeferenced to monitor changes over time (Croft & Chow-Fraser 2007). Non-motility also results in their ability to integrate the chemical, biological, physical, spatial and temporal dynamics of the wetland system (Cronk & Fennessy 2001). Similarly, Wei (2006) also asserts that plant communities integrate the effects of many factors that act simultaneously on the assemblage over a long period of time, therefore repetitive monitoring programs can use macrophyte indices as a reasonably cost-effective method to monitor wetlands for evidence of anthropogenic disturbance, and follow-up with a more specific and intensive sampling for water quality conditions. Furthermore, macrophyte surveys can be completed without specialized and expensive equipment and few researchers, compared to fish surveys that require electrofishing boats or series of paired fyke nets (Seilheimer & Chow-Fraser 2006). Also, most plant surveys can be completed in a day, unlike fish and benthic invertebrate surveys that at a minimum, require overnight traps. Macrophyte sampling techniques are well-developed and extensively documented, and most sampling results are available immediately with limited need for further processing (Croft & Chow-Fraser 2007). Furthermore, despite the effectiveness of actually measuring water quality degradation through metrics like a WQI (Croft & Chow-Fraser 2007), the effort required to collect and analyze copious water quality parameters, such as different nutrient forms, suspended solids, chlorophyll concentrations, etc., reduces the likelihood of its use by most environmental agencies.

Disadvantages of utilizing macrophytes as biological indicators include a potential lag in macrophyte response to stressors, especially in long-lived species; difficulty in identifying taxa to the species level, depending on the season; challenging sampling conditions and restrictions (such as only being able to sample during the growing season). Despite these limited disadvantages, the development of a cost-effective strategy that can be used to indicate the degree of anthropogenic impact and the resultant influence on wetland habitat can be an important contribution for conservation and management.

Significant work has been accomplished in the Great Lakes region that has studied coastal wetland macrophyte community responses to degraded water quality caused by anthropogenic stressors, yet no such investigation or index has been accomplished for Lake Simcoe's fringe wetlands. Chapter 3 showed that anthropogenic disturbances corresponding to each sampling site altered wetland water quality, which was reflected in the emergent macrophyte community composition and its metrics, including richness, density, diversity, and above-ground biomass. Several objectives and hypotheses were tested in this study. The objective of this chapter was to explore the relationship between macrophyte species and specific water quality parameters. The above literature review shows that certain macrophyte species and their responses and/or tolerances are considered to be indicators of water quality. Indicator species in this study and their responses will be assessed to verify whether Lake Simcoe macrophytes can be used as bio-indicators of water quality. A summary of indicator species, their responses to water quality in relation to anthropogenic disturbances in this study will be addressed in the discussion. Therefore, the hypothesis tested was, “Do the indicator species have a relationship with the water quality parameters measured in this study?” It is anticipated that the responses and tolerance of indicator species to water quality will be consistent with the literature and will substantiate the use of emergent macrophytes in Lake Simcoe’s fringe wetlands of indicators of water quality, and wetland integrity.

4.2 Methods

Detailed descriptions of methods are provided in Chapter 2. Multiple regression analysis was carried out to study the variation of species diversity and richness of the low, moderate, and highly disturbed sites with sampling period (SP), wetland site (SITE), temperature (TEMP), pH, dissolved oxygen (DO), conductivity (COND), water depth (DEPTH), chlorophyll *a* (CHL *a*), total nitrogen (TN) and total phosphorus (TP). The data distribution met the assumptions of linearity, independence of errors, homoscedasticity, unusual points and normality of residuals. If multicollinearity was present, variables were removed from the analysis, starting with those of the highest variance inflation factor values, until these values were less than 10 (Lund & Lund

2013). Sampling sites and sampling periods were used in the regression and ordination analyses as explanatory variables to evaluate variation in biological community with season and sampling location.

Species densities and above-ground biomasses at sites of varying disturbance levels were ordinated using Canonical Correspondence Analysis (CCA) and Redundancy Analysis (RDA) using CANOCO 4.5 software to examine their relationship to different limnologic parameters described above (ter Braak & Smilauer 2002).

A Spearman's correlation was carried out to calculate a correlation coefficient (ρ) to measure the relationship between macrophyte community parameters (species richness, density, diversity, and above-ground biomass) and water parameters (CHL *a*, TEMP, pH, DO, COND, TN, TP, and DEPTH). The assumptions of variables on a continuous scale, paired observations, and monotonic relationships were met for each of these pairwise correlation analyses (Lund & Lund 2013).

4.3 Results

4.3.1 All Sites

In order to understand the relationship between water quality parameters and species composition, a CCA between these parameters from all sampling locations combined was carried out. Table 12 lists the species and their abbreviations used in all bi-plots. Eigenvalues along the first and second axes were 0.592 and 0.562 respectively. CCA axis 1 explained 28.1% of the variance while axis 2 explained an additional 26.7%. 54% of the variation in macrophyte distribution could be explained by the first two environmental gradients (Figure 20). Weighted correlation matrix values are presented in Table 13; the most important predictors of macrophyte distribution, as indicated by their correlation with CCA axis 1, were COND (r= 0.41), DO (r=

0.38), and TP (r= -0.35), variables indicative of water quality. The second CCA axis was correlated with variables indicative of location, being SITE (r= -0.67), and DEPTH (r= 0.59).

Table 12. Species name and abbreviations utilized for CCA and RDA bi-plot interpretation.

Taxa	Abbreviation	Taxa	Abbreviation
Acer negundo	AcerN	*Persicaria lapathifolia*	PersL
Acer spp.	AcerSpp	*Phalaris arundinacea*	PhalA
Alnus incana	AlnuI	*Phragmites australis*	PhraA
Betula papyrifera	BetuP	*Pontederia cordata*	PontC
Bidens cernua	BideC	*Prunus spp.*	PrunSpp
Calamagrotsis canadensis	CalaC	*Rhamnus frangula*	RhamF
Calystegia sepium	CalyS	*Ribes spp.*	RibeSpp
Carex hystercina	CareH	*Rosa palustris*	RosaP
Carex lacustris	CareL	*Rubus idaeus L. var. strigosus*	RubuI
Cornus obliqua	CornO	*Rumex orbiculatus*	RumeO
Cornus spp.	CornSpp	*Sagittaria graminea graminea*	SagiG
Cornus stolonifera	CornS	*Sagittaria graminea latifolia*	SagiL
Eleocharis smallii	EleoS	*Salix spp.*	SaliSpp
Epilobium ciliatum spp glandulsum	EpilC	*Salix* x *rubens*	SaliR
Equisetum arvense	EquiA	*Sambucus canadensis*	SambC
Equisetum fluviatile	EquiF	*Scirpus acutus*	ScriA
Euthamia graminifolia	EuthG	*Scirpus pungens*	ScriP
Galium trifidum	GaliT	*Scirpus cyperinus*	ScriC
Glyceria striata	GlycS	*Scutellaria galericulata*	ScutG
Ilex verticillata	IlexV	*Scutellaria lateriflora*	ScutL
Impatiens capensis	ImpaC	*Sium suave*	SiumS
Leersia oryzoides	LeerO	*Solanum dulcamara*	SolaD
Lysimachia terrestris	LysiT	*Solidago canadensis*	SoliC
Lythrum salicaria	LythS	*Sparagenium americanum*	SparA
Melilotus albus	MeliA	*Sparganium eurycarpum*	SparE
Mimulus ringens	MimuR	*Sparganium spp.*	SparSpp
Myrica gale	MyriG	*Thelypteris palustris*	ThelP
Onoclea sensibilis	OnocS	*Typha angustifolia*	*Typha*
Persicaria lapathifolia	PersL	*Typha latifolia*	TyphL
Phalaris arundinacea	PhalA	*Typha spp.*	TyphSpp
Phragmites australis	PhraA	*Typha* x *glauca*	TyphG
Pontederia cordata	PontC	*UnID twigs*	UnId1

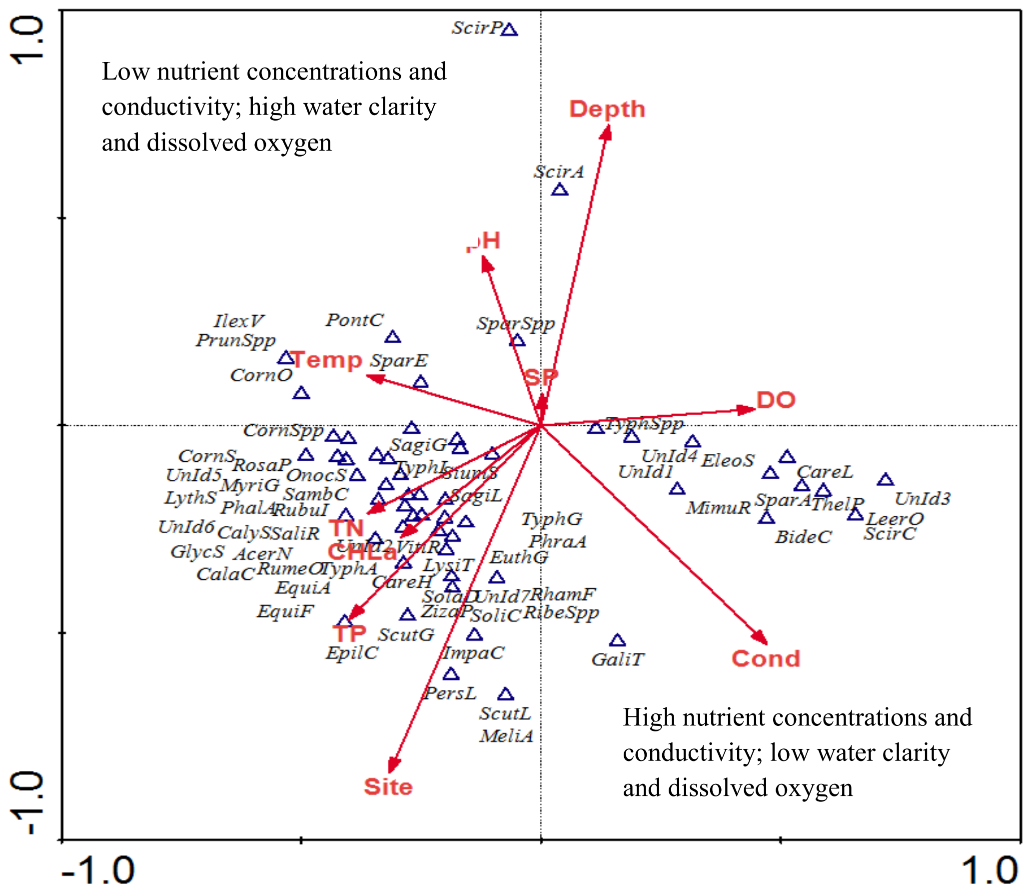
Prunus spp.	PrunSpp	*UnID vines*	Unid2
Rhamnus frangula	RhamF	*Unidentifiable emergent*	UnId3
Ribes spp.	RibeSpp	*Unidentifiable Wood Stems*	UnId4
Rosa palustris	RosaP	*Unidentified plant*	UnId5
Rubus idaeus L. var. strigosus	RubuI	*Unidentified plant*	UnId6
Rumex orbiculatus	RumeO	*Unidentified plant*	UnId7
Myrica gale	MyriG	*Vitis riparia*	VitiR
Onoclea sensibilis	OnocS	*Zizania palustris*	ZizaP
Persicaria lapathifolia	PersL	*Unidentified plant*	UnId6
Phalaris arundinacea	PhalA	*Unidentified plant*	UnId7
Myrica gale	MyriG	*Vitis riparia*	VitiR
Onoclea sensibilis	OnocS	*Zizania palustris*	ZizaP

Table 13. Weighted correlation matrix values for corresponding environmental parameters in CCA and RDA analysis of species density in all sites, the least disturbed site (DP), moderately disturbed sites (BC, VP, and LC) and highly disturbed sites (CB and HM).

Variable	SPEC AX1	SPEC AX 2
All Sites		
SITE	-0.27	-0.67
SP	0	0.06
TEMP	-0.31	0.1
pH	-0.1	0.32
DO	0.38	0.03
COND	0.41	-0.42
DEPTH	0.12	0.58
CHL *a*	-0.25	-0.22
TN	-0.31	-0.17
TP	-0.35	-0.37
Highly Disturbed Sites		
SITE	-0.85	0.19
TEMP	-0.34	-0.73
pH	-0.5	0.39
COND	0.32	0.07
TP	0.06	-0.64
Moderately Disturbed Sites		
SITE	0.94	-0.05

SP	0.05	0.12
TEMP	0.31	0.1
pH	0.11	-0.35
TN	0.28	0.71
Least Disturbed Site		
TEMP	0.59	0.21
pH	-0.48	0.25
DO	-0.43	0.11
COND	-0.02	-0.23
DEPTH	0.73	0.07
CHL *a*	0.55	0.11
TP	0.16	0.11

Figure 20. CCA bi-plot of species density with corresponding environmental factors for all sites (DP, BC, VP, LC, CB and HM). Eigenvalues along the first and second axes were 0.592 and 0.562 respectively. The species names have been abbreviated to fit the bi-plot.



4.3.2 Least Disturbed Site (DP)

4.3.2.1 Richness

Richness was significantly correlated with TEMP (ρ=0.52, p<0.01), DEPTH (ρ=0.52, p<0.01), CHL *a* (ρ=0.47, p<0.05), pH (ρ=-0.42, p<0.05), and TN (ρ=0.39, p<0.05) (Table 13). The results of the multiple regression analysis showed that only pH, DEPTH, and TP significantly predicted species richness, $F_{3, 24}$= 3.727, $p < 0.05$, adj. $R^2 = 0.233$.

Table 14. Spearman's rho (ρ) correlation coefficients at each sampling site. N=28. *Correlation is significant at the 0.05 level (2-tailed). ** Correlation is significant at the 0.01 level (2-tailed).

LEAST DISTURBED SITE					
DP	**CHL *a***	**RICHNESS**	**DENSITY**	**BIOMASS**	**DIVERSITY**
pH	-0.40*	-0.42*	-0.26	0.31	-0.40*
DO	0.40*	-0.17	-0.09	0.21	-0.04
COND	- 0.63**	0	0.01	-0.12	-0.16
TEMP	0.80**	0.52**	0.23	-0.25	0.60**
TN	0.11	0.39*	0.17	-0.27	0.34
TP	0.40*	-0.17	-0.09	0.21	-0.04
DEPTH	0.40*	0.52**	0.24	-0.33	0.50**
CHL *a*		0.47*	0.24	-0.21	0.60**
RICHNESS			0.50**	-0.08	0.77**
DENSITY				0.51**	0.34
BIOMASS					-0.05
MODERATELY DISTURBED SITES					
BC	**CHL *a***	**RICHNESS**	**DENSITY**	**BIOMASS**	**DIVERSITY**
pH	-0.80*	0.01	-0.14	-0.34	0.12
DO	-0.8	0.14	0	0.35*	0.41*
COND	-0.20*	-0.07	0.1	-0.04	0.08
TEMP	-0.40*	-0.13	-0.08	-0.59**	-0.22
TN	0.4	-0.14	0	-0.35*	-0.41*
TP	-0.4	0.14	-0.22	-0.06*	0.11
DEPTH	0.40*	-0.14	0	-0.35*	-0.41*
CHL *a*		-0.21	0.24	-0.05*	-0.45*

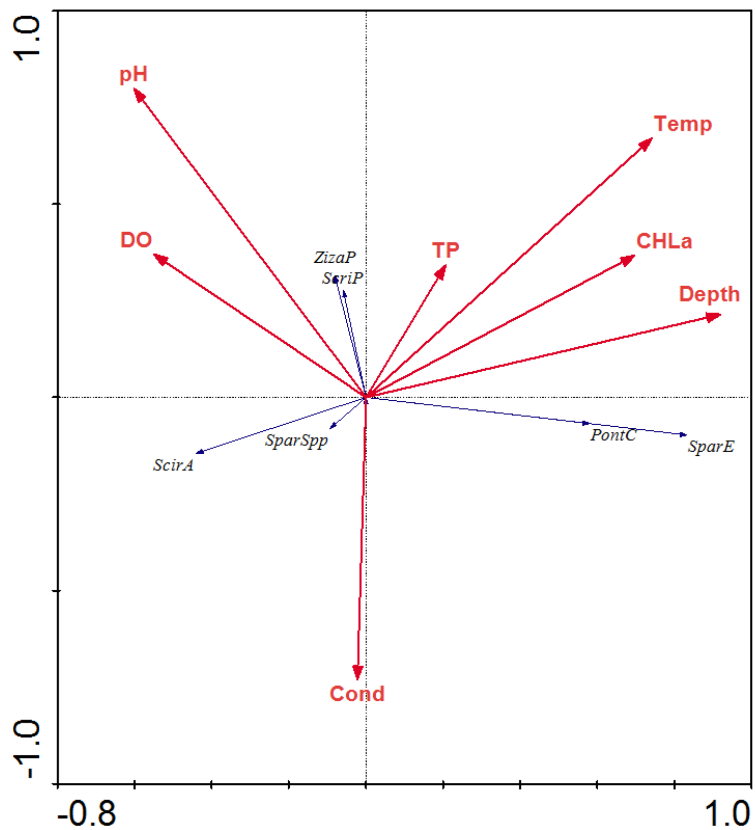
RICHNESS			0.44	-0.07	0.45*
DENSITY				-0.03	-0.26*
BIOMASS					0.25
VP	**CHL *a***	**RICHNESS**	**DENSITY**	**BIOMASS**	**DIVERSITY**
pH	0.6	-0.22	0.28	-0.06	-0.29
DO	0.40*	-0.03	-0.1	0.39*	0.12
COND	1.00*	-0.17	0.17	0.33*	-0.01
TEMP	0.40*	0.08	0.09	0.08	0.24
TN	-0.40*	-0.08	-0.09	-0.08	-0.24
TP	-0.2	-0.2	0.08	-0.21**	-0.45*
DEPTH	-0.2	0.15	0.03	-0.18	0.17
CHL *a*		-0.17	0.17	0.33*	-0.01
RICHNESS			0.18	0.15	0.55**
DENSITY				-0.24	-0.37**
BIOMASS					0.25
LC	**CHL *a***	**RICHNESS**	**DENSITY**	**BIOMASS**	**DIVERSITY**
pH	-0.4	-0.49**	0.19**	-0.63**	-0.32
DO	-0.60*	0.52**	0.35**	0.46**	0.65**
COND	-0.8	0.28	0.41	0.10*	0.51**
TEMP	-1.00*	0.02	0.52	-0.06	0.24
TN	1.00*	-0.02	-0.52	0.06**	-0.28
TP	-0.60*	0.52**	0.35**	0.46**	0.65**
DEPTH	-0.95**	-0.18	0.48	-0.26	0.08
CHL *a*		-0.02	-0.52	0.07**	-0.28
RICHNESS			0.45	0.49**	0.80**
DENSITY				0.07**	0.56**
BIOMASS					0.44*
HIGHLY DISTURBED SITES					
CB	**CHL *a***	**RICHNESS**	**DENSITY**	**BIOMASS**	**DIVERSITY**
pH	-0.40*	0.14	0.06	0.22	0.23
DO	-0.2	0.08	0.04	0.03	0.23
COND	0.32	0.01	0.35	0.2	-0.01
TEMP	-0.40*	0.06	-0.21	0.11	-0.07
TN	0.40*	-0.06	0.21	-0.11	0.07
TP	0	-0.07	-0.25	-0.26	-0.07
DEPTH	1.00**	-0.2	0.36	-0.15	-0.23
CHL *a*		-0.2	0.36	-0.15	-0.23

RICHNESS				-0.05	0.78**
DENSITY				0.17	0.19
BIOMASS					0.05
HM	**CHL *a***	**RICHNESS**	**DENSITY**	**BIOMASS**	**DIVERSITY**
pH	1.00**	0.68**	0.71**	-0.11	0.65**
DO	-0.40*	-0.61**	-0.57**	0.22	-0.45*
COND	0.40*	0.45*	0.34	-0.34	0.21
TEMP	0	0.32	0.22	-0.29	0.09
TN	0	-0.32	-0.22	0.29	-0.09
TP	-0.80**	-0.49**	-0.59**	-0.08	-0.60**
DEPTH	-0.80**	-0.73**	-0.70**	0.26	-0.58**
CHL *a*		0.68**	0.70**	-0.11	0.65**
RICHNESS			0.68**	-0.14	0.84**
DENSITY				0.17	0.452*
BIOMASS					-0.16

4.3.2.2 Density

Table 13 shows no significant correlations in terms of Spearman's correlation coefficient between macrophyte density and any of the limnologic parameters. The RDA results showed eigenvalues along the first and second axes were 0.321 and 0.028, respectively. RDA axis 1 explained 88.9% of the variance while axis 2 explained only an additional 7.9% (Figure 21). Axis 1 was correlated with DEPTH (*r*= 0.73), TEMP (*r*= 0.59), and CHL *a* (*r*= 0.55), while axis 2 was correlated with pH (*r*= 0.23) and COND (*r*=-0.23) (Table 13), and Figure 21 displays these relationships.

Figure 21. RDA bi-plot of species density with corresponding environmental factors for least disturbed site (DP). Eigenvalues along the first and second axes were 0.321 and 0.028 respectively. The species names have been abbreviated to fit the bi-plot.



4.3.2.3 Diversity

Species diversity was significantly correlated with TEMP (ρ=0.60, p<.01), CHL a (ρ=0.60, p<.01), DEPTH (ρ=0.50, p<.01), pH (ρ=-0.40, p<.05) and was also significantly correlated with species richness, (ρ=0.77, p<0.01) (Table 13). The results of multiple regression analysis did not show any significant relationship between the parameters tested ($F_{3, 23}$ = 1.787, p= 0.178, adj. R^2 = 0.083).

4.3.2.4 Above-Ground Biomass

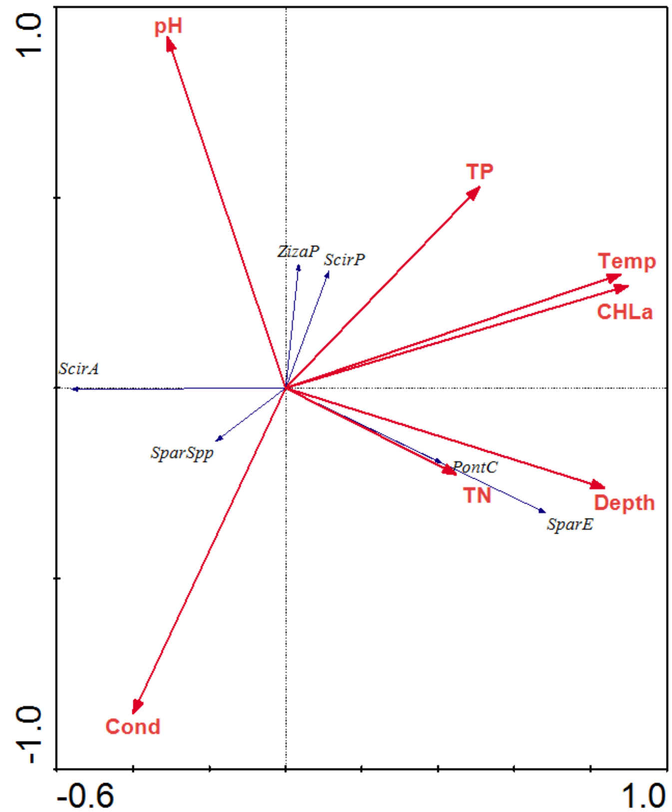
Above-ground biomass was not significantly correlated with any of the limnologic parameters, but was significantly correlated with density (ρ=-0.51, p<.01) (Table 13).

Eigenvalues along the first and second axes were 0.264 and 0.030 respectively. CCA axis 1 explained 89.7% of the variance while axis 2 explained an additional 10.3% (Figure 22). Axis 1 was correlated with CHL *a* (r= 0.52), TEMP (r= 0.51), and DEPTH (r= 0.49), while axis 2 was correlated with pH (r= 0.54) and COND (r= -0.50) and TP (r= 0.31) (Table 14).

Table 14. Weighted correlation matrix values for corresponding environmental parameters in CCA and RDA analysis of above-ground biomass in the least disturbed site (DP), moderately disturbed sites (BC, VP, and LC) and highly disturbed sites (CB and HM).

Variable	SPEC AX1	SPEC AX 2
Least Disturbed Site		
TEMP	0.51	0.17
pH	-0.18	0.54
COND	-0.23	-0.50
DEPTH	0.49	-0.15
CHL *a*	0.52	0.16
TN	0.26	-0.13
TP	0.29	0.31
Moderately Disturbed Sites		
SITE	0.13	0.44
SP	0.21	-0.07
TEMP	-0.09	0.24
pH	0.60	0.17
CHL *a*	0.39	-0.05
TN	-0.72	0.15
Highly Disturbed Sites		
SITE	0.56	-0.10
TEMP	0.03	-0.56
pH	0.45	0.25
DEPTH	-0.28	-0.08
CHL *a*	0.24	0.14

Figure 22. RDA bi-plot of above-ground biomass with corresponding environmental factors for least disturbed site (DP). Eigenvalues along the first and second axes were 0.321 and 0.028 respectively. The species names have been abbreviated to fit the bi-plot.



4.3.3 Moderately Disturbed Sites

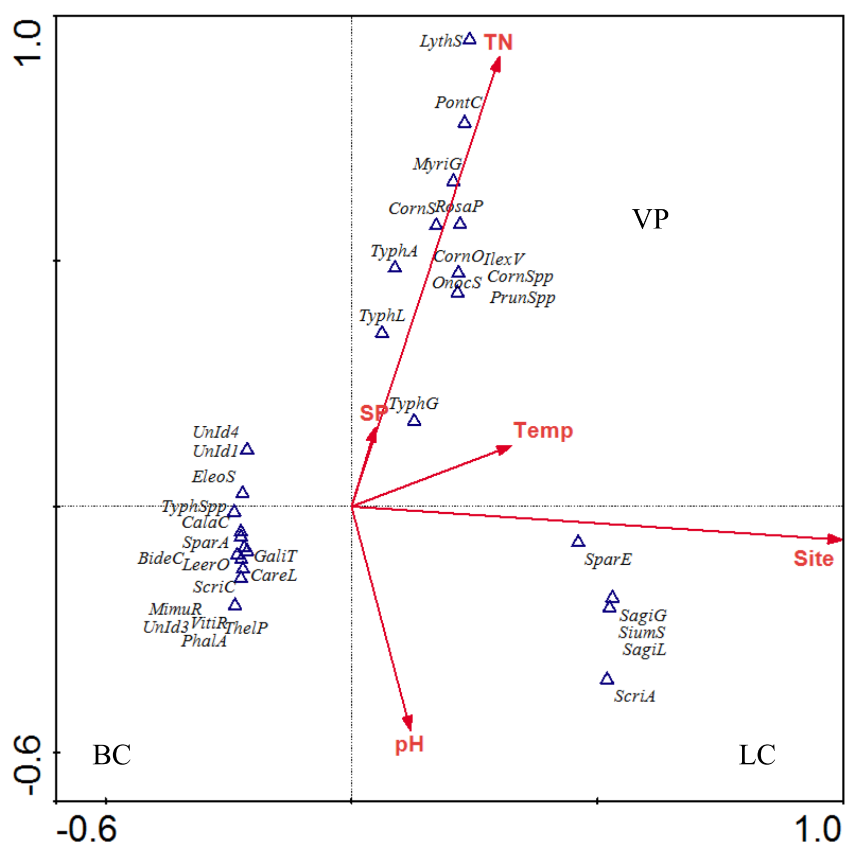
4.3.3.1 Species Richness

Richness was not significantly correlated with any limnologic parameters in BC or VP. In LC, richness was significantly correlated with DO (ρ=0.52, p<.01), TP (ρ=0.52, p<.01), and pH (ρ=-0.49, p<.01) (Table 13). Multiple regression analysis revealed that limnologic variables significantly influenced species richness, $F_{8, 75} = 7.20$, $p < 0.0005$.

4.3.3.2 Density

Similar to richness, density was not significantly correlated with any limnologic parameters in BC or VP. In LC, richness was significantly correlated with DO (ρ=0.35, p<0.01), TP (ρ=0.35, p<0.01), and pH (ρ=0.19, p<0.01) (Table 13). CCA analysis showed eigenvalues along the first and second axes were 0.796 and 0.435 respectively. CCA axis 1 explained 50.8% of the variance while axis 2 explained an additional 27.8% (Figure 23). Axis 1 was correlated with SITE (r= 0.94), TEMP (r= 0.31), and TN (r= 0.28), while axis 2 was correlated with TN (r= 0.71) and pH (r= -0.35) (Table 13). Although the largest variation among the three sites was attributed to the first CCA axis, the second CCA axis also exhibited strong correlations with the environmental variables. Figure 23 shows the position of BC on the left side of bi-plot, VP is on the upper right quadrant and LC is in the lower right quadrant. VP exhibited the highest TN concentrations and lowest pH values of these three sites (Table 3).

Figure 23. CCA bi-plot of species density with corresponding environmental factors for moderately disturbed sites (BC, VP, and LC). Eigenvalues along the first and second axes were 0.796 and 0.435 respectively. The species names have been abbreviated to fit the bi-plot.



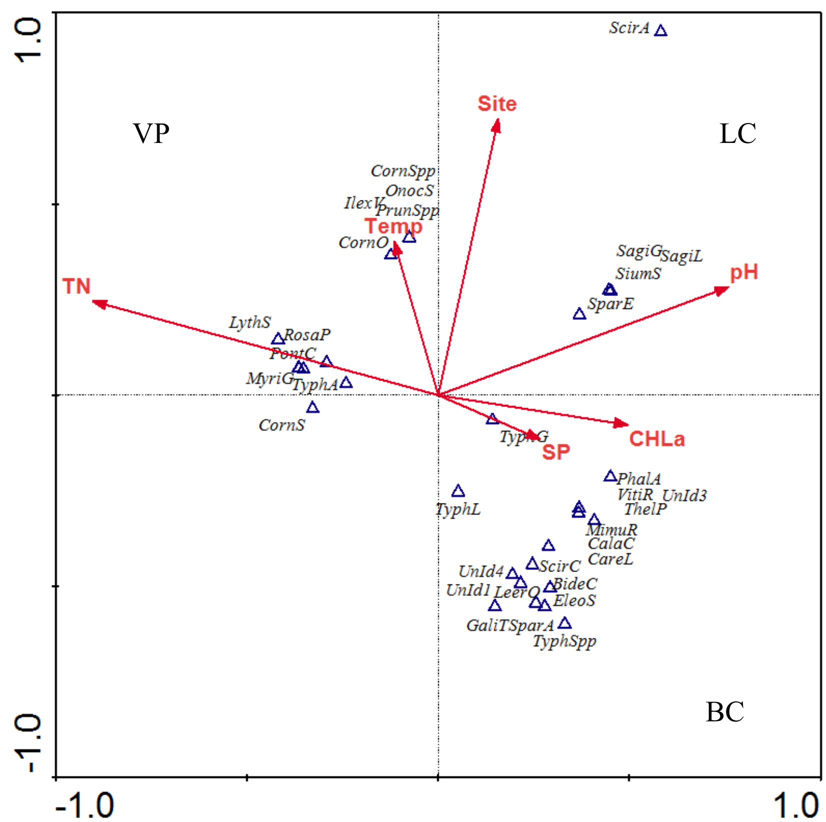
4.3.3.3 Diversity

Diversity was significantly correlated with CHL *a* (ρ=-0.45, p<0.05) , DO (ρ=0.41, p<0.05), TN (ρ=-0.41, p<0.05), DEPTH (ρ=-0.41, p<0.05) in BC; in VP it was significantly correlated with TP (ρ=-0.45, p<0.05); and at LC diversity was significantly correlated with TP (ρ=0.65, p<0.01), DO (ρ=0.65, p<0.01), and COND (ρ=0.51, p<0.01) (Table 13). Multiple regression analysis revealed that SP, pH, CHL *a* and TP significantly influenced species diversity, $F_{8, 75} = 6.022$, $p < 0.0005$, adj. $R^2 = 0.326$.

4.3.3.4 Above-ground Biomass

Biomass was significantly correlated with TEMP (ρ=-0.59, p<0.01), DEPTH (ρ=-0.35, p<0.05), DO (ρ=0.35, p<0.05), TN (ρ=-0.35, p<0.05), TP (ρ=-0.06, p<0.05), and CHL *a* (ρ=-0.05, p<0.05) in BC; in VP it was significantly correlated with DO (ρ=0.39, p<0.05), COND (ρ=0.33, p<0.05), TP (ρ=-0.21, p<0.01); and in LC biomass was significantly correlated with pH (ρ=-0.63, p<0.01), DO (ρ=0.46, p<0.01), TP (ρ=0.46, p<0.01), COND (ρ=0.10, p<0.05), TN (ρ=0.06, p<0.01), and CHL *a* (ρ=0.06, p<0.01) (Table 13). In the CCA analysis, eigenvalues along the first and second axes were 0.404 and 0.239 respectively (Figure 24). CCA axis 1 explained 43.1% of the variance while axis 2 explained an additional 25.5%. Axis 1 was correlated with TN (r= -0.72), pH (r= 0.60), and CHL *a* (r= 0.39), while axis 2 was correlated with SITE (r= 0.44) and TEMP (r= 0.24) (Table 14). Figure 24 shows the position of VP on the left side of the bi-plot, LC on the right in the upper quadrant, and BC in the lower quadrant. VP exhibited the highest TN concentrations and temperatures, and lowest CHL *a* concentrations of these moderately disturbed sites (Table 3). BC had the lowest temperatures but had the highest pH, despite the fact that this site corresponded on the bi-plot to the axis for CHL *a*. Several of the species in the lower right quadrant were exclusive to BC; the same was with LC and VP and the species located in their quadrants.

Figure 24. CCA bi-plot of above-ground biomass with corresponding environmental factors for moderately disturbed sites (BC, VP, and LC). Eigenvalues along the first and second axes were 0.404 and 0.239 respectively. The species names have been abbreviated to fit the bi-plot.



4.3.4 Highly Disturbed Sites

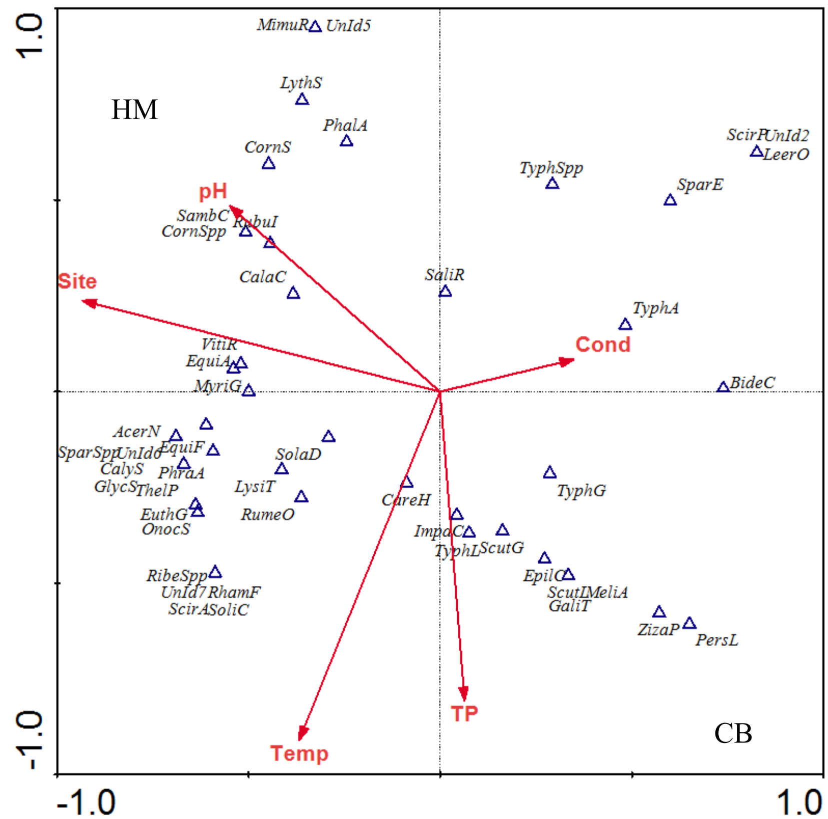
4.3.4.1 Richness

At HM, species richness was significantly correlated with DEPTH (ρ=-0.73, p<0.01), pH (ρ=0.68, p<0.01), DO (ρ=0-.61, p<0.01), COND (ρ=0.45, p<0.05), and TP (ρ=-0.49, p<0.01), while in CB no significant correlations between richness and water quality parameters were observed (Table 13). Multiple regression results showed that SITE, DO, DEPTH, and CHL *a* significantly predicted species richness, $F_{7, 48} = 9.115$, $p < 0.001$, adj. $R^2 = 0.508$.

4.3.4.2 Density

Similar to richness, no limnologic parameters were significantly correlated to density in CB, however in HM density was significantly correlated with pH (ρ=-0.71, p<0.01), CHL *a* (ρ=0.71, p<0.01), DEPTH (ρ=-0.70, p<0.01), TP (ρ=-0.59, p<0.01) and DO (ρ=-0.57, p<0.01) (Table 13). CCA showed eigenvalues along the first and second axes were 0.535 and 0.262, respectively. CCA axis 1 explained 50.1% of the variance while axis 2 explained an additional 24.5% (Figure 25). Axis 1 was correlated with SITE (r= -0.85), pH (r= -0.50), and TEMP (r= -0.34), while axis 2 was correlated with TEMP (r= -0.73), TP (r= -0.64), and pH (r= 0.39) (Table 13). Although the largest variation among the three sites was attributed to the first CCA axis, the second CCA axis also exhibited strong correlations with the environmental variables. Figure 25 shows HM plotted on the left side of the bi-plot, corresponding to pH and TEMP, while CB is plotted on the right side and associated with TP and COND. HM had higher pH and water temperatures than CB, and CB exhibited higher TP concentrations (Table 3). HM did have higher COND values than CB despite this parameter's location on the right side of the bi-plot, however, conductivity was not a strong explanatory variable of species density (Figure 25). Furthermore, several of the species on the left side of the bi-lot were exclusive to HM, and several exclusive to CB were located on the right.

Figure 25. CCA bi-plot of species density with corresponding environmental factors for highly disturbed sites (CB and HM). Eigenvalues along the first and second axes were 0.404 and 0.239 respectively. The species names have been abbreviated to fit the bi-plot.



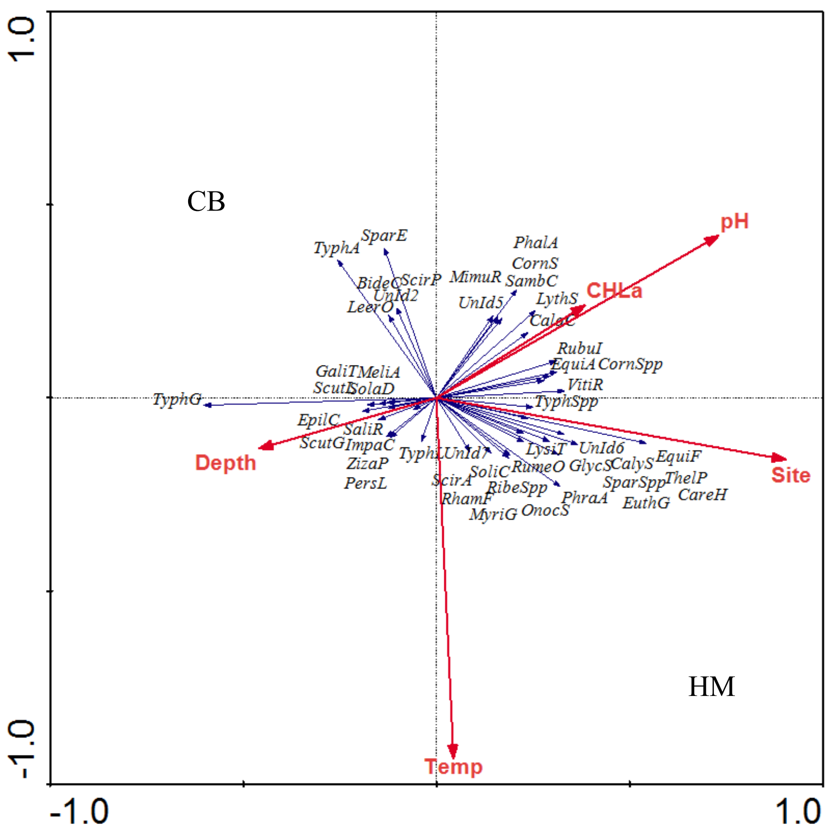
4.3.4.3 Diversity

No water quality parameters were significantly correlated with diversity in CB, however in HM diversity was significantly correlated with pH (ρ=0.65, p<0.01), CHL *a* (ρ=0.65, p<0.01), TP (ρ=-0.60, p<0.01), DEPTH (ρ=-0.58, p<0.01), and DO (ρ=-0.45, p<0.05) (Table 13). Multiple regression analysis revealed that limnologic variables significantly predicted species diversity, $F_{7, 48} = 4.414$, $p < 0.05$, adj. $R^2 = 0.303$.

4.3.4.4 Above-ground Biomass

Table 13 revealed that above-ground biomass was not significantly correlated with any of the limnologic parameters studied. RDA showed eigenvalues along the first and second axes were 0.279 and 0.019 respectively. RDA axis 1 explained 88.4% of the variance while axis 2 explained an additional 5.8% (Figure 26). Axis 1 was correlated with SITE (*r*= 0.56), pH (*r*= 0.45), and DEPTH (*r*= -0.28), while axis 2 was correlated with TEMP (*r*= -0.56) and pH (*r*= 0.25) (Table 14). Figure 26 shows CB plotted on the left side of the bi-plot, corresponding to DEPTH, while HM is located on the right side and is associated with CHL *a*, TEMP, and pH. HM had higher CHL *a* concentrations, water temperature, and pH throughout the study compared to CB, and water depth was greater at CB (Table 3). Furthermore, several of the species on the left of the bi-plot were exclusive to CB, and on the right side many were found only in HM.

Figure 26. RDA bi-plot of above-ground biomass with corresponding environmental factors for highly disturbed sites (CB and HM). Eigenvalues along the first and second axes were 0.279 and 0.019 respectively. The species names have been abbreviated to fit the bi-plot.



4.4 Discussion

4.4.1 All Sites

Constrained ordination allows one to describe the major ecological patterns in one set of response variables that can be explained by another set of explanatory variables. CCA and RDA are both constrained ordination techniques; RDA assumes a linear response function while CCA assumes a unimodal response function of the species along linear gradients defined by the water quality parameters. Both techniques use species and environmental data with the premise that each species thrives under specific environmental conditions (Croft & Chow-Fraser 2007). Therefore, their bi-plots are similarly interpreted, but with a few differences. In CCA, perpendiculars drawn from each species to the appropriate environmental arrow gives approximate ranking of species responses to that variable, and indicates whether a species has higher than-average or lower-than average optimum on that environmental variable. In RDA, the direction of an arrow drawn from the origin to species and environmental arrows gives the approximate relationship between species and environmental variables, including whether the species has positive or negative relationship with it (University of Massachusetts, n.d.). As an initial step to understand the relationship between water quality and species composition, a CCA with the limnologic parameters and macrophyte density data from all wetland sites was carried out. Indicators of wetland degradation such as high nutrient levels and conductivity, as well as low DO were found to ordinate along the first CCA axis. In CCA, the location of a species along an axis is referred to as its “centroid,” and the spatial position of a species’ centroid provide useful information (Leps & Smilauer 2003). Figure 20 shows that the majority of centroids located in the lower quadrants of the bi-plot corresponded to species found in the sites that had high nutrient levels and are considered to be moderately to highly impacted and degraded, whereas those in the upper quadrants species corresponded to sites that had lower nutrient levels and higher DO concentrations, and are considered minimally impacted and least degraded. SITE was moderately correlated with both CCA axes and therefore confirms the distinction between minimally impacted sites (less nutrients) and highly disturbed sites (nutrient rich) (Lougheed et al. 2001). The principal goal of this chapter was to assess the use of the macrophyte community to reflect water quality conditions, including nutrient enrichment and clarity, as these are the usual human-induced impacts on wetland environments. However, as recognized in this study and by Croft & Chow-Fraser (2007) some changes in the species dynamics are unrelated to altered water quality, but nevertheless reflect other effects of human-induced disturbance (e.g. recreational/boating, dredging and filling, etc.). As a result, including SITE in the ordination analysis incorporates the additional disturbances that impact each wetland. SP was the least important variable in describing species density patterns (Figure 20). This is consistent with the one-way ANOVA results which revealed that HM was the only site that showed significant variation between sampling periods ($F_{3, 27} = 18.404$, $p < 0.001$).

In Figure 20, the location of certain plant taxa to the lower left of the origin in the bi-plot suggests that these plants tolerate a turbid, nutrient-rich water column. These included *Typha latifolia*, *Typha* x *glauca*, *Lythrum salicaria*, *Phalaris arundinacea*, and *Phragmites australis*. Those taxa that were least tolerant of turbidity and high nutrient concentration included *Carex lacustris*, *Scirpus acutus*, *Scirpus cyperinus*, and *Sparganium americanum*. In general CCA axis 1 separates the nutrient-rich wetlands in the highly agricultural and urbanized areas from the clearer, more oligotrophic wetlands. Different macrophyte community types are apparent in the bi-plot. In the upper quadrants of the CCA bi-plot are the wetlands where *Scirpus* spp. had higher densities, which is indicative of DP, the least disturbed site. In contrast, *Typha angustifolia*, *Typha* x *glauca*, *Phragmites australis, Calamagrostis canadensis*, and *Phalaris arundinacea* were some of the dominant emergent macrophytes in the highly disturbed sites, which are seen in the lower half of the bi-plot. Species located in the lower quadrants corresponded with moderately to highly nutrient rich wetlands. These results thus suggest that the separation of the six studied wetland sites into three degrees of anthropogenic disturbances (DP as least disturbed site; BC, VP and LC as moderately disturbed sites; and CB and HM as highly disturbed sites) is reliable and valid, and allows the ordination analysis of each level of disturbance together.

Lougheed et al. (2001) performed a CCA to determine the association between environmental variables and the distribution of macrophytes. 62% of the variation in macrophyte distribution could be explained by the first two environmental gradients. The most important predictors of macrophyte distribution, as indicated by their correlation with CCA axis 1, were total suspended solids (TSS) (r= 0.79), TP (r= 0.73), CHL *a* (r= 0.70), TN (r= 0.65), and COND (r= 0.61). The second CCA axis was correlated with variables indicative of sediment quality, being TP in sediment (TPSED) (r= –0.54), inorganic sediment (INORGSED) (r= 0.34), and pH (r= 0.41)). Similarly, in this study it was also found that COND, TP, TN, CHL *a*, as well as DO and TEMP were important predictors of distribution. TSS, TPSED, and INORGSED were not included in and thus present a limitation of this study, despite their ability to predict macrophyte distribution as demonstrated by Lougheed et al. (2001). Macrophyte growth is limited by both water quality and sediment quality (Day et al. 1988; Barko et al. 1991; as cited by Lougheed et al. 2001). Wetland soils/sediments are the primary storage of available chemicals for emergent macrophytes, such as TP or TN (Mitsch & Gosselink 2007). Furthermore, the presence of high clay and silt content (INORGSED) in the sediment is undesirable, since plants grow better in organic than in inorganic substrates and small inorganic particles in the sediment can become easily re-suspended and remain in suspension thus keeping the water column turbid (Lougheed et al. 2001). Although their study also included parameters indicative of sediment quality, this only explained an additional 8% of the variance in macrophyte distribution (62% compared to 54% in this study). Results of their CCA also suggested that *Typha*, *Sagittaria graminea*, and *Lythrum* tolerate turbid, nutrient rich water, while those taxa that were least tolerant of turbidity and high nutrient concentrations included the emergent taxa *Pontederia cordata* and *Sparganium* spp., which was consistent with this study (Figure 20).

The community structure of emergent macrophytes in all six of the fringing wetlands studied in Lake Simcoe responded predictably to water quality parameters and appeared to be a function of the water quality, which is a proxy for site disturbance. These potential indicator species or groups showed identifiable relationships to known variation in degrees of anthropogenic stress. The analyses demonstrated that types of disturbance varied between wetlands and that our proposed indicators showed predictable response to water quality and disturbance.

4.4.2 Relationship of Macrophytes and Water Quality Parameters at the Least Disturbed Site (DP)

Species richness was somewhat correlated with pH, TEMP, CHL *a*, pH and DEPTH, and was weakly correlated with TN (Table 13). Regression analysis also revealed that these parameters significantly predicted species richness at DP, and explained 38% of the variation. Diversity at the least disturbed site was strongly correlated with temperature and CHL *a*, and moderately correlated with pH and DEPTH (Table 13), but regression analysis showed that none of the limnologic parameters significantly predicted species diversity. The strongest explanatory variables of species density and above-ground biomass were DEPTH, TEMP, CHL *a*, pH and

COND (Tables 12 and 15). *Sparganium eurycarpum*, *Scirpus acutus*, and *Scirpus pungens* were the dominant species in the least disturbed site, and responded predictability to the limnologic parameters. *Sparganium eurycarpum* and *Pontederia cordata* biomass was positively correlated with TN concentrations and deep water. These species preferred lower pH and were only moderately influenced by COND and TP concentrations. These associations explained the ubiquity of *Sparganium eurycarpum* in the study. *Scirpus acutus* biomass showed a negative correlation with TP and CHL *a*, indicating a tolerance of low nutrient and turbidity levels. Its location on the bi-plot relative to pH and COND suggests its biomass was influenced most by high to moderate pH and conductivity concentrations (Figure 21). The density of *Sparganium eurycarpum* and *Pontederia cordata* was influenced most by DEPTH. The location of these species relative to pH and DO suggested they prefer low pH and DO concentrations, which is indicative of tolerance of degraded water quality. *Scirpus acutus* was negatively correlated with TP, TEMP, CHL *a*, and is found in shallower water. *Scirpus pungens* and *Zizania palustris* density was influenced most by increasing pH and DO concentrations, which reflects a preference for healthy water quality conditions (Figure 22). Overall, the species composition and their relationship with the water quality parameters revealed by RDA demonstrate that the responses of these species reflected the water quality and low anthropogenic stress at the least disturbed site, DP.

4.4.3 Relationship of Macrophytes and Limnologic Parameters at the Moderately Disturbed Sites (BC, VP, LC)

In BC and VP, species richness was not correlated with any of the limnologic parameters. At LC, Spearman's correlation revealed that species richness was moderately positively correlated with DO and TP, and moderately negatively correlated with pH (Table 13). Multiple regression analysis showed that SITE, SP, CHL *a,* and TP significantly predicted 38% of the variation in species richness of all moderately disturbed sites. Species diversity was moderately negatively correlated with CHL *a*, DEPTH, and TN, and was moderately positively correlated with DO. At LC, species diversity was moderately positively correlated with COND, and strongly correlated with DO and TP; in VP diversity was moderately negatively correlated with TP (Table 13). Regression analysis showed that SP, pH, CHL *a,* and TP were significant predictors of diversity and explained 32% of the variation. The most important predictors of species biomass and density were SITE, TEMP, TN, and pH (Tables 12 and 15). The biomass of *Lythrum salicaria, Pontederia cordata* and the dominant *Typha angustifolia* was positively associated with increasing TN, reflecting a tolerance and positive response to increased nutrient concentrations. The dominant species *Typha* x *glauca*, *Leersia oryzoides*, and *Eleocharis smallii*, as well as *Phalaris arundinacea*, and *Calamagrostis canadensis* were positively associated with CHL *a*, an indication that these species tolerated turbid water. Unexpectedly, *Typha* x *glauca* biomass was negatively correlated with TN, which suggests that this species biomass will increase with decreasing nutrients. This finding reflects this species' exploitation of available nutrients (Tuchman et al. 2009) to support its large amount of standing biomass. *Typha latifolia*, the only *Typha* spp. not considered invasive, was negatively correlated with TN, suggesting a preference for lower nutrient levels. The biomass of cosmopolitan *Sparganium eurycarpum* showed a higher than average optimum of pH, demonstrating a positive response to alkaline water (Figure 24). The position of the centroid for *Scirpus acutus* on the bi-plot relative to SITE showed that this species was influenced by location and human disturbance rather than specific water quality parameters. Chapter 2 described in detail the disturbances associated with each moderately disturbed site. Furthermore, as mentioned in the Results section of this chapter (section 4.3), certain species were exclusive to each of the moderately disturbed sites. *Lythrum salicaria* and *Rosa palustris* were only found in VP, and *Myrica gale* had the highest biomass at this site, therefore it was plotted on the left side of the bi-plot. BC was plotted in the lower right quadrat, and *Phalaris arundinacea*, *Vitis riparia*, *Galium trifidum* (three-petaled bedstraw), *Calamagrostis canadensis* and *Carex lacustris* were all exclusive to BC. *Sparganium eurycarpum* and *Scirpus acutus* were associated with LC as these species had the highest biomass at this site; however they were also present at BC and VP. *Scutellaria galericulata*, *Scutellaria latifolia*, and *Sium suave* were exclusive to LC and therefore it was plotted in the upper right quadrant of the bi-plot (Figure 24). SITE was also associated with species density along CCA axis 1 (Figure 23), again suggesting that density was also influenced by the differences in location of the moderately disturbed sites and the specific human-induced disturbances impacting each wetland. Similar to biomass, the exclusively of the species in different sites allowed for LC to be plotted in the upper right quadrant, BC in the lower right, and VP was located to the upper left of the origin (Figure 23).

The densities of invasive, introduced species such as *Lythrum salicaria*, *Typha angustifolia*, *Typha* x *glauca,* and native *Pontederia cordata* and *Typha latifolia* were positively associated with TN, indicating a positive response to nutrient enrichment, that is, their densities increase with increasing TN. SP was positively correlated with TN, suggesting that this nutrient's concentration changed with different sampling periods (Figure 23). However, no statistically significant differences in TN were identified between sampling periods (one-way ANOVA $F_{3,\,10}$=0.731, p = 0.556). The position of the centroids of *Calamagrostis canadensis*, *Phalaris arundinacea*, *Bidens cernua*, and *Leersia oryzoides* in relation to TN suggests these species preferred lower than average optimum TN concentration, which was unexpected for these species. *Sparganium eurycarpum* and *Scirpus acutus* corresponded with pH and SITE, suggesting that these species preferred a higher water column pH, reflecting the differences in the densities of these species in each of the moderately disturbed sites (Figure 23). Overall, the species' density and biomass and their relationship with the water quality parameters revealed by CCA demonstrated that the responses of these species reflected the water quality and moderate anthropogenic degradation impacting BC, VP and LC wetlands.

4.4.4 Relationship of Macrophytes and Water Quality Parameters at the Highly Disturbed Sites

In CB, species richness was not significantly associated with any of the water quality parameters. However, in HM species richness was strongly associated with DEPTH, pH, and DO, and moderately associated with TP and COND. Species richness was found to be negatively correlated with DEPTH (Table 13). Deeper water is often associated with lower emergent cover, as plants intolerant of flooding are killed by insufficient root aeration (Tulbure et al. 2007), which may explain this result. Multiple regression analysis revealed that SITE, DO, DEPTH, and CHL *a* were significant in explaining 50% of the variation in species richness at the highly disturbed sites. Diversity also was strongly correlated with pH, as well as CHL *a,* and TP, and was moderately correlated with DEPTH and DO (Table 13). However, multiple regression analysis showed that only DEPTH and CHL *a* were significant predictors of diversity and explained 30% of the variation at the highly disturbed sites. Evidently, only two of the water quality parameters predicted richness and diversity in the highly disturbed sites. This suggests that there are other water (or sediment) quality variables that may better predict these dynamics in highly disturbed sites; or it may be that ordination analysis is more useful in determining how macrophyte community composition and dynamics relate to water quality, but less useful in regard to species richness.

Species richness at the highly disturbed sites was higher than both the moderately and least disturbed sites, and observation of the above-ground biomass and density bi-plots (Figures 23 and 24) showed very different species composition and species-environment relationships compared to the RDA and CCA results of the least and moderately disturbed sites. The dominant species at the highly disturbed sites included *Sparganium eurycarpum*, *Typha* x *glauca*, *Phragmites australis*, and *Calamagrostis canadensis*. SITE was an important predictor of macrophyte above-ground biomass and density, reflecting the different human-induced disturbances impacting each wetland described in Chapter 2. While both are subjected to intense agricultural and urbanization activities, HM has additionally experienced significant ecological structural breakdown and physical modification through in-filling and draining of the wetland in the early twentieth century. The biomass of *Phragmites australis* and *Sparganium* spp. were positively correlated with SITE and TEMP (Figure 26). *Phragmites australis* has been reported to demonstrate a positive response to physical degradation of the area (Albert & Minc 2004), further reflecting the physical modifications of HM and CB. Additionally, similar to the moderately disturbed sites, several species were exclusive to either CB or HM. For example, *Phalaris arundinacea*, *Scirpus acutus*, *Ribes* spp., *Myrica gale*, and *Sambucus canadensis*, were just some of the species exclusive to HM, while *Bidens cernua*, *Epilobium ciliatum* spp. *glandulosum* (willowherb)*, Galium trifidum*, *Melilotus albus* (sweet clover) *and Scutellaria lateriflora* were found only in CB. Therefore, HM was plotted on the right side of the bi-plot, and CB on the left (Figure 26).

pH and CHL *a* were highly correlated, and the biomass of several species were positively correlated with these parameters, including the aggressive species *Phalaris arundinacea*, *Calamagrostis canadensis*, and *Lythrum salicaria*, suggesting a positive response to these water quality parameters and tolerance of increasingly turbid and alkaline water (Figure 26). CB and HM had some of the highest mean CHL *a* concentrations (3.02 mg·m^{-3} and 3.26 mg·m^{-3}, respectively), and lowest pH values (6.9 and 7.13, respectively; though these pH values are still close to neutral) (Table 3). These species were negatively correlated with DEPTH and therefore tolerated shallower water, and reflects these species' known preference for moist soils (Darris 2005) (Figure 26). Similar to *Phragmites australis*, *Lythrum salicaria* was also reported to respond positively to physical degradation of the area (Albert & Minc 2004). *Typha angustifolia*, *Sparganium eurycarpum*, as well as *Bidens cernua*, *Leersia oryzoides*, and *Scirpus pungens* were negatively correlated with SITE, but not positively correlated with any specific water quality parameter. This could be interpreted as suggesting that more than one water quality parameter had a strong influence on the biomass of these species, or that another water quality parameter not included in this study best describes the species-environment relationship, possibly one that indicates sediment quality as discussed by Lougheed et al. (2001). *Typha* x *glauca* biomass was highly correlated with DEPTH, reflecting a tolerance of deeper water depths compared to *Phalaris arundinacea, Lythrum salicaria,* and *Calamagrostis canadensis.*

Typha x *glauca* and *Typha latifolia* were associated with increasing TP concentrations, suggesting a positive response to nutrient enrichment; *Typha* x *glauca* was also associated with COND according the position of its centroid (Figure 25). At the moderately disturbed sites, however, the density of *Typha* x *glauca* showed a negative correlation with TN, again reflecting this species' exploitation and uptake of surplus nutrients (Tuchman et al. 2009). Moreover, these species were negatively associated with pH, revealing that they preferred a lower than average optimum pH and could therefore tolerate a more acidic water column. Aggressive *Typha angustifolia, Bidens cernua,* and *Leersia oryzoides*, cosmopolitan *Sparganium eurycarpum* and native *Scirpus pungens* corresponded to COND and demonstrated a preference for lower water temperatures, reflecting a tolerance of greater ion concentrations and thus degraded water quality. The density of *Lythrum salicaria*, *Calamagrostis canadensis*, and *Phalaris arundinacea* was positively associated with pH, an indication that they prefer somewhat alkaline pH levels. They also appeared to be negatively correlated with TP, an unexpected result as these species are tolerant of higher nutrient levels and are indicators of stress tolerance (US EPA 2000). The density of *Phragmites australis* was associated with SITE, and the position of its centroid relative to COND suggested that it cannot tolerate higher concentrations of ions in the water (Figure 25). This result may actually be a reflection of its preference for shallower water, as the above-ground biomass bi-plot suggests; if it cannot tolerate deep water then a negative association with the amount of ions in the water column would be expected. Again, overall the results showed that the species' relationship with the water quality parameters revealed by CCA and RDA demonstrate that the responses of these species reflect the water quality and high degree of anthropogenic degradation impacting CB and HM wetlands.

4.4.5 Indicator Species Responses and Utility as Bio-indicators of Water Quality

Several indicator species reported in the literature were present in Lake Simcoe's fringe wetlands, and the responses of these indicator species to the limnologic parameters measured in this study are consistent with the literature and demonstrate the potential of their utility as biological indicators of wetland water quality and by extension, wetland integrity (Albert & Minc 2004; Cronk & Fennessy 2001).

In agreement with Croft & Chow-Fraser (2007), *Sparganium eurycarpum* was present in all six wetland sites and its relationship with different limnologic parameters in each varying of degree of degradation confirmed its tolerance of a wide range of conditions. In the least disturbed sites, the distribution of *Sparganium eurycarpum* was influenced most by water depth, a limnologic parameter but not one that relates directly to water quality. At the moderately disturbed sites, the distribution was largely influenced by pH and site, and at the highly disturbed sites it was influenced by conductivity. According to the WMI, *Sparganium eurycarpum* is "neutral" in terms of its tolerance of water quality degradation and it was found that this species enjoys a cosmopolitan distribution (Croft & Chow-Fraser 2007). An increase in dominance of this species has also been reported as a response to increased N and P concentrations (Gernes & Helgen 1999; Neely & Davis 1985; Schwartz 1985; Srivastava et al. 1995; as cited by US EPA 2000). In this Lake Simcoe study, *Sparganium eurycarpum* was either moderately correlated or showed a preference for average nutrient concentrations. Furthermore, the relative density of this species in sites with higher nutrient concentrations was less than those sites with lower nutrient concentrations (Table 9), conflicting with the response reported by the US EPA (2000). However, Lougheed et al. (2001) found that *Sparganium* spp. was less tolerant of turbidity and eutrophication, which this study was consistent with.

A variety of responses and tolerances of *Pontederia cordata* to water quality degradation have been described in the literature, and in this study the relationship of this species with the limnologic parameters reflects this range. Like *Sparganium eurycarpum*, Croft & Chow-Fraser (2007) found this species to be neutral and tolerant of many different conditions. In this study, *Pontederia cordata* was present in only the least disturbed and moderately disturbed sites, suggesting that it is not as cosmopolitan as these authors claim, and couldn't tolerate highly degraded water quality. Lougheed et al. (2001) determined that *Pontederia cordata* was intolerant of nutrient enrichment and turbidity, which is supported by Hough et al. (1991) (as cited by US EPA 2000) in the *Wetland Plant Sensitivities Database* that its dominance will increase as nutrient concentrations decrease. In this study, in low and moderately disturbed sites the above-ground biomass of *Pontederia cordata* would increase with increasing TN, a result inconsistent with Hough et al. (1991) (US EPA 2000), yet its absence at the highly disturbed sites suggest that its biomass will increase with increasing nutrients to a concentration somewhere between what was measured at the moderately disturbed sites and highly disturbed sites (Table 3). Consistent with Lougheed et al. (2001) was that this species was either moderately correlated or negatively associated with chlorophyll *a*, indirectly suggesting the use and competition for TN between *Pontederia cordata* and algae in the water column. Other studies included in the US EPA (2000) *Wetland Plant Sensitives Database* by Andreas & Lichvar (1995); Gernes & Helgen (1999); Michner (1990); and Srivastava et al. (1995) assert that *Pontederia cordata* was somewhat to moderately tolerant of general pollution and nutrient enrichment. Similarly, the present study also showed an increase in species density with decreasing pH and DO, variables that can reflect water pollution and eutrophication.

The least disturbed site was dominated by sensitive species including *Scirpus acutus*. In the WMI, Croft & Chow-Fraser (2007) assigned a U-value of 4 to this species, stating that it was intolerant of degradation and an indicator of good water quality conditions. In this study, even though *Scirpus acutus* was present in low, moderate, and highly degraded wetlands, the responses of *Scirpus acutus* to the limnologic parameters in Lake Simcoe fringe wetlands support Croft & Chow-Fraser's (2007) finding that this species indicated good water quality conditions. Despite its presence in moderately and highly degraded sites, its relative density and frequency were low compared to the least disturbed site In the least disturbed site, the density of *Scirpus acutus* was shown to decrease with increasing nutrient concentrations. In the moderately disturbed sites, it demonstrated a preference for increasing pH and water temperatures, and was influenced by SITE more than any other limnologic parameter. Also in the highly disturbed sites this species was associated with temperature and its density was influenced by lower conductivity readings. Evidently, there was no strong association or positive correlation of *Scirpus acutus* distribution with increases in nutrient concentration, DO, turbidity, etc. Therefore this study disagrees with Croft & Chow-Fraser (2007) in that *Scirpus acutus* is intolerant of environmental degradation, but does support their finding in that it is indicative of good water quality conditions. The responses of *Scirpus acutus* in this study are strongly supported by the finding of this species to be somewhat tolerant of nutrient increase (Michner 1990; as cited by US EPA 2000). Lougheed et al. (2001) found *Scirpus* spp. to be common in infertile soils and its dominance at DP where oligotrophic conditions occur is comparable to this result.

Typha x *glauca* and *Typha angustifolia* are invasive cattail species, and their tolerance of and response to water quality degradation and anthropogenic stress is widely discussed in the literature (Albert & Minc 2004; Croft & Chow-Fraser 2007; Lougheed et al. 2001; US EPA 2000). In the present study, either *Typha* x *glauca* or *Typha angustifolia* was found to be one of the dominant species with consistently higher IVIs in every moderately and highly disturbed site (Table 7). Both were found to be indicative of degraded water quality in coastal wetlands of the Great Lakes as they were very tolerant of degradation (Croft & Chow-Fraser 2007), and the same result was found in Lake Simcoe. Consistent with Croft & Chow-Fraser (2007), Lougheed et al. (2001) also found in coastal as well as inland Great Lakes basin wetlands *Typha* tolerated a turbid, nutrient-rich water column. The relationship with different limnologic parameters and environmental degradation at sampling sites showed the responses of *Typha* x *glauca* and *Typha angustifolia* to water quality degradation. This validates their utility as an indicator species of environmental degradation. In the moderately disturbed sites, a positive association with TN and

CHL *a* was revealed, and the densities of these species in the highly disturbed sites also showed positive associations with TP and COND, suggesting that as concentrations of these water quality parameters increase, distribution of these species will also increase, thus reflecting a positive response to increased nutrients and ions from runoff. The above-ground biomass of *Typha* x *glauca* showed a positive response to water depth, and this as well as its response to nutrient enrichment is consistent with Albert & Minc (2004), who stated that *Typha* spp. was an indicator of both nutrient enrichment and reduced fluctuations in water level in the Great Lakes coastal wetlands. Manipulation of lake levels disrupts the natural cycle, which favours species intolerant of water-depth change and associated stresses, and/or excludes those species that require periodic exposure to fertile substrates (Albert & Minc 2004). Consequently, the result of hydrologic flow modification as reported by Keddy (1989) and observed at CB and HM is a dense stand of *Typha* that will eventually replace a diversity of macrophytes. Albert & Minc (2004) proposed that the relative dominance of *Typha* spp., as well as measures of species diversity, provides a strong indicator of water level regulation in the Great Lakes, and the same was found in this study in Lake Simcoe. Additionally, nutrient enrichment probably played an important role in the dominance of *Typha* in these sites, for as described in Chapter 2, agricultural land use and urbanization in these wetland areas is extensive. In the Great Lakes, *Typha* spp. demonstrated a positive response to nutrient stress, as was found in this study also. Furthermore, *Typha angustifolia* demonstrated a positive response to conductivity in highly disturbed sites, which according to Minc & Albert (2004) is a good indicator of urban-runoff.

Along with *Typha* spp., invasive *Phragmites australis* was also found to demonstrate a positive response nutrient enrichment, increased sedimentation and turbidity, and physical degradation (Albert & Minc 2004). A positive response to nutrient enrichment was also supported by Michner (1990) as stated in the *Wetland Plant Sensitives Database* (US EPA 2000), which described *Phragmites australis* as very tolerant to nutrient increases. In this Lake Simcoe study*, Phragmites australis* was only present at HM, and no strong influence of TN and TP on this species' distribution was revealed, which was inconsistent with the literature. Instead, this species could be a more useful indicator of increased turbidity and physical degradation in Lake Simcoe fringe wetlands. Sedimentation was not measured in this study; however, the presence of *Phragmites australis* only at HM and the intensive agricultural stress impacting this wetland that can increase sedimentation substantiates the potential of this species to be an indicator of turbid water conditions. What this species appears to be an indicator of in this study is not chemical degradation of water quality, but physical degradation of the wetland, consistent with Albert & Minc (2004). The anthropogenic disturbances previously discussed impacting HM were removal of riparian vegetation, channelization, fragmentation, and filling-in of the wetland complex (Louis Berger Group 2006; O'Connor 2014).

Phalaris arundinacea, present but not dominant in moderately and highly disturbed Lake Simcoe fringe wetlands, has been reported to be an indicator of degradation by positively responding to increased turbidity and physical degradation in Great Lakes wetlands (Albert & Minc 2004). The relationship of this species with the limnologic parameters is consistent with the responses determined by Albert and Minc (2004). In this study, *Phalaris arundinacea* was consistently positively correlated with CHL *a*, a measure of the particulate matter in the water column that expresses turbidity, and the presence of this species in only moderately to highly disturbed sites is an indicator of the physical degradation that impacts these wetlands. Furthermore, studies from the US EPA (2000) *Wetland Plant Sensitivities Database* stated that *Phalaris arundinacea* was moderately tolerant of nutrient increases, (Michner 1990) and an increase in growth with the addition of increasing N concentrations was observed by Bernard (1995). Results of this study were consistent with Bernard (1995) as *Phalaris arundinacea* demonstrated a negative association with TN, as well as TP, which demonstrates that nutrient concentrations decrease as this species uptakes and exploits their availability.

Akin to *Phalaris arundinacea*, *Lythrum salicaria* was not a dominant species in Lake Simcoe, but its significance as an indicator species can be found throughout the literature. Croft & Chow-Fraser (2007) stated that this species was dominant in polluted sites and an indicator of degraded water due to its high tolerance. The presence of *Lythrum salicaria* in moderately and highly disturbed sites in Lake Simcoe agrees with the above report. This result is also supported by Michner (1990) who reported that this species is tolerant of nutrient increase (US EPA 2000). An examination of the relationship between this species and TN revealed a positive trend, however in the highly disturbed sites *Lythrum salicaria* demonstrated a preference for low TP concentrations, which is consistent with its reported low nutrient requirements. It may also reflect its preference moist, not inundated organic soils (Urbatsch 2000). *Lythrum salicaria* was reported by Albert & Minc (2004) to respond positively to sedimentation, and physical degradation in the Great Lakes. As previously mentioned, sedimentation was not measured in this study, but the utility of this species as an indicator of physical modification is reflected by its presence in moderately and highly disturbed sites with significant human impacts, as reported by Albert & Minc (2004).

Eleocharis smallii was a prominent species in the moderately disturbed sites. In the WMI, Croft & Chow-Fraser (2007) assigned a U-value of 4 to this species, stating that it was intolerant of environmental degradation and an indicator of good water quality conditions. Lougheed et al. (2001) reported a similar finding that in the Great Lakes *Eleocharis smallii* was present in wetlands with relatively low water turbidity and nutrient concentrations. The presence of *Eleocharis smallii* in moderately disturbed sites in this study is strongly supported by the finding of this species to be somewhat tolerant of nutrient increases (Michner 1990; as cited by US EPA 2000). Still, among the moderately disturbed sites, BC, which *Eleocharis smallii* was exclusive to, had the lowest nutrient and CHL *a*, concentrations, thus supporting the assertions of Croft & Chow-Fraser (2007) and Lougheed et al. (2001), and validating its use as an indicator species in Lake Simcoe wetlands. Other species exclusive to moderately disturbed sites, specifically LC, that are reported by the literature to be effective indicator species included *Sagittaria graminea* and *Sagittaria latifolia*. Overall, the literature shows that these species are tolerant of degraded conditions, but no specific responses to degradation are described. In Lake Simcoe, these species demonstrated a positive association to pH and SITE, suggesting an intolerance of acidic water, and thus degraded conditions. Croft and Chow-Fraser (2007) determined these species to be tolerant to water quality degradation; similarly, Lougheed et al. (2001) found *Sagittaria graminea* spp. to tolerate turbid, nutrient rich water in Great Lakes wetlands. In the US EPA (2000) *Wetland Plant Sensitives Database* no specific responses of this species were provided; only Gernes & Helgen (1999) determined that *Sagittaria graminea* was moderately tolerant of increases of an unspecified stressor, that is, general pollution increases. Ultimately, the response of *Sagittaria graminea* spp. to water quality parameters was it was intolerant of degraded conditions, however its presence at moderately disturbed sites conflicts with this result and mirrors the tolerance described above.

Several of the indicator species present in the study were introduced or exotic species. Trebitz & Taylor (2007) surveyed 58 coastal wetlands across a gradient of anthropogenic impacts in the 5 Great Lakes to describe the distribution of introduced and invasive taxa. The authors found that the number of introduced and invasive species increased with increasing levels of agricultural intensity in the watershed. The same finding was evident in Lake Simcoe also, as the highly degraded sites with the most intense agricultural activities were dominated by, and had the highest richness of, invasive species such as *Typha angustifolia*, *Typha* x *glauca*, and *Phragmites australis*. In the Great Lakes, *Phalaris arundinacea*, and *Lythrum salicaria* were more widespread than in Lake Simcoe (Albert & Minc 2004). In accordance with the literature, these exotic plant species responded to physically modified wetlands and water quality degradation and thus are potentially are good indicators of disturbance (Minc & Albert 2004; Simon et al. 2001). McNair (2006) has shown that impacted coastal marshes tend to be more susceptible to exotic invasions than are un-impacted wetlands, and over time, the native species in impacted sites can lose ground to introduced species. Consequently, it is expected that in moderately and highly disturbed sites the presence of introduced species will continue to increase. Wetlands found in urban or industrial areas are often dominated by introduced species, even though the number of these may be relatively low (Albert & Minc 2004). In this study, in CB and HM where agricultural and urbanization are intense, only *Typha* spp., and *Phalaris arundinacea*, and a limited number of other different introduced and invasive species were present, yet their dominance in the community was clear. Still, the coverage of introduced species may not be predictable over time, as they can rapidly respond to fluctuations in water level (Minc 1997).

Some research suggests that several native colonizing species respond in way similar to introduced species, in that these species can quickly expand in response to rapid reductions in water level and increased sedimentation. These early successional native annuals included *Bidens cernua*, *Impatiens capensis*, and *Leersia oryzoides* (Albert & Minc 2004; Minc & Albert

2004). Only the latter species was a dominant species in this study; *Bidens cernua* and *Impatiens capensis* were present in the moderately and highly disturbed sites. Only *Impatiens capensis* showed a response to decreased water depth that was consistent with the literature, as the biomass of this species was positively correlated with DEPTH. Instead, these species demonstrated a variety of responses to the limnologic parameters. In the moderately disturbed sites, *Leersia oryzoides* and *Bidens cernua* appeared to be negatively associated with TN, reflecting an intolerance to nutrient enrichment. In the highly disturbed sites, *Impatiens capensis* was positively associated with TP, and negatively associated with CHL *a*, while *Leersia oryzoides* and *Bidens cernua* were associated with COND. Despite the variety of these responses, they exhibit tolerances reported by the literature, that *Impatiens capensis* and *Leersia oryzoides* were moderately tolerant of nutrient increases, and *Bidens cernua* was only somewhat tolerant of nutrient increases (Michner 1990; as cited by US EPA 2000).

Evidently, some of the species in this study were indicative of more than one type of degradation, for example *Typha* spp. demonstrated a positive response to nutrient enrichment and water level regulation. Having a species or group of species that indicates more than one type of disturbance can complicate their value as an environmental indicator, possibly reducing their usefulness for making specific mitigation and restoration decisions. Also, it has been asserted that a better indicator of wetland quality is the type of community rather than the presence/absence of certain indicator species (Lougheed et al. 2001). Nevertheless, despite these minor obstacles in biological assessment using indicator species, several studies support the use of indicator species to determine wetland water quality and ecological integrity (Albert & Minc 2004; Croft & Chow-Fraser 2007) and the results of this study support their use.

4.5 Conclusions

This chapter explored whether Lake Simcoe's fringe wetland macrophytes function as indicator species of anthropogenic degradation and its impacts on wetland integrity. The relationship between macrophyte species and specific water quality parameters was studied to explore the responses and potential differences in tolerance of different macrophyte species to various water quality parameters. Certain species and their responses are considered to be indicators of water quality, (Albert & Minc 2004; Croft & Chow-Fraser 2007; US EPA 2000) which in this site was a proxy of site disturbance. Some of the widely reported indicator species present in Lake Simcoe's fringe wetlands included *Scirpus acutus*, *Sparganium eurycarpum*, *Pontederia cordata*, *Phalaris arundinacea*, *Lythrum salicaria*, and *Typha* spp. Overall, it was determined that the responses and tolerance of indicator species to water quality parameters were consistent with previous reports in the literature. In a few cases the species better indicated the physical degradation impacting the wetland rather than the chemical degradation affecting water quality. Therefore, it can be said that macrophytes in this study generally reflect the quality of the water and can potentially be used as indicators of it. Furthermore, these results will aid in the creation of a specific macrophyte index, such as the WMI by Croft & Chow-Fraser (2007), as a cost effective management strategy for Lake Simcoe.

Chapter 5 Conclusions

Emergent macrophyte communities are reliable indicators of water quality changes in aquatic ecosystems. Macrophytes can reflect the changes in water quality because they are affected by the temporal, spatial, chemical, biological, and physical dynamics of the water. Hence, impairments in wetland water quality by anthropogenic stressors could potentially be reflected by taxonomic composition and dynamics of the plant community (Croft & Chow-Fraser 2007; Cronk & Fennessy 2001; NWWG 1997). Using the data from six wetlands across the Lake Simcoe basin, it was established that the macrophytes responded predictably to water quality degradation through alterations in community composition and metrics (richness, density, diversity and above-ground biomass), supporting the assumption that plants are indeed good indicators of water quality conditions and site disturbance in wetlands.

In order to understand the spatial and temporal variation of water quality and macrophyte composition and dynamics, data were collected from six different fringe wetlands around Lake Simcoe subject to varying degrees of exposure to anthropogenic stressors. The study was repeated four times to represent different seasons of a year. Several objectives and hypothesis were tested in this study. The objectives related to water quality parameters, which were measured as a proxy of site disturbance, were to analyze the variance of water chemistry at all sites and seasons to determine if water quality reflects the degree of anthropogenic disturbance impacting the wetland. The objective of the macrophyte survey was to determine dominant species at each site, and analyze the variation in species richness, diversity, density, and above-ground biomass between sites and seasons.

It was expected that the water quality parameters and macrophyte community and dynamics would vary both spatially and temporally. Correspondingly, it was hypothesized that water quality will reflect the degree of anthropogenic degradation influencing the wetland. The least anthropogenically disturbed wetland (DP) will have the greatest water quality; the highly anthropogenically disturbed sites (CB and HM) will demonstrate the poorest water quality, while moderately disturbed sites (BC, VP, and LC) will reflect an intermediate of this gradient. It was also hypothesized that emergent macrophyte community/composition and metrics including richness, density, diversity, and above-ground biomass; as well as indicator characteristics will reflect the quality of the water in the wetland. Specifically, the site with the highest water quality (and thus least anthropogenically disturbed, DP), was expected to have higher species richness and diversity, but lower densities and above-ground biomass. Conversely, the sites with the poorest water quality, and thus highly degraded (CB and HM) will have the lowest species richness and diversity, but greater densities and above-ground biomass.

The results demonstrated that wetlands in this study represented a wide range of environmental conditions, ranging from very clear and nutrient poor (e.g., TP =11.25 μg/L, TN = 345.25 μg/L, CHL *a* =1.37 $mg\cdot m^{-3}$) to turbid and eutrophic (e.g., TP = 47.25 μg/L, TN = 2285 μg/L, CHL *a* = 3.26 $mg\cdot m^{-3}$). The limnologic parameters demonstrated temporal and spatial variation, and overall indicated that water quality reflected the degree of anthropogenic degradation in the wetland. The least anthropogenically disturbed wetland DP had the highest water quality; the most anthropogenically disturbed sites, CB and HM, had the poorest water quality, while moderately disturbed sites BC, VP, and LC reflected an intermediate water quality. Species richness, density, diversity, and above ground biomass varied among sites and between seasons, and the dominant vegetation reflected the water quality of each wetland. The least disturbed site DP was dominated by the native species *Sparganium eurycarpum*, *Scirpus acutus* and *Scirpus pungens*. The moderately disturbed sites were dominated by both native species with varying aggressiveness, and introduced species, including *Leersia oryzoides*, *Eleocharis smallii*, *Scirpus acutus*, *Sparganium eurycarpum*, *Typha* x *glauca*, and *Typha angustifolia*. Highly disturbed sites were dominated by only one native species, *Calamagrostis canadensis*, as well as several introduced species- *Typha* x *glauca*, *Typha angustifolia*, and *Phragmites australis.* All three of these are considered to be invasive, tolerant of degradation, and/or able to compete with native species, form monocultures and reduce species diversity (Albert & Minc 2004; Croft & Chow-Fraser 2007; Hill 2008; Tuchman et al. 2009; Tulbure et al. 2007; US EPA 2000). *Sparganium eurycarpum* was the only species that was present in all sites, supporting the perspective that this species is cosmopolitan and tolerant of varying environmental conditions (Croft & Chow-Fraser 2007; US EPA 2000). Thus, the results from this study provide important and useful information to ecosystem managers to consider macrophyte community dynamics as an environmentally benign method of assessment of water quality in this area.

Additional objectives of this study were to explore the relationship between macrophyte species and specific limnologic parameters to investigate the responses of different species to various limnologic parameters, and to assess their utility as indicators of water quality. The hypothesis tested was the indicator species will have a relationship with the limnologicparameters measured in this study. The responses and tolerance of indicator species to water quality was consistent with the literature and substantiated the use of emergent macrophytes in Lake Simcoe's fringe wetlands as potential indicators of water quality, and wetland integrity.

Some research suggests that certain emergent macrophyte species and their responses and/or tolerances are reflections of water quality (Albert & Minc 2004; Croft & Chow-Fraser

2007; Lougheed et al. 2001; Rothrock & Simon 2007; Wilcox et al. 2002). Some of the widely reported indicator species present in Lake Simcoe's fringe wetlands included *Scirpus acutus*, *Sparganium eurycarpum*, *Pontederia cordata*, *Phalaris arundinacea*, *Lythrum salicaria*, and *Typha* spp. Overall, it was determined that the responses and tolerance of indicator species to water quality parameters were consistent with the literature (Albert & Minc 2004; Croft & Chow-Fraser 2007; Lougheed et al. 2001; US EPA 2000). In a few cases the species better indicated the physical degradation impacting the wetland rather than the chemical degradation affecting water quality. Therefore, it can be said that macrophytes in this study generally reflected the limnologic conditions influenced by disturbance, and hold promising potential to be indicators of water quality.

With a large local human population and further significant growth being planned (LSRCA 2013b), it is important that the Lake Simcoe watershed continues to support high quality natural heritage and shoreline areas. Conservation efforts need to ensure that adequate research and funding are allocated to monitor and protect these habitats so that further degradation is prevented and their integrity is maintained. Croft & Chow-Fraser (2007) developed a WMI that can be used to rank the degree of human-induced disturbance among a wide range of coastal wetlands. A modification of this index might also benefit the monitoring and future management of coastal areas of Lake Simcoe. A major objective of this study was to develop baseline data that can be used to develop a WMI that could be used by environmental agencies. That approach must be easy to implement, cost-effective, and able to detect year-to-year changes in wetland conditions. In future studies, the WMI is an index that can be used to track the impact of human-induced disturbances and its effects on macrophytes in fringe wetlands. There are a limited number of minimally impacted wetlands along the Lake Simcoe shoreline, and a WMI can be one of the useful biotic assessment tools available for use by volunteers, researchers, and government personnel to monitor the status, and ensure the ongoing health of coastal wetlands. It is imperative to continually remind members of the community, present and future, of the fact that coastal wetlands provide us with irreplaceable ecosystem services, without which biodiversity, particularly of native species, in the Lake Simcoe watershed will be placed at serious risk (Cvetkovic & Chow-Fraser 2011).

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