Understanding the Role of Aquatic Plants in Stormwater Management Pond Performance in Oshawa, Ontario, Canada

By

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The above committee determined that the thesis is acceptable in form and content and that a satisfactory knowledge of the field covered by the thesis was demonstrated by the candidate during an oral examination. A signed copy of the Certificate of Approval is available from the School of Graduate and Postdoctoral Studies.

ABSTRACT

Stormwater Management Ponds (SMPs) are engineered to receive, store, and treat stormwater runoff before it enters receiving waters in urbanizing landscapes. While these systems are not considered natural, they are typically colonized by aquatic plants. Although submergent and emergent vegetation is common in SMPs, not much is known about their potential impacts on SMP performance. The aim of my thesis project was to investigate the effect of aquatic plants on the water treatment capacity of 15 SMPs in Oshawa, Ontario, Canada, over two years (2018-2019). I determined that overall, SMPs serve as sinks for certain water quality parameters including chloride and nitrogen, while being a net source of phosphorus to tributaries. The effect of plants on SMP performance was mixed. Increasing submergent plant biomass was associated with decreasing nitrogen concentrations at outflow locations ($p = 0.002$, cor = -0.316). Emergent vegetation had no significant impact on stormwater treatment overall, but the invasive species, *P. australis* was associated with decreasing outflow nitrogen concentrations. Overall, I determined that pond characteristics, including pond size, age, and drainage area are significant drivers of established plant profiles.

Keywords: urban ecology; stormwater management ponds; water quality; aquatic vegetation

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I hereby certify that I am the sole author of this thesis and that no part of this thesis has been published or submitted for publication. I have used standard referencing practices to acknowledge ideas, research techniques, or other materials that belong to others. Furthermore, I hereby certify that I am the sole source of the creative works and/or inventive knowledge described in this thesis.

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

Appendix A: Supplementary Material for Chapter 2

A.1 List of Figures

A.2 List of Tables

Appendix B: Supplementary Material for Chapter 3

B.1 List of Figures

CHAPTER 1: GENERAL INTRODUCTION

1.1 INTRODUCTION

1.1.1 Urbanization and impacts to freshwater

Urbanization is becoming the basic framework for developing countries. It is predicted that by the year 2030, there will be over 2 billion new urban global residents (McDonald et al., 2008). Patterns of urban landscape development revolves around maximizing developed space, consequently limiting natural surfaces. North America has been recognized as one of the most developed landscapes around the globe (Elmqvist et al., 2013). Within Canada and the United States, over 80% of the population is classified as living in an urban area (Elmqvist et al., 2013). In Ontario specifically, the population is expected to increase by 38% over the next 28 years (Ontario Ministry of Finance, 2019). This rapid expansion will have the most devastating impacts on biodiversity in developing areas (McDonald et al., 2008).

Rapid urban development can greatly alter landscapes by removing natural soils and plant cover, consequently changing the natural topography (Rhea et al., 2015). This phenomenon is known as landscape transformation. One of the defining features of urban development and landscape transformation is a shift from pervious to impervious surfaces (Gallagher et al., 2011). This includes any surfaces that are non-porous, such as roads, driveways, parking lots, as well as lawns with shallow soil profiles. These types of surfaces limit the ability of stormwater to percolate into naturally porous soils, forcing it directly into natural water bodies. This can have overwhelming repercussions on aquatic communities receiving this runoff water for two main reasons. Firstly, alterations to soil characteristics can increase stormwater runoff velocity and volume (Rhea et al., 2015). These high velocities can result in localized flooding, as well as erosion to natural soils. Due to the lack of percolation, incoming surface water is also given more time to accumulate a variety of pollutants, such as bacteria, nutrients, and debris (Rhea et al., 2015). This can be detrimental to aquatic communities receiving this stormwater runoff, especially since freshwater habitats tend to undergo greater biodiversity declines compared to terrestrial habitats (Hassall, 2014).

Unfortunately, the repercussions of stormwater have been enhanced in recent years due to increased rain events and global climate change. An increase in the occurrence of 100-year storms has been noted, with the likelihood of occurrence increasing to 1 in 30 years (Marsooli et al., 2019). These now regular storm events have forced urbanizing regions to develop innovative ways to manage their stormwater runoff, and limit pollution into naturalized systems.

1.1.2 Major constituents of urban stormwater

Urban water systems are regularly exposed to a variety of anthropogenically sourced contaminants. Nutrients are key components of aquatic communities, but high concentrations can have devastating effects on ecosystems. Phosphorus is typically a limiting nutrient in natural aquatic environments, and plays an important role in plant and algal growth. Phosphorus concentrations in urban freshwater systems is sourced from fertilizers, animal waste, soil loadings, and atmospheric deposition (Yang & Lusk, 2018). Phosphorus can enter aquatic systems in particulate or dissolved forms, leading to eutrophication of inland waters. Freshwater systems are considered eutrophic if phosphorus levels exceed 35 μg/L (Government of Canada, 2015). Nitrogen is another essential nutrient for plant and algal growth, which is sourced from fertilizers and animal waste washed off surrounding landscapes. Both inorganic and organic forms of nitrogen can be found in surface waters, however individual types show various toxic effects to organisms (Casey & Klaine, 2001; Massal et al., 2007). All forms of nitrogen which enter freshwater environments can be directly or indirectly bioavailable, and may further influence eutrophication effects.

Salt (mainly in the form of NaCl) is also of major concern in developing landscapes, due to its application for de-icing of paved surfaces. Both current applications and legacy salt concentrations are easily mobile during storm events and winter melts (Marsalek, 2003; Dugan et al., 2017). Freshwater systems can become saltier in areas where excess salt is directly washed into ecosystems. Chloride ions in road salt is of particular concern for freshwater organisms, due to its ability to induce toxicity in a variety of species (Gillis, 2011; Hintz & Relyea, 2017; Jones et al., 2017). The Canadian Water Quality Guidelines for the Protection of Aquatic Life recognizes chloride levels above 120 mg/L to be toxic for organisms with long-term exposure (Canadian Council of Ministers of the Environment, 2011).

Urban freshwater environments can also be exposed to various metals, sourced from both industrial and transportation activities. Metals are commonly found in dust forms across urban landscapes, making them well transported via stormwater runoff and capable of settling into freshwater sediments along with suspended solids. Both copper and zinc have been noted in high concentrations throughout urbanized sites in Oshawa Creek (Kirkwood, 2016). While the majority of metals found in aquatic environments are considered micronutrients, excess concentrations can have toxic effects on organisms.

Organic contaminants are also common pollutants to urban freshwater environments, especially in well-developed residential areas. Organic contaminants can be sourced from lawn maintenance practices such as the use of herbicides, as well as atmospheric deposition of pesticides and hydrocarbons. Persistence of organic pollutants in communities can result in toxic effects to aquatic wildlife due to their ability to bioconcentrate and accumulate (Katagi, 2010). Due to these and other contaminants affecting freshwater communities, urbanizing cities must mitigate the impacts of stormwater on downstream ecosystems.

1.1.3 Design and functionality of Stormwater Management Ponds

Over the last 30 years, Stormwater Management Ponds (SMPs) have become a ‘best management practice’ for runoff surface water across North America (Casey et al., 2006; Drake & Guo, 2008; Williams et al., 2013; Frost et al., 2015). They are engineered waterbodies, that are becoming increasingly common in both residential and commercial areas. The initial introduction of SMPs into Canadian stormwater management practices occurred in the early 1970’s due to a notable increase in runoff peaks (Watt et al., 2001). The original design of these ponds greatly reduced peak flows as well as flooding potential and drainage expenses caused by excess runoff (Marsalek et al., 1992). Early research on these original ponds however, highlighted their potential to cause damage to receiving waters (Marsalek et al., 1992). As such, their recognition as a ‘best management practice’ did not occur until the 1990’s when new pond designs included measures to not only reduce peak flows, but also improve water quality (Marsalek et al., 1992; Watt et al., 2003). These measures included adding elements such as forebays, which capture and hold sediment from inlet locations, as well as planting vegetation along the pond embankment to reduce erosion (Government of Ontario, 2003). Currently, the construction of these ponds in Ontario must ensure five key factors are met (Government of Ontario, 2003), this includes,

1. The preservation of groundwater,
2. The protection of water quality,
3. The resulting watercourse will not cause any geomorphic change,
4. There is no increase in flooding potential, and,
5. An appropriate diversity of aquatic life is maintained.

The overall design of SMPs varies greatly depending on pond location, physical site characteristics (i.e. topography and soil substrate), as well as surrounding drainage area (i.e. total area surrounding the pond which collects incoming precipitation) (Government of Ontario, 2003). However, the designed purpose of urban ponds remains consistent since they are considered an “end of pipe control” for their watershed. In this way, the overall functionality of these ponds is to resolve the two major hydrological problems that arise with surface runoff: water quality and quantity.

SMPs are regularly exposed to multiple anthropogenic stressors, including physical (i.e. high-water volumes), chemical (i.e. nutrients, pollutants), and biological (i.e. invasive species, bacterial contamination) factors (Tixier et al., 2011). However, their primary function is to handle physical stressors, and reduce the velocity of incoming surface water (Casey et al., 2006). This functionality ultimately slows the release of stormwater into receiving waters and reduces peak flow potential (Drake & Guo, 2008; Song et al., 2013; Miró et al., 2018). The introduction of SMPs into urban areas has greatly minimized the repercussions of more frequent 100-year storm events. Physical barriers to reduce water velocity can be included in a number of ways, including tiling or gravel at inflow locations, addition of sediment forebays, as well as introduction of aquatic vegetation (Government of Ontario, 2003). By slowing stormwater velocity, urban ponds limit erosion to surrounding water bodies, as well as the possibility for flooding and increased stream velocities in urbanized settings (Olding et al., 2004; Miró et al., 2018).

A secondary function of SMPs includes their ability to improve outflow water quality (Tixier et al., 2011). Urban surface water runoff has been noted as a major source of pollution to surrounding freshwater systems (Davis et al., 2001; Walaszek et al., 2018). These ponds are therefore engineered to enable the settling of particulates (and adsorbed contaminants), limiting its release into the environment and reducing nonpoint pollutant loadings (Wu et al., 1996; Gallagher et al., 2011). SMPs utilize naturally occurring processes (i.e. sedimentation) which are capable of removing material common to surface waters, including suspended solids, heavy metals, nutrients, bacteria, and hydrocarbons (Marsalek et al., 1997; Olding et al., 2004; Frost et al., 2015; Ivanovsky et al., 2018). These settling processes occur during an engineered retention time, which varies depending on pond depth and width to length ratios. In general, optimal retention times are in the range of 24-48 hours during a storm event.

Urban SMPs show impressive water column reductions of pollutants, limiting the impact of contaminated discharge on downstream biological communities, and potentially functioning as contaminant sinks (Olding et al., 2004; Frost et al., 2015). Efficiencies ranging from 60% to 90% have been noted for the removal of suspended solids from runoff water (Marsalek et al., 1997). The removal and accumulation of heavy metals in SMP sediments has also been thoroughly reviewed (Van Buren et al., 1996; Davis et al., 2001; Weiss et al., 2006). Removal of zinc and iron has been recorded at efficiencies of 80% and 87% respectively (Davis et al., 2001). Other studies have highlighted that in ponds where zero or limited removal of dissolved constituents occurs (i.e. chloride and nutrients), complete removal of metals and organics is still possible (Van Buren et al., 1996). It should also be noted that part of the water treatment which occurs in SMPs, is due to dilution from the permanent pool within the pond. In this way, incoming suspended solids and contaminants from surface water is diluted prior to leaving the pond. However, improvements to water quality can be variable between ponds, due to discrepancies in sediment maintenance practices.

The overall maintenance of urban ponds can be quite extensive, and is the responsibility of Ontario municipalities (who own the majority of SMPs in their jurisdictions). Due to their designed features, maximal performance of SMPs greatly decreases over time as sediment accumulates and decreases water holding capacities (Drake & Guo, 2008). Several factors can influence sediment accumulation including surrounding land use, construction, and SMP design. This reduction of total water volume may result in localized flooding, and decreased ability to capture incoming particulates. As a result, it is the responsibility of the municipality to regularly remove all sediment and associated vegetation within the pond in order to maintain original pond depth (Drake & Guo, 2008). This process, known as ‘dredging’, is suggested as common practice for all cities maintaining their ponds, however cost tends to limit regular upkeep. The mechanical removal of sediment is a relatively cheap process, however, the sediment itself can be highly contaminated with hazardous constituents, and therefore must be disposed of appropriately (Drake & Guo, 2008). In Ontario, SMPs are designed to last approximately 10 to 15 years without sediment maintenance, but cases vary by pond (Drake & Guo, 2008). However, it is not uncommon for SMPs to remain unmaintained well beyond their expected performance life. In fact, many municipalities assume their established SMPs are meeting performance requirements, and therefore do not monitor local ponds for changes in water quantity or quality.

1.1.4 Biodiversity of Stormwater Management Ponds

Although they are engineered systems, SMPs can also serve as refuge for local fauna and flora. Even with potential exposure to excess nutrients, bacteria, and other pollutants, many species can still inhabit and even thrive in a variety of urban pond habitats (Foltz & Dodson, 2009). SMPs have been noted to support diverse aquatic and terrestrial species, and may function as essential wildlife refuge in areas where natural ponds and wetlands are lost due to urbanization (Casey et al., 2006; Gallagher et al., 2011; Miró et al., 2018). These ponds also act to improve opportunities to enhance local biodiversity, and are considered crucial biodiversity "hotspots" in urban areas (Tixier et al., 2011; Holtmann et al., 2018; Miró et al., 2018). Freshwater habitats have been noted to undergo greater biodiversity declines compared to terrestrial environments (Hassall, 2014), therefore SMPs and other small freshwater systems may contribute a great deal to improving biodiversity in urban settings.

It has been recognized that smaller water bodies, such as urban ponds, are generally more biologically active than larger waterbodies (Williams et al., 2013). Furthermore, these systems can provide an opportunity to enhance and conserve freshwater biodiversity, while simultaneously utilizing key ecosystems services including basic stormwater treatment and storage (Hassall & Anderson, 2015; Hill et al., 2017). In this way, enhanced biodiversity within these ponds may also act to greatly improve water treatment processes by taking advantage of differences between individual taxa (Leto et al., 2013). Various growth and life cycles may provide a greater number of water treatment possibilities, thereby maximizing pond efficiency. Urban ponds may in fact be a 'Jack-of-all-Trades', providing essential water treatment services while acting as functional habitats for local species. However, it should be noted that all taxa found within SMPs colonize these ponds through natural dispersal mechanisms, and are not purposefully introduced. For plants specifically, design plans suggest regular planting within and surrounding local SMPs (Government of Ontario, 2003). However, in Ontario, most municipalities only incorporate upland planting in the riparian zone surround the pond, and do not plant aquatic macrophytes. In this way, species that colonize SMP habitats must be naturally resistant to variable water conditions and potentially high pollutant levels.

1.1.5 Aquatic vegetation in Stormwater Management Ponds and potential for water treatment

Regardless of the high productivity of these dynamic ecosystems, very little is understood about their biological function, and its effects on water quality treatment (Williams et al., 2013). Early studies completed on SMPs recognized the possibility for in-pond biological processing to improve outflow water quality (Marsalek et al., 1992). Similar studies completed in constructed wetlands illustrate the potential for aquatic plants (both emergent and submergent species) to play a significant role in physically improving water treatment processes at these locations (Lee & Scholz, 2007). It has been noted that macrophyte biomass in freshwater systems can enhance processes such as sedimentation and filtering (Vymazal, 2011). Aquatic plant growth may also decrease water velocities, ultimately lengthening retention times and improving particulate removal capabilities (Pettecrew & Kalff, 1992; Lee & Scholz, 2006). In fact, the ability of constructed wetlands to remove suspended solids was notably higher in sites containing macrophytes compared to those without (Karathanasis et al., 2003). In this way, the presence of established plant communities and resulting physical barriers may further enhance water treatment.

The interactions of macrophytes with microorganisms found in pond sediment and water may also significantly contribute to stormwater treatment (Leto et al., 2013). Biofilms are responsible for a large portion of the microbial water treatment processes which occur in constructed wetlands and urban ponds (Leto et al., 2013). Their presence within freshwater environments is positively associated with increasing macrophyte biomass (Leto et al., 2013). Furthermore, aquatic plants can enhance the production of nitrifying bacteria via oxygen transport to the rhizosphere (Reddy et al., 1989). In this way, the presence of macrophytes in a system may encourage aerobic decomposition and the removal of stormwater pollutants.

Aquatic vegetation may also directly contribute to pollutant removal in stormwater. It was highlighted that within urban SMPs, two types of biological treatment may occur. This includes treatment via suspended plant biomass, but also through rooted vegetation (Marsalek et al., 1992). Some studies have shown that a variety of both terrestrial and aquatic plant species are capable of removing contaminants from stormwater (Fritioff & Greger, 2003; Ivanovsky et al., 2018). Specifically, aquatic vegetation has been noted to uptake zinc, copper, and lead from stormwater in constructed wetlands (Fritioff & Greger, 2003). Certain species of aquatic grasses have also been noted to remove heavy metals from the sediments of urban ponds (Weiss et al., 2006). Rooted plants especially are capable of facilitating pollutant adsorption, as well as uptake through both the plant-sediment and plant-water interface (Marsalek et al., 1992). Free-floating macrophytes have also been noted as an effective way to directly remove nutrients from stormwater inflows (Chang et al., 2012). SMPs can be made up of plant communities established by a variety of free-floating, submergent and emergent macrophytes. Aquatic plant type, abundance, and community structure in a SMP may enhance its ability to treat stormwater and improve quality prior to discharge.

1.2 GOALS AND OBJECTIVES

The main goal of my thesis research was to understand the functional role of aquatic vegetation in Oshawa SMPs, including their potential effects on water quality treatment. This study also aimed to understand the role of surface runoff in influencing the structure of established plant communities in SMPs. To achieve these goals, the following research objectives were completed:

1. Assess the water treatment performance of SMPs in Oshawa, Ontario reflecting variations in age and vegetation cover.
2. Determine the effect of aquatic plant abundance, type (i.e. species, emergent or submergent), and diversity on the water quality profiles of 15 SMPs.
3. Determine the effect of inflow water quality on defining aquatic plant abundance, type (i.e. species, emergent or submergent), and diversity in Oshawa SMPs.

1.3 SIGNIFICANCE

SMPs are becoming a necessary reality of urbanizing regions in the Great Lakes Basin, including those found within the Durham Region. Although they are becoming more prevalent, there is still a lack of knowledge surrounding the water treatment processes occurring within these ponds, and the effect of macrophytes on stormwater quality. This study provides critical information on the health and efficacy of 15 SMPs located throughout the city of Oshawa, ON. Aquatic plant type and abundance was identified for the selected SMPs, which marks the first-time complete plant profiles have been described for SMPs in Canada. The function of these plant communities was assessed in relation to water treatment processes occurring between in and out locations. Furthermore, the influence of inflow water quality on established macrophyte communities was also addressed. This information will provide direction for future SMP construction and maintenance to promote optimal water treatment performance.

The following chapters summarize the results obtained from data collected over a two-year study period (2018-2019). Chapter 2 focuses on pond performance within Oshawa SMPs, and the ability of the selected sites to function as sources or sinks of stormwater constituents. Chapter 3 highlights the structure of aquatic plant communities in Oshawa SMPs, and the effect of macrophyte abundance, diversity, and type on outflow water quality. Chapter 4 examines the influence of inflowing stormwater quality and specific pond design elements on aquatic plant communities established in SMPs. Finally, Chapter 5 summarizes the results obtained from this study, and offers recommendations for future SMP maintenance in the City of Oshawa. Potential endeavors for future research on water treatment processes in SMPs are also highlighted.

CHAPTER 2: STORMWATER MANAGEMENT POND PERFORMANCE IN OSHAWA, ONTARIO, CANADA

2.1 INTRODUCTION

Stormwater management ponds (SMPs) are an essential aspect of developing landscapes. Over recent decades, they have become a predominant feature in growing residential and commercial areas (Casey et al., 2006; Williams et al., 2013; Frost et al., 2015). These ponds are designed as a simple yet effective way of reducing runoff velocity and decreasing stormwater suspended solids (Wu et al., 1996; Olding et al., 2004; Walaszek et al., 2018). However, their ability to consistently remove stormwater pollutants from runoff has been questioned. In fact, research has shown that SMPs can have high variability in terms of their water treatment processes.

In general, SMPs are primarily constructed to maximize water holding capacity and minimize flood potential in urban settings (Casey et al., 2006). The physical barrier provided by SMPs between natural systems and stormwater runoff is an essential functionality in urbanized settings. In fact, initial introduction of SMP facilities into developing areas resulted in major decreases to peak flows and flooding potential in natural streams (Marsalek et al., 1992). Urban ponds also reduce the risk of erosion to natural systems, by reducing the velocity of runoff. A number of specific pond design traits can contribute to further reducing runoff velocity including pond size, aquatic vegetation, as well as the addition of physical barriers such as sediment forebays.

A secondary function of SMPs is their ability to improve water clarity, and quality to some extent. In general, it has been accepted that SMPs are fairly sufficient in removing suspended particulates from incoming surface water (Marsalek et al., 1992;

Marsalek et al., 1997; Gallagher et al., 2011). Urban SMPs are engineered to maximize retention time, ultimately providing runoff particles sufficient time to settle into pond sediments (Wu et al., 1996; Gallagher et al., 2011). Pond depth also plays an essential role in maximizing stormwater residence time, and can have outstanding effects on particulate removal (Marsalek et al., 1992). Efficiencies ranging from 60% to 90% have been noted for the removal of suspended solids from runoff water (Marsalek et al., 1997). Due to this natural accumulation of particulates, urban ponds require sediment maintenance via dredging, typically every 10-15 years (Drake & Guo, 2008). This process is completed in order to maintain pond depth and maximize sedimentation of particulates. The overall effects of dredging on water quality changes from inflow to outflow locations has not been addressed.

Nutrients (phosphorus and nitrogen) are common constituents to urban aquatic environments, and are sourced from a variety of anthropogenic factors including fertilizers. It has been suggested that removal of nitrogen in stormwater management facilities can be highly variable, ranging from ponds acting as sources to complete removal of nitrogen (Koch et al., 2014). However, it was highlighted that wet ponds (i.e. retention ponds, SMPs) show more effective nitrogen removal capabilities compared to dry ponds (i.e. detention ponds). Furthermore, small and shallow ponds have been noted to more efficiently remove all forms of nitrogen compared to larger facilities (Koch et al., 2014). Other studies have shown opposing trends for phosphorus removal, by which ponds that maximize length to width ratios and macrophyte cover, undergo optimal nutrient removal (Mallin et al., 2002). These discrepancies highlight the lack of knowledge surrounding nutrient removal processes in SMPs, as well as design characteristics which maximize water treatment.

For aquatic ecosystems located in urbanized settings, road salt is a major source of toxicity to established communities. Since SMPs act as intermediaries between natural systems and urbanized landscapes, they tend to receive the brunt of excess salt application. It has been noted that SMPs can undergo stratification from high salt concentrations (Marsalek, 2003). In this way, salt concentrations can vary with pond depth, ultimately trapping the saltiest water at the sediment-water interface. These patterns however are dependent on seasonality and salt application regimes. Research completed on SMPs suggest that while they may act to slow the release of chloride, they are not sufficient in reducing loadings to naturalized systems (Snodgrass et al., 2017).

This chapter focuses on the functional performance of 15 SMPs located in Oshawa, Ontario, Canada. By comparing inflowing stormwater quality to outflowing water quality, I aimed to assess the ability of these ponds to function as sinks and/or sources of a variety of water quality parameters. Furthermore, the water treatment variations across study ponds has been assessed based on a variety of defining characteristics including pond size (length, width, area, depth), pond age, drainage area, surrounding impervious cover, and sediment maintenance via dredging.

2.2 MATERIALS AND METHODS

2.2.1 Study Location

Oshawa, Ontario, Canada, is a growing urban city located in Southern Ontario, approximately 60 km east of Toronto. Noted as the eastern anchor of the Greater Toronto

Area, it is the largest municipality in Durham Region, reflecting a land-use gradient of older industrial zones in the south and newer residential zones in the north. Fifteen stormwater management ponds within the city of Oshawa were selected for this study in order to assess the effects of aquatic vegetation on water quality treatment (Figure 1). These ponds were chosen based on their various ages, sediment maintenance, surrounding land use, and accessibility (Table 1). The selected ponds represent a variety of urbanized landscapes, including well established residential zones, newly developed areas, and active construction sites. Special consideration in pond selection was also placed on relative vegetation cover (for both submergent and emergent aquatic plants), as well as their location across the city's latitudinal gradient. For this study, a wide range of aquatic vegetation coverage was selected in order to capture changes in water quality dynamics with various plant communities, densities, and types.

Notably, no SMPs were selected in the South West portion of the city of Oshawa. This represents the downtown portion of the city, which is old enough that SMPs were not included in original design plans. In this way, no ponds are located in the downtown core. For this reason, the majority of ponds are located further North, where newer construction (i.e. past 30 years) contains SMPs in development designs.

Of the 15 selected ponds, three underwent sediment maintenance dredging (ponds 4, 6, 11) in the late winter / early spring of 2018. Through this process, all excess sediment and associated aquatic vegetation is mechanically removed from the body of the pond. Dredging is completed in order to maximize water holding capacity, and maintain original pond depth.

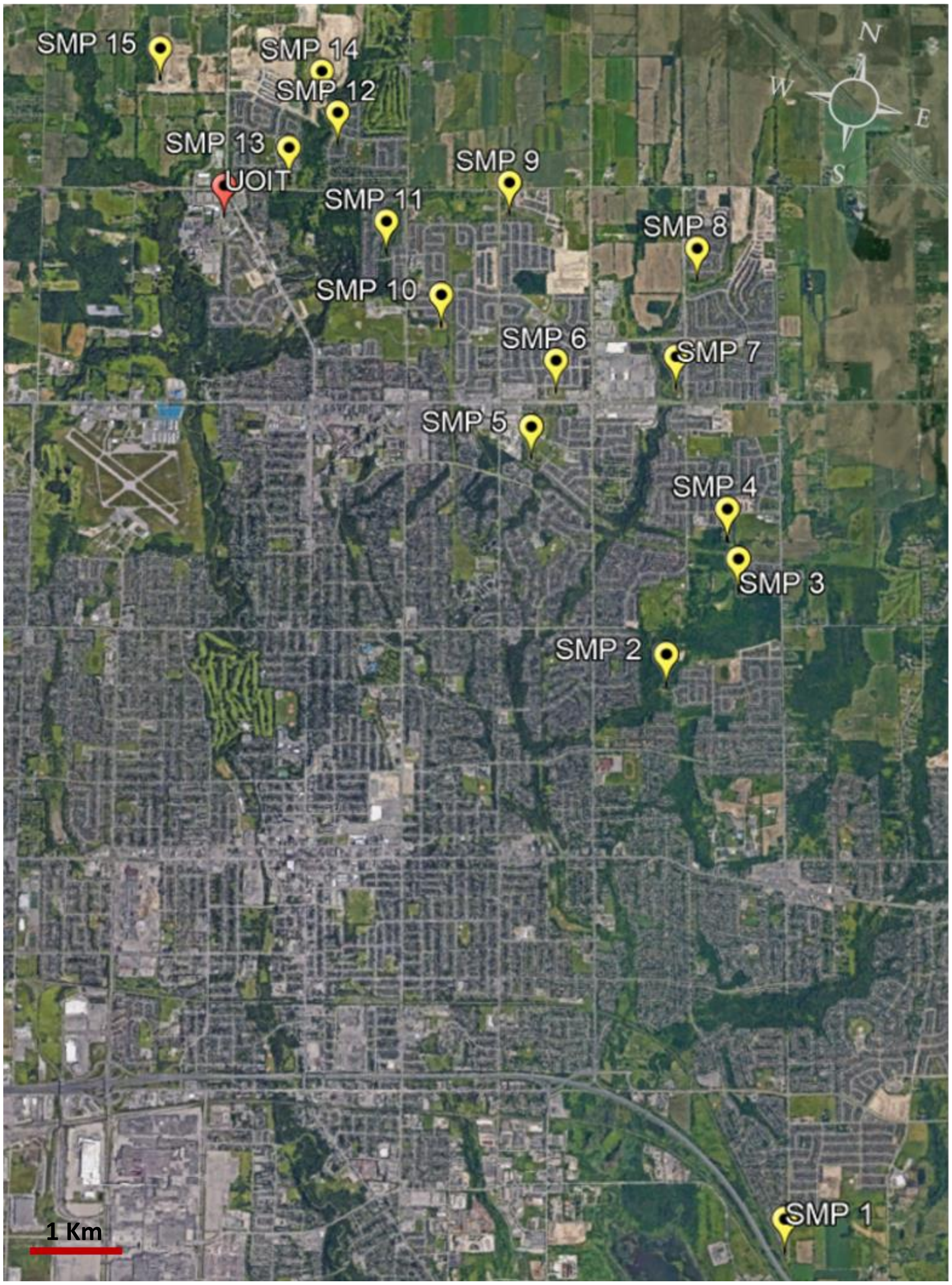

Figure 1. Locations of 15 selected stormwater management ponds in Oshawa, Ontario, Canada.

Table 1. Characteristics of 15 sampled stormwater management ponds in Oshawa, Ontario, Canada.

Pond	Pond Age (Years)	Drainage Area (ha)	Impervious Surface (%)	Forebay Present	Permanent Pool Depth (m)	Adjacent Landuse
1	13	11.32	NA	Yes	1.2	Residential
2	17	28.65	48	Yes	2.5	Residential
3	13	9.38	NA	Yes	1.5	Residential
4*	17	26.99	NA	Yes	1.2	Residential
5	18	142.9	NA	Yes	1.2	Residential
6*	13	42.25	61	Yes	1.05	Residential/commercial
7	19	69.8	48	Yes	2	Residential/commercial
8	12	43.1	NA	Yes	1.27	Residential
9	14	62.62	40	Yes	2.2	Residential
10	20	47.63	NA	Yes	1.85	Residential
11*	26	30.9	NA	No	0.1	Residential
12	13	20.28	45	Yes	3	Residential
13	14	54.06	42	Yes	3	Residential
14	3	39.48	NA	NA	NA	NA
15	5	26.42	58	NA	NA	NA

* These ponds were dredged in early 2018.

2.2.2 Water Sample Collection

Water samples were collected bi-weekly from both the inflow and outflow locations at each of the 15 SMPs from June to September 2018, and June to September 2019. Three additional sampling dates were included in the fall of 2019 (two dates in October and one date in November) to capture the period of aquatic plant senescence. See Appendix A, Figure A1 for cross section of water sampling locations.

Field parameters measured at the inflow, outflow, and vegetation collection sites included: pH, conductivity, temperature, and dissolved oxygen using a YSI multi-meter probe. Unfortunately, the YSI probe could not be used at the outflow sites for ponds 4, 10 and 14 due to inaccessibility to the SMP outfalls, however, water samples could be collected using a suspended tygon tube and peristaltic pump. At the inflow and outflow locations of each pond, two acid-washed 1-L Nalgene bottles served as technical replicates to store SMP sample water. One of the 1-L bottles was sterile for the collection of coliform samples. All water samples were placed on ice, until further laboratory processing within 24-hrs of collection, but typically on the same day of collection.

Water samples collected in sterile bottles were immediately poured for coliform analysis, using Coliplates™ (Bluewater Biosciences, Mississauga, ON). Following a 24-hour incubation at 37°C, blue-stained wells (indicating coliform presence) were counted. Using a UV-lamp, wells that fluoresced (indicating *E. coli* presence) were also counted. Total coliforms and total *E. coli* concentrations (colony forming units per 100 mL of water sampled) are calculated based on the most probable number (MPN) method. Water samples were also tested for chloride (mg/L), using a Cole-Parmer chloride ion electrode probe (Cole-Parmer, 2019). Chlorophyll α is used as a proxy measure for algal biomass, and was collected by filtering 250 mL of sampled water through 47 mm glass-fibre GFA filters, wrapping in aluminum foil and freezing until extraction. Further extraction was completed using 90% acetone, as described by Su *et al*. (2010). Total suspended solids (g) was measured by filtering 250 mL of collected water samples through pre-weighed dry GFA filters. The filters were then weighed, oven dried at 60°C for 24-hours, and reweighed. Total organic suspended solids (g) was calculated by drying the total suspended solid filters in a muffle furnace at 550°C for 2-hours, and reweighing. Weight by difference was used to calculate both total suspended solids and total organic suspended solids.

Water samples for total phosphorus (µg/L), and total nitrogen suite (ammonia/ammonium, nitrite/nitrate, and total kjeldahl nitrogen) (mg/L), were immediately collected using acid-washed 50 mL Falcon tubes, and then frozen until further analysis. Total dissolved phosphorus samples (µg/L) were collected by filtering water samples through 0.2 µm Nylon membrane filters, and freezing in 50 mL acid-washed Falcon tubes until analysis. Phosphorus samples were measured using methods previously described by Murphy and Riley (1962) and the Ontario Ministry of Environment (1983). Nitrogen suite analysis, including kjeldahl nitrogen (mg/L), ammonia and ammonium (mg/L), nitrite (mg/L), and nitrate (mg/L) were analyzed by an accredited lab (SGS Canada), in Lakefield, Ontario.

2.2.3 Sediment Collection

Sediment samples were collected once from each of the 15 SMPs on August 26 and 28, 2019 to determine pore-water phosphorus concentrations. Samples were collected using a WILDCO 2424-A and 2424-B 20” hand corer. All samples were collected in acid washed sample cups and immediately stored on ice until further analysis. Sediment samples were portioned and centrifuged for 10 minutes to separate pore water from sediment. Separated pore water was isolated and frozen until further analysis. Pore water was later analyzed for total phosphorus using methods previously described above for water column phosphorus measurements.

2.2.4 Data Analysis

All t-tests, one-way analyses of variance, post-hoc tests, correlation analyses, and principal component analyses were completed using RStudio v1.1.463 (RStudio, Boston, USA). All water quality parameters and biological data were non-normal, and thus were transformed to improve normality, when possible. All other parametric assumptions were met, therefore due to the robustness of such a large dataset, parametric tests were used. For multivariate ordination analyses, water quality parameters were center-standardized.

2.3 RESULTS

2.3.1 Assessing changes in water quality between inflow and outflow locations

Welch two sample t-tests were completed to assess differences in water quality variables between sampling locations (Table 2). T-tests were also completed for individual ponds comparing inflow and outflow locations (See Appendix A, Tables A1-A15). Combined, the 15 study ponds do not show any significant decrease in turbidity, total suspended solids, or total organic suspended solids between locations (Table 2). However, the selected ponds show decreasing trends between in and out locations for both chloride (and its proxy conductivity) as well as nitrogen. On the contrary, total phosphorus concentrations tend to increase from inflow to outflow sites. These trends for total phosphorus were further assessed by comparing sediment pore-water phosphorus to water column concentrations. One-way Analysis of Variance and corresponding post-hoc Tukey tests were performed to statistically compare mean total phosphorus levels between sediment pore water, and water at the inflow and outflow locations (Figure 2). Pore-water phosphorus concentrations are significantly higher, compared to inflow and outflow concentrations (Figure 2).

Table 2. Water quality parameters for inflow and outflow locations for all 15 SMPs and combined sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test where significant differences are denoted by $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (abs @ 440 nm) ***	0.005 (0.01)	0	0.034	0.008 (0.004)	0	0.03
Turbidity (abs @ 750 nm)	0.024 (0.13)	0	1.59	0.021 (0.077)	0	1.065
Total Suspended Solids (g/L)	0.0584 (0.46)	0	6.75	0.0331 (0.115)	0	1.6272
Total Organic Suspended Solids (g/L)	0.0188 (0.11)	0	1.625	0.0108 (0.0094)	0	0.1124
Total Coliforms (CFU/100 mL)	55.78 (200.23)	0	2424	47.66 (159.45)	0	1696
Total *E. coli* (CFU/100 mL)	21.97 (53.54)	0	587	19.02 (47.81)	0	375
Chlorophyll α (g/L)	0.0108 (0.04)	0	0.3728	0.0075 (0.01)	0	0.0634
Chloride (mg/L) *	369.5 (253.9)	0	858.8	319.4 (246)	0	900.2
Conductivity (µs/cm) ***	1295.22 (706.26)	21.44	4061	1064.4 (626.38)	89.6	4386
Total Phosphorus (µg/L) **	41.99 (72.17)	0	637.21	60.52 (76.76)	4.67	803.75
Total Dissolved Phosphorus (µg/L)	8.54 (32.96)	0	501.39	11.23 (40.07)	0	576.11
Total Kjeldahl Nitrogen (mg/L) ***	0.19 (0.24)	0	1.31	0.33 (0.37)	0	3.42
Ammonia + Ammonium (mg/L) **	0.078 (0.12)	0	1.0	0.123 (0.238)	0	2.2
Nitrite (mg/L)	0.053 (0.58)	0	8.91	0.012 (0.017)	0	0.096
Nitrate (mg/L) ***	1.3 (1.33)	0	8.23	0.52 (0.76)	0	5.01
Total Nitrogen (mg/L) ***	1.55 (1.39)	0.07	11.48	0.85 (0.75)	0.07	5.01
Temperature (°c) **	18.36 (3.88)	10.5	28.5	19.52 (3.51)	11.2	27.5
Dissolved Oxygen (mg/L) ***	8.59 (2.27)	8.77	10.15	7.84 (2.26)	0.87	17.11
pH	7.79 (0.42)	6.78	9.58	7.76 (0.41)	6.87	9.53

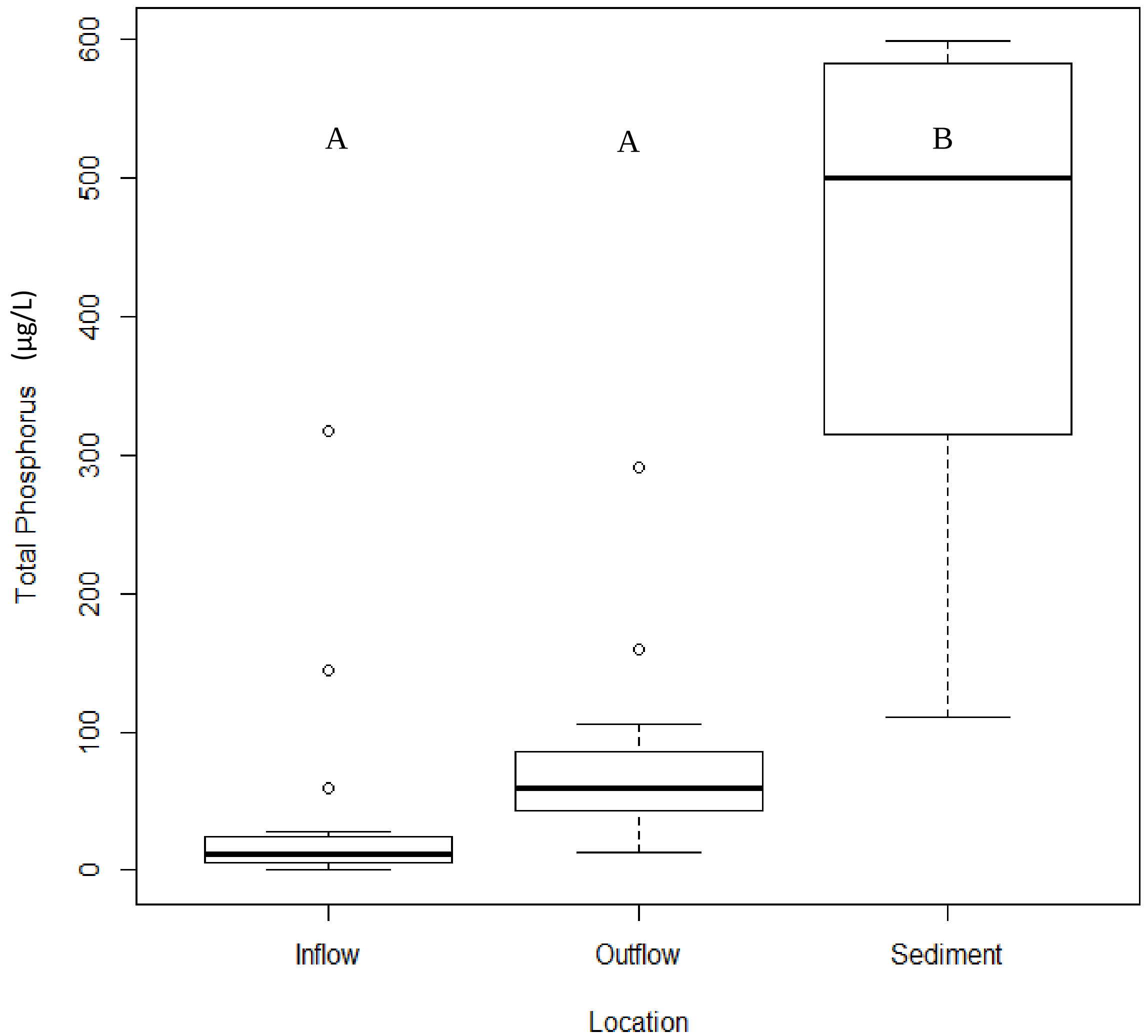

Figure 2. Total phosphorus concentrations from 15 SMPs collected from three different locations (inflow, outflow and sediment pore-water) in August of 2019. One-way Analysis of Variance with post-hoc Tukey Test, where inflow and outflow (A) are significantly different from sediment pore-water (B) phosphorus concentrations ($p < 0.001$).

Relationships between various water quality parameters was assessed to determine potential trends occurring at in and out locations. In this way, significant relationships between water quality variables may highlight similar sources (i.e. from the landscape), or similar removal processes within SMPs. Furthermore, significant relationships may indicate important interactions between water quality variables. Differences between inflow and outflow locations are illustrated using Pearson correlations between water quality variables (Figures 3, 4, 5, 6). Pearson correlations were performed for inflow water quality parameters (Figure 3). Due to missing YSI-probe field data at some sites because of issues of probe access, inflow field data (pH, temperature, conductivity, dissolved oxygen) analyses (Figure 4) were kept separate from the larger and more complete dataset for water sample parameters (colour, turbidity, total suspended solids, chloride, chlorophyll α, total phosphorus, total dissolved phosphorus, total nitrogen, coliforms) (Figure 3). At the inflow location, nutrients show variable relationships with other water quality parameters (Figure 3). Total nitrogen is positively correlated with incoming chloride concentrations, however is significantly associated with decreasing phosphorus and total dissolved phosphorus at inflow sites. Total phosphorus on the other hand is positively associated with increased chlorophyll α concentrations (i.e. phytoplankton biomass), and suspended solids from incoming stormwater. Interestingly, suspended solids are also positively associated with water turbidity, but show no significant relationship with phytoplankton concentrations. Significant relationships were also found for inflow YSI parameters (Figure 4). Notably, incoming surface water temperature is correlated with increasing pH, as well as decreasing conductivity and dissolved oxygen levels.

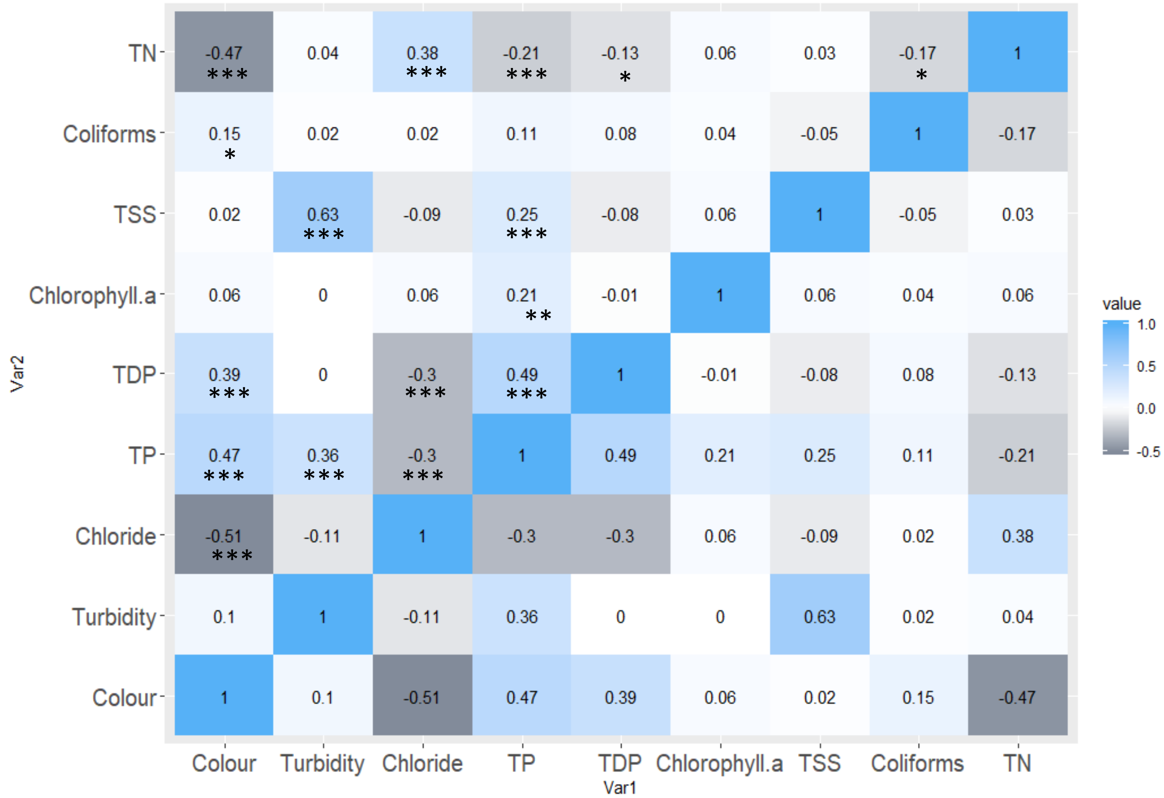

Figure 3. Pearson correlation for **inflow** water quality parameters. Water quality parameters include all dates (except Fall 2019) for both 2018 and 2019. Significant relationships are denoted with p < 0.05 *, p < 0.01 **, p < 0.001 ***.

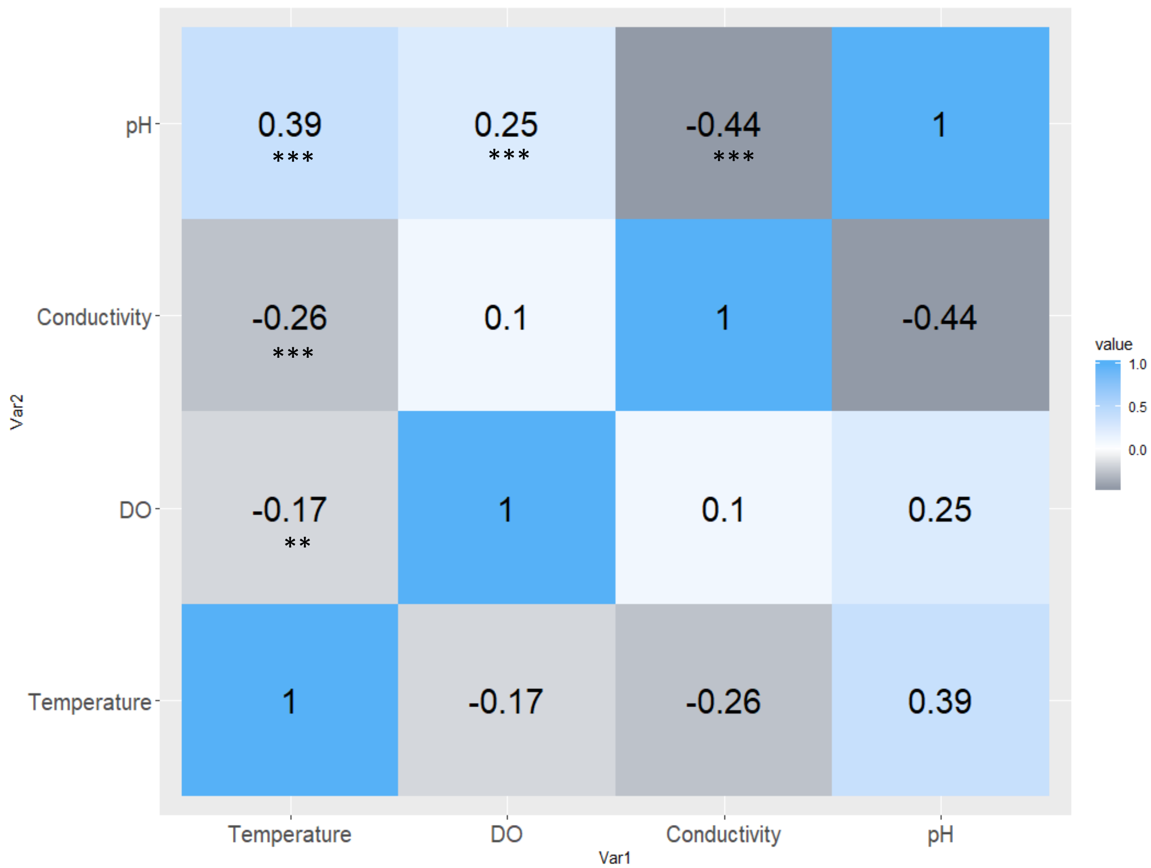

Figure 4. Pearson correlation for **inflow** water conditions. Water parameters include all dates (except Fall 2019) for both 2018 and 2019. Significant relationships denoted are with p < 0.05 *, p < 0.01 **, p < 0.001 ***.

Pearson correlations were also performed for outflow water quality parameters (Figure 5). Similar to the issues described above for inflow on-site YSI readings, YSI field data analysis was done separately from water quality parameters measured with water samples (Figure 6). Trends for nutrient concentrations differ at the outflow location, compared to inflow sites (Figure 5). In this case, outgoing nitrogen concentrations are significantly related to increased phosphorus, as well as turbidity, and chloride concentrations. Interestingly, total phosphorus concentrations at the outflow sites is positively correlated with outgoing suspended solids, and chlorophyll α levels. Turbidity and suspended solids at the outflow locations remains positively correlated with one another, however in this case, suspended solids are also significantly related to increasing phytoplankton biomass. Significant relationships are also found for outflow YSI parameters (Figure 6). Notably, outflow water temperature is correlated with increasing pH, as well as decreasing dissolved oxygen levels.

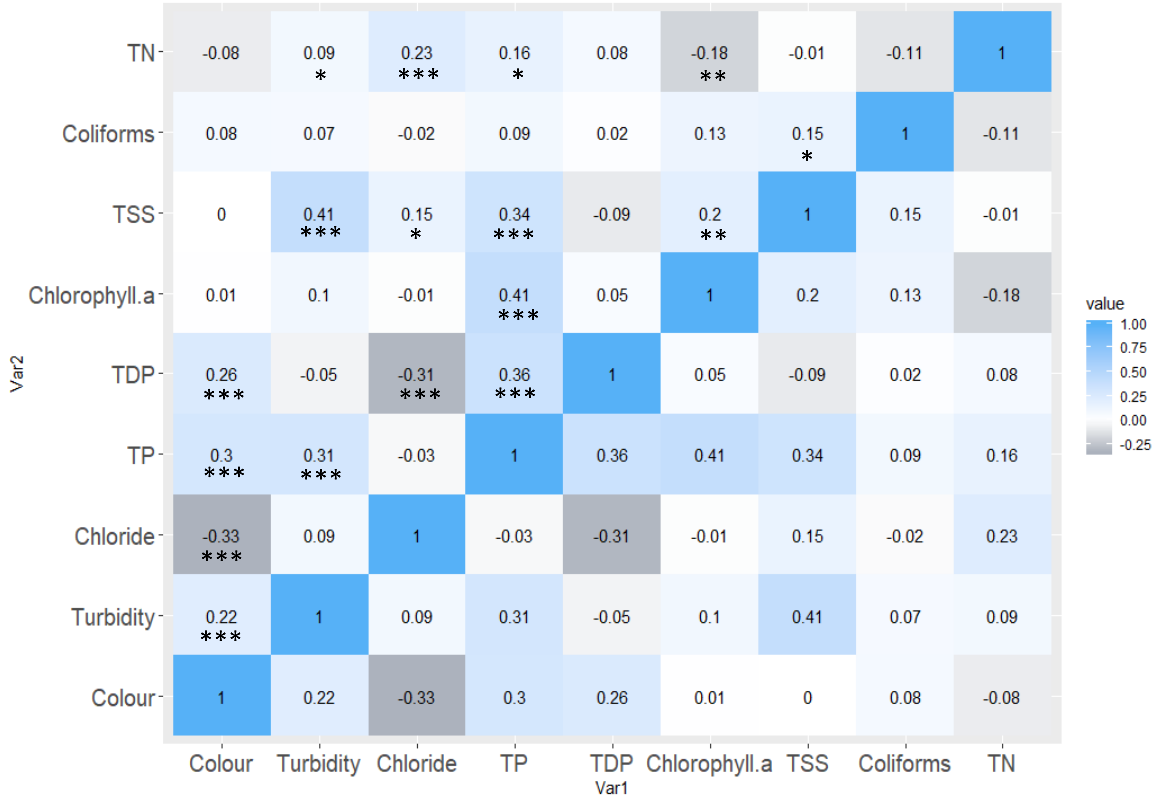

Figure 5. Pearson correlation for **outflow** water quality parameters. Water quality parameters include all dates (except Fall 2019) for both 2018 and 2019. Significant relationships are denoted with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

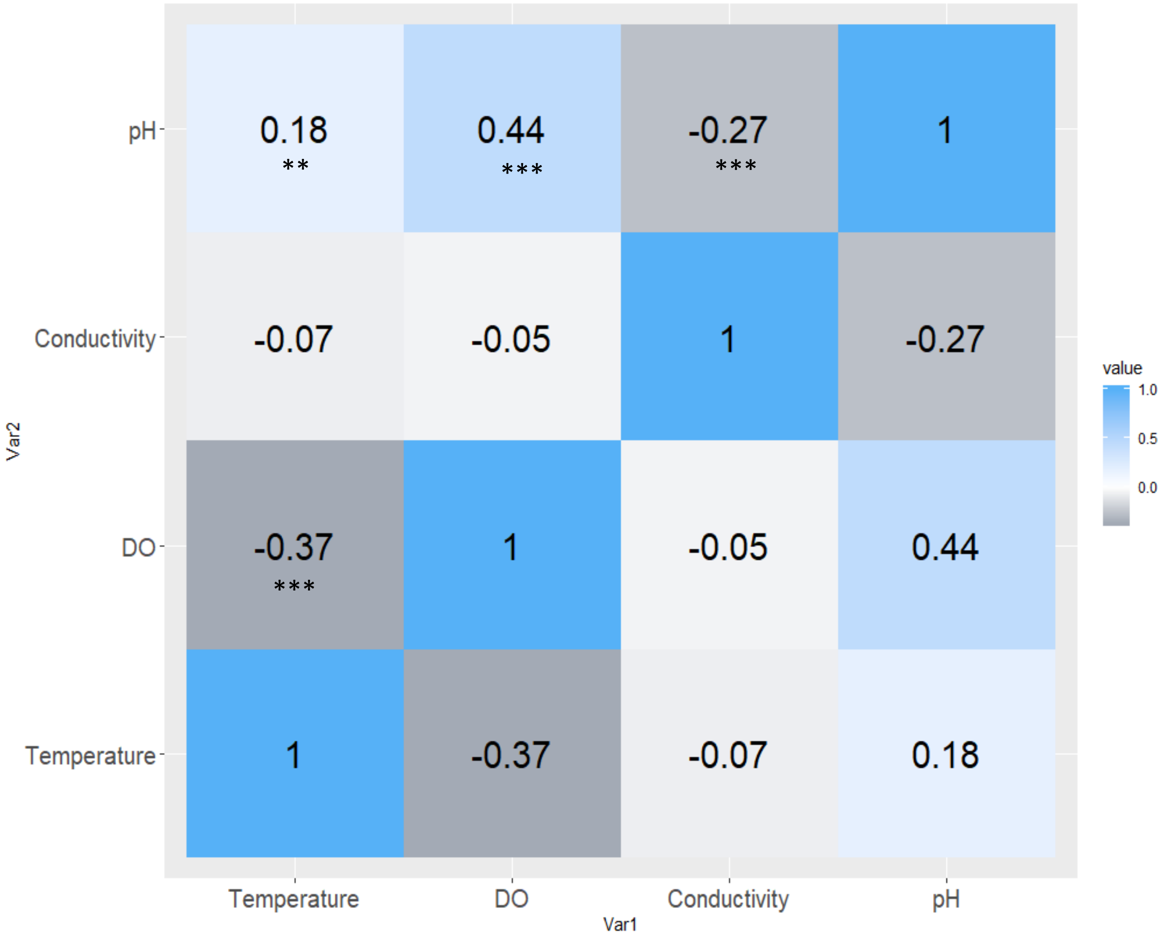

Figure 6. Pearson correlation for **outflow** water conditions. Water parameters include all dates (except Fall 2019) for both 2018 and 2019. Significant relationships are denoted with p < 0.05 *, p < 0.01 **, p < 0.001 ***.

A principal component analysis was completed for inflow water quality parameters for all sampled dates, excluding fall of 2019 (Figure 7). A gradient along PC1 axis separates ponds with high chloride, conductivity, and total nitrogen, from ponds with high total phosphorus, turbidity (TSS), and temperature. There are noticeable outliers at the inflow locations, specifically pond 15, which has exceptionally high phosphorus and chloride concentrations for some sampling dates. With the exception of occasional outliers, it should be noted that there is no remarkable variation in water quality across inflow locations for all 15 study sites. This is made apparent by the clustering of samples located along the PC1 axis gradient.

A principal component analysis was also completed for outflow water quality parameters including all sampling dates except fall 2019 (Figure 8). There appears to be greater variation in the quality of water at the outflow sites across the 15 study ponds. Notably, extreme values are less frequent, with the exception of pond 10 which has high phosphorus levels from one of the sampling dates. Patterns of outflow water quality across the two-year study period can be easily noted for individual ponds. Pond 15, for example, has outflow water quality readings that cluster closely across the duration of the study. Pond 15 had particularly high chloride, and phosphorus levels at the inflow site (Figure 7), however shows opposite trends at the outflow site (Figure 8). In this case, pond 15 shows relatively low phosphorus concentrations and decreased suspended particulates at the outfall compared to the other studied ponds.

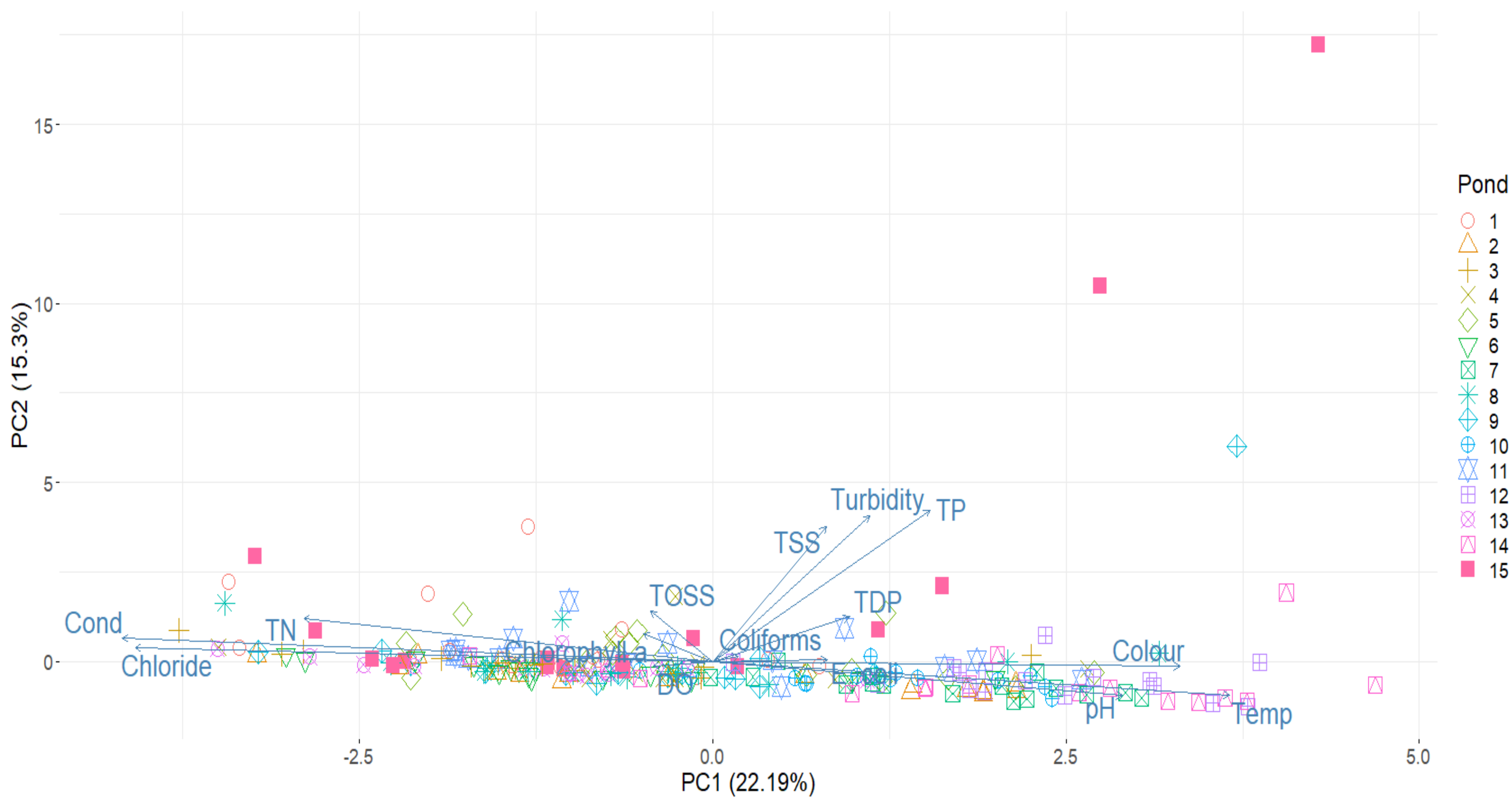

Figure 7. Principal component analysis for **inflow** water quality variables. Water quality parameters include all dates collected in 2018 and 2019 (except fall 2019). PC1 represents 22.19% of the variability and PC2 represents 15.3% of the variability.

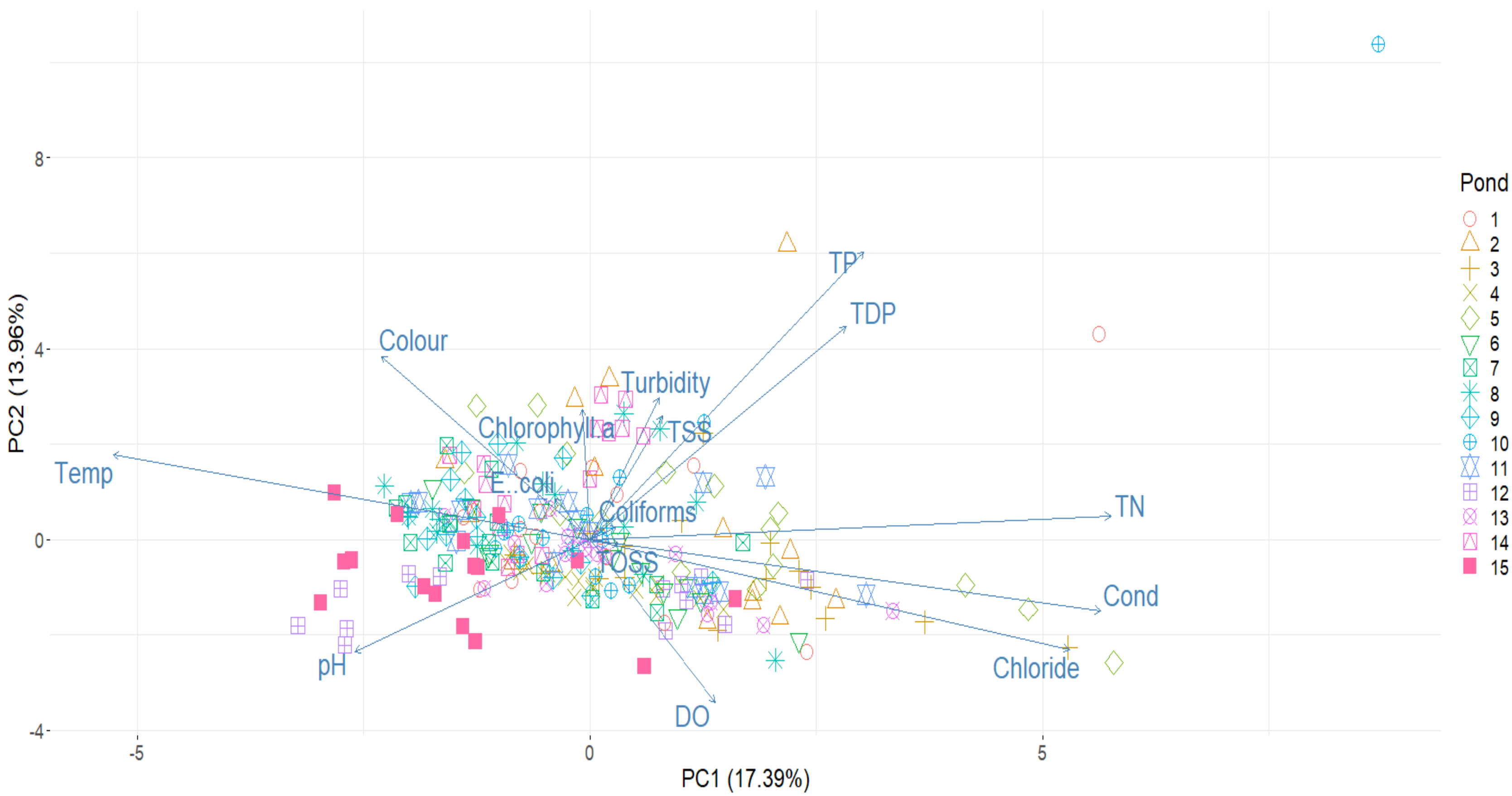

Figure 8. Principal component analysis for **outflow** water quality variables. Water quality parameters include all dates collected in 2018 and 2019 (except fall 2019). PC1 represents 17.39% of the variability and PC2 represents 13.96% of the variability.

2.3.2 Effect of seasonality on water quality between inflow and outflow locations

Due to the duration of the study spanning multiple months and seasons, it is important to visualize differences in sampling dates across the two-year study period. A principal component analysis was completed for inflow (Figure 9) and outflow (Figure 10) locations for all sampling dates in 2018 and 2019 (including fall 2019). There is limited variation in terms of water quality changes at inflow locations across sampling seasons (Figure 9). The majority of sites cluster along a gradient represented by chloride, conductivity, and nitrogen at one end, and temperature at the other. However, patterns of seasonality at the outflow locations are evident (Figure 10). It appears that most June and Fall (October and November) dates tend to cluster in similar areas, with characteristically high dissolved oxygen, conductivity, and chloride concentrations. The remaining summer months, July, August, and September tend to show lower salt and oxygen concentrations but are higher in nutrients, suspended solids, and algal biomass. Seasonal trends between inflow and outflow locations are further illustrated using line graphs by year and month sampled (see Appendix A: Figures A7-A12).

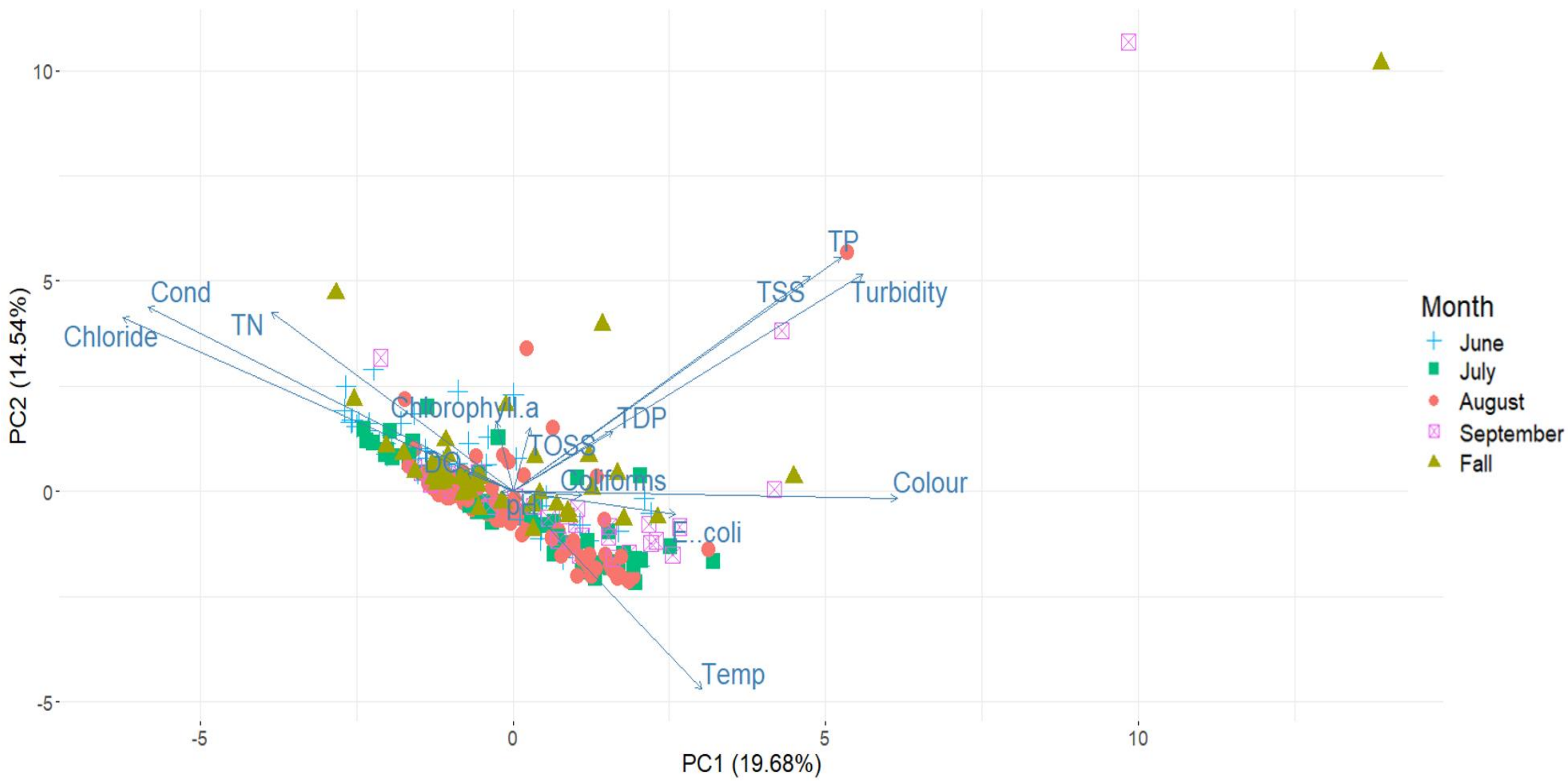

Figure 9. Principal component analysis for **inflow** water quality variables. Water quality parameters include all sampling dates from 2018 and 2019. PC1 represents 19.68% of the variability and PC2 represents 14.54% of the variability.

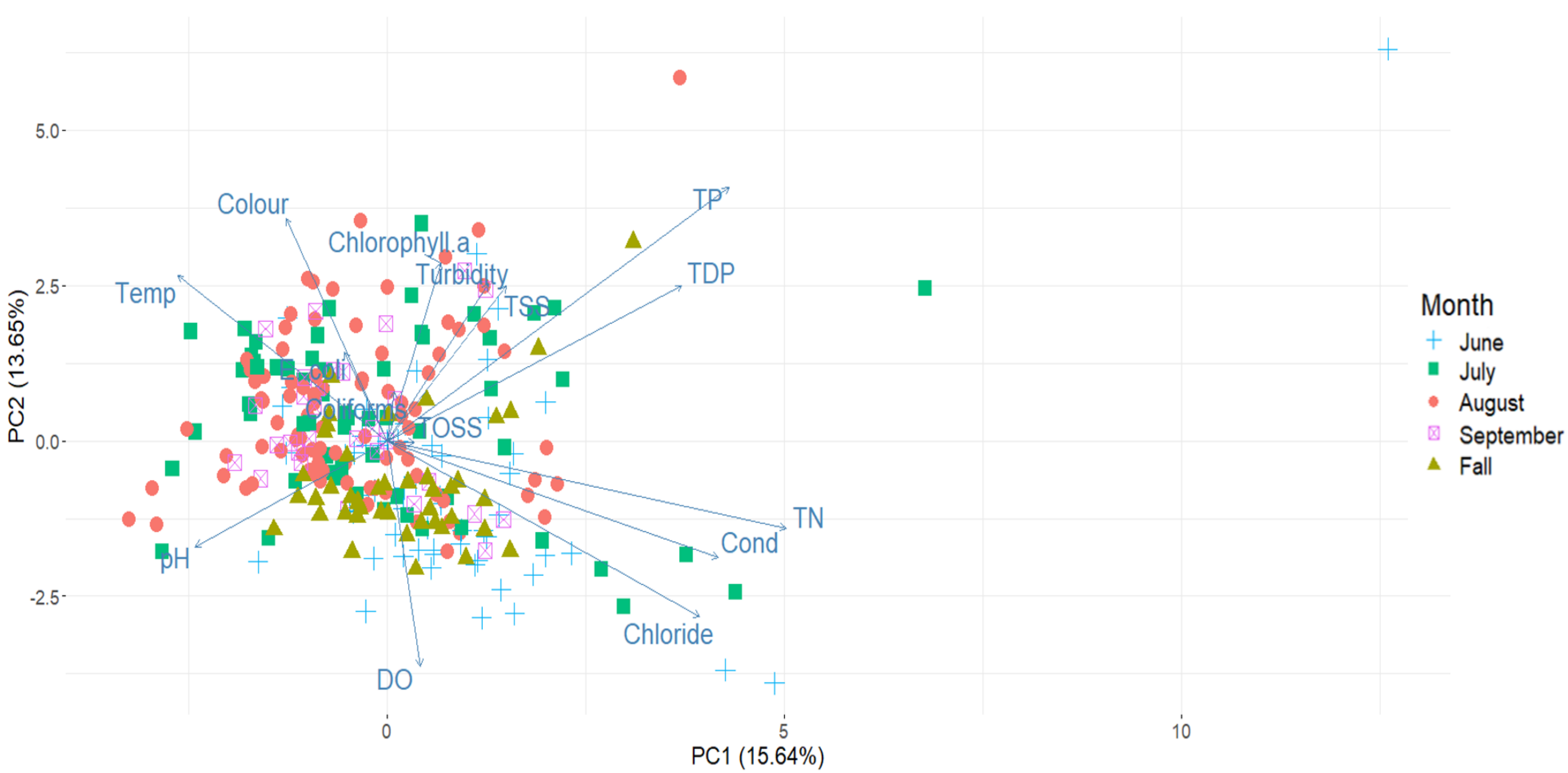

Figure 10. Principal component analysis for **outflow** water quality variables. Water quality parameters include all sampling dates from 2018 and 2019. PC1 represents 15.64% of the variability and PC2 represents 13.65% of the variability.

2.3.3 Effect of pond characteristics on water quality between inflow and outflow locations

Pond design and characteristics of the surrounding SMP landscape can be critical in defining water quality and water treatment processes within these systems. In order to highlight potential relationships between location specific water quality parameters and pond design traits, correlation analysis was used. Pearson correlation analysis was completed for inflow water quality variables and pond characteristics (Figure 11). For inflow water quality parameters, focus was placed on surrounding pond characteristics, which influence the quality of stormwater runoff. It was noted that as surrounding drainage area increases, there is a significant increase in total phosphorus, and significant decrease in total nitrogen concentrations. Looking at specific impervious surface levels within pond catchment areas reveals that with increasing imperviousness, there is a significant increase in turbidity, and suspended solids at the inflow location. However, with increasing impervious surfaces, there is also a significant decrease in total dissolved phosphorus concentrations.

A Pearson correlation analysis was also completed for outflow water quality variables and pond characteristics (Figure 12). For outflow water quality parameters, focus was placed on specific pond design characteristics, which may influence the quality of water leaving the facility. In this case, pond size (area, length, width, parameter) are all positively associated with total phosphorus and chlorophyll α concentrations. Therefore, as ponds increase in size, their outgoing phosphorus and algal concentrations may also increase. Interestingly, the opposite trend is seen with nitrogen, whereby as pond size increases there is a significant decrease in total nitrogen concentrations. The selected

SMPs vary greatly in age, however, only outflow chloride concentrations seem to be positively associated with increasing pond age.

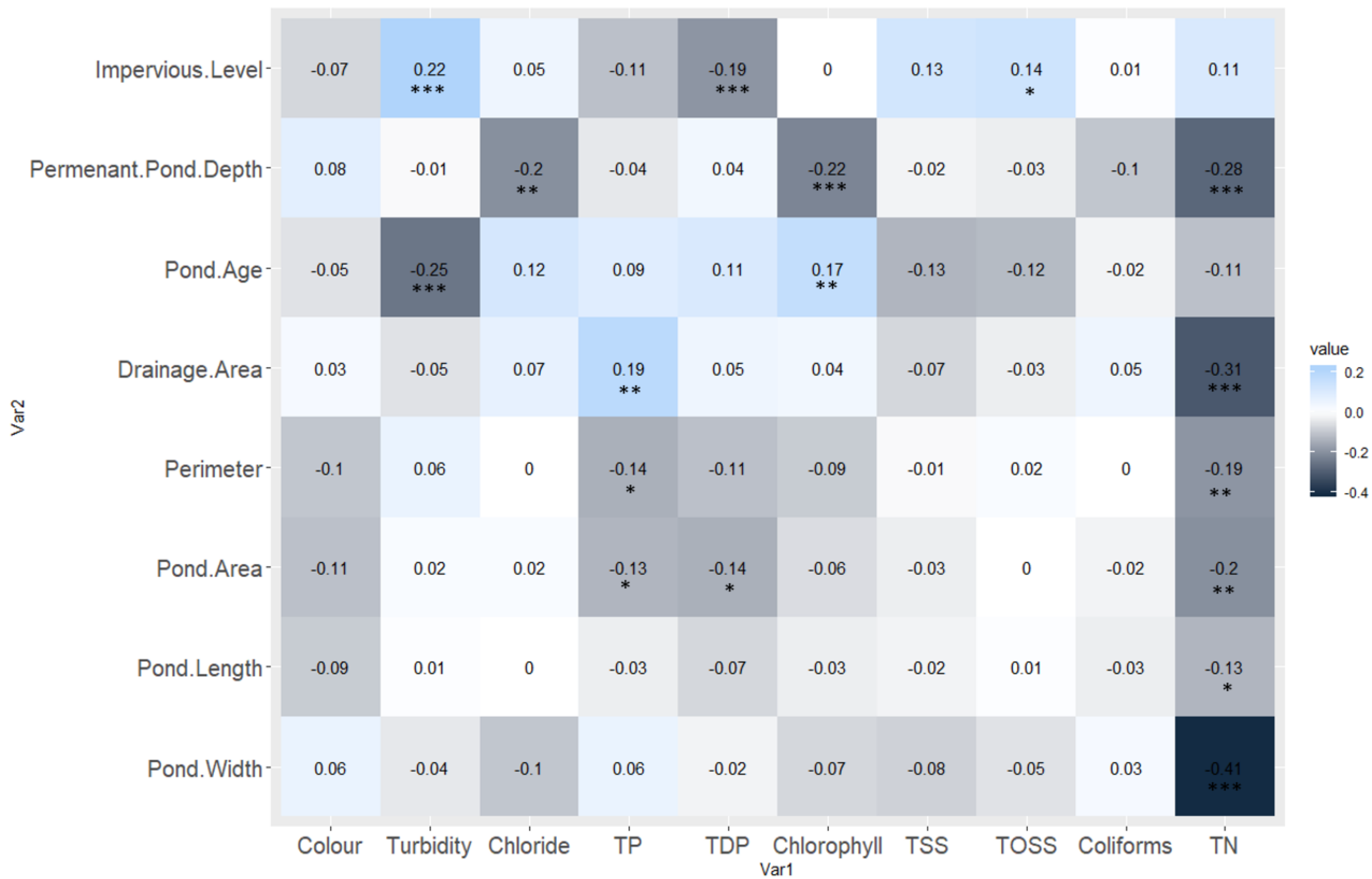

Figure 11. Pearson correlation for **inflow** water quality parameters and pond characteristics. Water quality parameters include all dates for 2018 and 2019 (except Fall 2019). Due to missing information only 7 SMPs were included for impervious surface levels, and only 13 SMPs were included for pond depth. Significant relationships are denoted with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

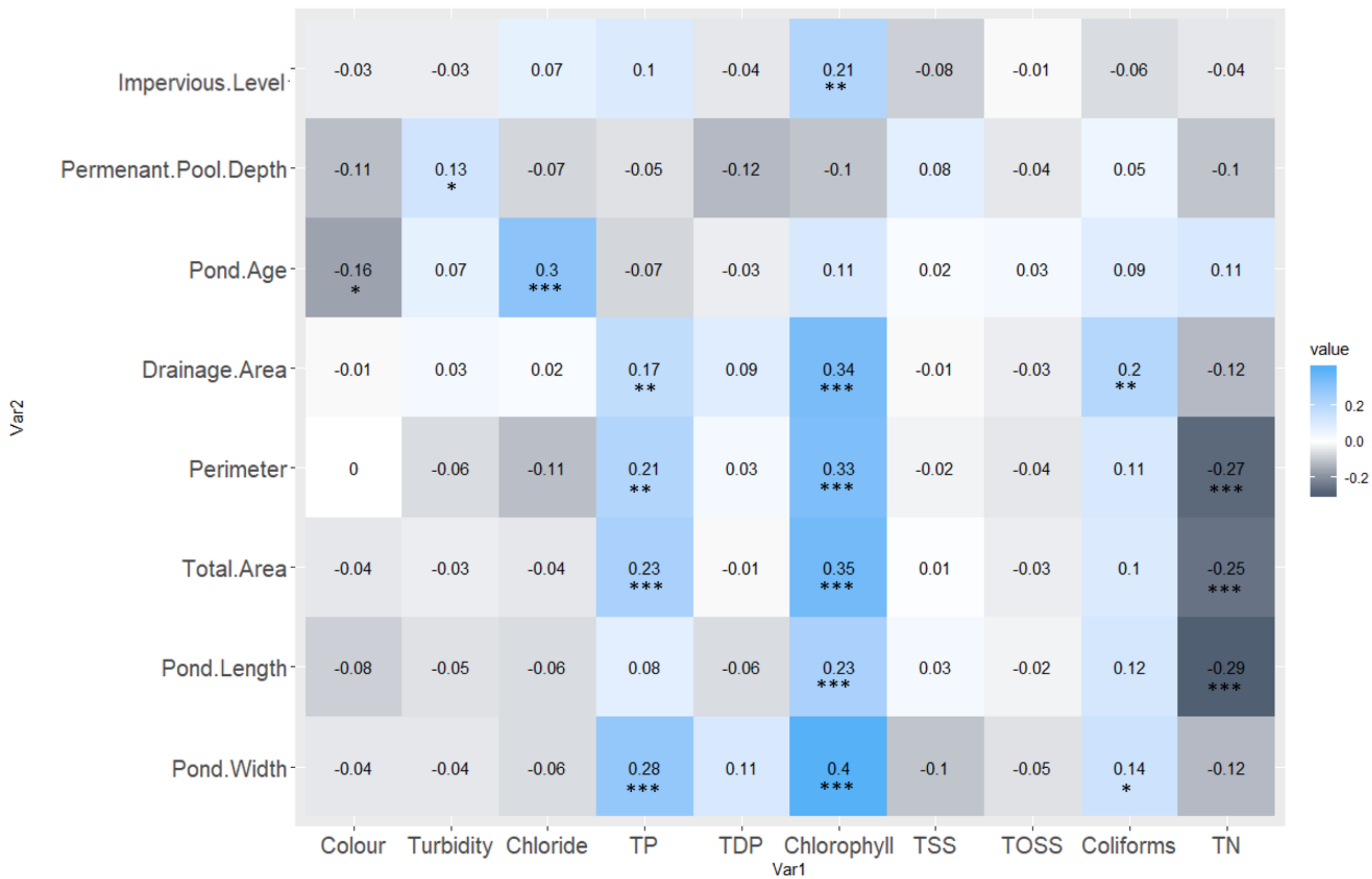

Figure 12. Pearson correlation for **outflow** water quality parameters and pond characteristics. Water quality parameters include all dates for 2018 and 2019 (except Fall 2019). Due to missing information only 7 SMPs were included for impervious surface levels, and only 13 SMPs were included for pond depth. Significant relationships are denoted with p < 0.05 *, p < 0.01 **, p < 0.001 ***.

2.3.4 Effect of sediment removal via maintenance dredging on outflow water quality

A Welch two sampled t-test was used to address the effects of sediment removal at both the inflow and outflow locations (Figure 13, 14, 15, 16). For this analysis three dredged (ponds 4, 6, 11) were compared to three undredged ponds (ponds 2, 3, 9). The selected undredged ponds have characteristically high emergent and submergent cover, and are candidates for the 'to be dredged' list. During the process of dredging, all sediment and vegetation within the pond is mechanically removed. This process is used to maximize the water holding capacity of the pond. There are no significant differences between dredged and undredged ponds in terms of their water quality parameters at the inflow location (Figure 13, 14, 15, 16). However, there are significant differences in water quality at outflow locations between ponds with different maintenance histories. Both turbidity and total suspended solids are significantly different between dredged and undredged ponds (Figures 13, 14). In both cases, pond clarity is significantly higher for ponds that have undergone dredging. In terms of nutrients, total dissolved phosphorus is higher for dredged ponds at the outflow site, however total nitrogen is lower for dredged ponds at outflow locations (Figure 14, 15). Seasonal trends between dredged and undredged ponds at the outflow location are further illustrated using line graphs by month sampled (see Appendix A: Figures A13-A15).

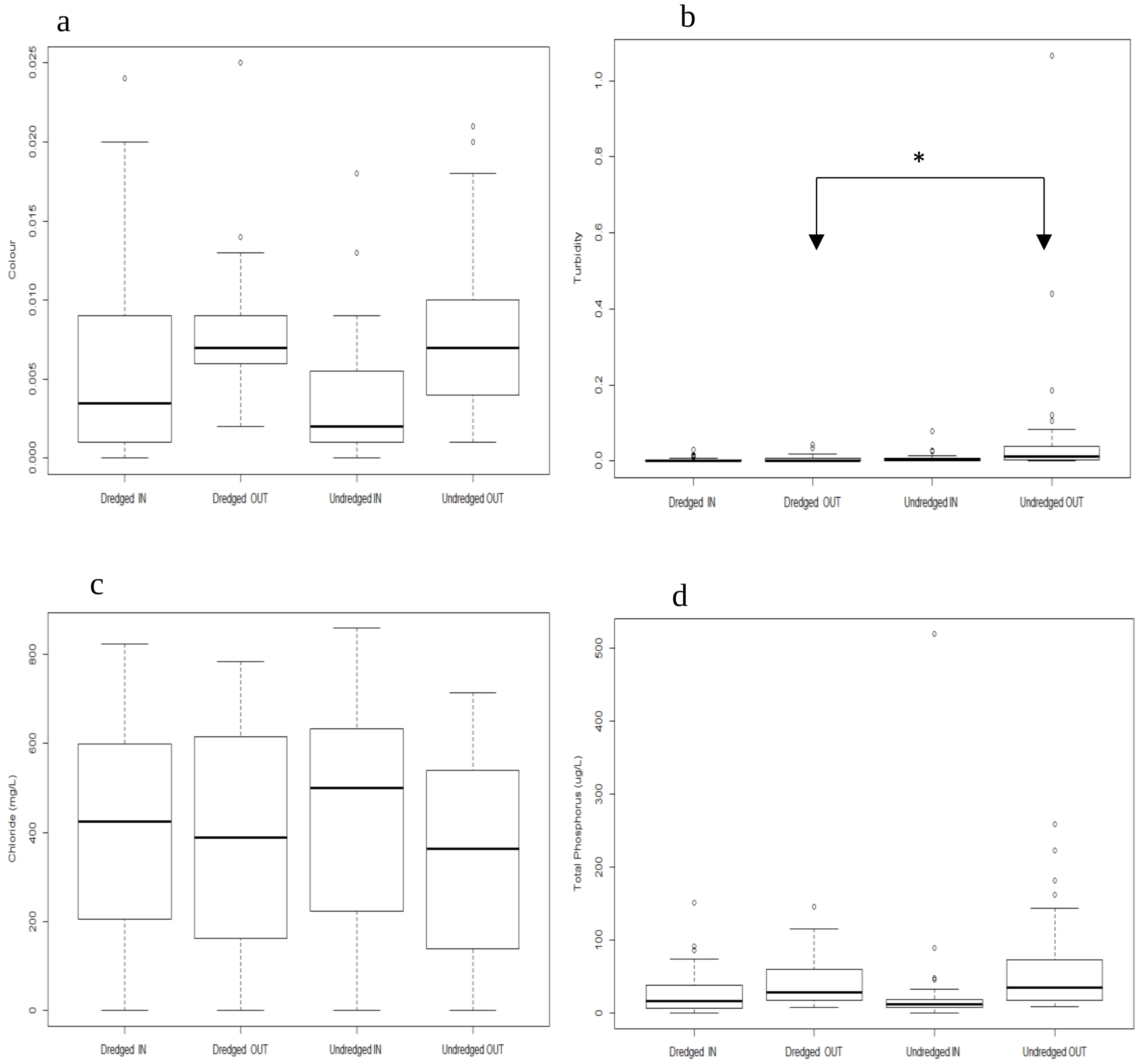

Figure 13. Colour (a), turbidity (b), chloride (c), and total phosphorus (d) for three dredged (ponds 4, 6, 11) and three undredged (ponds 2, 3, 9) ponds divided by location (inflow and outflow) for all sampled dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

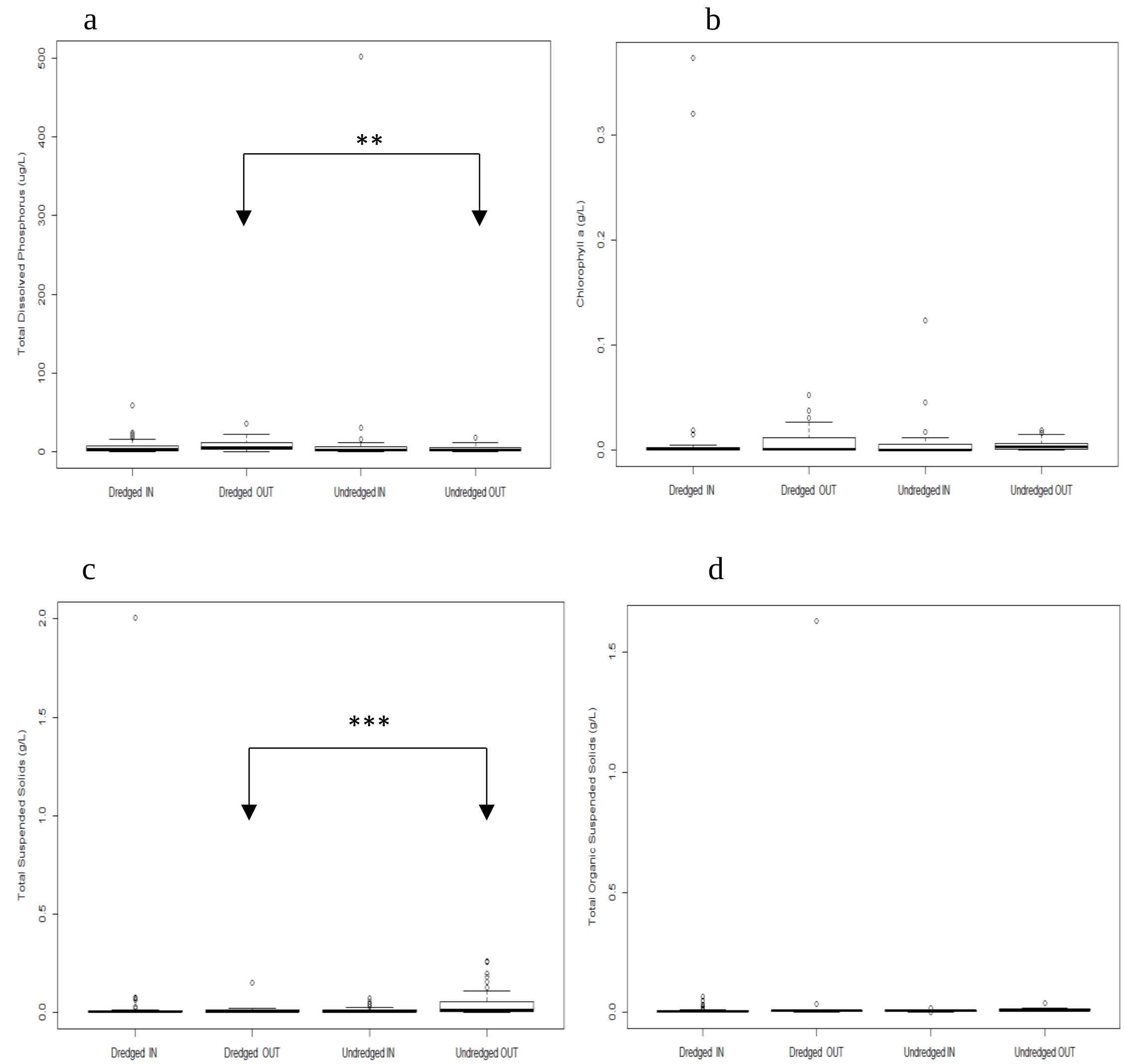

Figure 14. Total dissolved phosphorus (a), chlorophyll α (b), total suspended solids (c), and total organic suspended solids (d) for three dredged (ponds 4, 6, 11) and three undredged (ponds 2, 3, 9) ponds divided by location (inflow and outflow) for all sampled dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

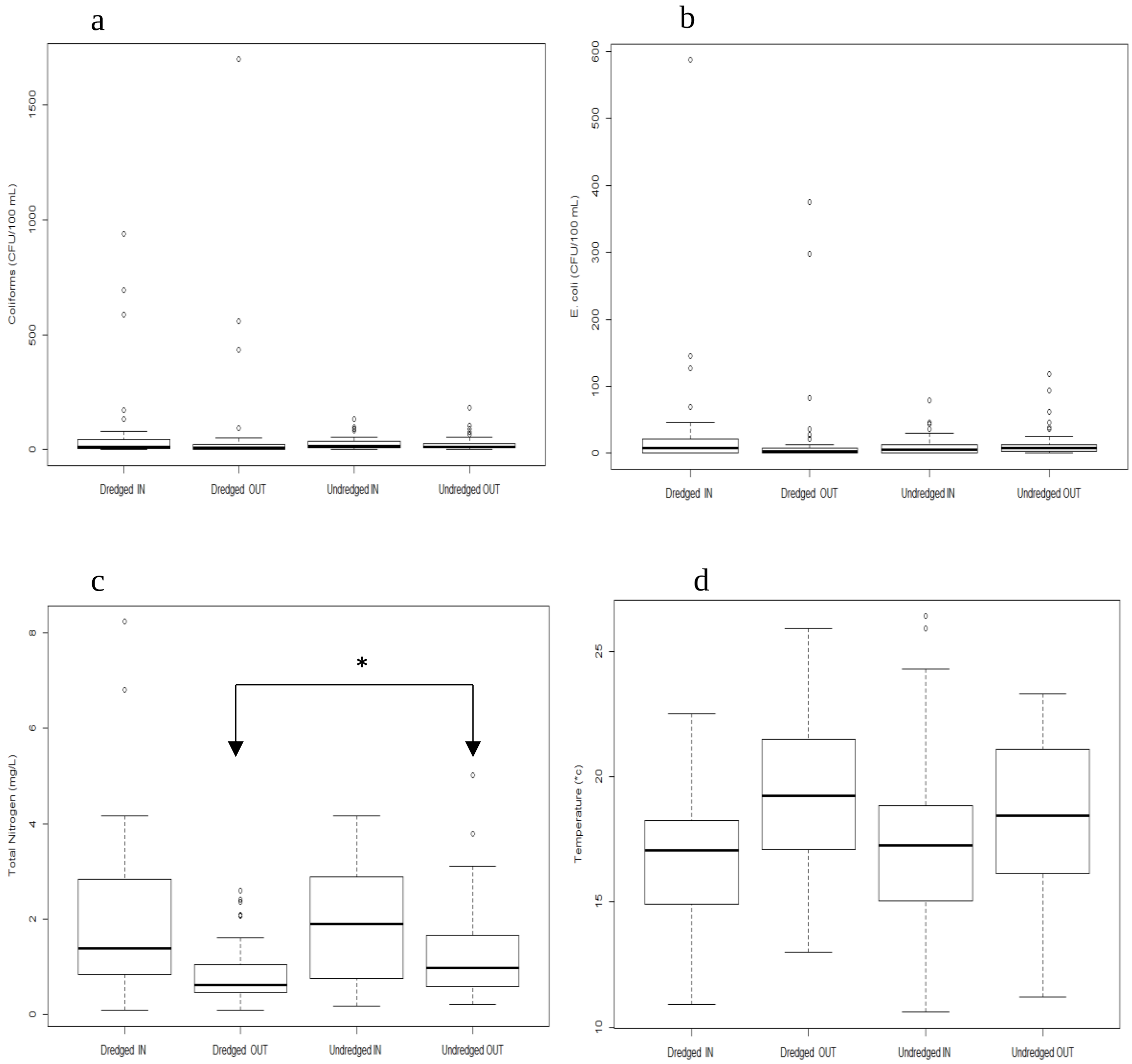

Figure 15. Coliforms (a), total *E. coli* (b), total nitrogen (c), and temperature (d) for three dredged (ponds 4, 6, 11) and three undredged (ponds 2, 3, 9) ponds divided by location (inflow and outflow) for all sampled dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

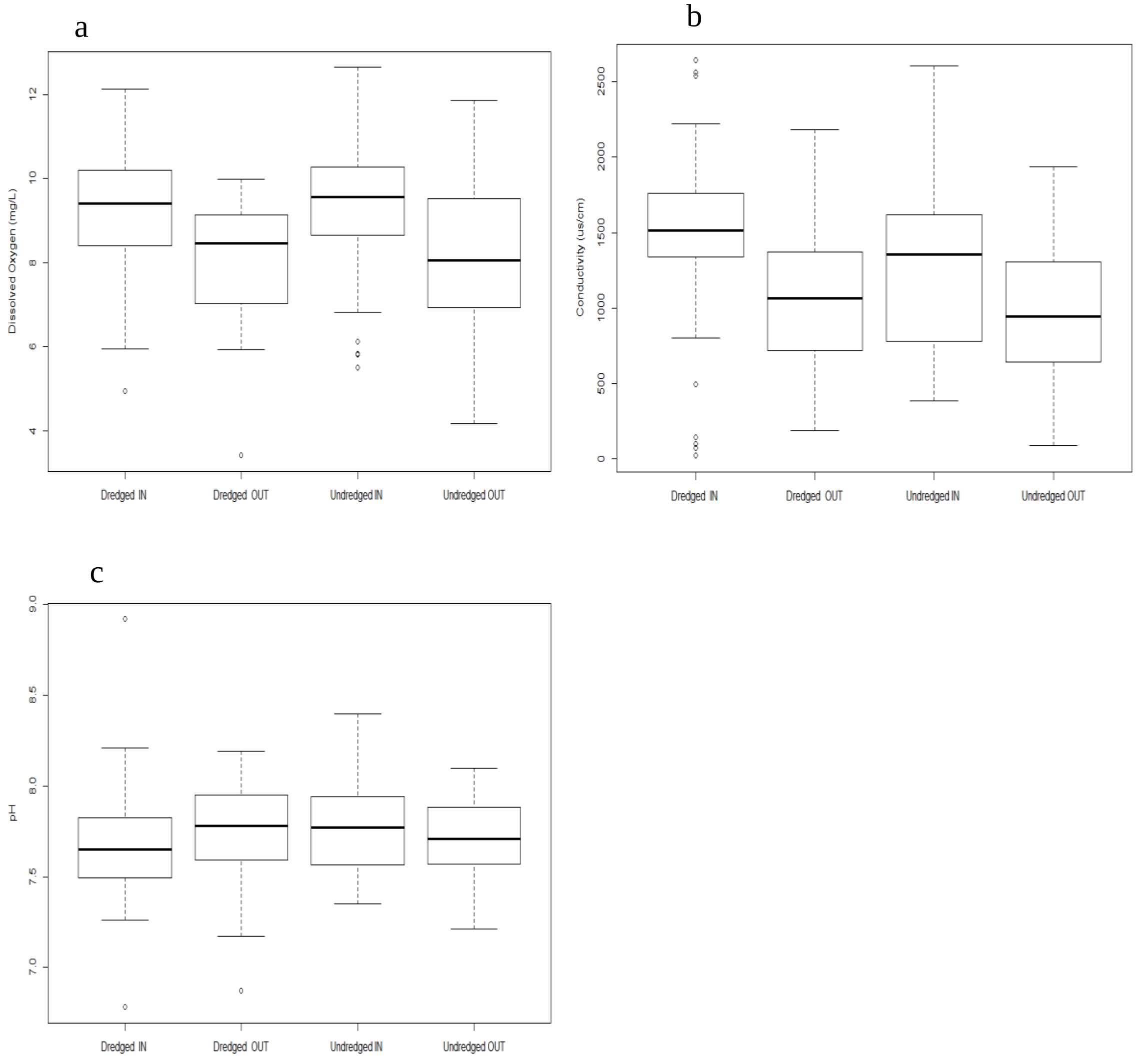

Figure 16. Dissolved oxygen (a), conductivity (b), and pH (c) for three dredged (ponds 4, 6, 11) and three undredged (ponds 2, 3, 9) ponds divided by location (inflow and outflow) for all sampled dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

2.4 DISCUSSION

2.4.1 Water quality differences between inflow and outflow locations

The main design function of SMPs is to reduce water velocity and remove suspended particulates from incoming stormwater (Casey et al., 2006; Tixier et al., 2011). In this way, stormwater suspended solids and turbidity can be greatly reduced, and any adsorbed contaminants are sequestered within the sediments thereby limiting exposure to natural systems (Wu et al., 1996; Gallagher et al., 2011). However, the selected study ponds show no significant differences in terms of their water clarity between inflow and outflow locations (Table 2). This is especially concerning in developing landscapes where inflow locations are highly contaminated with runoff from construction sites. These results may suggest that water clarity at inflow and outflow sites is dictated by different sources. Incoming stormwater clarity tends to be defined by suspended particulates and debris from the surrounding landscape. However, outflow water clarity may instead be defined by algal biomass. This is consistent with outflow water parameters, in which chlorophyll α concentrations are positively associated with total suspended solids (Figure 5). However, there was no significant difference found between inflow and outflow chlorophyll α concentrations (Table 2), highlighting the potential for other factors to drive water clarity.

This lack of water clarity improvement from in to out locations may also be the consequence of limited sediment maintenance via dredging. Comparing the three selected dredged ponds (ponds 4, 6, 11) and three undredged ponds (ponds 2, 3, 9), significant differences are noted in terms of their ability to improve water clarity (Figures 13, 14).

This may suggest that regular sediment maintenance of Oshawa SMPs will not only retain pond depth, but also ensure their ability to remove suspended solids is kept.

Along with sediment settling, the ability of these ponds to act as sinks and sequester incoming contaminants from the landscape is an important function (Olding et al., 2004; Frost et al., 2015). Due to the variety of contaminants being washed into SMPs, there is also a variety of removal processes that can occur. It appears that the selected SMPs do in fact remove both salt and nitrogen from inflow water, thereby limiting its release at outflow locations (Table 2). Nitrogen in stormwater can be found in a variety of forms, including inorganic (ammonium, nitrite, and nitrate), dissolved organic, and particulate organic. The removal of nitrogen from these ponds may be the result of a variety of processes, including assimilation, adsorption, and denitrification (Collins et al., 2010). Through the process of assimilation, inorganic nitrogen is transformed into microbial or plant biomass as a temporary storage of organic nitrogen (Collins et al., 2010). Through adsorption processes, ammonium-nitrogen (NH_4^+) can also be temporarily removed when it becomes attached to negatively charged sediment particles (Collins et al., 2010). Denitrification on the other hand, can permanently remove nitrogen from a system by transforming it into nitrogen gas, which is directly released into the atmosphere (Collins et al., 2010). This process can occur in stormwater systems, however specific anoxic conditions must be met (Collins et al., 2010). All of these processes may contribute to the decreasing nitrogen concentrations witnessed between SMP inflow and outflow sites.

Another major contributor to urban water pollution is road salt. Salt application in North America is widespread over the winter months in temperate regions. However, excess road salt can be washed off impervious surfaces directly into natural systems (Tiwari & Rachlin, 2018). Since SMPs act as an intermediary between surface runoff and natural freshwater systems, they receive the burden of high salt concentrations. It has been noted that chloride ions are in fact a major contributor to aquatic toxicity from road salt applications (Gallagher et al., 2011). Other cation species associated with road salt, including calcium and potassium, can also result in the mobilization of other toxicants (Gallagher et al., 2011). Therefore, the ability of SMPs to remove salt from incoming surface water prior to its release into natural systems is essential for the well-being of downstream aquatic communities.

Chloride ions have been noted to be conserved in freshwater environments (Tiwari & Rachlin, 2018). However, both chloride and sodium ions have also been noted to form complexes with heavy metals, resulting in the accumulation and precipitation of salt ions into pond sediments (Tiwari & Rachlin, 2018). The selected SMPs do in fact show the ability to remove chloride from surface waters (Table 2), reducing the risk of creating toxic freshwater environments. However, it should be noted that pond age is positively related to outgoing chloride concentrations (Figure 12). The age of SMPs can dictate not only the types of plants established, but also sediment chemistry and volume (Egemose et al., 2015). In this way, older ponds may not be sufficient in sequestering salt from incoming stormwater, compared to newer SMPs. This may suggest that as chloride builds up over time, excess or “legacy” chloride can leak out of the SMP via the pond outfall. This may in part be due to limited maintenance of older ponds (with built up sediment levels), but may also be reflective of surrounding salt use. Many older ponds are found in well-established residential/commercial areas that may utilize larger volumes of road salt, compared to newer developments.

What is of particular concern in the study SMPs is their apparent role as phosphorus point-sources to receiving waters, rather than serving as a phosphorus sink (Table 2). In general, permanent pool depths of SMPs are designed to be between 1-2 m (Government of Ontario, 2003). Due to their shallow depths and relatively small sizes, SMPs were therefore thought to be well mixed ecosystems (Song et al., 2013). This mixing ensures minimal thermal stratification is occurring, and therefore reduced anoxic conditions in the sediment. However, studies show that due to seasonal patterns and local weather conditions, SMPs can undergo long periods of stratification and therefore result in anoxic conditions at the sediment-water interface (Song et al., 2013; Chiandet & Xenopoulos, 2016). In this case, the ponds become stratified during the summer months resulting in cold water (which is denser) being trapped on the bottom. This cold water holds more oxygen, which is slowly depleted throughout the summer due to the decomposition of organic matter. As a result, these anoxic conditions can force phosphorus to be released from pond sediment, ultimately "loading" it into the water column.

Repeated patterns of stratification followed by mixing can lead to the release of nutrients previously stored in the sediments of SMPs, potentially affecting downstream aquatic communities and water quality (Song et al., 2013). The selected SMPs in this study show statistically significant increases in total phosphorus concentrations from inflow to outflow locations (Table 2). As a result, the water leaving these ponds is high in dissolved and particulate phosphorus, which would contribute to excess algal growth downstream. The eutrophication of freshwater systems can lead to dramatic changes in community structure and composition (Smith et al., 2006). Nutrient enrichment tends to cause a shift from macrophyte dominated to algae dominated communities, including harmful cyanobacteria (Smith et al., 2006). These conditions result in limited resources for native fish and macroinvertebrate communities. In many cases, freshwater environments maintaining a prolonged eutrophic state will result in mortality of aquatic organisms due to decreased habitat (i.e. aquatic plants), edible algae, and oxygen concentrations. Therefore, it is especially concerning that the selected study ponds show potential to function as sources of phosphorus to natural systems.

Interestingly, there was no significant difference between coliform or *E. coli* levels at inflow or outflow locations (Table 2). The presence of microbial pathogens in surface waters poses a serious threat to both water quality and human health. In urban ponds, major sources of coliforms include fecal matter from wildlife and pets, which is washed off the landscape into SMPs (Beutel & Larson, 2015). However, what is of particular concern is that coliform and *E. coli* levels in the studied ponds does not decrease between in and out locations (Table 2). Following a storm event, pond retention times will vary between 24-48 hours, however during baseflow conditions, water can remain within the pond for extended periods of time (days to weeks). Extended residence times, would likely result in the mortality of fecal bacteria washed into SMPs from the surrounding landscape. However, it has been highlighted that one of the key limitations of urban ponds is their inability to consistently remove pathogens prior to surface water entering naturalized systems (Beutel & Larson, 2015). Other studies have also concluded that a variety of factors contribute to levels of fecal contamination in urban ponds, and treatment is site specific (Petterson et al., 2016). However, suggestion has been made that waterfowl and other wildlife utilizing SMPs for habitat may also contribute to increasing fecal concentrations in urban ponds (Petterson et al., 2016). In this way, SMPs may ultimately function as reservoirs for bacteria, rather than treatment sites.

2.4.2 Effect of land use and pond characteristics on water quality

Due to the variations in contaminant removal capabilities, it is not surprising that there is greater variation in outflow water quality parameters (Figure 8) compared to inflow locations (Figure 7). Since all 15 ponds are located in the same geographical location with similar underlying geology, the dynamics of inflow water quality show limited variation between sites (Figure 7). This is made evident by the clustering of sites along the first axis of the principal component analysis biplot (Figure 7). Therefore, the properties of water entering the selected ponds is fairly consistent across measured water quality parameters. However, there is significant variation in terms of outflow water quality between ponds (Figure 8). This illustrates that although incoming surface water is relatively similar, the ability of each pond to treat stormwater runoff varies greatly between sites.

To understand the wide variation in water quality across SMPs, individual pond characteristics were assessed to determine whether morphometry or age play a role in influencing outflow water quality. Both phosphorus and chlorophyll α (i.e, phytoplankton) at outflow sites were positively influenced by several SMP characteristics including drainage area, total area, perimeter, and pond width (Figure 12). These findings are in line with natural systems, where catchment or watershed area positively correlates with nutrient inputs to ponds and lakes (Robertson & Saad, 2011; Soranno et al., 2015).

Basically, the larger the drainage area, the more land is available to contribute phosphorus loadings to run-off, as it makes its way into SMPs. With respect to pond size playing a role in phosphorus concentrations, larger ponds may allow for longer stratification periods (Song et al., 2013), which encourage phosphorus loading into the sediments. During anoxic periods, phosphorus can be loaded into the water column increasing its concentrations at outflow sites.

In contrast, total nitrogen was negatively correlated with pond perimeter, total area, and pond length (Figure 12). This finding is very interesting, but also perplexing. These metrics relate to pond size, and suggest that larger SMPs have lower total nitrogen concentrations. The underlying mechanism behind this relationship is not clear, but may have to do with larger biological processing capacity in larger ponds (e.g., more plants to assimilate nitrogen). It has been suggested that vegetation, pond length, and residence time are all key design factors that can maximize nitrogen removal in SMPs (Collins et al., 2010). In this way, larger ponds allow for greater establishment of plants, as well as increased retention times. Therefore, nitrogen removal processes, such as denitrification and assimilation, are given more time to occur thereby reducing concentrations before water leaves the pond.

Landuse and surrounding drainage areas can have major impacts on water quality (Hassal & Anderson, 2015). Urban freshwater systems can show large variations in water conditions based on the composition of their surrounding watershed (Hassal & Anderson, 2015). SMPs, although much smaller in size and considered engineered systems, can also be largely influenced by their surrounding landscapes. Previous research indicates that imperviousness of the SMP watershed is an important predictor of water quality (Vincent

& Kirkwood, 2014). Based on the study ponds, it appears that impervious surface levels have significant impacts on the water clarity of incoming stormwater runoff (Figure 11). Since stormwater cannot penetrate non-porous surfaces, it can accumulate particulates and debris prior to entering SMP facilities. In this way, with increased imperious surfaces, a greater volume and variety of particulates can be collected, thereby decreasing runoff water clarity.

However, increased imperviousness of surrounding urban areas does not seem to influence nutrient levels of SMPs at inflow locations (Figure 11). Drainage area on the other hand, was shown to have a significant positive relationship with phosphorus. In this way, the total area of captured stormwater appears to play a bigger role in influencing stormwater nutrients rather than the amount of impervious surface cover. This result may be indicative of varying phosphorus sources. Since phosphorus can be sourced from anthropogenic and natural sources, the total catchment area of an SMP may have a greater influence on phosphorus concentrations in urban SMPs, compared to the percent of developed (i.e., impervious) land. Furthermore, specific anthropogenic sources of phosphorus, such as fertilizers, are not necessarily applied to impervious surfaces, but rather are washed off of manicured lawns and gardens. Therefore, the total catchment area, including both pervious and impervious surfaces, may be more significant to increasing phosphorus levels in SMPs.

2.4.3 Effect of dredging on outflow water quality

Canadian SMPs are designed on average to operate for 10-15 years, beyond which sediment maintenance practices are required (Drake & Guo, 2013). However, dredging practices can be rather costly, depending on sediment contamination levels

(Drake & Guo, 2013). These expenses can result in ponds going unmaintained beyond their operational lifecycle. Dredging involves the removal of all sediment and vegetation from the SMP, returning the pond to its original depth and hard-liner bottom. Analyses between the dredged and undredged ponds confirmed that there was no significant difference in inflow water quality characteristics (Figures 13, 14, 15, 16). However, outflow water quality did vary between dredged and undredged sites. As previously mentioned, water clarity at outfall sites is significantly lower for undredged ponds (Figures 13, 14). This suggests that the increased settling volume created by dredging may be improving settling capacity, as designed. In this way, dredged ponds allow for increased sedimentation of suspended particulates, thereby improving the clarity of outgoing stormwater.

Stormwater leaving these undredged ponds also has significantly higher total nitrogen concentrations (Figure 15). In this way, undredged ponds have lower nitrogen removal capabilities compared to ponds which have been dredged. Dredged ponds on the other hand, have higher levels of total dissolved phosphorus at outflow locations (Figure 14). This increase in phosphorus at the outflow sites of dredged ponds may be the result of “loaded” phosphorus from the sediment being disturbed and released into the water column.

2.5 CONCLUSION

SMPs are an increasingly prevalent feature across developing landscapes, however their performance as water quality treatment facilities remains largely assumed. Although SMPs are designed for suspended solids removal, there was no significant reduction in total suspended solids or coliform bacteria found in this study. I also determined that SMPs can be significant sources of phosphorus to natural receiving waters. In contrast, the study SMPs were effective in removing chloride and nitrogen from stormwater runoff.

Drainage area size influenced phosphorus concentrations in stormwater runoff, however, water clarity was largely influenced by impervious cover. Pond design elements, such as size and age did influence SMP performance. Specifically, larger ponds had higher phosphorus concentrations, and older ponds had higher salt concentrations in outflowing water. Dredged SMPs were found to have lower water turbidity and nitrogen levels compared to ponds earmarked for dredging. However, this dredging was also found to increase phosphorus loadings in outfall samples. Understanding how dredging exacerbates phosphorus release into receiving waters would be an important line of research to pursue to develop mitigation measures. Overall, SMPs still remain an essential component of urbanizing landscapes for flood control and have great potential to be optimized for water quality treatment as well. With increased monitoring of stormwater quality entering and leaving these ponds, cities can gain a better understanding of pond performance. Regular monitoring of these unique systems will also ensure maintenance practices are completed in a timely fashion, so that ponds do not surpass their expected performance lifetime.

CHAPTER 3: THE IMPACT OF AQUATIC VEGETATION ON OUTFLOW WATER QUALITY IN STORMWATER MANAGEMENT PONDS

3.1 INTRODUCTION

The inclusion of SMPs in urban areas minimizes the effects of increased runoff water velocity, and degraded quality on natural systems. These engineered systems are designed to significantly improve water clarity and limit flooding in urban areas (Casey et al., 2006). While these systems are not considered 'natural', they do become naturalized via colonization by a variety of flora and fauna. In fact, aquatic plants commonly establish and infill SMPs, yet very little is known about the types of species that naturally colonize, or their effect on water treatment performance in these urban ecosystems. Although the provincial SMP design manual suggests regular planting of submergent and emergent aquatic species (Government of Ontario, 2003), municipalities rarely if ever implement this recommendation. As such, all plants established and growing in SMPs must be able to tolerate wide fluctuations in water levels and water quality in these artificial systems. Although aquatic plants commonly occupy SMPs, there is a lack of knowledge surrounding the effects of aquatic vegetation in mitigating stormwater treatment in these urban ponds.

It has been noted that SMPs contribute a great deal to enhancing local biodiversity in urban areas (Tixier et al., 2011; Holtmann et al., 2018; Miró et al., 2018). When there is a wide variety of species, it ultimately allows an ecosystem to take advantage of significant differences between individuals (Leto et al., 2013). In settings such as SMPs, this ensures a variety of water treatment processes are occurring due to variations in plant growth, root structures, purification capacity, and so on. It has been noted that constructed wetlands can in fact enhance nutrient removal by maximizing macrophyte diversity (Greenway, 2005). In this way, it may be possible for cities to improve the water treatment potential of SMPs by ensuring diverse plant communities are established.

While research on SMPs and aquatic vegetation is limited, many studies have been completed on constructed wetlands and wastewater treatment sites. It has been broadly accepted that aquatic plants play a significant role in physically improving water treatment processes at these locations (Lee & Scholz, 2007). The abundance of macrophytes in aquatic systems can encourage processes such as sedimentation and filtering, and can also decrease the likelihood of particulate resuspension (Vymazal, 2011). Aquatic plants may also slow down flowing water velocity, ultimately lengthening retention times and improving the potential for contaminant removal (Pettecrew & Kalff, 1992; Lee & Scholz, 2006). In fact, the ability of constructed wetlands to remove suspended solids was 34% higher in sites containing both emergent and submergent macrophytes compared to those without (Karathanasis et al., 2003). Emergent vegetation can also provide windbreaks to freshwater systems, further reducing resuspension of sediment (Vymazal, 2011). In this way, the physical barriers and root systems established by aquatic vegetation is significant in improving water quality by increasing water clarity.

In Ontario, there are extensive protocols and management plans surrounding urban SMP design. However, in terms of vegetation the only regularly practiced maintenance strategy includes riparian planting (i.e. the terrestrial periphery of the SMP). In this case, various trees and shrubs are planted near the shorelines and flood zones to help shade the pond and minimize water temperatures (Government of Ontario, 2003).

Riparian planting has also been noted to reduce erosion, and provide long-term stability along pond banks (Government of Ontario, 2003). In Ontario, the planting of both submergent and emergent aquatic vegetation is also recommended to improve water quality and enhance local biodiversity (Government of Ontario, 2003).

However, planting of aquatic vegetation is not typically done in Canadian SMPs. Nonetheless, aquatic vegetation establishes in most SMPs due to natural dispersal and colonization mechanisms. Aquatic vegetation has shown its ability to remove a variety of contaminants from stormwater in both constructed and natural wetland systems (Marsalek et al., 1992; Fritioff & Greger, 2003; Ivanovsky et al., 2018). Both rooted and free-floating species have illustrated their potential to remove heavy metals (such as zinc, copper, and lead), as well as nutrients from stormwater runoff and sediments in constructed wetlands (Fritioff & Greger, 2003; Weiss et al., 2006; Chang et al., 2012). However, little is known about the relationship between SMP water quality treatment and total aquatic plant abundance in these systems.

Possible modes of action for aquatic macrophytes in SMP performance may involve their role in stabilizing sediments, and offering habitat and organic substrate for microbial degraders (Leto et al., 2013). Large volumes of macrophytes provides surface for the production of biofilms, which are largely responsible for microbial water treatment processes (Leto et al., 2013). Furthermore, aquatic plants are responsible for transporting over 90% of oxygen available in the rhizosphere, acting to enhance the growth of nitrifying bacteria and encouraging aerobic decomposition (Reddy et al., 1989). While the ability of macrophytes to directly reduce nitrogen has been deemed relatively low compared to microbial processes, recently the ability of aquatic plants to contribute to salt phytoremediation has been noted (Shelef et al., 2013). This may further highlight the potential for specific plant species and established communities to improve water treatment process in aquatic urban environments.

This chapter focuses on the role of aquatic plant abundance, type, and diversity on stormwater treatment in 15 SMPs located in Oshawa, ON. The main research objective was to assess outflowing water quality as a function of submergent and emergent plant communities. Variation in plant amount, type, and species richness across the study sites was compared to outflow water quality profiles for the study sites, in order to determine the role of aquatic vegetation in mitigating water quality in SMPs.

3.2 MATERIALS AND METHODS

3.2.1 Study sites and water sample collection

See Chapter 2, section 2.2 Materials and Methods, for complete description of study sites and water sampling methods used.

3.2.2 Vegetation sample collection

The 15 SMPs were sampled monthly for submergent aquatic vegetation from June to September 2018 and 2019. Initial submergent plant sampling in June 2018 was completed using a lake rake, which was unsuccessful. All remaining submergent vegetation samples were collected using a 1 m^2 quadrat. Due to these various sampling techniques and low total biomass in 2019, submergent plant samples for June from both years was not included in subsequent analyses. Sites where submergent plants were collected were primarily selected based on accessibility, but were also consistently collected in the same general location in each pond. The quadrat was never placed in the same sampling site, but rather placed adjacent to previous visits to ensure vegetation removed was not regrowth from prior collection periods. All vegetation within the quadrat boundaries was hand pulled, and kept on ice until further analysis. Only defined portions of the quadrat were used during August and September collection dates for both years due to high vegetation biomass (typically 0.5 m^2). Once in the laboratory, plants were sorted, identified, weighed, and dried to determine plant biomass and relative abundances for each pond. See Appendix A, Figure A1 for cross section of plant sampling locations.

Three emergent vegetation transects were also completed for each pond, on August 24, 2018 as well as August 22, 2019. Emergent vegetation type and areal coverage was estimated using both drone images and point intercept transects. High resolution drone images of each pond were taken on September 6, 2018 and used to estimate percent emergent vegetation coverage. This was estimated using ImageJ software, and calculated relative to total pond area. It has been suggested that while emergent cover varies across long time periods, plant communities are well established and do not vary greatly in the short term (Grosshans et al., 2004). Therefore, emergent cover estimates act as a proxy for both 2018 and 2019 sampling seasons. See Appendix B, Figure B1 for a sample drone image used to estimate percent emergent plant coverage. Frequency of species occurrence for emergent vegetation was calculated using a point intercept method. Three 15 m transects were placed at equal distance surrounding each pond. Species located at each 1 m mark along the transect line were identified, and counted. The three transects were combined to determine the relative frequency of occurrence for species recorded.

3.2.3 Data Analysis

All relative abundance plots, t-tests, correlation analyses, and linear regressions were completed using RStudio v1.1.463 (RStudio, Boston, USA). All constructed correspondence analyses and canonical correspondence analyses were completed using Paleontological Statistics (PAST) version 4.0 (Hammer et al., 2001). All water quality parameters and biological data were non-normal, and thus were transformed to improve normality, when possible. All other parametric assumptions were met, therefore due to the robustness of such a large dataset, parametric tests were used. For multivariate ordination analyses, water quality parameters were center-standardized.

Diversity indices were calculated for all 15 study ponds based on their emergent and submergent plant species. The Shannon-Wiener Diversity Index was calculated using the following formula:

$$H' = \sum_{i=1}^{s} (p_i)(\ln p_i)$$

Where,

H' = Shannon-Wiener index of species diversity.

S = Number of species in the community (species richness).

P_i = Proportion of total abundance represented by i^{th} species.

3.3 RESULTS

3.3.1 Characterization of emergent and submergent plant communities

Emergent transect sampling captured a mixture of both terrestrial and aquatic species. Figure 17 illustrates the species composition and relative abundance for emergent vegetation, including terrestrial species, for each pond in 2018 and 2019. The majority of ponds are fairly stable in their species composition over the two-year study period. Notably, species with the highest frequencies, such as aquatic plant *T. latifolia* and terrestrial plant *S. canadensis*, are recognized as “pioneer” species. These species readily colonize freshly disturbed landscapes, and are opportunistic in disrupted locations. However, many invasive plants are also recognized as pioneer species. The only invasive plants identified across the two-year study period were included in the emergent vegetation category. These included *P. australis* (aquatic), *L. salicaria* (aquatic), *A. lappa* (terrestrial) and *V. rossicum* (terrestrial).

Since terrestrial emergent species do not play a significant role in directly mitigating water treatment in SMPs, they were removed from the remainder of analyses. Figure 18 illustrates relative abundances for aquatic emergent plants that were identified in 2018 and 2019. These plants were defined as ‘aquatic emergent vegetation’ based on their tendency to be rooted in water and pierce the surface, so that the majority of the plant is exposed to air. When terrestrial species are removed from the emergent community, it becomes clear that SMPs tend to show relatively stable communities between years, however, they are comprised of dominant monocultures. Many SMPs are heavily dominated by one of two species, *T. latifolia* and *P. australis*. These species are both considered opportunistic and can readily out compete other species for space and nutrients. However, in high volumes, these species can also drastically impact the hydrology of a system.

Submergent plant communities were also documented for all 15 ponds over a four-month sampling season from June to September each study year. Due to variations in sampling protocols in 2018 and low plant biomass in 2019, June was removed from the submergent relative plant abundance plot (Figure 19). There is limited variation in submergent plant diversity temporally across the summer season and spatially across the study sites. Dominant species did tend to shift over the sampling season, however very few ponds have established communities with greater than three species present at a given time. In fact, the majority of sampled SMPs show consistent monocultures that do not vary greatly between sampling dates. In early summer, well established monocultures of *S. pectinata* and *P. pusillus* are noted for both sampling years. While some community structures are stable, others tended to shift to *N. flexilis* and *N. guadalupensis* dominated systems during late summer.

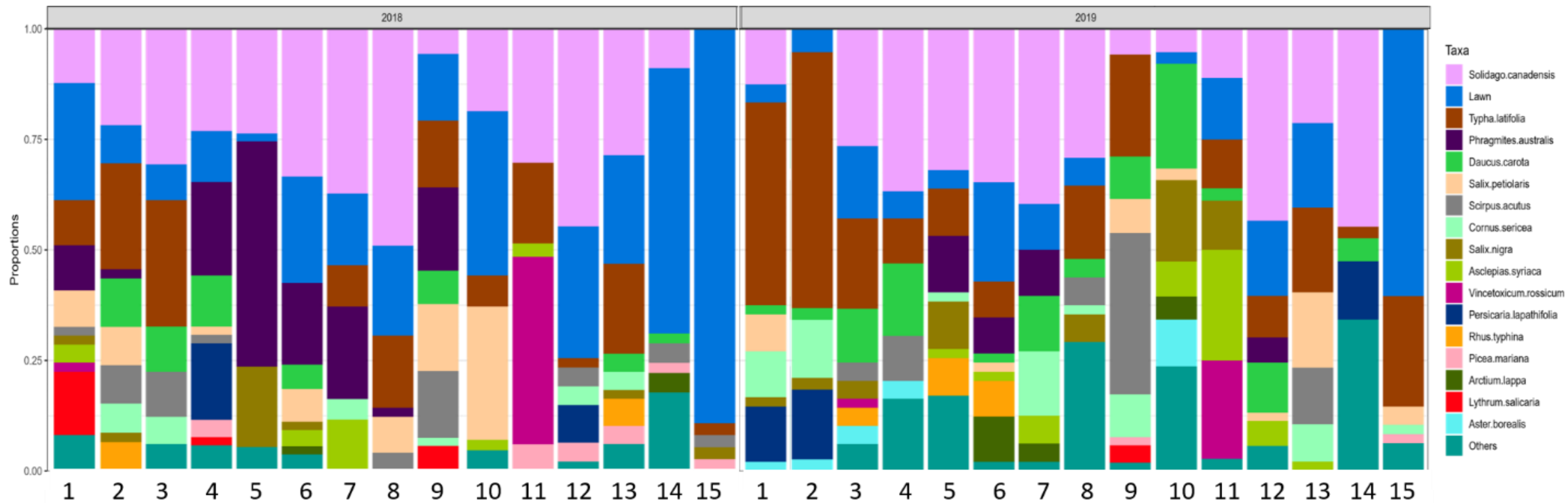

Figure 17. Relative abundances for emergent plants (aquatic and terrestrial) for 2018 and 2019. Ponds are labelled by increasing pond number (1-15).

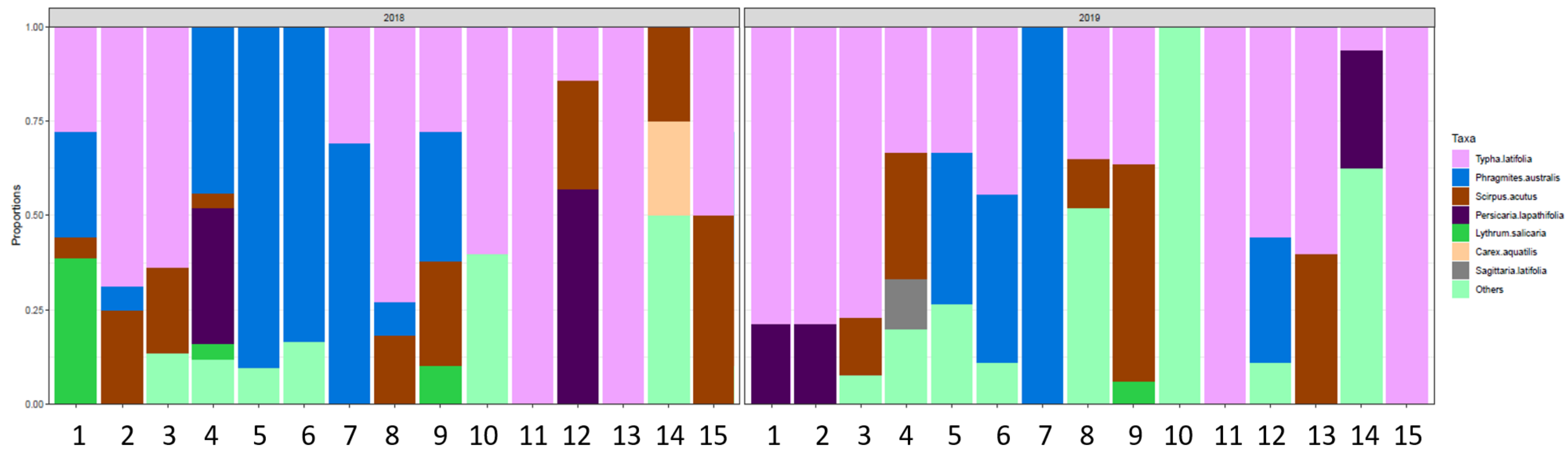

Figure 18. Relative abundances for emergent plants (aquatic only) for 2018 and 2019. Ponds are labelled by increasing pond number (1-15).

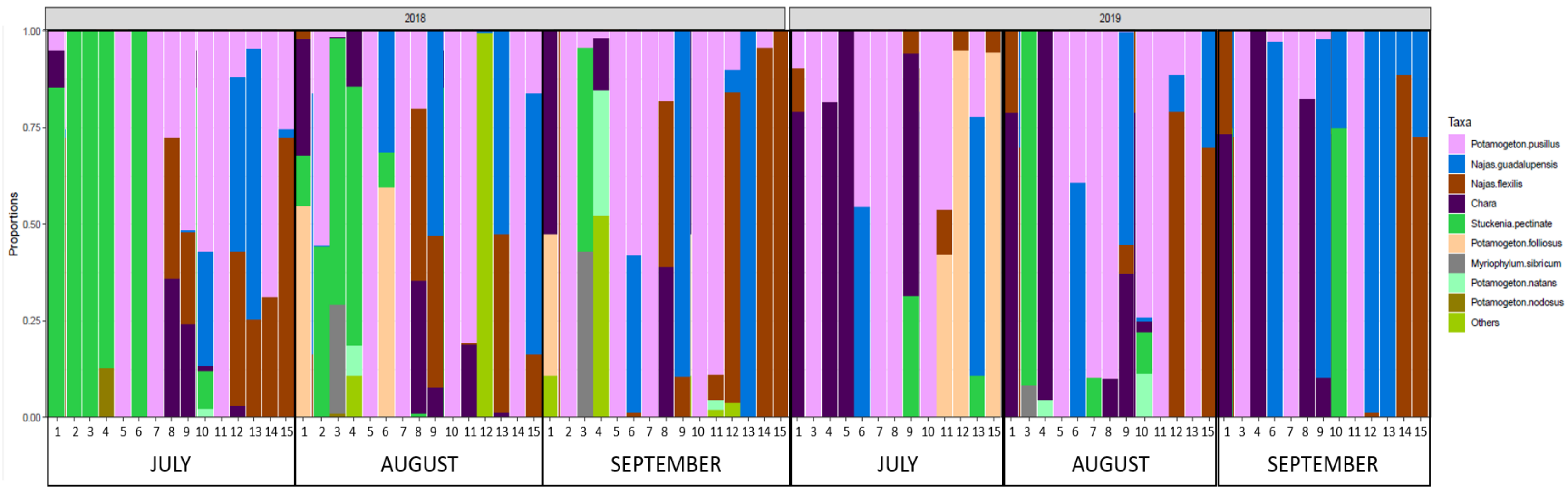

Figure 19. Relative abundances for submergent vegetation sampled from July to September for 2018 and 2019. Ponds are labelled by increasing pond number (1-15). No vegetation was collected in July 2019 from ponds 2 and 14, August 2019 from ponds 2 and 14, as well as September 2019 from ponds 2 and 5.

3.3.2 Effect of aquatic plant abundance on outflow water quality

Welch two sample t t-tests were performed to compare mean submergent plant biomass collected each month between study years (Figure 20). Total submergent biomass for all 15 SMPs does not vary greatly across months or between years, with the exception of July. Peak submergent plant biomass occurs in September for 2018 and August for 2019, with a median submergent biomass reflecting 500 g/m^2. The range of submergent plant biomass measured from 2018 and 2019 varied greatly across all 15 study ponds (Figure 21). This variation is further illustrated when submergent biomass is normalized by total open water area for each pond (Figure 22). There is also wide variation in emergent cover in 2018 across all 15 study sites (Figure 23). It should also be noted that no significant relationship was found between submergent plant biomass and emergent plant cover (Pearson Correlation, $p > 0.05$).

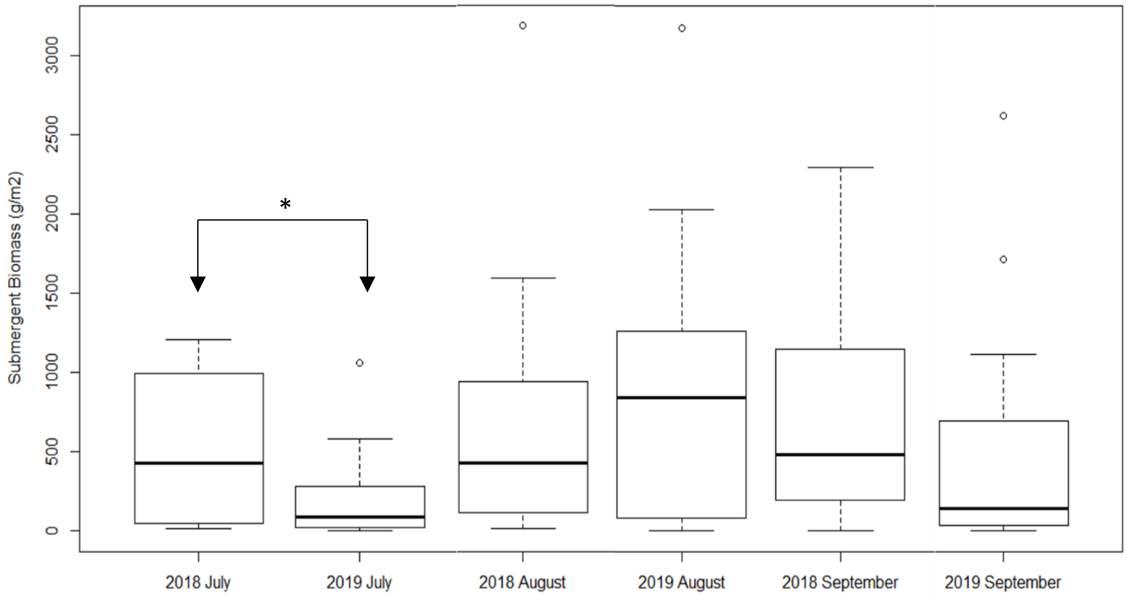

Figure 20. Total submergent biomass (g/m^2) by year and month for all 15 ponds. Welch two sample t-test, significant differences are denoted by $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

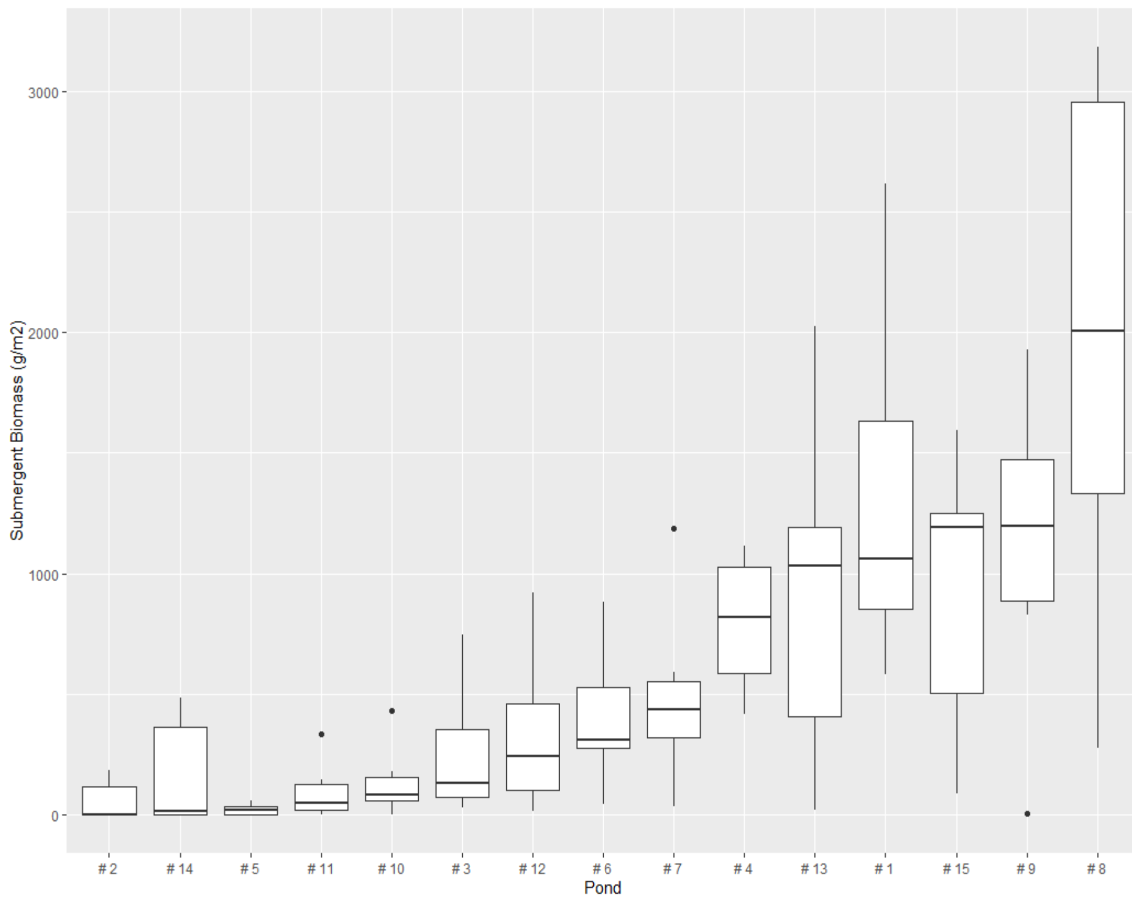

Figure 21. Total submergent biomass (g/m^2) for all 15 ponds combined for 2018 and 2019. Ponds are sorted by increasing median values.

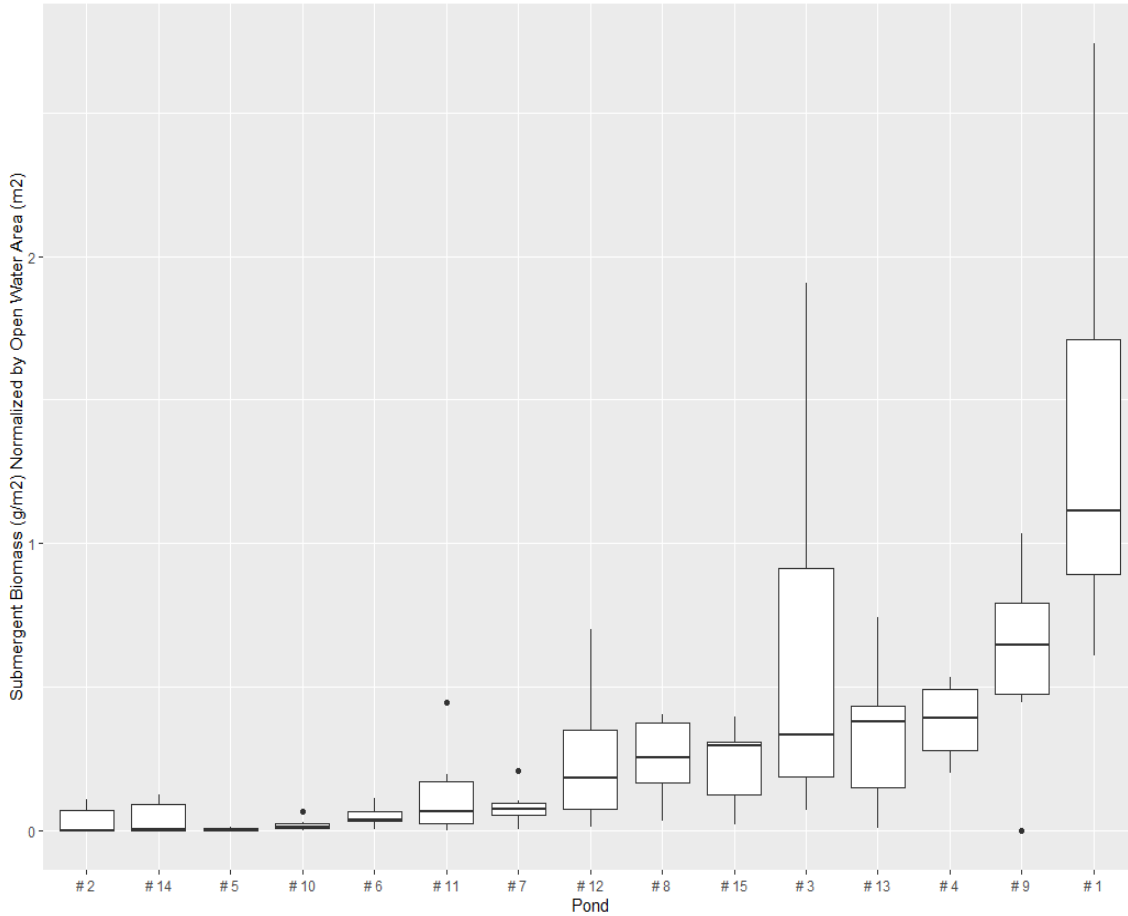

Figure 22. Total submergent biomass (g/m^2) for all 15 ponds from 2018 and 2019 normalized by total open water area (m^2) for each pond. Ponds are sorted by increasing median values.

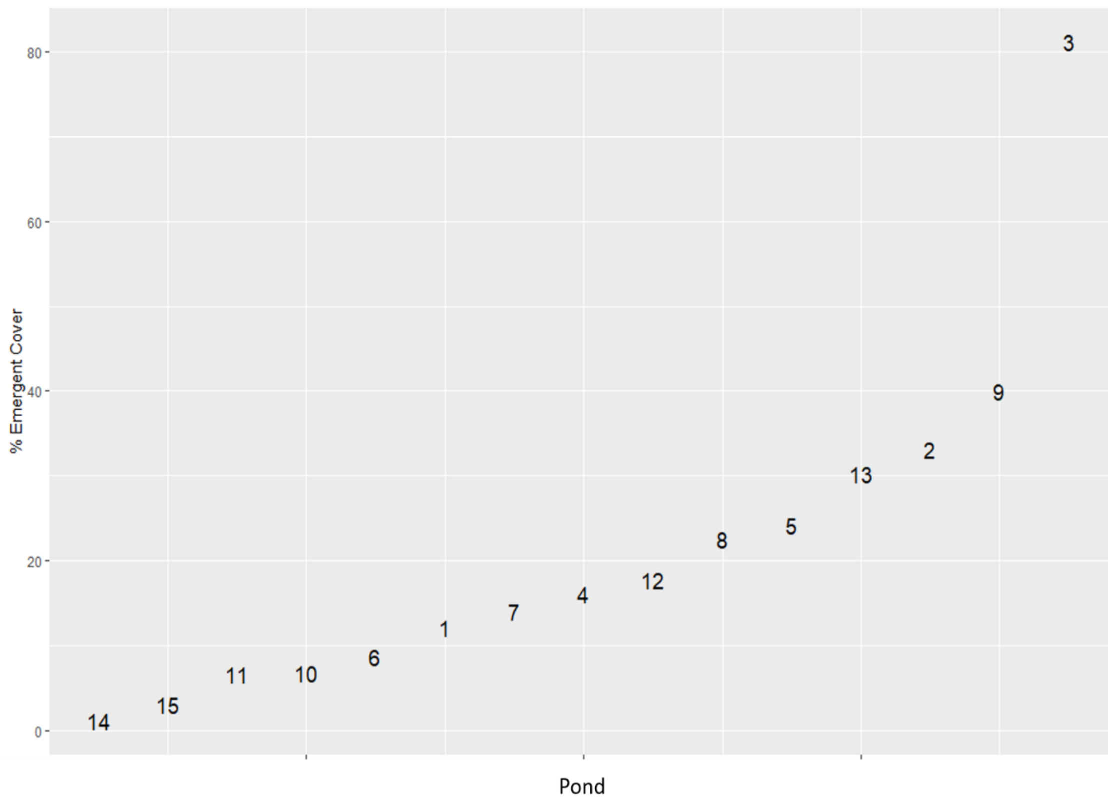

Figure 23. Percent emergent plant cover for each pond completed using drone images in 2018.

Relationships between outflow water quality variables and aquatic plant biomass may highlight specific water treatment processes occurring either directly or indirectly due to aquatic plant presence. Pearson correlation analysis was completed for outflow water quality parameters and aquatic plant (submergent and emergent) abundances (Table 3). With increasing submergent plant biomass, there is a notable decrease in both total nitrogen and phosphorus (marginally significant) concentrations at outflow locations. As submergent biomass increases, there is also a notable increase in outflow temperature (marginally significant), and a significant decrease in dissolved oxygen. Emergent vegetation also has significant relationships with certain water quality parameters (Table 3). As emergent vegetation increases, there are notable increases in total nitrogen (marginally significant) and total suspended solids. Emergent plant cover also shows a significant relationship with temperature, but unlike submergent plants, as abundance increases temperature tends to decrease. Significant correlations are further illustrated in Appendix B, Figures B5-B8.

To further address these relationships, a linear regression analysis was completed for submergent plant biomass and outflow water quality parameters (Table 4). Selected water quality variables were based on significant and marginally significant relationships from the completed Pearson correlation (Table 3). Increasing submergent biomass is significantly related to decreasing nitrogen concentrations at the outflow site ($p = 0.004$, $R^2 = 0.078$). Increasing submergent plant biomass is also significantly related to decreasing dissolved oxygen concentrations at outflow locations ($p = 0.015$, $R^2 = 0.067$).

A linear regression analysis was also completed for emergent plant cover and outflow water quality parameters (Table 5). Selected water quality variables were based on significant and marginally significant relationships from the completed Pearson correlation (Table 3). Increasing emergent cover is significantly related to increasing total suspended solid concentrations at the outflow site ($p = 0.019$, $R^2 = 0.149$). Increasing emergent cover is also significantly related to decreasing outflow temperature ($p = 0.004$, $R^2 = 0.278$).

Table 3. Pearson correlation analysis between **outflow** water quality parameters and aquatic plant abundance (percent emergent cover and submergent biomass). Dates for emergent plant cover included August 2018 and 2019 only, dates for submergent biomass included all dates when submergent vegetation was sampled in 2018 and 2019. Significant relationships are bolded.

	Percent Emergent Cover		**Submergent Biomass (g/m^2)**	
Parameter	**Cor.**	**p-value**	**Cor.**	**p-value**
Colour (A @ 440 nm)	-0.161	0.397	-0.075	0.479
Turbidity (A @ 750 nm)	0.245	0.191	-0.113	0.288
Total Suspended Solids (g/L)	0.423	**0.019**	-0.112	0.291
Total Organic Suspended Solids (g/L)	0.134	0.478	0.029	0.783
Total Coliforms (CFU/100 mL)	0.095	0.617	-0.008	0.939
Total E. coli (CFU/100 mL)	0.161	0.394	-0.071	0.545
Chlorophyll α (g/L)	-0.145	0.446	-0.055	0.603
Chloride (mg/L)	0.024	0.899	-0.117	0.271
Conductivity (µs/cm)	-0.276	0.192	-0.134	0.261
Total Phosphorus (µg/L)	-0.081	0.672	-0.2	0.059*
Total Dissolved Phosphorus (µg/L)	-0.198	0.293	-0.118	0.267
Total Nitrogen (mg/L)	0.338	0.068*	-0.316	**0.002**
Temperature (°c)	-0.563	**0.005**	0.227	0.055*
Dissolved Oxygen (mg/L)	0.092	0.668	-0.281	**0.017**
pH	-0.027	0.899	-0.097	0.417

*Not significant, but notable relationships.

Table 4. Summary statistics for least-squares linear regression using submergent plant biomass as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Total phosphorus	-1.81	1.03	-1.76	0.023	0.082
Total Nitrogen	-312.31	106.7	-2.93	0.078	**0.004**
Temperature	55.24	29.68	1.86	0.033	0.067
Dissolved oxygen	-100.12	40.32	-2.48	0.067	**0.015**

Table 5. Summary statistics for least-squares linear regression using emergent plant cover as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Total suspended solids	282.89	114.43	2.47	0.149	**0.019**
Total Nitrogen	16.07	8.51	1.89	0.081	0.069
Temperature	-5.95	1.89	-3.14	0.278	**0.004**

3.3.3 Effect of aquatic plant type on outflow water quality

Relationships between plant communities and water quality variations between ponds are important in fully understanding the dynamics within SMP facilities. Understanding how individual plant types relate to one another, as well as how individual plant types relate to various outflow water variables can highlight potential community profiles that have the greatest influence on water quality. A canonical correspondence analysis was completed for all submergent plant species and outflow water quality parameters (See Appendix B, Figure B9). Due to the presence of many species which occur in low frequencies, a second canonical correspondence analysis for outflow water quality was completed for submergent plant biomass with rare species removed (Figure 24). In this case, all rare species which were responsible for less than 1% of the total submergent plant biomass, were removed from analysis (Figure 24). Ponds with higher *Chara* biomass tend to have high organic suspended solids, and salt concentrations at the outflow locations, this includes ponds 1 and 5. Plant communities including *S. pectinata* and *P. pusillus* represent outflow locations that are higher in nutrients (total phosphorus and nitrogen), including ponds 3 and 10. Three plant species, including *P. folliosus*, *N. flexilis* and *N. guadalupensis*, represent ponds with outflow locations that have high dissolved oxygen, temperature and pH levels, including ponds 9, 14 and 15.

A canonical correspondence analysis was also completed for emergent aquatic vegetation and outflow water quality parameters (Figure 25). *P. australis* is an invasive species that tends to dominate systems in which it occurs. In this case, outflow water from ponds with high *P. australis* abundances, such as ponds 4 and 6, tend to have higher temperatures and turbidity levels. Ponds with high *P. lapathifolia* coverage, show increases in nutrients and chloride at outflow locations. Communities including aquatic emergent species such as, *L. salicaria, S. acutus* and *S. latifolia*, show decreased chloride and nutrient concentrations at outflow sites. Ponds with communities including *C. aquatilis, T. latifolia* and *A. plantago-aquatica* show decreased turbidity and temperature at outflow locations.

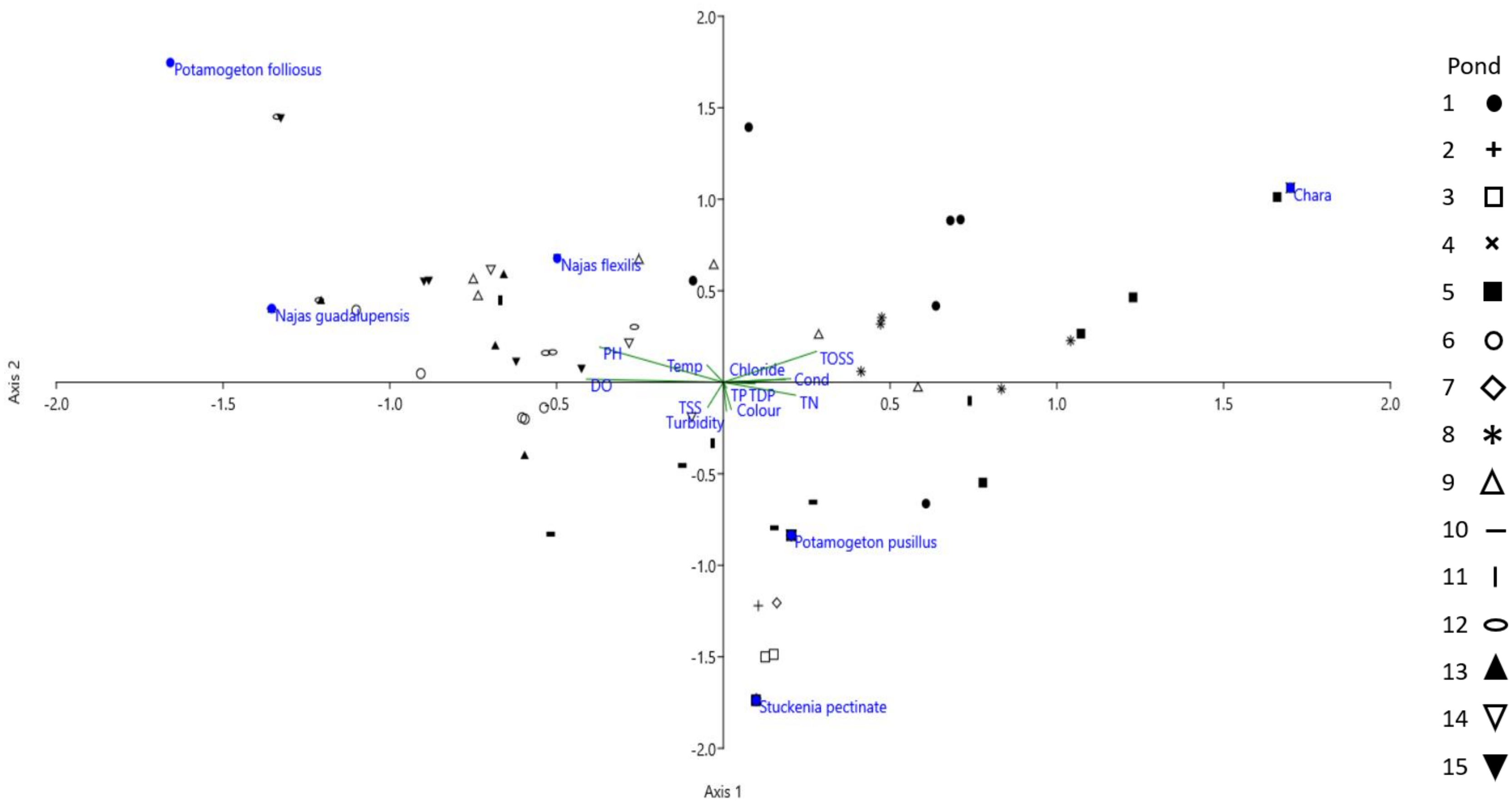

Figure 24. Canonical correspondence analysis for submergent plant biomass (excluding rare species) and **outflow** water quality parameters for 2018 and 2019. Axis 1 represents 58.08% of the variation and axis 2 represents 20.25% of the variation.

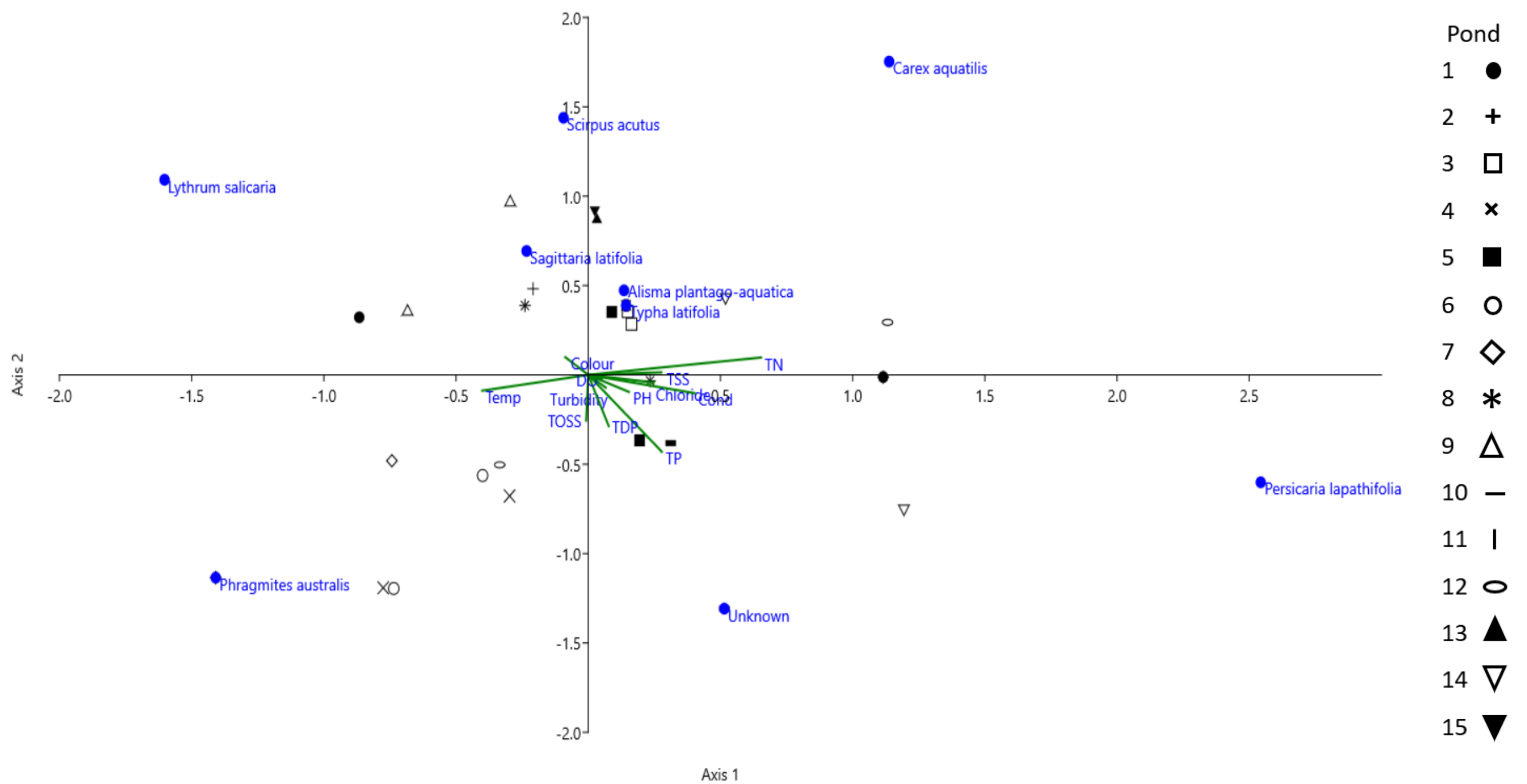

Figure 25. Canonical correspondence analysis for emergent plant cover and **outflow** water quality parameters for August 2018 and 2019. Axis 1 represents 39.56% of the variation and axis 2 represents 29.7% of the variation.

Relationships between individual plant species and outgoing water quality parameters can further highlight taxa that are associated with improved stormwater quality. A Pearson correlation was completed for submergent vegetation and water quality parameters at the outflow locations (Figure 26). Specific submergent plant species are significantly related to multiple water quality parameters. *Chara* is a native and relatively common species to Oshawa SMPs. Its presence in urban ponds significantly correlates with increased organic suspended solids, as well as decreased dissolved oxygen and pH at outflow locations. *M. sibricum* was found at only one SMP throughout the duration of the study (pond 3) and is associated with increased nitrogen and deceased temperature at SMP outfalls. *N. flexilis* is a member of the water nymph family, commonly found in a variety of the study ponds. Its presence in urban ponds is significantly correlated with decreased chloride and nitrogen concentrations, as well as increased pH at outflow locations. A number of other submergent species show significant relationships with single water quality variables. Notably two Potamogeton species, *P. natans* and *P. pusillus* respectively are associated with increased organic suspended solids and increased chlorophyll α (i.e. phytoplankton biomass) concentrations at outfall locations.

To further address these relationships, a linear regression analysis was completed for submergent plant species and outflow water quality parameters. Selected water quality variables were based on significant relationships from the completed Pearson correlation (Figure 26). *Chara* biomass is significantly associated with increased organic suspended solids, as well as decreased dissolved oxygen and pH at the outflow site (Table 6). Biomass of *M. sibricum* is significantly related to increasing nitrogen and decreasing temperature at the outflow site (Table 7). *N. flexilis* abundance is significantly associated with increasing pH, as well as decreasing chloride and nitrogen at the outflow locations (Table 8). Biomass of *N. guadalupensis* is significantly related to increasing oxygen concentrations at outflow sites (Table 9). *P. foliosus* abundance is significantly related to increased outflow pH (Table 10). *P. natans* biomass is significantly related to increased organic suspended solids at outflow locations (Table 11). Another Potamogeton species, *P. pusillus,* is significantly associated with increasing outflow chlorophyll α concentrations (Table 12). Finally, *S. pectinate* is significantly associated with increasing chloride concentrations at outflow locations (Table 13).

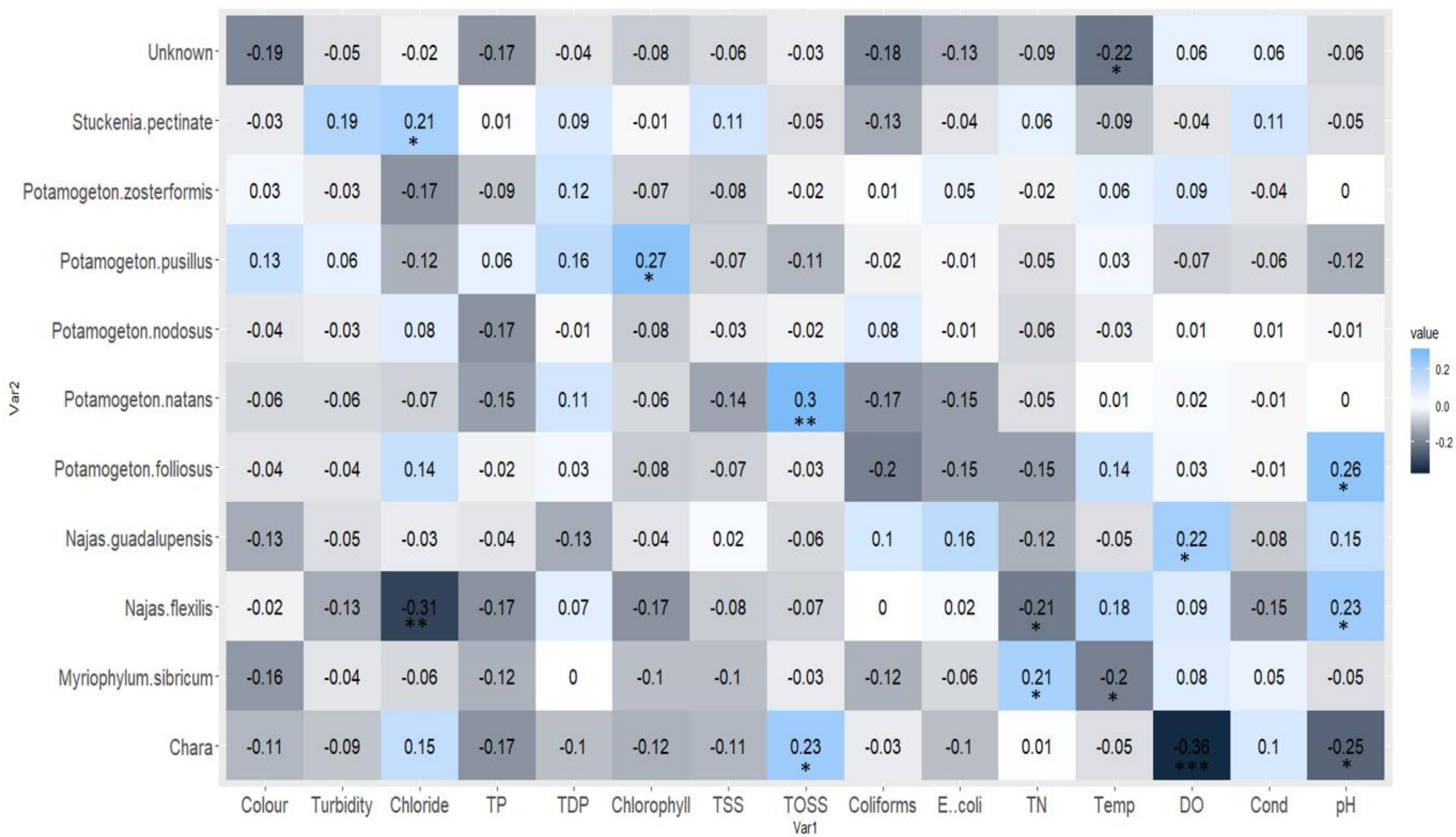

Figure 26. Pearson correlation for submergent plant abundance and **outflow** water quality parameters from 2018 and 2019. Significant relationships are denoted by p < 0.05 *, p < 0.01 **, p < 0.001 ***.

Table 6. Summary statistics for least-squares linear regression using submergent species *Chara* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Total organic suspended solids	3.76	1.66	2.26	0.044	**0.026**
Dissolved oxygen	-2.04	0.57	-3.59	0.118	**< 0.001**
pH	-8.75	3.69	-2.37	0.049	**0.019**

Table 7. Summary statistics for least-squares linear regression using submergent species *Myriophyllum sibricum* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Total Nitrogen	0.39	0.2	1.98	0.031	**0.05**
Temperature	-0.91	0.47	-1.93	0.029	0.057

Table 8. Summary statistics for least-squares linear regression using submergent species *Najas flexilis* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Chloride	-0.19	0.06	-3.04	0.084	**0.003**
Total nitrogen	-1.14	0.56	-2.01	0.033	**0.047**
pH	8.36	3.73	2.24	0.043	**0.027**

Table 9. Summary statistics for least-squares linear regression using submergent species *Najas guadalupensis* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Dissolved oxygen	1.32	0.63	2.09	0.037	**0.039**

Table 10. Summary statistics for least-squares linear regression using submergent species *Potamogeton foliosus* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
pH	5.87	2.35	2.5	0.056	**0.014**

Table 11. Summary statistics for least-squares linear regression using submergent species *Potamogeton natans* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Total organic suspended solids	1.64	0.56	2.93	0.078	**0.004**

Table 12. Summary statistics for least-squares linear regression using submergent species *Potamogeton pusillus* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Chlorophyll α	46.38	17.87	2.59	0.061	**0.011**

Table 13. Summary statistics for least-squares linear regression using submergent species *Stuckenia pectinate* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Chloride	0.12	0.06	1.99	0.032	**0.049**

A Pearson correlation analysis was also completed for aquatic emergent vegetation and water quality parameters at the outflow locations for all 15 SMPs (Figure 27). Very few aquatic emergent species show significant trends with outflow water quality variables. *P. australis* has been noted as one of the most common macrophyte species to colonize both constructed and natural wetland systems. Its presence in SMPs is correlated with decreased nitrogen, and increased chlorophyll α concentrations at outflow locations. On the other hand, *S. acutus* which is another common macrophyte to wetland systems, shows a significant inverse relationship with chlorophyll (i.e. phytoplankton) concentrations at outfall sites. Interestingly, *T. latifolia* which is a dominant species in the study SMPs, as well as urban ponds across Canada, is not significantly correlated with any water quality variables. A number of rare emergent species (i.e. represent < 1% of total emergent cover) are correlated with specific water quality parameters. This includes *S. latifolia* and *P. lapathifolia* which are respectively associated with decreased phosphorus and increased nitrogen at outflow locations.

A linear regression analysis was also completed for emergent plant species and outflow water quality parameters. Selected water quality variables were based on significant relationships from the completed Pearson correlation (Figure 27). Coverage of emergent species *A. aquatica* is significantly related to decreased coliform levels at the outflow site (Table 14). *P. lapathifolia* abundance is significantly associated with increasing outflow nitrogen concentrations (Table 15). Invasive species *P. australis* coverage is significantly related to increasing chlorophyll α concentrations, as well as decreasing nitrogen concentrations at outflow locations (Table 16). *S. latifolia* coverage is significantly related to decreasing outflow total phosphorus concentrations (Table 17).

Finally, common wetland species *S. acutus* is significantly associated with decreased chlorophyll α levels at the outflow site (Table 18).

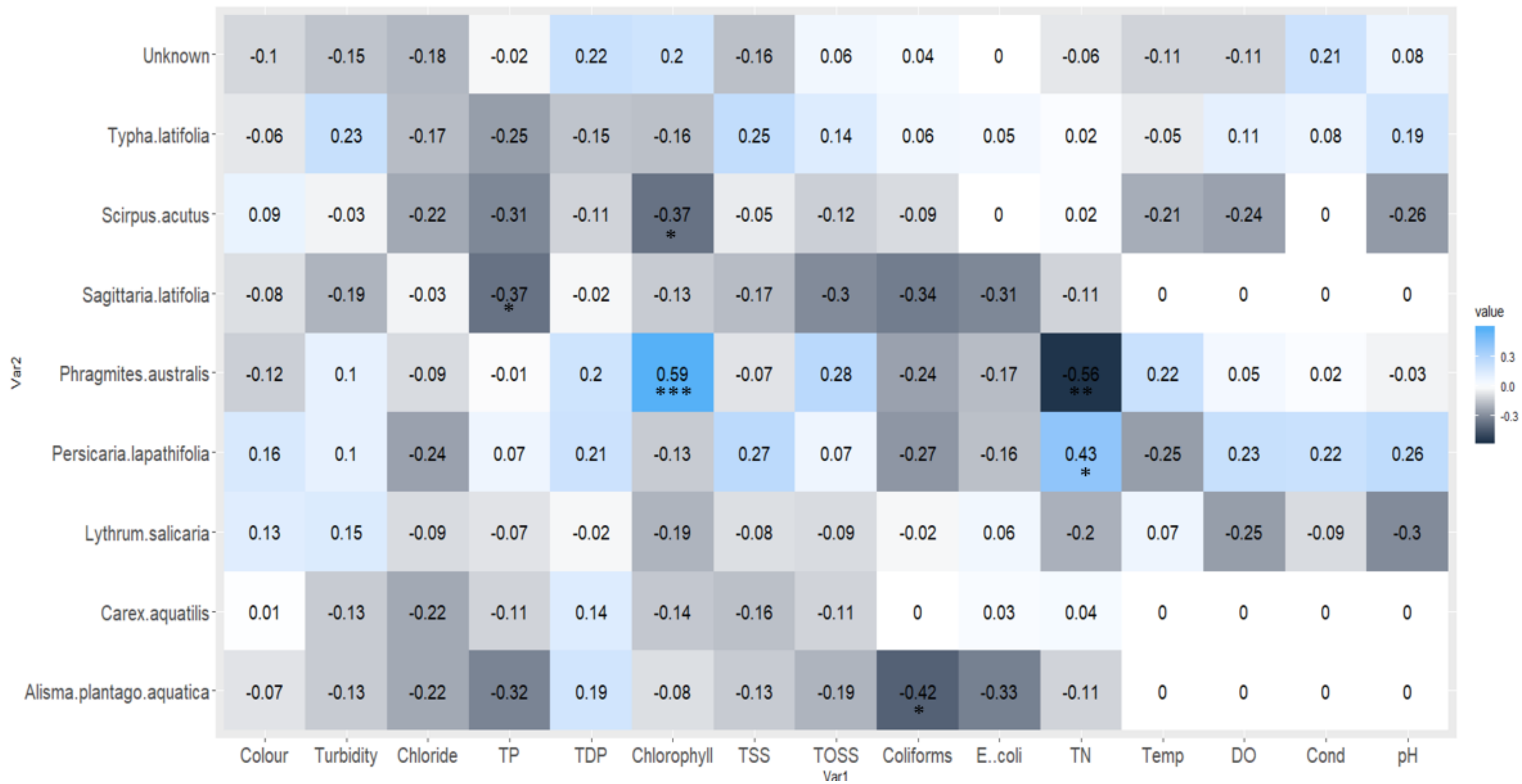

Figure 27. Pearson correlation for emergent plant cover and **outflow** water quality parameters from 2018 and 2019. Significant relationships are denoted by $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

Table 14. Summary statistics for least-squares linear regression using emergent species *Alisma-plantago aquatica* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Coliforms	-0.06	0.03	-2.42	0.143	**0.022**

Table 15. Summary statistics for least-squares linear regression using emergent species *Persicaria lapathifolia* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Total Nitrogen	2.05	0.82	2.51	0.154	**0.018**

Table 16. Summary statistics for least-squares linear regression using emergent species *Phragmites australis* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Chlorophyll α	58.45	15.12	3.87	0.324	< **0.001**
Total Nitrogen	-3.61	1.02	-3.54	0.284	**0.001**

Table 17. Summary statistics for least-squares linear regression using emergent species *Sagittaria latifolia* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Total phosphorus	-0.23	0.11	-2.08	0.103	**0.047**

Table 18. Summary statistics for least-squares linear regression using emergent species *Scirpus acutus* as the model predictor of select **outflow** environmental variables. Significant relationships are bolded.

Variable	**Estimate**	**Standard Error**	**T-value**	**Adjusted R^2**	**p-value**
Chlorophyll α	-33.68	15.86	-2.12	0.108	**0.043**

Since the variety of emergent aquatic vegetation is low, the majority of study ponds are well colonized by one of two species, *P. australis* or *T. latifolia*. Six ponds were selected based on their relative emergent cover, to compare differences between *T. latifolia* and *P. australis* dominated systems. The selected ponds showed 50% or more coverage of the dominant taxa. T-tests between *T. latifolia* dominated ponds (ponds 2, 3, 13) and *P. australis* dominated ponds (ponds 5, 6, 7) were completed for all outflow water quality parameters (Figures 28, 29, 30, 31). Significant differences were noted for chlorophyll α, total suspended solids, total nitrogen, temperature, and dissolved oxygen. *T. latifolia* dominated systems have notably higher suspended solids, total nitrogen, and dissolved oxygen at the outflow locations (Figures 29, 30, 31), while *P. australis* dominated ponds have higher temperatures and chlorophyll α concentrations (Figures 29, 30).

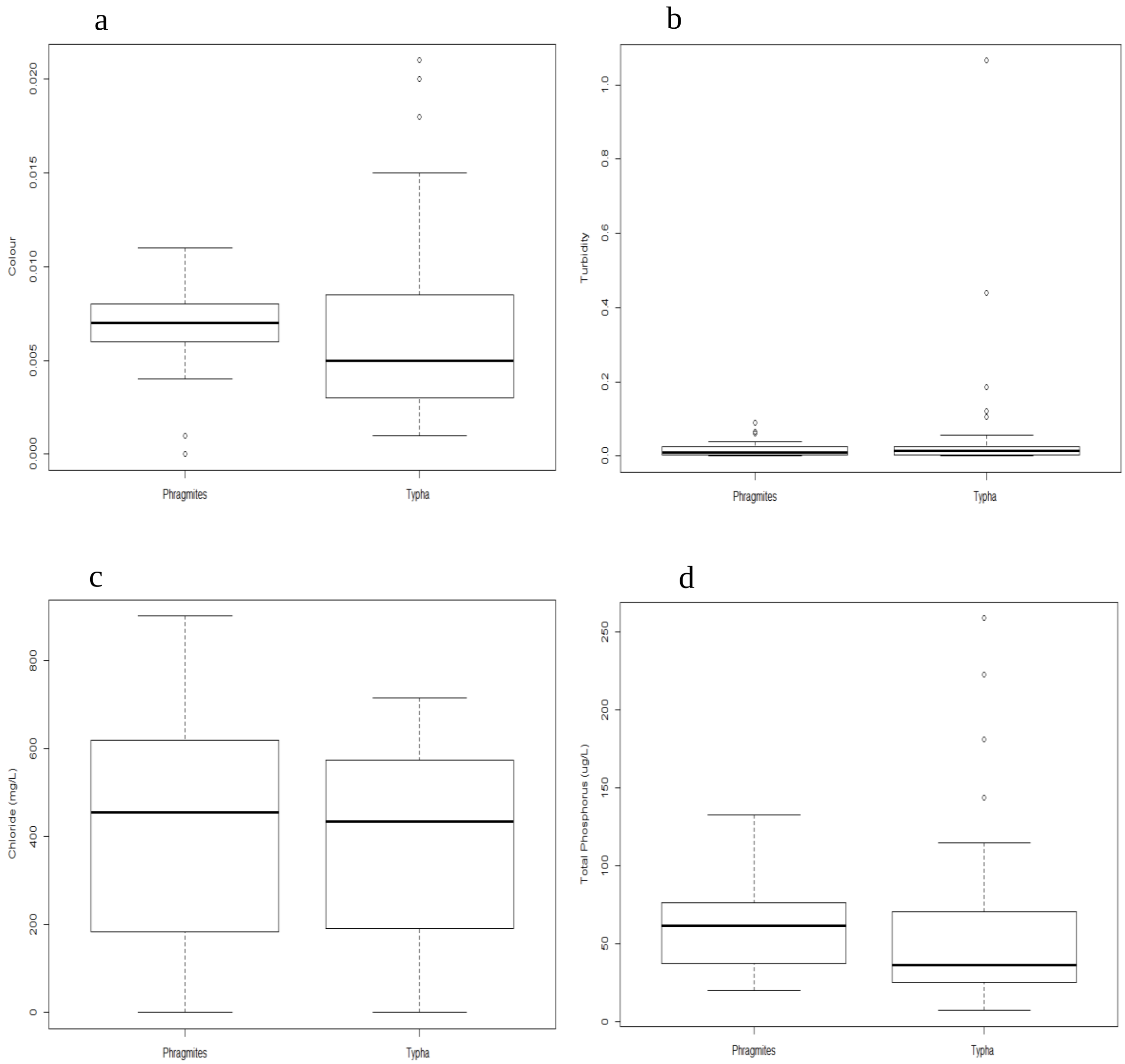

Figure 28. Colour (a), tubidity (b), chloride (c) and total phosphorus (d) for **outflow** locations of three *Typha latifolia* dominated ponds (2, 3, 13) and three *Phragmites australis* dominated ponds (5, 6, 7). All collection dates for 2018 and 2019 were included, except fall 2019. Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

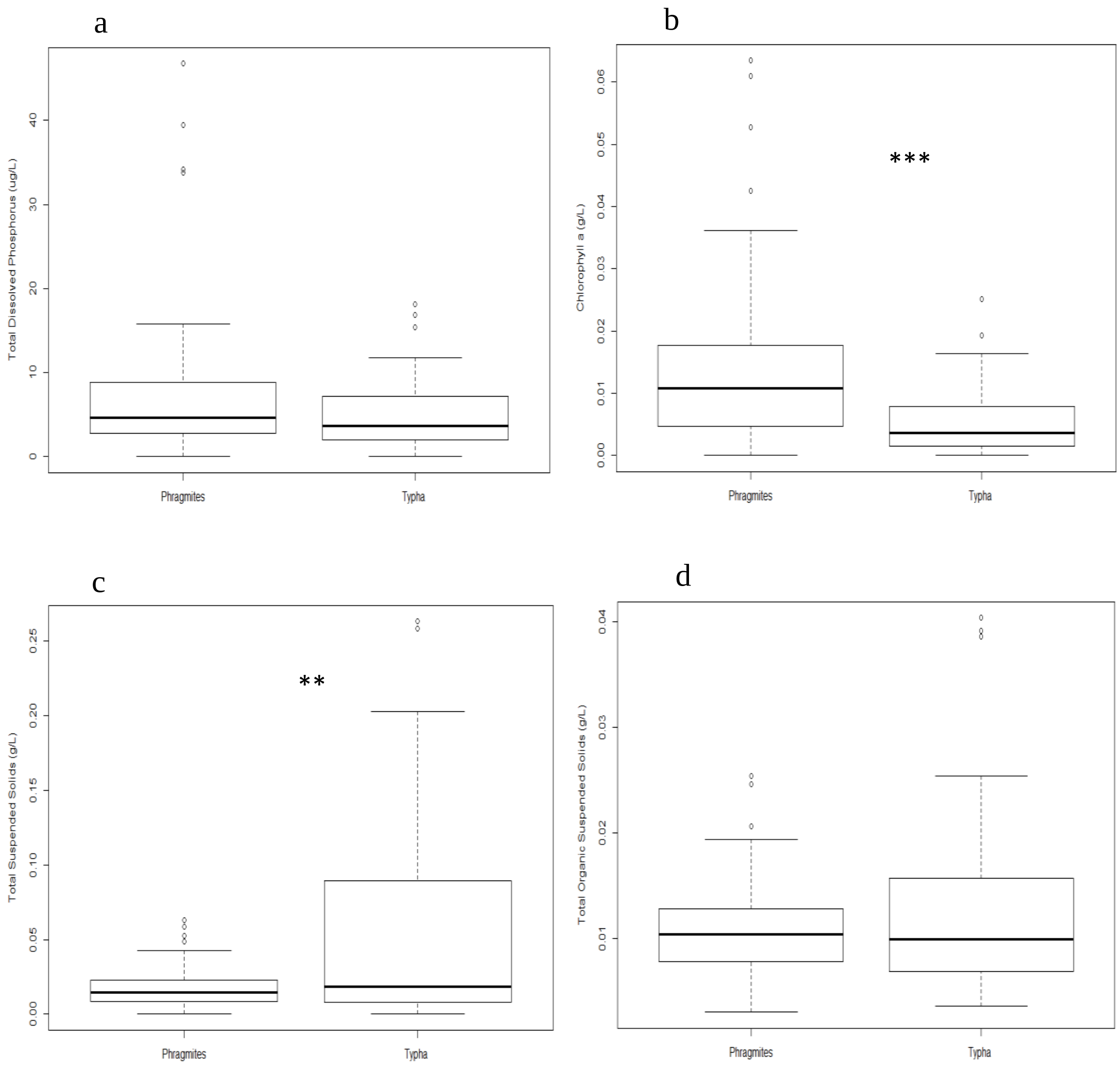

Figure 29. Total dissolved phosphorus (a), chlorophyll α (b), total suspended solids (c) and total organic suspended solids (d) for **outflow** locations of three *Typha latifolia* dominated ponds (2, 3, 13) and three *Phragmites australis* dominated ponds (5, 6, 7). All collection dates for 2018 and 2019 were included, except fall 2019. Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

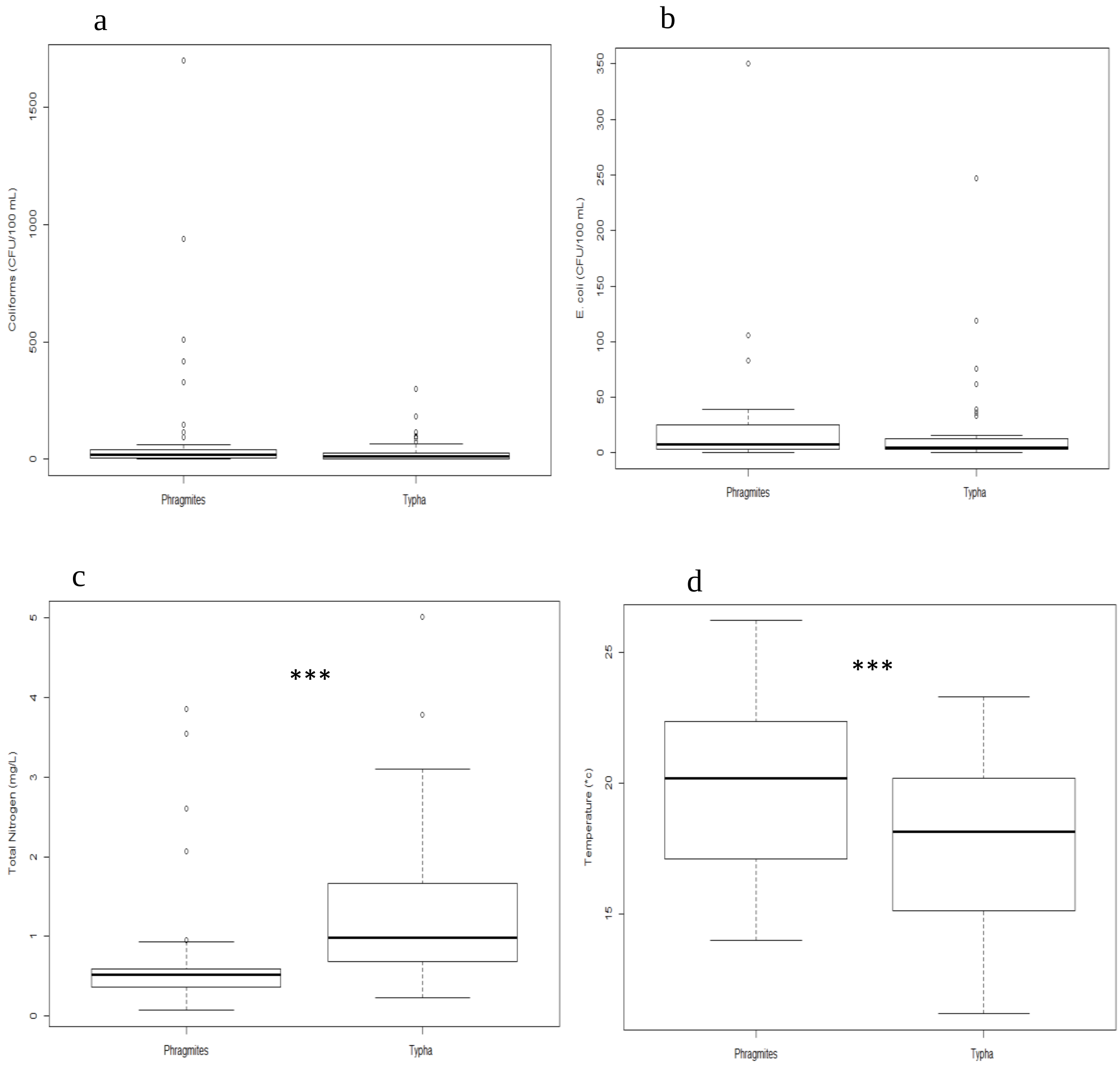

Figure 30. Coliforms (a), *E. coli* (b), total nitrogen (c) and temperature (d) for **outflow** locations of three *Typha latifolia* dominated ponds (2, 3, 13) and three *Phragmites australis* dominated ponds (5, 6, 7). All collection dates for 2018 and 2019 were included, except fall 2019. Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

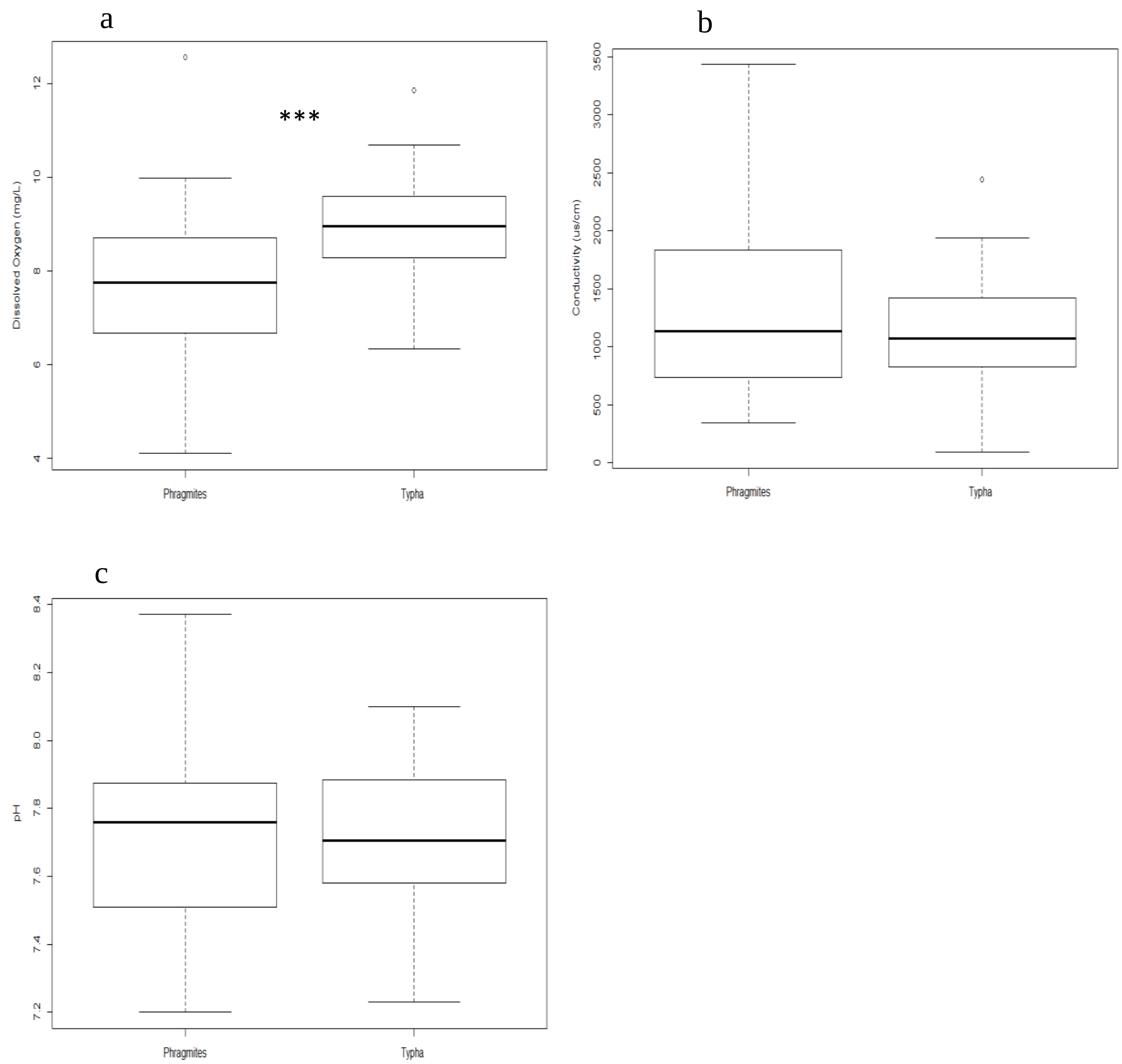

Figure 31. Dissolved oxygen (a), conductivity (b) and pH (c) for **outflow** locations of three *Typha latifolia* dominated ponds (2, 3, 13) and three *Phragmites australis* dominated ponds (5, 6, 7). All collection dates for 2018 and 2019 were included, except fall 2019. Welch two sample t-test, significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

3.3.4. Effect of aquatic plant species richness on outflow water quality

The relationship between plant diversity and water quality at outflow locations may highlight the importance of variable species performance in SMP systems. In this way, the relationship between species richness and outgoing water quality may reveal the effect of community diversity on water treatment. Overall, the diversity of the study ponds is relatively low for both submergent and emergent aquatic plant communities (See Appendix B, Figures B11-B13). A correlation analysis was completed for submergent vegetation species richness and outflow water quality parameters (Figure 32). There are no significant relationships between aquatic submergent plant richness and water quality at the outflow sites.

A correlation analysis was also completed for emergent vegetation (aquatic only) and outflow water quality parameters (Figure 33). There are no significant relationships between aquatic emergent plant richness and water quality at the outflow sites. A correlation analysis was also completed for emergent vegetation (terrestrial and aquatic) and outflow water quality parameters (See Appendix B, Figure B14).

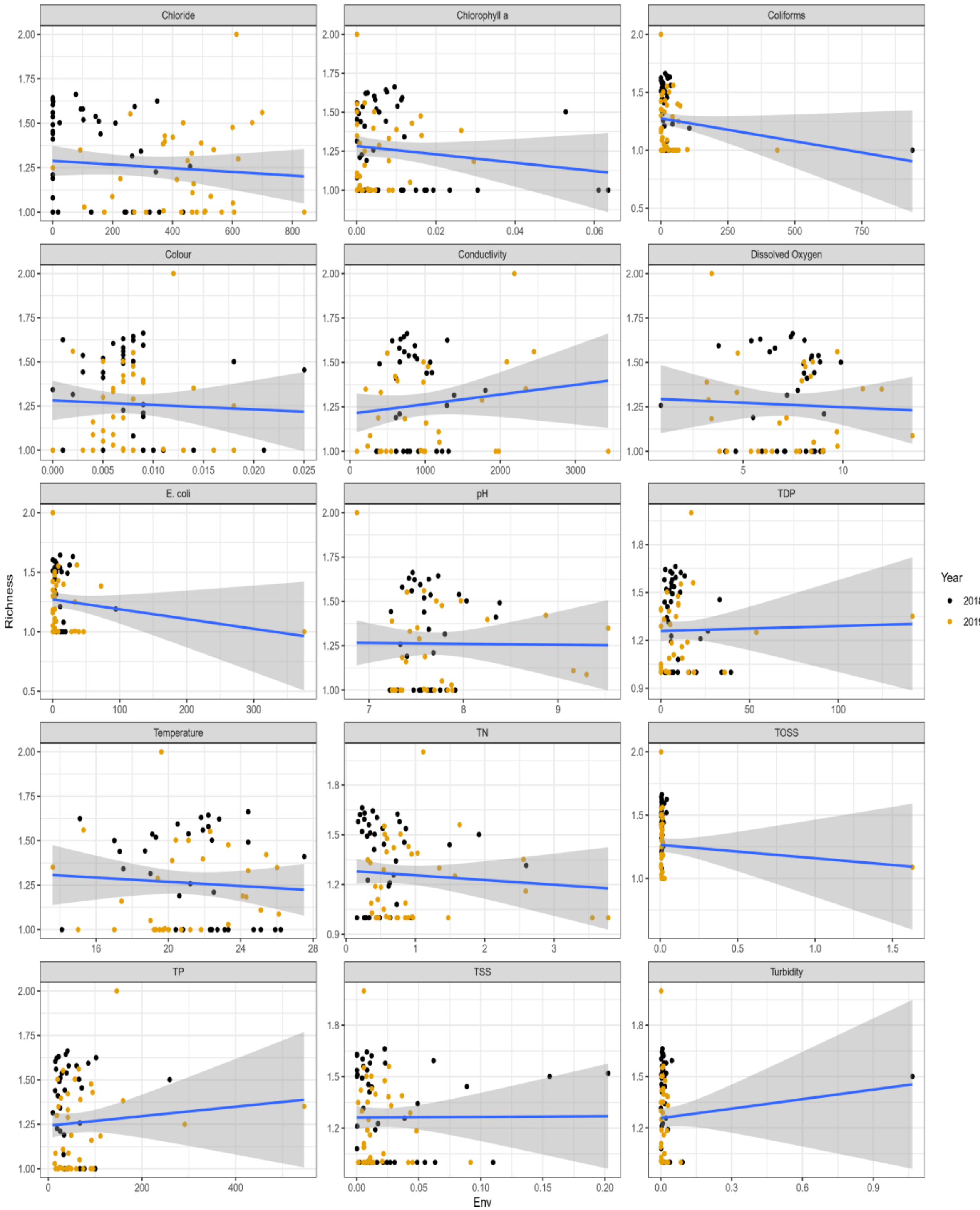

Figure 32. Species richness plots for submergent plant biomass and **outflow** water quality parameters. Significant correlation analyses are denoted by p < 0.05 *, p < 0.01 **, p <0.001 ***.

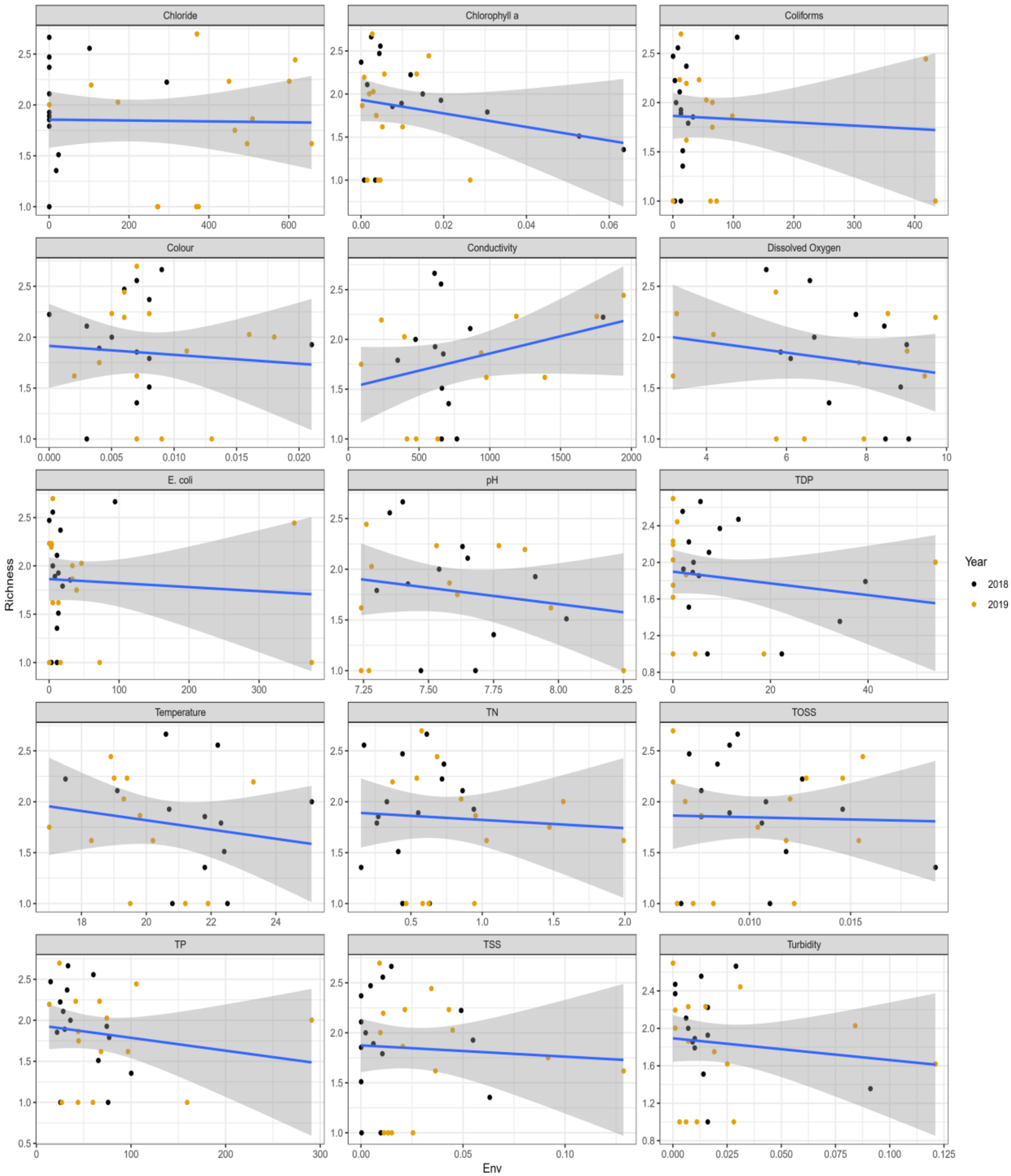

Figure 33. Species richness plots for emergent plant cover (aquatic only) and **outflow** water quality parameters. Significant correlation analyses are denoted by $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

3.4 DISCUSSION

3.4.1 Effect of aquatic plant abundance on outflow water quality

The ability of aquatic vegetation to remove contaminants from freshwater systems has been confirmed by several authors (Marsalek et al., 1992; Fritioff & Greger, 2003). In fact, specific emergent and submergent plant species have been shown to remove pollutants from stormwater in constructed wetlands, and wastewater treatment sites (Marsalek et al., 1992; Weiss et al., 2006; Chang et al., 2012). The communities established within SMPs vary greatly from naturalized freshwater systems, mainly due to the variability in water quality and potential toxicity from urban contaminants. Nonetheless, it is apparent that aquatic vegetation does colonize these artificial systems, and some species may in fact thrive in these variable conditions. Looking at both submergent and emergent aquatic vegetation, it is clear that plant biomass does in fact play a role in determining water quality changes between in and out locations (Table 3). Submergent vegetation plays an essential role in removing nitrogen (and possibly phosphorus, however statistically non-significant) from stormwater runoff (Table 4). This may highlight the ability of submergent aquatic plants to assimilate nitrogen and phosphorus into biomass thereby removing it from the water column. This is consistent with studies on constructed wetlands, which highlight the ability of submergent aquatic plants to reduce phosphorus levels from highly polluted inflow waters (Gu, 2008). This process in urban SMPs may be essential in reducing the exposure of natural systems to excess nutrient levels.

However, emergent vegetation does not show a clear role in improving water quality between in and out locations, but rather may act to decrease water clarity and increase nitrogen concentrations (Table 3). Previous studies have noted weak relationships between emergent vegetation and water quality parameters in SMPs, with exception of temperature which notably decreases with increasing coverage (Vincent & Kirkwood, 2014). However, studies completed on constructed wetlands show the potential for a variety of emergent aquatic species to play crucial roles in nutrient removal (Tanner, 1995). In fact, emergent vegetation is recommended in constructed wetland design to maximize physical (i.e. shoreline stabilization, wind break) as well as biological (i.e. nutrient uptake) water treatment processes (Lee & Scholz, 2006; Leto et al., 2013). As such, further investigation is needed to fully understand the role of emergent aquatic vegetation in water treatment in SMPs.

3.4.2 Effect of emergent plant type on outflow water quality

SMPs are engineered urban systems that differ in terms of their morphometry and hydraulics compared to natural ponds. Due to potentially large fluctuations of incoming water quality, biota that inhabit SMPs must be highly tolerant to changes in nutrient, turbidity, and contaminant levels. Nonetheless, a variety of species have been noted to colonize urban ponds, many of which seem to thrive in these highly variable environments. Many terrestrial species are specifically planted in riparian and flood zones of SMPs to reduce erosion from embankments surrounding the pond (Government of Ontario, 2003). However, planting of aquatic emergent species is very uncommon for both old and new urban SMPs in Canada. In fact, the city of Oshawa has never planted aquatic vegetation in any of their maintained ponds. The aquatic emergent plants found in SMPs all naturally colonize, and therefore tend to be dominated by species that are tolerant to potentially polluted water. The studied Oshawa SMPs show low diversity in terms of their aquatic emergent communities, and tend to be heavily dominated by pioneer species. This includes *P. australis* and *T. latifolia,* both of which are notable competitors in wetland and marsh type environments. However, SMPs may in fact promote the spread of specific invasive species, such as *P. australis,* due to the ever-changing water conditions, which many native plants cannot tolerate.

Emergent aquatic vegetation has been noted to remove contaminants from stormwater via particulate uptake through root systems (Lee & Scholz, 2006; Leto et al., 2013). More importantly however, rooted emergents can also act to stabilize shorelines, ultimately preventing erosion, decreasing water turbidity and limiting contaminant leaching from the soil (Lee & Scholz, 2007). It was noted that increasing abundance of *P. australis,* a highly dominant species in SMP systems, is significantly associated with decreased nitrogen concentrations at outflow locations, ultimately limiting its release into natural systems (Figure 27, Table 16). However, *T. latifolia* which is another dominant species in urban ponds, did not show any significant relationships with outgoing stormwater parameters (Figure 27). *T. latifolia* is commonly used in constructed wetlands and systems used to treat wastewater (Leto et al., 2013). It has been shown to outperform other emergent macrophytes in terms of its nitrogen uptake and ability to produce high biomass yields (Leto et al., 2013). Its heightened performances in treating wastewater is most likely due to its aggressive and competitive nature, as well as its ability to adapt to changing conditions (Leto et al., 2013). Due to the nature of *T. latifolia*, it is also more adapted to monoculture environments (Leto et al., 2013). Nonetheless, no significant trends were noted between *T. latifolia* and outgoing water quality parameters (Figure 27), suggesting further research is needed to fully understand the effect of this species on SMP water quality.

P. australis on the other hand, is one of the most common wetland plants across the globe (Lee & Scholz, 2006). Its stem density, height, and broad salinity tolerance allows it to outgrow native species and thrive in a variety of environments (Meyerson et al., 2000). It has been noted to provide excellent filtration conditions and significantly contribute to nitrogen removal via plant uptake in constructed wetlands (Lee & Scholz, 2006). Comparing sites that are dominated by these two species, ponds with high *P. australis* coverage have significantly lower suspended solids, nitrogen, and dissolved oxygen levels at outflow sites (Figures 28, 29, 30, 31). *T. latifolia* dominated systems however, have overall lower temperatures and chlorophyll α concentrations at outflow locations. Based on these findings, it appears that SMPs dominated by *P. australis*, rather than *T. latifola*, may provide greater water treatment for incoming stormwater.

Looking at established community structures reveals overarching patterns to outflow water quality. It seems that ponds with established communities including *L. salicaria, S. latifolia*, and *S. acutus*, show decreasing trends of phosphorus and chloride at outflow locations (Figure 25). Similarly, ponds with communities including *C. aquatilis, A. plantago-aquatica,* and *T. latifolia* show decreases in water temperature and turbidity at outflow sites (Figure 25). This may highlight the potential for specific emergent plant communities to improve stormwater treatment. It has been noted that with a greater variety of species, differences between taxa can be advantageous in habitats with changing environmental characteristics (Leto et al., 2013), such as SMPs. High levels of diversity can also ensure that maximal water treatment processes can occur due to variations in purification capacities between species (Leto et al., 2013). It has been previously noted that in constructed wetlands, designed to mitigate wastewater, nutrient removal was improved with a greater diversity of aquatic macrophytes (Greenway, 2005). However, there was no significant relationship noted between increased emergent species richness and outflow water quality parameters in the studied ponds (Figure 33). In this way, the introduction of a greater variety of emergent species in SMP systems, may not significantly improve stormwater treatment processes in these ponds.

3.4.3 Effect of submergent plant type on outflow water quality

Submergent aquatic vegetation has been noted to naturally colonize urban ponds, even though SMP design manuals suggest seasonal planting (Government of Ontario, 2003). Similar to aquatic emergent plants, submergent macrophytes tend to show little variation in terms of community profiles. This being that many urban ponds are densely populated by one or two dominant species and show low overall diversity (Figure 19). These plants tend to be tolerant to eutrophic conditions, and are capable of colonizing freshly disturbed sites, such as following sediment maintenance via dredging. In early summer months, the majority of studied SMPs show communities well established by *S. pectinata*, *P. pusillus*, or *Chara*. Charophytes especially have been noted as good colonizers which are tolerant to poor water conditions (Lambert-Servien et al., 2006). The early establishment of these species in SMPs may represent poor water conditions following snowmelt and spring washouts. Progressing through the summer, a notable shift in community structure is made towards ponds dominated by *N. flexilis* and *N. guadalupensis*. Both species are members of the water nymph family, and have been documented throughout North America. In marshes of the Great Lakes, *N. flexilis* was noted as being intolerant to high nutrient and turbid conditions (Lougheed et al., 2001). Furthermore, disappearance of *N. flexilis* in freshwater lakes was shown to be the results of eutrophication (Wingfield et al., 2006). Within the studied Oshawa SMPs, these plants are established later in the sampling season, and may be reflective of improvements in water quality as the summer progresses, and plant biomass increases.

It has been recorded that within SMPs, two main types of water treatment via submergent vegetation can occur. This includes contaminant removal from suspended plant biomass, as well as rooted vegetation (Marsalek et al., 1992). Rooted plants are capable of facilitating pollutant adsorption, as well as uptake through both the plant-sediment and plant-water interface (Marsalek et al., 1992). Free-floating macrophytes have also been noted as an effective way to directly remove nutrients from stormwater (Chang et al., 2012). Specific submergent plant species can be associated with changes in water quality at the outflow locations. Both *P. natans* and *Chara* are associated with higher levels of suspended solids at outflow sites (Figure 26). Interestingly, *N. flexilis* is associated with decreasing chloride and nitrogen concentrations at outflow locations (Figure 26, Table 8). This water nymph species has been shown to not tolerate poor water conditions, especially environments with excess pollution (Lougheed et al., 2001; Wingfield et al., 2006). This particular species may be recognized as an indicator in SMPs, highlighting locations where sufficient water treatment processes are occurring or potential locations where incoming stormwater is less polluted.

Similarly, looking at community structures of submergent plants and outflow water quality, specific patterns can be recognized. Ponds with communities composed of *P. folliosus, N. guadalupensis,* and *N. flexilis* show increased dissolved oxygen concentrations, but also decreases in nutrient and chloride concentrations (Figure 24). These plants share common preferred environments, in which high pollution is not tolerated (Lougheed et al., 2001; Wingfield et al., 2006). The presence of these species in SMPs may highlight the ability of specific ponds to efficiently treat stormwater runoff.

3.4.4 Effect of species richness on outflow water quality

Biodiversity has previously been recognized as an essential component to maximizing ecosystem productivity in freshwater systems (Schultz et al., 2011). However, recently studies have suggested that plant composition is a stronger predictor for ecosystem productivity compared to species diversity in freshwater wetlands (Schultz et al., 2011). Plants are an essential component of aquatic environments, as they provide food and habitat for wildlife. The overall diversity of aquatic vegetation in Oshawa SMPs is relatively low. Unfortunately, the variations in water quality at these sites results in environments that few species can tolerate, some of which are invasive. Surprisingly, only two invasive aquatic plants were recorded for this study including *P. australis* and *L. salicaria*. Both of these species are considered emergent macrophytes and are common to North American freshwater systems. Nonetheless, a small subset of both submergent and emergent native species seem to be able to tolerate SMP environments. It was noted that species richness for both aquatic emergent and submergent vegetation has no significant impact on water quality variables at outflow locations (Figures 32, 33). However, this may be the result of very low species richness levels across all study ponds. In this way, since diversity of aquatic plants is low across the 15 study sites, no relationship to outflow water quality is recognized. Future studies should look to assess the impact of increased species richness on SMP systems, and the influence of biodiversity levels on water treatment.

3.5 CONCLUSION

Overall, it appears that the range of aquatic plant profiles for the selected SMPs does not vary greatly across sites or between sampling years. The abundance of aquatic emergent vegetation does not significantly influence outflow water quality, with the exception of decreasing water temperature with increased emergent cover. Aquatic emergent vegetation for all studied SMPs is heavily dominated by *T. latifolia* and *P. australis*, with little variation in terms of community structure across the two-year sampling period. The invasive species *P. australis* shows potential to limit nutrient concentrations, specifically nitrogen at outflow sites. However, overall species richness of emergent vegetation does not show any significant relationships with outflow water quality parameters.

Submergent macrophyte biomass is significantly associated with decreasing nitrogen concentrations at outflow sites. These communities tend to be dominated by two to three native species throughout the summer, however notable shifts in community structures does occur. The studied SMPs show seasonal shifts from pollutant tolerant species (i.e. *Chara* and *P. pusillus*) to pollutant intolerant species (i.e. *N. flexilis* and *N. guadalupensis*). These seasonal changes may reflect gradual improvements in water quality as aquatic plant biomass increases. Species richness of submergent aquatic plants is not significantly related to any outflow water quality parameters. However, limited diversity and species richness across the 15 study sites may have been unable to illustrate the full potential of water treatment in SMPs which have a greater variety of plants. In this way, municipalities should look to include regular aquatic planting regimes in their annual maintenance practices of local urban ponds. By regularly monitoring both water quality and plant accumulation (both type and abundance), municipalities can ensure a variety of native plants remain established in these ponds in order to increase stormwater treatment.

CHAPTER 4: THE ROLE OF HABITAT CONDITIONS IN STRUCTURING STORMWATER MANAGEMENT POND MACROPHYTE COMMUNITIES

4.1 INTRODUCTION

Aquatic environments can face a number of factors that alter and structure their established communities. In freshwater systems, such as ponds and wetlands, aquatic plants make up an important level in community composition. Macrophytes can function as indicators of water quality and highlight underlying effects from external sources. SMPs have been noted to support a variety of aquatic plants, and may function as essential sources of biodiversity in areas where natural ponds are lost due to urbanization (Casey et al., 2006; Gallagher et al., 2011; Miró et al., 2018). However, urban SMPs are designed to mitigate impacts of stormwater, and therefore can receive large varieties and loads of pollutants. Furthermore, pond characteristics, location, and composition of the surrounding landscape can have significant impacts on the communities of macrophytes established in these systems.

In freshwater lakes and ponds, the main drivers of aquatic plant biomass include light penetration, sediment substrate chemistry, lake morphometry, and trophic status (Duarte et al., 1986). Although SMPs are much smaller in size, aquatic plants are likely controlled by the same environmental factors in natural lentic systems. Firstly, morphometry of the water body can have significant effects on macrophyte establishment. In this case, depth, perimeter, and area all control the amount of available habitat for plant colonization (Duarte et al., 1986). More shallow water bodies with gradual slopes will allow for greater light penetration, therefore increasing growth potential of plants (Duarte et al., 1986). Overall, increased pond size results in increased habitat availability for plants, and therefore increases to biomass and diversity.

The trophic status of a freshwater system may also drive plant abundance. Although nutrients are essential for macrophyte growth, excess loads of phosphorus to aquatic systems can result in eutrophication. In this way, a shift from macrophyte dominated to phytoplankton dominated communities can occur (Balls et al., 1989; Bakker et al., 2010). These highly productive states will also result in decreased light penetration and therefore reduced macrophyte growth (Duarte et al., 1986). Eutrophic conditions of freshwater systems has been directly linked to surrounding land use, including drainage area and impervious surface levels (Robertson & Saad, 2011; Soranno et al., 2015). In this way, with increased drainage area, runoff is given a longer time to accumulate particulates and pollutants before being washed into naturalized systems, or in many cases SMP facilities. Excess loading of nutrients, and other constituents into these environments can greatly alter macrophyte establishment.

Specific water profiles and environment characteristics may also allow for the establishment of invasive species. Invasive species can result in decreases to biodiversity, productivity, and alterations to habitat structure (Zedler & Kercher, 2004). Urban freshwater environments, such as SMPs have been noted to enhance the spread of invasive species in two main ways. Firstly, many invasive species take advantage of existing habitats which are disturbed by human activities in urban settings (such as dredging). These disturbances can include excess garbage and pollution, as well as decreased permeability of the surrounding landscape (Hassall, 2014). Secondly, it has been argued that wetlands, similar to urban ponds, function as 'sinks' which accumulate a variety of material from runoff, including nutrients, salts, suspended solids, and metals (Zedler & Kercher, 2004). These conditions combined with frequent disturbances of the surrounding landscape, make for optimal environments for the invasion of non-native species (Zedler & Kercher, 2004). In this way, both the quality of incoming stormwater and characteristics of the surrounding landscape may function as drivers for the establishment of invasive species.

This chapter focuses on role of SMP habitat conditions in structuring aquatic plant communities in 15 SMPs studied in Oshawa, Ontario. Habitat conditions include, drainage area characteristics, pond size and dimensions, and water quality profiles. In natural systems, habitat conditions are known to influence plant communities, therefore it was hypothesized that the unique habitat features of SMPs would influence aquatic plant abundance and composition. It was predicted that plant diversity would decrease with increasing water quality degradation in SMPs (i.e. increased chloride, conductivity, turbidity, nutrients, and decreased dissolved oxygen). Additionally, it was hypothesized that the amount of impervious surface cover in the watershed would cause higher amounts of contaminated run-off to enter SMPs, and as such, predicted that aquatic plant diversity would decrease in response to increased impervious surface cover. Finally, characterization of invasive species occurrence in SMP aquatic plant communities was completed. Urban habitats are known to expedite the dispersal and spread of invasive species, therefore understanding the role of SMPs in invasive species dynamics is important for their management.

4.2 MATERIALS AND METHODS

4.2.1 Study sites and sampling methods

See Chapter 2, section 2.2 Materials and Methods for complete description of study sites and water sampling methods used. See Chapter 3, section 3.2 Materials and Methods for complete description of vegetation sample collection protocols.

4.2.2 Data analysis

All correlation analyses, and multiple linear regressions were completed using RStudio v1.1.463 (RStudio, Boston, USA). All constructed canonical correspondence analyses were completed using Paleontological Statistics (PAST) version 4.0 (Hammer *et al.*, 2001). All water quality parameters and biological data were non-normal, and thus were transformed to improve normality, when possible. All other parametric assumptions were met, therefore due to the robustness of such a large dataset, parametric tests were used. For multivariate ordination analyses, water quality parameters were center-standardized.

4.3 RESULTS

4.3.1 Effect of inflow water quality and pond characteristics on aquatic plant abundance

To determine potential relationships between aquatic plant amount and incoming stormwater quality, correlation and multiple linear regression analysis was used. Pearson correlation analysis was completed for all inflow water quality parameters and aquatic plant (submergent and emergent) abundances (Table 19). Increased emergent plant coverage is significantly associated with decreased coliform and *E. coli* levels at the inflow location. Aquatic emergent cover is also marginally related to decreased water temperature at inflow sites. Submergent plant biomass is also significantly related to a number of inflow water quality parameters. Increased submergent vegetation is significantly correlated to decreasing colour, temperature, and pH at inflow locations. Submergent biomass is also significantly related to increased nitrogen, and marginally associated with increased conductivity at inflow sites.

A multiple linear regression analysis was completed for submergent plant biomass and inflow water quality parameters (Table 20). Selected independent variables were based on significant and marginally significant relationships from the completed Pearson correlation (Table 19). All non-significant variables from the multiple linear regression were removed to establish the final model (Table 20). Increased submergent plant biomass can partially be explained (8.3%) by decreasing colour and pH at inflow locations.

Table 19. Pearson correlation analysis between **inflow** water quality parameters and aquatic plant abundance (percent emergent cover and submergent biomass). Dates for emergent plant cover included August 2018 and 2019 only, dates for submergent biomass included all dates when submergent vegetation was sampled in 2018 and 2019. Significant relationships are bolded.

	Percent Emergent Cover		**Submergent Biomass (g/m^2)**	
Parameter	**Cor.**	**p-value**	**Cor.**	**p-value**
Colour (A @ 440 nm)	-0.159	0.403	-0.27	**0.01**
Turbidity (A @ 750 nm)	-0.195	0.301	0.167	0.115
Total Suspended Solids (g/L)	-0.158	0.404	0.102	0.339
Total Organic Suspended Solids (g/L)	-0.119	0.532	0.143	0.179
Total Coliforms (CFU/100 mL)	-0.412	**0.024**	0.031	0.769
Total E. coli (CFU/100 mL)	-0.361	**0.049**	0.088	0.408
Chlorophyll α (g/L)	-0.120	0.526	0.147	0.167
Chloride (mg/L)	0.176	0.352	0.162	0.127
Conductivity (µs/cm)	0.224	0.235	0.177	0.095*
Total Phosphorus (µg/L)	-0.237	0.207	-0.06	0.573
Total Dissolved Phosphorus (µg/L)	-0.141	0.457	-0.005	0.965
Total Nitrogen (mg/L)	0.086	0.649	0.213	**0.044**
Temperature (°c)	-0.335	0.07*	-0.234	**0.026**
Dissolved Oxygen (mg/L)	0.183	0.334	-0.018	0.864
pH	-0.177	0.349	-0.218	**0.039**

* Not significant, but notable relationships.

Table 20. Multiple linear regression for submergent plant biomass predicted by **inflow** colour, and pH. Final model: Multiple $R^2 = 0.104$, adjusted $R^2 = 0.083$, F-statistic 5.047, DF = 87, p-value = 0.008.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	6022	2998	2.01	0.048
Colour	-92001	39300	-2.34	0.022
pH	-5520	3191	-1.73	0.087

To determine potential relationships between aquatic plant amount and specific pond characteristics, correlation and multiple linear regression analysis was used. Pearson correlation analysis was completed for defining pond characteristics and aquatic plant (submergent and emergent) abundances (Table 21). Increased emergent plant cover is significantly correlated with decreased impervious levels of the surrounding drainage area. Increased submergent plant abundance is significantly related to decreased pond width and pond age. Submergent plant biomass is also significantly correlated to increased pond length and total pond area.

A multiple linear regression analysis was completed for submergent plant biomass and defining pond characteristics (Table 22). Selected independent variables were based on significant relationships from the completed Pearson correlation (Table 21). All non-significant variables from the multiple linear regression were removed to establish the final model (Table 22). Submergent plant biomass can be partially explained (34.2%) by decreased pond width, as well as increased pond length and total area.

A multiple linear regression analysis was completed for emergent plant cover and defining pond characteristics (Table 23). Selected independent variables were based on significant relationships from the completed Pearson correlation (Table 21). All non-significant variables from the multiple linear regression were removed to establish the final model (Table 23). Increased emergent vegetation abundance can be marginally explained (10.6%) by decreased impervious level of the surrounding SMP watershed.

Table 21. Pearson correlation analysis between pond characteristics and aquatic plant abundance (percent emergent cover and submergent biomass). Dates for emergent plant cover included August 2018 and 2019 only, dates for submergent biomass included all dates when submergent vegetation was sampled in 2018 and 2019. Due to missing information only 7 SMPs were included for impervious surface levels, and only 13 SMPs were included for pond depth. Significant relationships are bolded.

	Percent Emergent Cover		**Submergent Biomass (g/m^2)**	
Parameter	**Cor.**	**p-value**	**Cor.**	**p-value**
Width (m)	-0.244	0.195	-0.246	**0.019**
Length (m)	-0.066	0.728	0.321	**0.002**
Depth (m)	0.214	0.257	0.005	0.964
Perimeter (m)	-0.229	0.223	0.172	0.106
Total Area (m^2)	-0.211	0.262	0.21	**0.047**
Percent Impervious Level	-0.37	**0.044**	-0.155	0.326
Drainage Area (ha)	-0.073	0.702	-0.14	0.187
Age (years)	0.051	0.787	-0.232	**0.028**

Table 22. Multiple linear regression for submergent plant biomass predicted by pond width, length, and area. Final model: Multiple $R^2 = 0.364$, adjusted $R^2 = 0.342$, F-statistic 16.4, DF = 86, p-value = < 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	647.91	207.49	3.12	0.002
Width	-22.1	3.8	-5.81	< 0.001
Length	5.13	2.65	1.94	0.056
Total Area	0.12	0.05	2.27	0.026

Table 23. Multiple linear regression for emergent plant cover predicted by pond impervious level. Final model: Multiple $R^2 = 0.137$, adjusted $R^2 = 0.106$, F-statistic = 4.45, DF = 28, p-value = 0.044.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	91.63	33.59	2.73	0.011
Impervious Level	-1.44	0.68	-2.11	0.044

4.3.2 Effect of inflow water quality and pond characteristics on aquatic plant type

A canonical correspondence analysis (CCA) was completed for submergent plant biomass (with rare species removed) and inflow water quality parameters (Figure 34). In this case, all rare species which were responsible for less than 1% of the total submergent plant biomass, were removed from analysis. Based on the short lines represented by water quality parameters in the CCA biplot, inflow water quality of the study ponds has low variability. However, it appears that specific plant communities may be well defined by various inflow characteristics. Ponds with notably high abundances of *P. foliosus, Chara,* and *S. pectinate* tend to have higher nutrient (phosphorus and nitrogen) and salt concentrations at inflow locations, including ponds 1 and 8. However, ponds dominated by *P. pusillus, N. guadalupensis,* and *N. flexilis* tend to be located in ponds with characteristically high inflow temperatures and turbidity levels, such as ponds 14 and 15.

A canonical correspondence analysis was also completed for emergent aquatic vegetation and inflow water quality parameters (Figure 35). There is limited spread in terms of the variation across inflow water quality parameters, however a well-defined gradient can be noted across axis 1. Emergent plant communities dominated by *P. australis, P. lapathifolia,* and *C. aquatilis* are defined by incoming stormwater that is high in dissolved oxygen, temperature, and pH. These community profiles can be seen in ponds 4, 6, and 14. On the other hand, ponds well established by *T. latifolia*, *S. acutus,* and *S. latifolia*, are associated with increased nutrients (nitrogen and phosphorus), as well as suspended solids and chloride at inflow locations. Ponds 8, 9 and 13 are well defined by these types of emergent community structures.

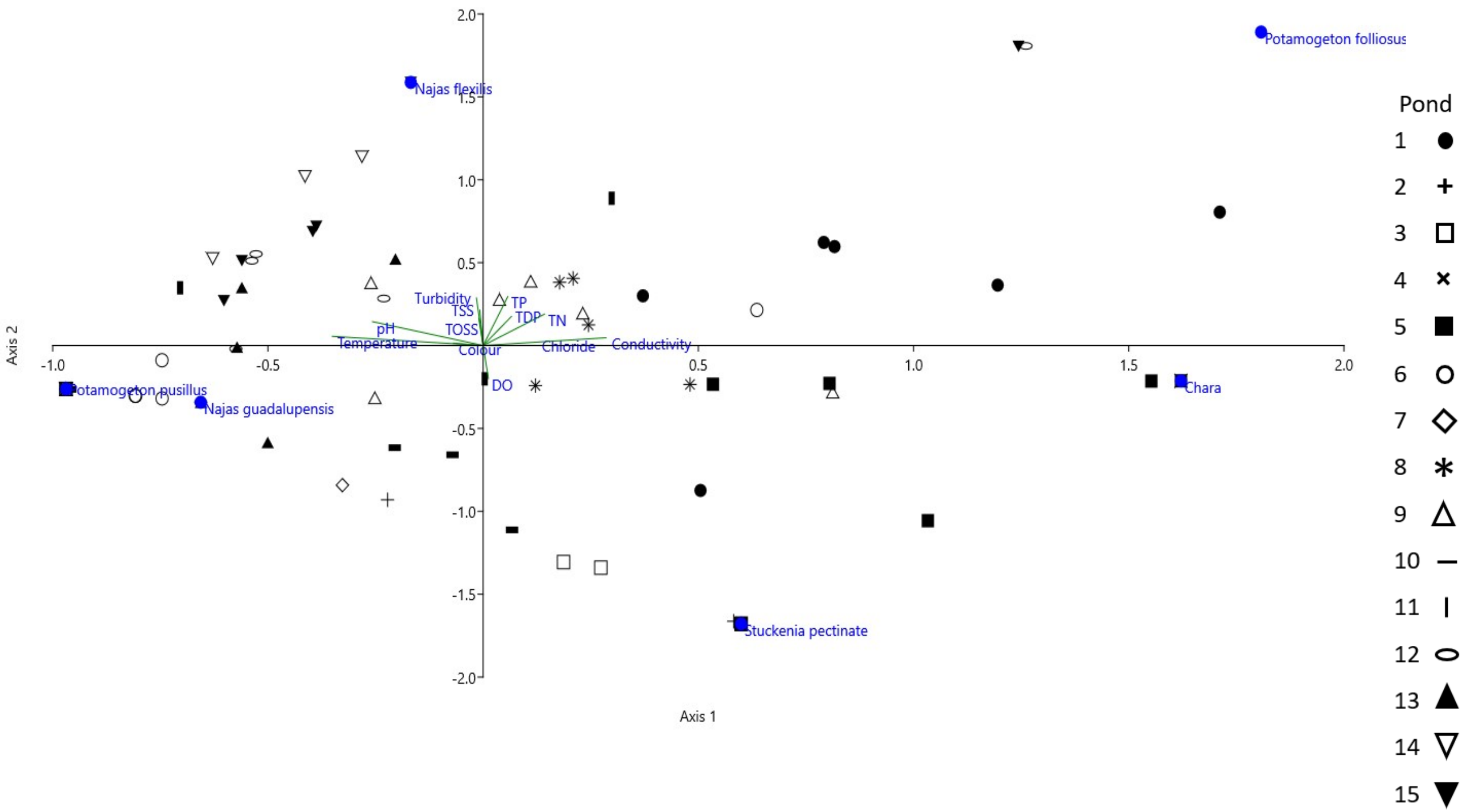

Figure 34. Canonical correspondence analysis for submergent plant biomass (excluding rare species) and **inflow** water quality parameters for 2018 and 2019. Axis 1 represents 33.82% of the variation and axis 2 represents 29.68% of the variation.

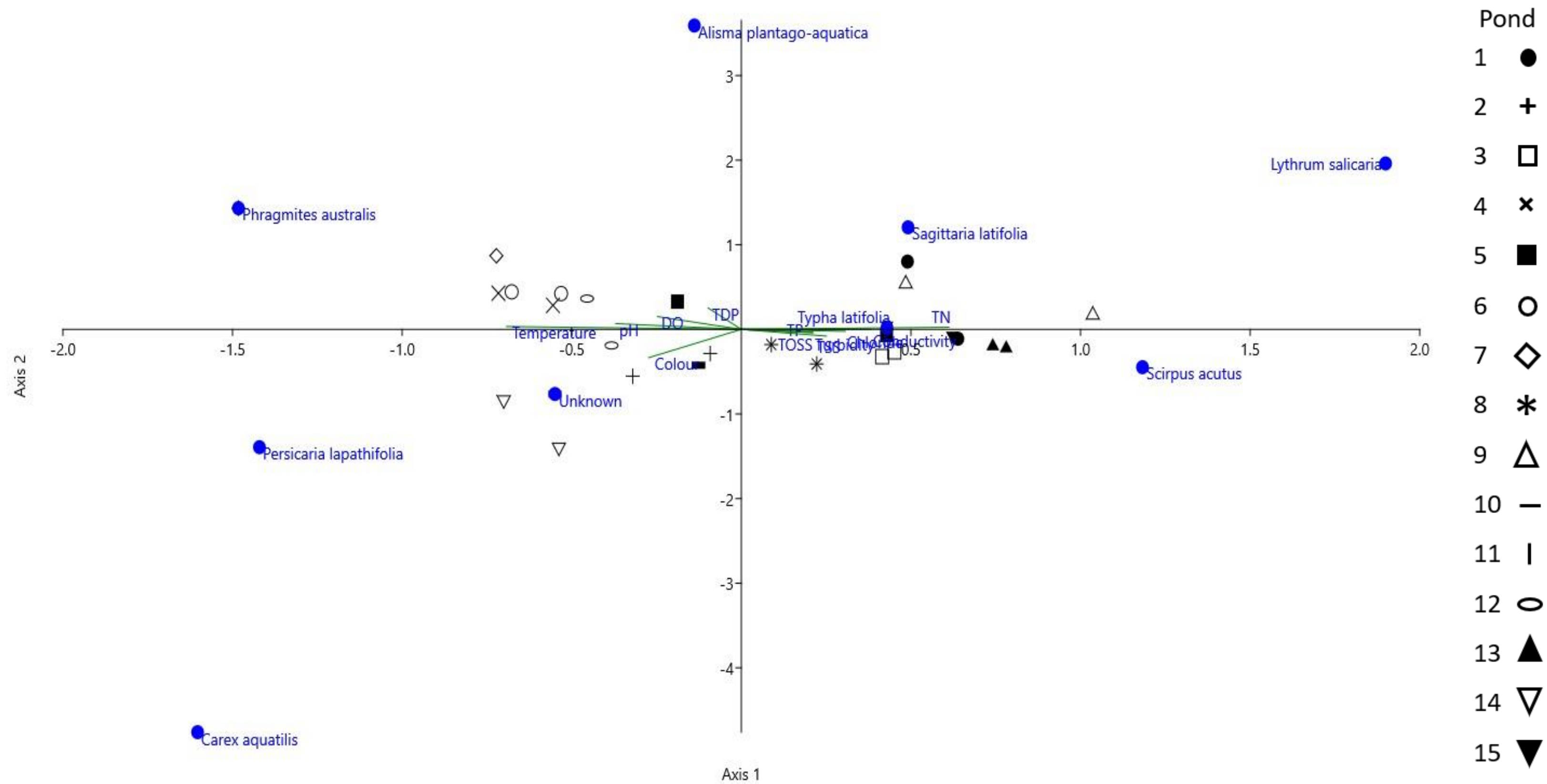

Figure 35. Canonical correspondence analysis for emergent plant cover and **inflow** water quality parameters for August 2018 and 2019. Axis 1 represents 37.53% of the variation and axis 2 represents 27.5% of the variation.

In order to determine potential relationships between incoming water quality and the presence of specific aquatic plant species, correlation and multiple linear regression analysis was used. A Pearson correlation was completed for submergent vegetation and inflow water quality parameters (Figure 36). A variety of submergent species are associated with multiple water quality variables. Within the study SMPs, *Chara* biomass is positively correlated with increased chloride, conductivity, and nitrogen concentrations at inflow sites. Like *Chara*, N. *flexilis* is common to many of the studied urban ponds. Increasing *N. flexilis* abundance is strongly associated with decreases to water clarity (i.e. increased turbidity, suspended solids, and organic suspended solids). Its presence is also significantly related to decreased oxygen concentrations of incoming stormwater. *P. pusillus* is a native free-floating macrophyte which is common to a variety of the study systems. Increasing biomass of *P. pusillus* is associated with increased dissolved phosphorus and temperature, as well as decreased chloride and conductivity at inflow locations. A number of low frequency submergent species also show significant relationships with specific water quality variables. *P. folliosus* for example is a submergent macrophyte which has leaves that float on the waters surface. Increased abundance of *P. folliosus* is significantly correlated to decreased coliform levels of incoming stormwater

A multiple linear regression analysis was completed for inflow water quality parameters and submergent plant species showing significant correlations with two or more variables. Selected independent variables were based on significant relationships from the completed Pearson correlation (Figure 36). All non-significant variables from the multiple linear regression were removed to establish the final model. Increased *Chara* biomass can be partially explained (11.1%) by decreased temperatures of incoming stormwater (Table 24). *N. flexilis* abundance can be marginally explained (13.4%) by increased suspended solids and decreased oxygen concentrations of incoming surface waters (Table 25). Finally, *P. pusillus* biomass can be partially explained (6.4%) by increased temperatures at inflow locations (Table 26).

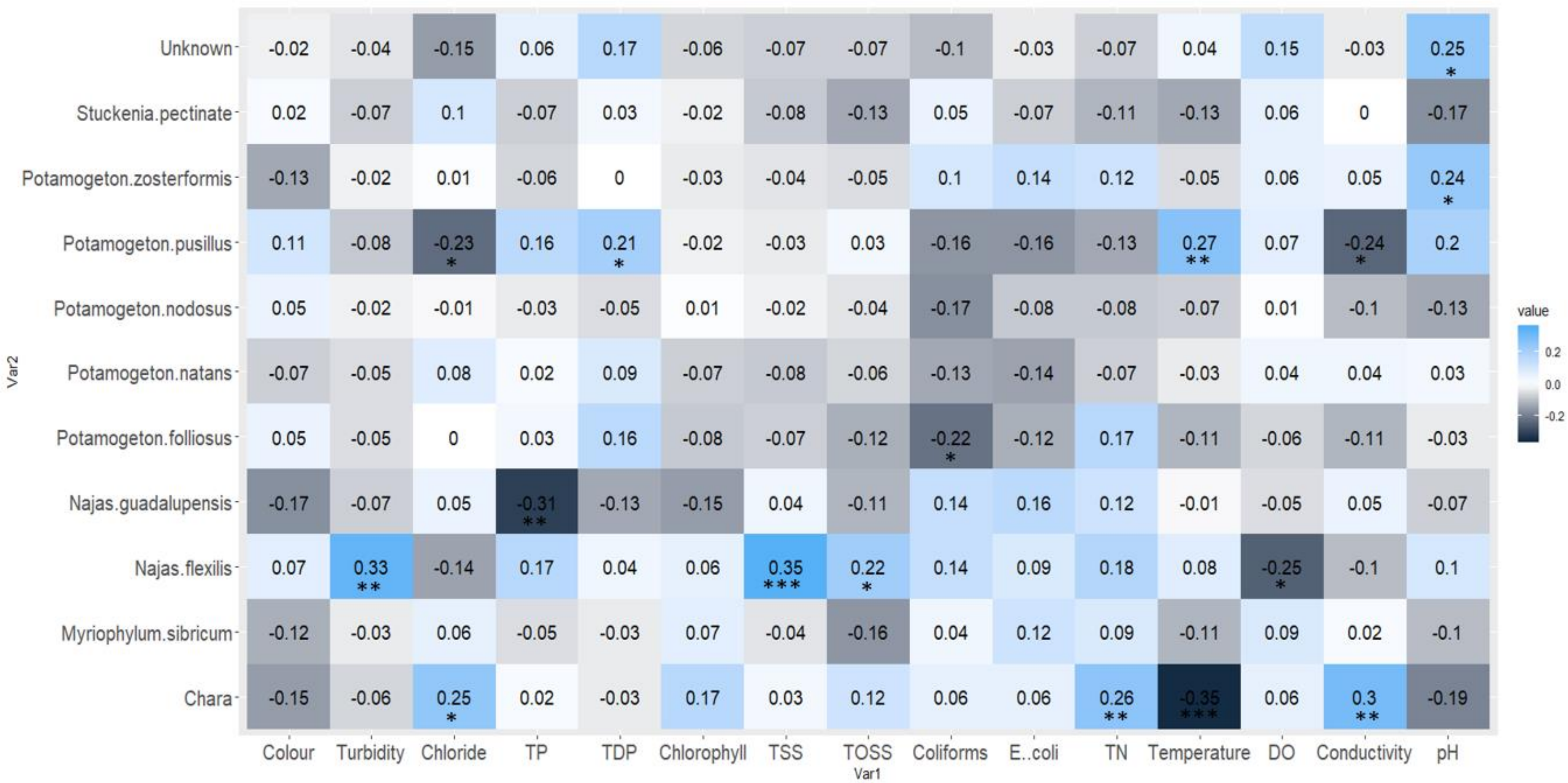

Figure 36. Pearson correlation for submergent plant abundance and **inflow** water quality parameters from 2018 and 2019. Significant relationships are denoted with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

Table 24. Multiple linear regression for total biomass of submergent plant species *Chara* predicted by **inflow** temperature. Final model: Multiple $R^2 = 0.121$, adjusted $R^2 = 0.111$, F-statistic 12.07, DF = 88, p-value = < 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	4.72	1.24	3.81	< 0.001
Temperature	-3.26	0.94	-3.47	< 0.001

Table 25. Multiple linear regression for total biomass of submergent plant species *Najas flexilis* predicted by **inflow** total suspended solids and dissolved oxygen. Final model: Multiple $R^2 = 0.154$, adjusted $R^2 = 0.134$, F-statistic 7.91, DF = 87, p-value = < 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	1.21	0.49	2.5	0.014
Total suspended solids	7.95	2.58	3.08	0.003
Dissolved Oxygen	-0.87	0.49	-1.76	0.083

Table 26. Multiple linear regression for total biomass of submergent plant species *Potamogeton pusillus* predicted by **inflow** temperature. Final model: Multiple $R^2 = 0.075$, adjusted $R^2 = 0.064$, F-statistic 7.102, DF = 88, p-value = 0.009.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	-3.25	1.58	-2.06	0.042
Temperature	3.18	1.19	2.67	0.009

A Pearson correlation analysis was also completed for aquatic emergent vegetation and water quality parameters at the inflow locations for all 15 SMPs (Figure 37). A variety of aquatic emergent species show significant trends with outflow water quality variables. *C. aquatilis* abundance in SMPs is associated with increasing colour, temperature, and pH of incoming stormwater. Increasing cover of this emergent species is also correlated with decreased chloride concentrations at inflow locations. *T. latifolia* is a common wetland species, native to North America. Its presence in urban ponds is related to decreased colour and temperature, as well as increased nitrogen and conductivity at inflow sites. Along with *T. latifolia, P. australis* is very common to urban stormwater facilities. Its abundance in SMPs is strongly associated with increased dissolved phosphorus concentrations of incoming stormwater.

A multiple linear regression analysis was completed for inflow water quality parameters and aquatic emergent plant species showing significant correlations with two or more variables. Selected independent variables were based on significant relationships from the completed Pearson correlation (Figure 37). All non-significant variables from the multiple linear regression were removed to establish the final model. Increased *C. aquatilis* abundance in urban ponds can be partially explained (24.2%) by increased colour and pH at SMP inflows (Table 27). Increased biomass of *T. latifolia* can also be marginally explained (27.3%) by decreased colour and increased conductivity of incoming stormwater runoff (Table 28).

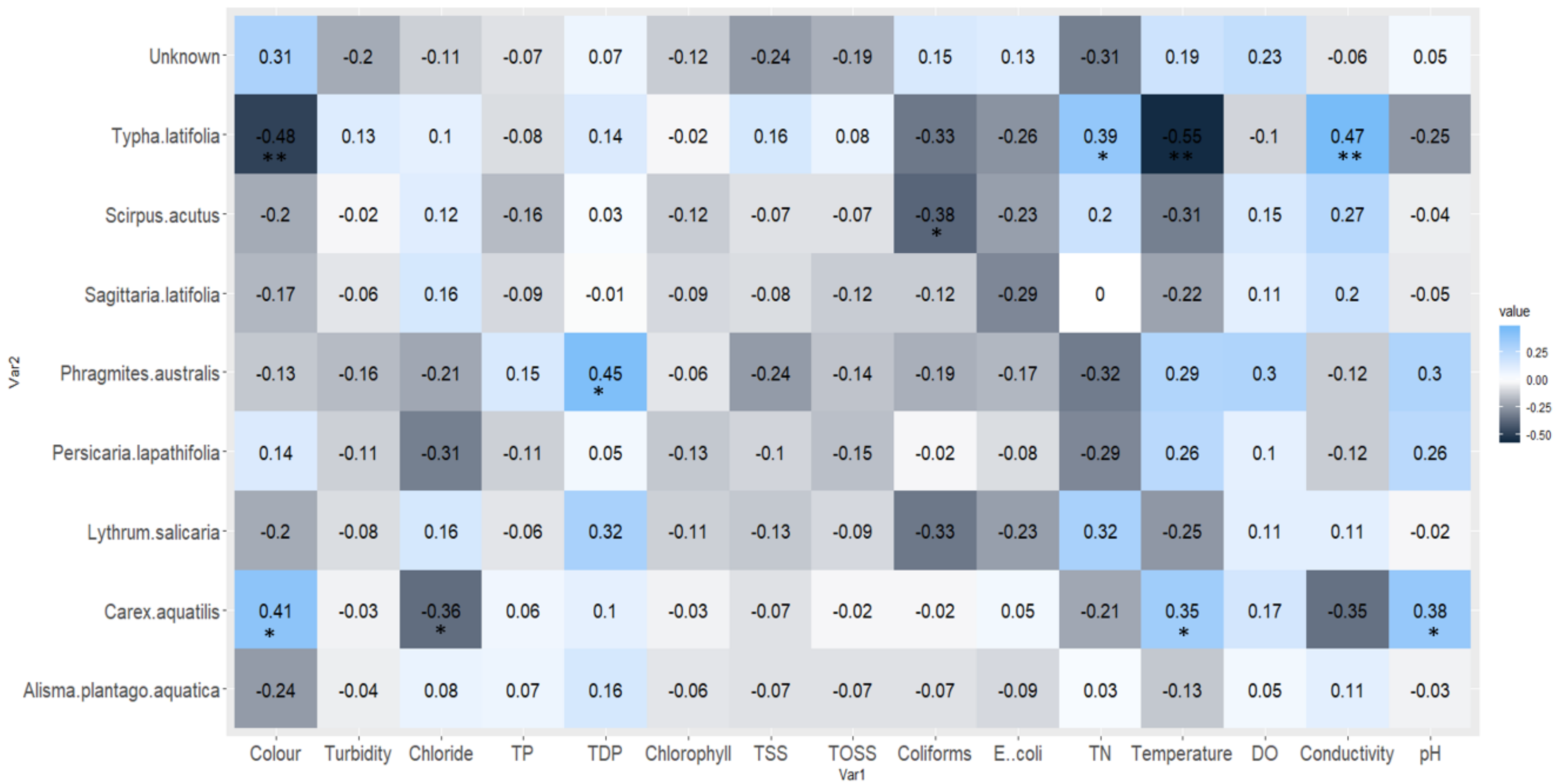

Figure 37. Pearson correlation for emergent plant cover and **inflow** water quality parameters from 2018 and 2019. Significant relationships are denoted by p < 0.05 *, p < 0.01 **, p < 0.001 ***.

Table 27. Multiple linear regression for relative abundance of aquatic emergent plant species *Carex aquatilis* predicted by **inflow** colour, and pH. Final model: Multiple $R^2 = 0.294$, adjusted $R^2 = 0.242$, F-statistic 5.62, DF = 27, p-value = 0.009.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	-2.05	0.9	-2.27	0.032
Colour	37.11	15.71	2.36	0.026
pH	2.11	0.96	2.2	0.036

Table 28. Multiple linear regression for relative abundance of aquatic emergent plant species *Typha latifolia* predicted by **inflow** colour, and conductivity. Final model: Multiple $R^2 = 0.323$, adjusted $R^2 = 0.273$, F-statistic 6.44, DF = 27, p-value = 0.005.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	-0.38	0.98	-0.39	0.701
Colour	-112.39	56.92	-1.97	0.059
Conductivity	0.59	0.31	1.95	0.062

In order to determine potential relationships between pond design characteristics and the abundance of specific aquatic plant species, correlation and multiple linear regression analysis was used. A Pearson correlation was completed for submergent vegetation and defining pond characteristics (Figure 38). The presence of specific submergent species seems to be well correlated with pond design variables. *Chara* biomass is negatively associated with pond depth, width, and impervious level of the surrounding watershed. On the other hand, *N. guadalupensis* which is a member of the water nymph family, is significantly associated with increasing pond size (specifically pond depth and perimeter). Another member of the water nymph family, *N. flexilis*, shows increased abundance with decreased pond width and age. Pond size also plays a critical role in defining abundance of *P. pusillus*. In this case, increasing *P. pusillus* biomass is correlated with increasing pond width, length, perimeter, total area, and age. This species is also significantly associated with decreased pond depth. *P. zosterformis* shows similar trends to *P. pusillus*, in that decreased pond depth and increased age, is significantly associated to increases is plant abundance.

A multiple linear regression analysis was completed for specific pond characteristics and submergent plant species showing significant correlations with two or more variables. Selected independent variables were based on significant relationships from the completed Pearson correlation (Figure 38). All non-significant variables from the multiple linear regression were removed to establish the final model. Increased *Chara* biomass in the studied SMPs can be significantly explained (19.4%) by decreased pond width, depth, and impervious level (Table 29). Total biomass of *N. flexilis* in the studied ponds can be partially explained (25.9%) by decreased pond width and age (Table 30). *N.*

guadalupensis abundance can also be partially explained (17.5%) by pond size, including increased pond depth and perimeter (Table 31). The completed multiple linear regression analysis for *P. pusillus* shows that biomass of this specific submergent species can be explained (23.9%) by increased pond area and age (Table 32). Finally, *P. zosterformis* biomass in the studied SMPs can be marginally explained (4.1%) by decreased pond depth (Table 33).

Figure 38. Pearson correlation for submergent plant abundance and pond characteristics. Due to missing information only 7 SMPs were included for impervious surface levels, and only 13 SMPs were included for pond depth. Significant relationships are denoted with p < 0.05 *, p < 0.01 **, p < 0.001 ***.

Table 29. Multiple linear regression for total biomass of submergent plant species *Chara* predicted by pond characteristics width, depth, and impervious level. Final model: Multiple $R^2 = 0.221$, adjusted $R^2 = 0.194$, F-statistic 8.12, DF = 86, p-value = < 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	3.74	0.86	4.37	< 0.001
Width	-0.01	0.003	-2.01	0.048
Depth	-0.36	0.1	-3.41	< 0.001
Impervious level	-0.05	0.02	-3.09	0.003

Table 30. Multiple linear regression for total biomass of submergent plant species *Najas flexilis* predicted by pond characteristics width, and age. Final model: Multiple $R^2 = 0.276$, adjusted $R^2 = 0.259$, F-statistic 16.56, DF = 87, p-value = < 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	1.68	0.22	7.55	< 0.001
Width	-0.007	0.003	-2.58	0.012
Age	-0.06	0.01	-4.97	< 0.001

Table 31. Multiple linear regression for total biomass of submergent plant species *Najas guadalupensis* predicted by pond characteristics depth, and perimeter. Final model: Multiple $R^2 = 0.194$, adjusted $R^2 = 0.175$, F-statistic 10.43, DF = 87, p-value = < 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	-0.64	0.26	-2.46	0.016
Depth	0.37	0.09	3.85	< 0.001
Perimeter	0.002	0.001	2.48	0.015

Table 32. Multiple linear regression for total biomass of submergent plant species *Potamogeton pusillus* predicted by pond characteristics total pond area, and age. Final model: Multiple $R^2 = 0.256$, adjusted $R^2 = 0.239$, F-statistic 14.94, DF = 87, p-value = < 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	-0.46	0.27	-1.69	0.09
Total pond area	1.12×10^{-4}	2.99×10^{-5}	3.75	< 0.001
Age	0.07	0.05	4.36	< 0.001

Table 33. Multiple linear regression for total biomass of submergent plant species *Potamogeton zosterformis* predicted by pond characteristic depth. Final model: Multiple $R^2 = 0.051$, adjusted $R^2 = 0.041$, F-statistic 4.786, DF = 88, p-value = 0.031.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	0.03	0.01	2.41	0.018
Depth	-0.02	0.01	-2.19	0.031

A Pearson correlation analysis was also completed for aquatic emergent vegetation and defining pond characteristics (Figure 39). Very few emergent macrophytes can be associated with specific pond design traits. Interestingly, species labeled as ‘unknown’ strongly correlate with increasing pond size (i.e. width, length, perimeter, and area). However, pond size specifically width, is also associated with decreasing *T. latifolia* and *S. acutus* abundances in the studied SMPs. Impervious level of the surrounding watershed also has a significant negative relationship with *S. acutus* biomass. On the other hand, total drainage area is positively related to increased *P. australis* cover within the study ponds. Although pond age varies greatly between sites, *C. aquatilis* is the only emergent aquatic species to show a significant negative relationship between plant abundance and pond age.

A multiple linear regression analysis was completed for design characteristics and emergent plant species showing significant correlations with two or more variables. Selected independent variables were based on significant relationships from the completed Pearson correlation (Figure 39). All non-significant variables from the multiple linear regression were removed to establish the final model. The abundance of emergent plant species, *S. acutus* can be significantly explained (33.8%) by decreased pond width and impervious level of the surrounding drainage area (Table 34).

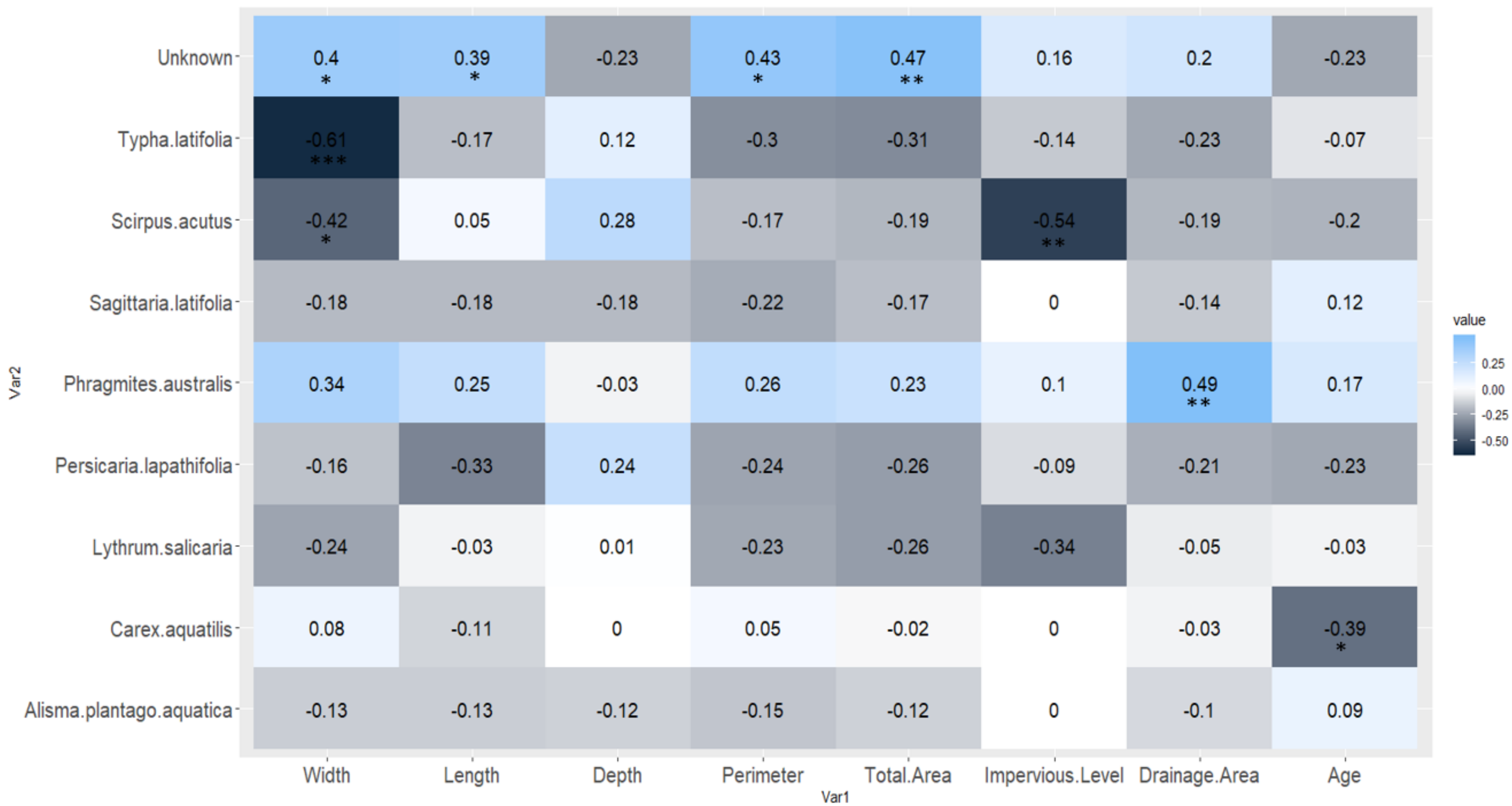

Figure 39. Pearson correlation for emergent plant cover and pond characteristics. Due to missing information only 7 SMPs were included for impervious surface levels, and only 13 SMPs were included for pond depth. Significant relationships are denoted with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

Table 34. Multiple linear regression for relative abundance of aquatic emergent plant species *Scirpus acutus* predicted by pond characteristics width and impervious level. Final model: Multiple $R^2 = 0.384$, adjusted $R^2 = 0.338$, F-statistic 8.416, DF = 27, p-value = 0.001.

Variables	**Estimate**	**Standard Error**	**t-value**	**p-value**
Intercept	3.501	0.838	4.181	< 0.001
Width	-0.008	0.004	-2.049	0.05
Impervious level	-0.053	0.017	0.017	0.005

4.3.3 Effect of inflow water quality and pond characteristics on aquatic plant richness

In order to determine the relationship between incoming water quality and species richness (for both emergent and submergent vegetation), correlation analysis was used. A Pearson correlation analysis was completed for submergent vegetation species richness and inflow water quality parameters (Figure 40). With increasing inflow dissolved phosphorus concentrations, there is a significant increase in submergent species richness ($p < 0.05$, cor = 0.27).

A correlation analysis was also completed for emergent plant species richness and inflow water quality parameters (Figure 41). There are no significant relationships between emergent plant richness and water quality at the inflow sites.

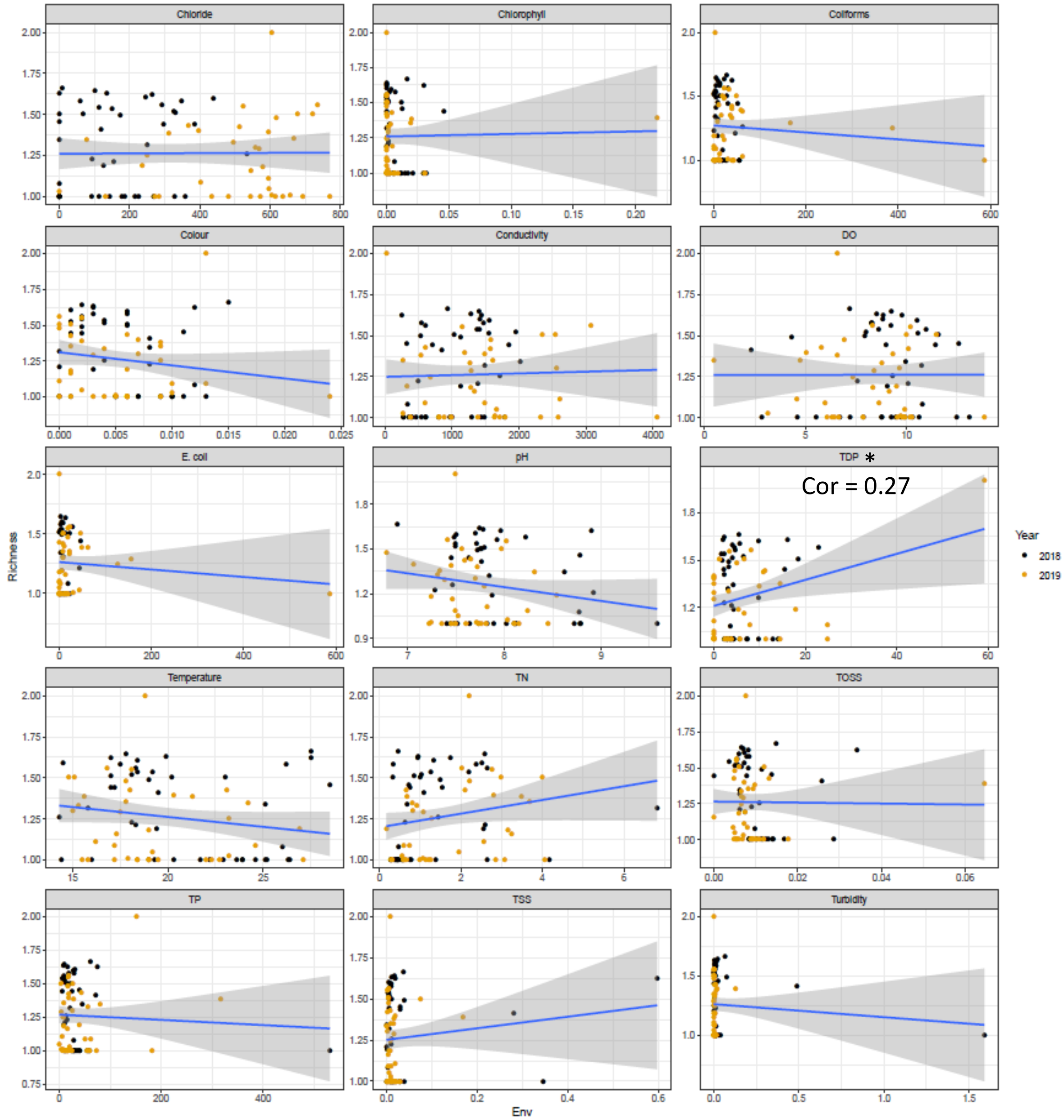

Figure 40. Species richness plots for submergent plant biomass and **inflow** water quality parameters. Significant correlation analyses are denoted by p < 0.05 *, p < 0.01 **, p <0.001 ***.

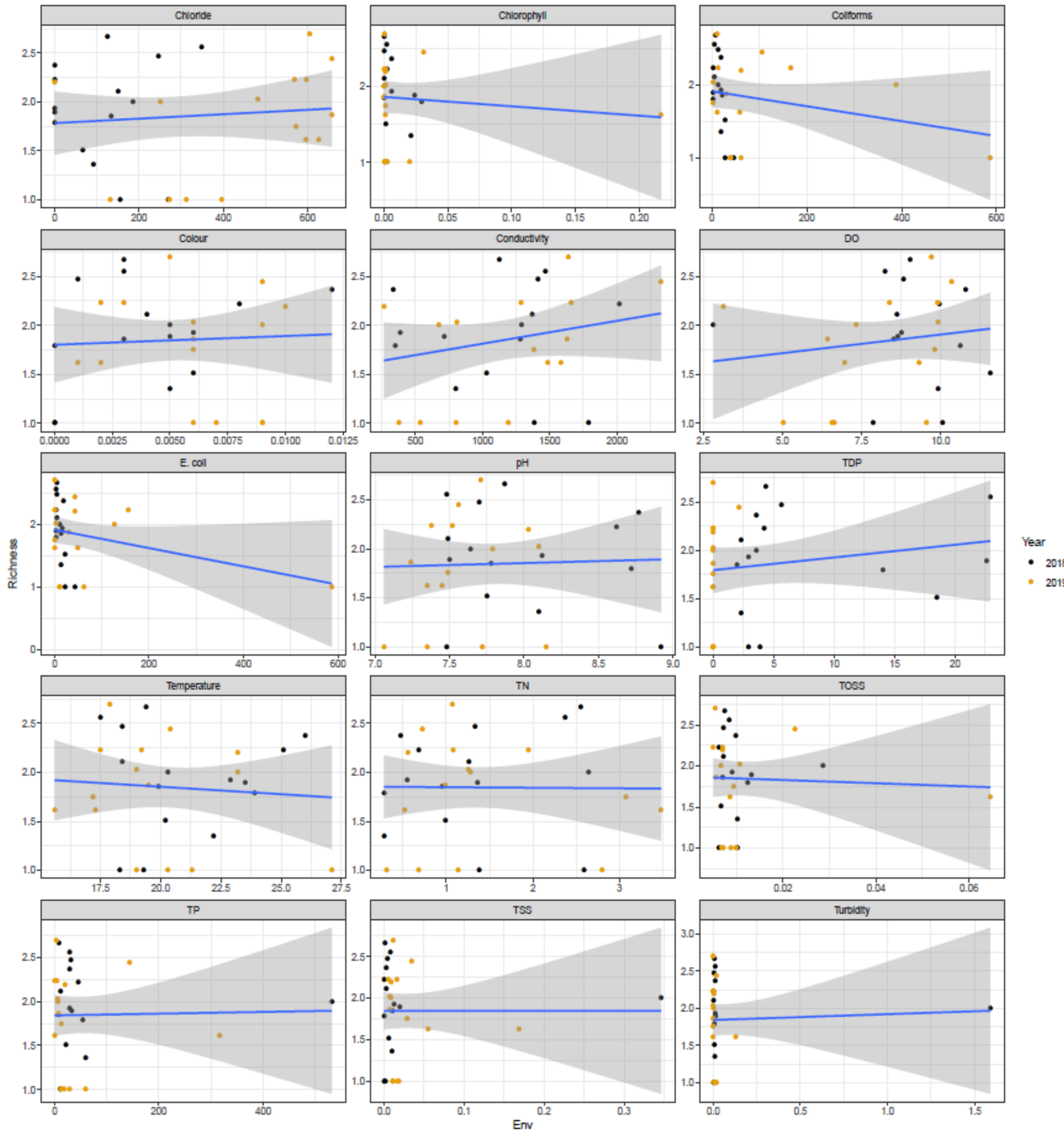

Figure 41. Species richness plots for emergent plant cover (aquatic only) and **inflow** water quality parameters. Significant correlation analyses are denoted by p < 0.05 *, p < 0.01 **, p <0.001 ***.

In order to determine potential relationships between pond characteristics and species richness (for both emergent and submergent vegetation), correlation analysis was used. A correlation analysis was completed for submergent vegetation species richness and defining pond characteristics (Figure 42). It was determined that increased species richness of submergent macrophytes is significantly associated with decreased pond drainage area ($p < 0.05$, cor = -0.24).

A correlation analysis was also completed for emergent plant species richness and pond design traits (Figure 43). There are no significant relationships between emergent plant richness and defining pond characteristics.

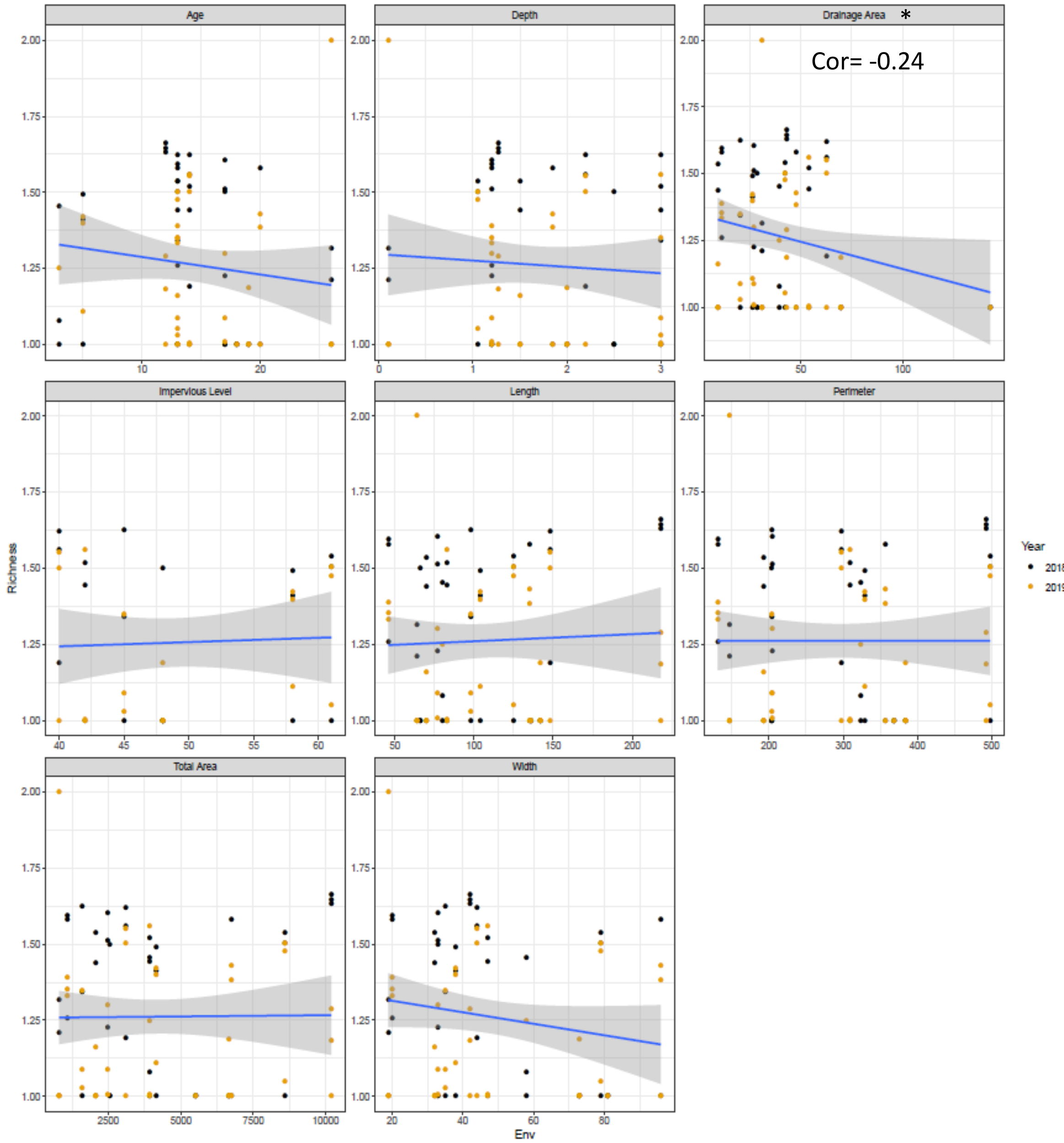

Figure 42. Species richness plots for submergent plant biomass and pond characteristics. Significant correlation analyses are denoted by p < 0.05 *, p < 0.01 **, p <0.001 ***.

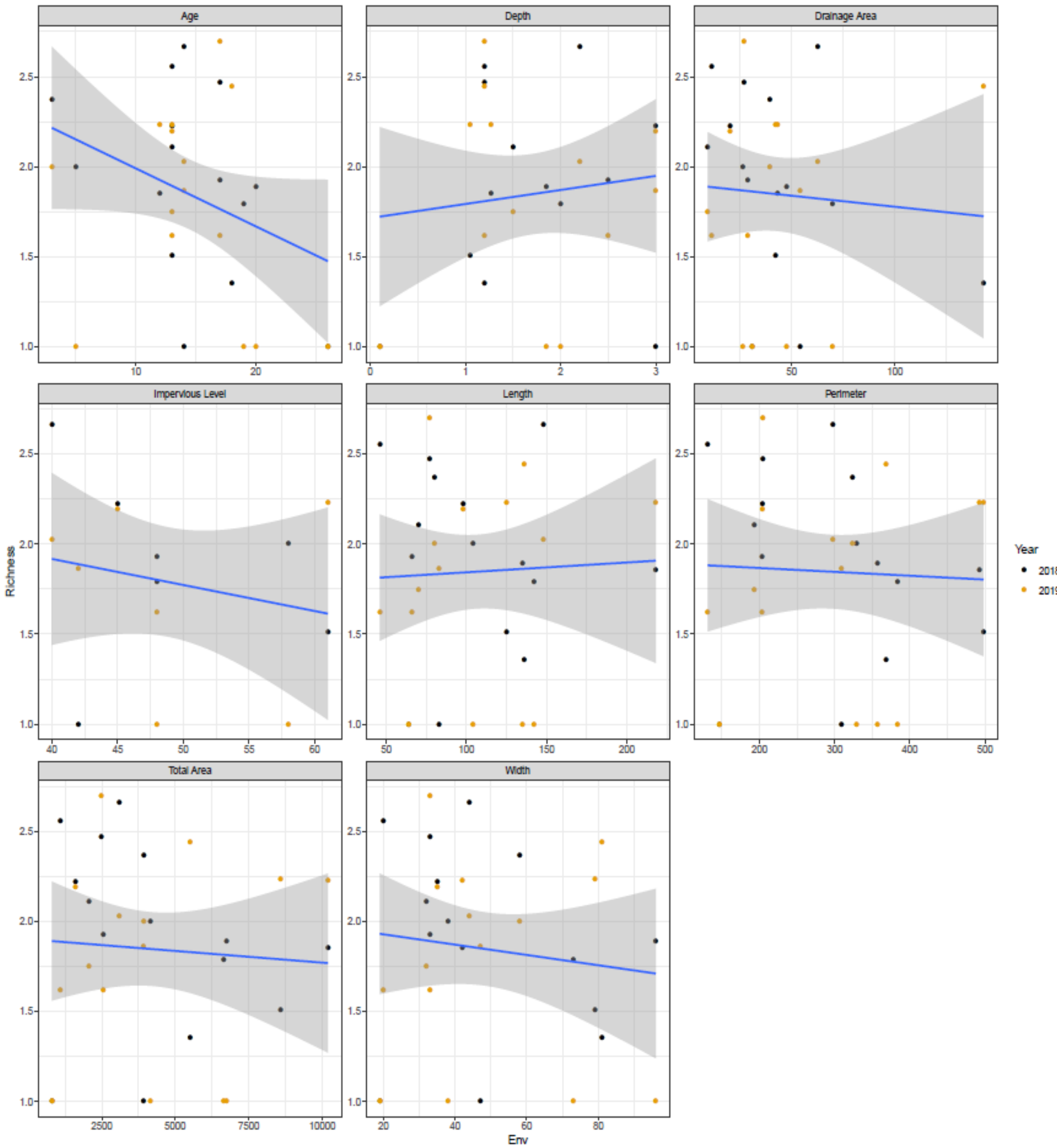

Figure 43. Species richness plots for emergent plant cover (aquatic only) and pond characteristics. Significant correlation analyses are denoted by p < 0.05 *, p < 0.01 **, p <0.001 ***.

4.4 DISCUSSION

4.4.1 Effect of inflow water quality and pond characteristics on emergent plant abundance and type

Emergent aquatic vegetation is an important element in a variety of freshwater systems including ponds, lakes, and wetlands. Emergent plants provide a number of services to established aquatic communities, including habitat and food resources for wildlife. While SMPs are not necessarily natural systems, a variety of emergent species can utilize these unique habitats. The relative variety of emergent species present within the study SMPs is relatively low (See Chapter 3, Figures 18, 19). In general, urban ponds tend to show aquatic emergent plant cover dominated by a single species, usually either *P. australis* or *T. latifolia*. However, the establishment of specific species within urban ponds may in fact be driven by the quality of water entering these systems. Furthermore, specific pond characteristics may also encourage establishment of specific species and their propagation into monocultures within these ponds.

Within the studied SMPs, there was large variation in terms of total emergent plant coverage (See Chapter 3, Figure 23). It has been previously suggested that cover of emergent macrophytes in lake systems is directly influenced by lake morphometry and size (Duarte et al., 1986). Furthermore, depth and slope of the system can strongly affect emergent plant coverage (Duarte et al., 1986). However, due to the relatively shallow depths of SMPs, emergent vegetation may not be limited in this way. It was determined that emergent cover is significantly related to decreasing imperviousness of the surrounding landscape (Table 21). However, decreased imperviousness can only explain 10.6% of why emergent vegetation increases throughout these sites (Table 23). These trends may highlight the importance of other factors in managing emergent plant cover throughout these systems. In fact, the importance of sediment characteristics, and light availability in freshwater lakes has been highlighted as important drivers of macrophyte coverage (Duarte et al., 1986).

Specific environmental and design characteristics of SMPs may also function in species selection. Certain species of emergent plants can tolerate larger variations in water quality compared to other taxa. Furthermore, permanent design traits of SMPs may also act to encourage or hinder the growth of individual taxa. *P. australis* is one of the most commonly found emergent macrophytes within the study ponds. Its presence within these systems is significantly associated with increased dissolved phosphorus at the inflow sites (Figure 37). Increased cover of this species is also associated with increased drainage area (Figure 39). This may highlight the importance of nutrient concentrations in incoming stormwater as a driver of *P. australis* colonization. In fact, a review completed on *P. australis* growth shows that increasing sediment nutrient concentrations, specifically nitrogen, can significantly increase plant density, height, and shoot diameter (Engloner, 2009).

Another common emergent macrophyte species in the study SMPs was *T. latifolia*. Increased cover of this species is correlated with decreased colour and temperature, as well as increased conductivity and nitrogen concentrations (Figure 37). In fact, over 27% of increased *T. latifolia* coverage can be explained by decreased colour and increased conductivity at the inflow locations (Table 28). This species is also correlated with decreasing pond width (Figure 39). Studies have shown that under nutrient rich conditions, *T. latifolia* can outgrow other common wetland species

(Svengsouk & Mitsch, 2001). This species can also tolerate moderately salty conditions, potentially giving it an advantage over other emergent taxa (Grace & Harrison, 1985). Water colour in this case, acts as an indicator for increased decomposition, and therefore decreased oxygen levels within a system. It has previously been suggested that *T. latifolia* is anoxia-tolerant, therefore allowing it to access habitats which other species may not tolerate (Crawford et al., 1989). These characteristics may allow this species to readily outcompete other emergents in SMP environments.

It should also be highlighted that the only invasive species documented throughout the duration of this study included emergent species *P. australis*, and *L. salicaria*. Both of these species are fairly prevalent in pond and wetland systems across Ontario and most of North America. In this study, *L. salicaria* was noted in very low abundances at a few sampled sites (See Chapter 3, Figures 18, 19). *P. australis* on the other hand, was frequently noted at the majority of sampled sites, and represented more than half the total emergent plant biomass in some ponds. In fact, it has been noted as one of the most common wetland plants across the globe (Lee & Scholz, 2006). In freshwater wetlands, this species has been noted to cause severe alterations to natural hydrology, and declines to macrophyte biodiversity (Ailstock et al., 2001). Its stem density, height, and broad salinity tolerance allows it to outgrow native species and thrive in a variety of environments (Meyerson et al., 2000). As previously mentioned, ponds trending towards excess nutrient levels (specifically phosphorus) may enhance the growth of this species (Figure 37). However, *P. australis* may also serve important nutrient removal processes within SMPs (See Chapter 3, 3.4 Discussion, 3.4.2 Effect of emergent plant type on outflow water quality), and therefore may not be considered a nuisance species in these systems. Overall, it appears that the studied urban ponds are not functioning as significant drivers for the dispersal and spread of invasive plants.

4.4.2 Effect of inflow water quality and pond characteristics on submergent plant abundance and type

The effect of inflow water quality and pond characteristics may be more influential on submergent macrophyte communities, compared to emergent plants. This is due to the growth and lifecycles of submergent plants, which are found completely submersed below the waters surface. Previous studies have identified light penetration, sediment substrate, and trophic status as significant drivers in predicting macrophyte cover (Duarte et al., 1986). It was originally thought lake productivity represented a direct relationship with macrophyte biomass. In this way, oligotrophic lakes should have lower aquatic plant coverage compared to eutrophic systems. However, high lake productivity will also result in decreased light penetration and therefore reduced macrophyte growth (Duarte et al., 1986). Furthermore, during these eutrophic states, aquatic plants tend to be replaced by phytoplankton communities (Balls et al., 1989; Bakker et al., 2010). However, submergent plant biomass is not significantly associated with chlorophyll α (i.e. phytoplankton biomass) concentrations (Table 19). This may in part be due to the eutrophic tolerance of many macrophyte species common to SMPs. However, within the study ponds, increased submergent plant biomass is significantly correlated with decreasing inflow colour, temperature, and pH (Table 19). It is also significantly related to increasing nitrogen concentrations at the inflow site (Table 19). However, decreasing colour and pH are only responsible for explaining 8.3% of the increase in submergent plant biomass within the study sites (Table 20). This may indicate that within urban ponds, stormwater quality is not a significant driver for submergent plant biomass.

The importance of lake morphometry in submergent plant growth has also been highlighted (Duarte et al., 1986). Within the studied SMPs, increased submergent biomass can be further explained by pond size, and age (Table 21). In this case, decreased pond width, as well as increased pond length and area can explain 34.2% of the increase in submergent biomass across the study sites (Table 22). This may highlight the importance of pond design in submergent plant maintenance, rather than water quality. Ponds which maximize overall size, provide a greater variety of useful habitat for submergent plants to establish. In this way, submergent species can take advantage of optimal habitat, therefore increasing coverage within SMPs. Interestingly, these results are inconsistent with research completed on macrophyte growth in lakes, whereby submergent plant growth is inversely related to lake size (Duarte et al., 1986). However, in lake systems slope plays an essential role in driving this relationship (Duarte et al., 1986). Since SMPs are relatively shallow, increasing pond size but not necessarily depth would increase available habitat for macrophyte growth.

Within the studied urban ponds, there is a greater variety of submergent aquatic vegetation, compared to emergent species. While many of the profiled submergent plant communities illustrated monocultures throughout the sampling season, some notable shifts in community structures were noted. These changes in plant profiles may highlight the influence of runoff water quality and pond characteristics on individual species. Similar to trends noted for total submergent biomass, the presence of individual species seems to be further explained by pond characteristics, rather than inflow water quality.

Submergent plant species including *Chara, N. flexilis*, and *P. pusillus*, all showed significant relationships with multiple independent water quality variables (Figure 36). However, when combined, significant water quality variables explained less than 10% of the increase noted for these species (Tables 24, 25, 26). Specific pond characteristics on the other hand, seem to provide further explanation for the presence of specific submergent taxa. Decreased pond width and depth explained 19.4% of *Chara* biomass in the SMPs (Table 29). Younger and shallower ponds also significantly explained (25.9%) the presence of *N. flexilis* in the study systems (Table 30). Furthermore, *P. pusillus* biomass can be explained (23.9%) by increased pond area and age in the selected SMPs (Table 32). In this way, pond morphometry and age seem to be more important in predicting biomass of common submergent plant species, compared to the quality of stormwater runoff.

4.4.3 Effect of inflow water quality and pond characteristics on aquatic plant richness

Generally, studies completed on SMPs recognize their potential as biodiversity hotspots (Tixier et al., 2011; Holtmann et al., 2018; Miró et al., 2018). In fact, the ability of these systems to function as essential locations of biodiversity enhancement in urban areas has been highlighted (Casey et al., 2006; Gallagher et al., 2011). However, the results from this study suggest that the selected Oshawa SMPs are low in aquatic plant diversity (See Chapter 3, Figures 18, 19). Major changes in aquatic environments can greatly alter community composition. In aquatic systems, shifts towards eutrophic conditions can result in devasting effects on plant diversity and richness (Arthaud et al., 2012). Environments such as urban ponds can undergo massive shifts in water quality within days, if not hours depending on storm event frequency. These changes to runoff water quality may largely influence the establishment and growth of species utilizing these habitats. No significant relationships were noted between emergent species richness and inflow water quality or pond characteristics (Figure 41, 43).

However, increased dissolved phosphorus concentrations at inflow locations is significantly correlated with increased submergent plant richness (Figure 40). Interestingly however, increasing drainage area of the surrounding SMP shows a significant negative relationship with submergent plant richness (Figure 42). Typically, increasing drainage area also increases nutrient concentrations, therefore providing essential resources for aquatic vegetation. However, high levels of nutrients may also result in eutrophic conditions within established SMPs. In this way, a community shift is made from macrophyte dominated to algae dominated systems (Balls et al., 1989; Bakker et al., 2010). Furthermore, increased drainage area may also result in increased water turbidity from particulates and debris accumulated from the landscape. Decreased water clarity in these systems results in lower light penetration, limiting habitat for optimal plant growth (Duarte et al., 1986).

4.5 CONCLUSION

There are a number of factors which can contribute to the community composition of a freshwater system. SMPs are an important example of the robustness and variety of aquatic plant species which can tolerate and even thrive in urban freshwater systems. Based on the evidence from this chapter, it appears that while pond characteristics and inflow water quality do a rather poor job of explaining total emergent plant cover, they can perhaps function in driving the establishment of specific emergent species. The colonization of two common emergent macrophytes, *P. australis* and *T. latifolia*, can be well explained by a variety of factors. Specifically, *P. australis* biomass is significantly related to increasing dissolved phosphorus concentrations at the inflow site, as well as total pond drainage area. This may highlight *P. australis* as an important indicator of ponds where nutrient enrichment from the surrounding landscape is occurring. Surface water quality provides weak explanations for total submergent plant biomass, as well as the presence of specific submergent species. However, defining pond traits, including pond age and size can explain up to 34% of the variation for submergent species, including *Chara*, *N. flexilis*, and *P. pusillus*.

Species richness and diversity of both submergent and emergent macrophytes is low across the studied sites. No significant relationships were detected between emergent species richness and inflow water quality or pond characteristics. Submergent species richness however is significantly associated with increased dissolved phosphorus at the inflow site, as well as decreased drainage area. These trends may indicate the possible shift in SMPs from plant dominated to algae dominated following prolonged nutrient exposure. Overall, it appears that urban pond characteristics may have a larger influence on established macrophyte communities, compared to inflow water quality. These findings may have significant impacts on SMP design and the establishment of future stormwater facilities.

CHAPTER 5: GENERAL CONCLUSION

5.1 INTRODUCTION

With rapid expansion of developing areas, urban ponds are becoming more prevalent across landscapes. While functioning as important physical dividers between natural systems and surface water runoff, their ability to improve water quality has been put into question. These ponds function as crucial freshwater habitats in urban areas, and rapidly colonize with aquatic vegetation. The functional role of aquatic plants in stormwater treatment within SMPs is unknown. Furthermore, the role of inflow water chemistry and pond characteristics in defining plant communities within urban ponds also remains unstudied. The previous chapters have provided insight into some of these major knowledge gaps surrounding SMP performance.

The previous chapters have highlighted the performance of 15 SMPs located in Oshawa, Ontario. Looking at differences between inflow and outflow water quality shows the potential for urban ponds to act as sinks of some runoff contaminants (i.e. chloride and nitrogen), while perhaps functioning as sources of others (i.e. phosphorus). Furthermore, sediment maintenance of SMPs may act to improve settling processes of suspended solids, however may also result in increased phosphorus loadings to outgoing stormwater.

The completed research has also shown that aquatic vegetation does in fact play a role in water treatment processes within SMPs. Overall diversity and species richness of both submergent and emergent species within Oshawa SMPs is low, and does not have a significant impact on improving outflow water quality. However, plant abundance, particularly of submergent plants, may play an important role in reducing downstream nutrient pollution of surface waters.

Finally, the previous chapters also highlighted the relationship between incoming water quality and pond characteristics in defining aquatic plant communities. Emergent plant cover is not well defined by water quality or pond traits, however the presence of specific species such as *P. australis*, can be largely explained by nutrient concentrations and surrounding drainage areas. Submergent biomass and species presence is well explained by pond size and age, however runoff water quality does not play a significant role in defining submergent plant communities.

5.2 RECOMMENDATIONS

Firstly, it is clear that Oshawa SMPs receive contaminated stormwater based on the elevated nutrients, suspended solids, coliform bacteria, and chloride measured during the study period. While this study did not directly assess impacts of stormwater constituents on aquatic communities, levels of nutrients and chloride were notably high at a number of SMP sites. In many cases, sampled water parameters (especially phosphorus and chloride) exceeded threshold levels put forth by Environment Canada and the Canadian Water Quality Guidelines. While urban ponds are designed to mitigate these impacts, these SMPs appear to be point-sources of phosphorus to natural waterways. Prolonged exposure to pollutants may have devastating effects on not only urban pond communities, but also downstream freshwater environments that are essential for fishing and other recreational activities.

Next, with the increasing numbers of SMPs going online in new residential and commercial areas, there should be regular monitoring of SMP effluent to ensure optimal performance. The Government of Ontario recommends annual monitoring of water quality and quantity within SMPs throughout their functional lifecycles. However, very few municipalities complete regular if any monitoring practices on established ponds. By providing consistent data on water conditions, quantity, and quality, there would be a better understanding of water treatment in SMPs, as well as their potential impact on receiving waters. Regular monitoring of SMPs will also ensure that dredging is appropriately timed. As illustrated, macrophytes can play an important role in water treatment. By regularly monitoring pond depth and sediment accumulation, municipalities can ensure that ponds are not dredged before or after water capacity limits are met. In this way, plant communities may be given more time to establish, thereby encouraging biological water treatment processes. Future studies should be done to determine appropriate amounts and type of plants for optimal treatment conditions.

Within the government of Ontario's guidelines for SMP design and management, it is recommended that municipalities complete regular planting of terrestrial and aquatic species. However, the inclusion of aquatic emergent and submergent planting during original pond construction does not regularly occur. By planting a variety of macrophytes in SMPs early in their lifecycles, enhancements to water treatment processes can be made. This will also enhance the biodiversity of established communities in an attempt to mitigate which species are inhabiting urban pond environments. Furthermore, species that have been shown to make notable improvements to surface water, such as *P australis* and *N. flexilis*, can be included in an attempt to maximize water treatment. Conversely, since macrophytes naturally colonize SMPs, cities can also focus on naturally established communities, and maintaining their presence in these systems.

5.3 CONCLUSION

Overall, SMPs are an essential part of developing landscapes, however there is a lack of knowledge surrounding the water treatment processes that occur within these ponds, as well as the effect of aquatic vegetation on water quality. Based on this research, it appears that SMPs do not remove all stormwater constituents. However, the presence of submergent macrophytes was found to play a key role in removing contaminants, including nutrients, which are common to stormwater runoff. It should be noted however that emergent aquatic plant abundance and diversity did not show a clear impact on outflow water quality. However, the presence of specific emergent species, specifically invasive species *P. australis*, may play a crucial role in improving stormwater quality by decreasing nitrogen levels. This research also marks the first time that aquatic plant communities have been assessed in Canadian SMPs, including their potential to serve as reservoirs for invasive species. It was determined that Oshawa SMPs do not seem to function as drivers for the colonization or spread of invasive macrophytes. Furthermore, the establishment of aquatic plant communities seems to be better explained by pond characteristics, rather than runoff water quality. Overall, it is clear that SMPs can function as unique habitats for macrophytes, including multiple native species. The plants established in these systems may act to further improve water treatment within SMPs, and should be treated as an essential element in pond design and performance.

REFERENCES

Ailstock, M.S., Norman, C.M., Bushmann, P.J. (2001). Common reed *Phragmites australis*: Control and effects upon biodiversity in freshwater nontidal wetlands. Restoration Ecology 9(1): 49-59.

Arthaud, F., Vallod, D., Robin, J., Bornette, G. (2012). Eutrophication and drought disturbance shape functional diversity and life-history traits of aquatic plants in shallow lakes. *Aquatic Sciences* 74: 471-481.

Bakker, E.S., Van Donk, E., Declerck, S.A.J., Helmsing, N.R., Hidding, B., Nolet, B.A. (2010). Effect of macrophyte community composition and nutrient enrichment on plant biomass and algal biomass. *Basic and Applied Ecology* 11: 432-439.

Balls, H., Moss, B., Irvine, K. (1989). The loss of submerged plants with eutrophication: Experimental design, water chemistry, aquatic plant and phytoplankton biomass in experiments carried out in ponds in the Norfolk Broadland. *Freshwater Biology* 22: 71-87.

Beutel, M.W., Larson, L. (2015). Pathogen removal from urban pond outflow using rock biofilters. *Ecological Engineering* 78: 72-28.

Canadian Council of Ministers of the Environment. (2011). Canadian water quality guidelines for the protection of aquatic life. Retrieved from http://ceqg-rcqe.ccme.ca/download/en/337/

Casey, R.E., Klaine, S.J. (2001). Nutrient attenuation by a riparian wetland during natural and artificial runoff events. Journal of Environmental Quality 30(5): 1720-1731.

Casey, R.E., Simon, J.A., Atueyi, S., Snodgrass, J.W., Renier, N.K., Sparling, D.W. (2006). Temporal trends of trace metals in sediment and invertebrates from stormwater management ponds. *Water, Air, Soil & Pollution* 178: 69-77.

Chang, N.B., Islam, M.K., Wanielista, M.P. (2012). Floating wetland mesocosm assessment of nutrient removal to reduce ecotoxicity in stormwater ponds. *International Journal of Environmental Science and Technology* 9(3): 453-462.

Chiandet, A.S., Xenopoulos, M.A. (2016). Landscape and morphometric controls on water quality in stormwater management ponds. *Urban Ecosystems* 19(4): 1645-1663.

Cole-Parmer Instrument Company. (2019). Cole-Parmer Scientific Experts. Retrieved from https://www.coleparmer.ca/ on 01/15/2019

Collins, K.A., Lawrence, T.J., Stander, E.K., Jontos, R.J., Kaushal, S.S., Newcomer, T.A., Grimm, N.B., Cole-Ekberg, M.L. (2010). Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis. Ecological Engineering 36: 1507-1519.

Crawford, R.M.M., Studer, C., Studer, K. (1989). Deprivation indifference as a survival strategy in competition: Advantages and disadvantages of anoxia tolerance in wetland vegetation. *Flora* 182(3-4): 189.201.

Davis, A.P., Shokouhian, M., Sharma, H., Minami, C. (2001). Laboratory study of biological retention for urban stormwater management. *Water Environment Research* 73(1): 5-14.

Drake, J., Guo, Y. (2008). Maintenance of wet stormwater ponds in Ontario. *Canadian Water Resources Journal* 33(4): 351-368.

Duarte, C.M., Kalff, J., Peters, R.H. (1986). Patterns of biomass and cover of aquatic macrophytes in lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 43: 1900-1908.

Dungan, H.A., Bartlett, S.L., Burke, S.M., Doubek, J.P., Krivak-Tetley, F.E., Skaff, N.K., Summers, J.C., Farrell, K.J., McCullough, I.M., Morales-Williams, A.M., Roberts, D.C., Ouyang, Z., Scordo, F., Hanson, P.C., & Weathers, K.C. (2017). Salting our freshwater lakes. *Proceedings of the National Academy of Sciences* 114(17): 4453-4458.

Egemose, S., Sønderup, M.J., Grudinina, A., Hansen, A.S., Flindt, M.R. (2015). Heavy metal composition in stormwater and retention ponds dependent on pond age, design and catchment type. *Environmental Technology* 36 (8): 959-969.

Elmqvist, T., Fragkias, M., Goodness, J., Güneralp, B., Marcotullio, P.J., McDonald, R.I., Parnell, S., Schewenius, M., Sendstad, M., Seto, K.C., Wilkinson, C. (2013). Urbanization, biodiversity and ecosystem services: Challenges and opportunities. *Springer.* DOI 10.1007/978-94-007-7088-1.

Engloner, A.I. (2010). Structure, growth dynamics and biomass of reed (*Phragmites australis*) – A review. *Flora* 204: 331-346.

Foltz, S.J., Dodson, S.L. (2009). Aquatic Hemiptera community structure in stormwater retention ponds: A watershed land cover approach. *Hydrobiologia* 621(1): 49-62.

Fritioff, A., Grerger, M. (2003). Aquatic and terrestrial plant species with potential to remove heavy metals from stormwater. *International Journal of Phytoremediation* 5(3): 211-224.

Frost, P.C., Song, K., Buttle, J.M., Marsalek, J., McDonald, A., Xenopoulos, A. (2015). Urban biogeochemistry of trace elements: What can the sediment of stormwater ponds tell us? *Urban Ecosystems* 18: 763-775.

Gallagher, M.T., Snodgrass, J.W., Ownby, D.R., Brand, A.B., Casey, R.E., Lev, S. (2011). Watershed-scale analysis of pollutant distributions in stormwater management ponds. *Urban Ecosystems* 14: 469-484.

Gillis, P.L. (2011). Assessing the toxicity of sodium chloride to the glochidia of freshwater mussels: Implications for salinization of surface waters. *Environmental Pollution* 159(6): 1702-1708.

Government of Canada. (2015). Phosphorus in aquatic ecosystems. Retrieved from https://www.canada.ca/en/environment-climate-change/services/freshwater-quality-monitoring/publications/phosphorus-aquatic-ecosystems/chapter-2.html

Government of Ontario. (2003). Stormwater management planning and design manual. Retrieved from: https://www.ontario.ca/document/stormwater-management-planning-and-design-manual-0.

Grace, J.B., Harrison, J.S. (1985). The biology of Canadian weeds, 73: *Typha latifolia* L., *Typha angustifolia* L., and *Typha xglauca* Godr. *Canadian Journal of Plant Science* 66: 361-379.

Greenway, M. (2005). The role of constructed wetlands in secondary effluent treatment and water reuse in subtropical and arid Australia. *Ecological Engineering* 25: 501-509.

Grosshans, R.E., Wrubleski, D.A., Goldsborough, L.G. (2004). Changes in the emergent plant community of Netley-Libau marsh between 1979 and 2001. Delta March Field Station (University of Manitoba) 4: 52.

Gu, B. (2008). Phosphorus removal in small constructed wetlands dominated by submersed aquatic vegetation in South Florida, USA. *Journal of Plant Ecology* 1(1): 67-74.

Hammer, Ø., Harper, D.A.T., and P. D. Ryan. (2001). PAST: Paleontological Statistics Software Package for Education and Data Analysis. *Palaeontologia Electronica* 4(1): 9pp.

Hassall, C. (2014). The ecology and biodiversity of urban ponds. *WIREs Water* 1:187-206.

Hassall, C., Anderson, S. (2015). Stormwater ponds can contain comparable biodiversity to unmanaged wetlands in urban areas. *Hydrobiologia* 745(1): 137-149.

Hill, M.J., Biggs, J., Thornhill, I., Briers, R.A., Gledhill, D.G., White, J.C., Wood, P.J., Hassall, C. (2017). Urban ponds as an aquatic biodiversity resource in modified landscapes. *Global Change Biology* 23: 986-999.

Hintz, W.D., Relyea, R.A. (2017). Impacts of road deicing salts on the early-life growth and development of a stream salmonid: Salt type matters. *Environmental Pollution* 223: 409-415.

Holtmann, L., Juchem, M., Brüggeshemke, J., Möhlmeyer, A., Fartmann, T. (2018). Stormwater ponds promote dragonfly (Odonata) species richness and density in urban areas. *Ecological Engineering* 118: 1-11.

Ivanovsky, A., Belles, A., Criquet, J., Dumoulin, D., Noble, P., Alary, C., Billon, G. (2018). Assessment of the treatment efficiency of an urban stormwater pond and its impact on the natural downstream watercourse. *Journal of Environmental Management* 226: 120-130.

Jones, D.K., Mattes, B.M., Hintz, W.D., Schuler, M.S., Stoler, A.B., Lind, L.A., Cooper, R.O., Relyea, R.A. (2017). Investigation of road salts and biotic stressors on freshwater wetland communities. *Environmental Pollution* 221: 159-167.

Karathanasis, A.D., Potter, C.L., Coyne, M.S. (2003). Vegetation effects on fecal bacteria, BOD, and suspended solid removal in constructed wetlands treating domestic wastewater. *Ecological Engineering* 20: 157-169.

Katagi, T. (2010). Bioconcentration, bioaccumulation, and metabolism of pesticides in aquatic organisms. Reviews of Environmental Contamination and Toxicology 204: 1-103.

Kirkwood, A.E. (2016). Assessment of water quality in Oshawa Creek and tributaries (2013-2015). A report prepared for the City of Oshawa, March, 2016.

Koch, B.J., Febria, C.M., Gevrey, M., Wainger, L.A., Palmer, M.A. (2014). Nitrogen removal by stormwater management structures: A data synthesis. *Journal of the American Water Resources Association* 50(6): 1594-1607.

Lambert-Servien, E., Clemenceau, G., Gabory, O., Douillard, E., Haury, J. (2006). Stoneworts (*Characeae*) and associated macrophyte species as indicators of water quality and human activities in the Pays-de-la-Loire region, France. *Hydrobiologia* 570: 107-115.

Lee, B.H., Scholz, M. (2007). What is the role of *Phragmites australis* in experimental constructed wetland filters treating urban runoff? *Ecological Engineering* 29: 87-95.

Leto, C., Tuttolomondo, T., La Bella, S., Leone, R., Licata, M. (2013). Effects of plant species in a horizontal subsurface flow constructed wetland – phytoremediation of treated urban wastewater with *Cyperus alternifolius L.* and *Typha latifolia L.* in the West of Sicily (Italy). *Ecological Engineering* 61: 282-291.

Lougheed, V.L., Crosbie, B., Chow-Fraser, P. (2001). Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes basin: latitude, land use, and water quality effects. *Canadian Journal of Fisheries and Aquatic Sciences* 58: 1603-1621.

Mallin, M.A., Ensign, S.H., Wheeler, T.L., Mayes, D.B. (2002). Pollutant removal efficacy of three wet detention ponds. *Journal of Environmental Quality* 31: 654-660.

Marsalek, J. (2003). Road salts in urban stormwater: An emerging issue in stormwater management in cold climates. Water Science and Technology 48(9): 61-70.

Marsalek, J., Watt, W.E., Anderson, B.C., Jaskot, C. (1997). Physical and chemical characteristics of sediments from a stormwater management pond. *Water Quality Research Journal of Canada* 32(1): 89-100.

Marsalek, J., Watt, W.E., Henry, D. (1992). Retrofitting stormwater ponds for water quality control. *Water Pollution Research Journal of Canada* 27(2): 403-422.

Marsooli, R., Lin, K., Emanuel, K., Feng, K. (2019). Climate change exacerbates hurricane flood hazards along US Atlantic and Gulf Coasts on spatially varying patterns. Nature Communications 10(3785): 1-9.

Massal, L., Snodgrass, J., Casey, R. (2007). Nitrogen pollution of stormwater ponds: Potential for toxic effects on amphibian embryos and larvae. *Applied Herpetology* 4(1): 19-29.

McDonald, R.I., Kareiva, P., Forman, R.T.T. (2008). The implications of current and future urbanization for global protected areas and biodiversity conservation. *Biological Conservation* 141: 1695-1703.

Meyerson, L.A., Saltonstall, K., Windham, L., Kiviat, E., Findlay, S. (2000). A comparison of *Phragmites australis* in freshwater and brackish marsh environments in North America. Wetlands Ecology and Management 8: 89-103.

Miró, A., Hall, J., Rae, M., O'Brien, D. (2018). Links between ecological and human wealth in drainage ponds in a fast-expanding city, and proposals for design and management. *Landscape and Urban Planning* 180: 93-102.

Murphy, J., Riley, J.P. (1962). A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimca Acta* 27: 31-36.

Olding, D.D., Steele, T.S., Nemeth, J.C. (2004). Operational monitoring of urban stormwater management facilities and receiving subwatersheds in Richmond Hill, Ontario. *Water Quality Research Journal of Canada* 39(4): 392-405.

Ontario Ministry of Environment. 1983. Handbook of Analytical Methods for Environmental Samples. Laboratory Services and Applied Research Branch, Ontario Ministry of Environment, Toronto, Ontario, Canada.

Ontario Ministry of Finance. (2019). Ontario Population Projections, 2018-2046. Retrieved from https://www.fin.gov.on.ca/en/economy/demographics/projections/projections2018-2046.pdf

Petterson, S.R., Mitchell, V.G., Davies, C.M., O'Connor, J., Kaucner, C., Roser, D., Ashbolt, N. (2016). Evaluation of three full-scale stormwater treatment systems with respect to water yield, pathogen removal efficacy and human health risk from faecal pathogens. *Science of the Total Environment* 543: 691-702.

Petticrew, E.L., Kalff, J. (1992). Water flow and clay retention in submerged macrophyte beds. *Canadian Journal of Fisheries and Aquatic Sciences* 49: 2483-2489.

Reddy, K.R., Patrick Jr., W.H., Lindau, C.W. (1989). Nitrification- denitrification at the plant root-sediment interface in wetlands. *Limnology and Oceanography* 34(6): 1004-1013.

Rhea, L., Jarnaigin, T., Hogan, D., Loperfido, J.V., Shuster, W. (2015). Effects of urbanization and stormwater control measures on streamflows in the vicinity of Claksburg, Maryland, USA. *Hydrological Processes* 29: 4413-4426.

Robertson, D.M., Saad, D.A. (2011). Nutrient inputs to the Laurentian Great Lakes by source and watershed estimated using SPARROW watershed models. *Journal of the American Water Resources Association* 47(5): 1011-1033.

Schultz, R., Andrews, S., O'Reilly, L., Bouchard, V., Frey, S. (2001). Plant community composition more predictive than diversity of carbon cycling in freshwater wetlands. *Wetlands* 31: 965-977.

Shelef, O., Gross, A., Rachmilevitch, S. (2013). Role of plants in a constructed wetland: Current and new perspectives. *Water* 5: 405-419.

Smith, V.H., Joye, S.B., Howarth, R.W. (2006). Eutrophication of freshwater and marine ecosystems. *American Society of Limnology and Oceanography* 51: 351-355.

Snodgrass, J.W., Moore, J., Lev, S.M., Casey, R.E., Ownby, D.R., Flora, R.F., Izzo, G. (2017). Influence of modern stormwater management practices on transport of road salt to surface waters. *Environmental Science and Technology* 51: 4165-4172.

Song, K., Xenopoulos, M.A., Buttle, J.M., Marsalek, J., Wagner, N.D., Pick, F.R., Frost, P.C. (2013). Thermal stratification patterns in urban ponds and their relationships with vertical nutrient gradients. *Journal of Environmental Management* 127: 317-323.

Soranno, P.A., Cheruvelil, K.S., Wagner, T., Webster, K.E., Bremigan, M.T. (2015). Effects of land use on lake nutrients: The importance of scale, hydrologic connectivity, and region. PLoS ONE 10(8): 1-22.

Su, S., Zhou, Y., Qin, J.G., Yao, W., Ma, Z. (2010). Optimization of the method for chlorophyll extraction in aquatic plants. *Journal of Freshwater Ecology* 25(4): 531-538.

Svengsouk, L.J., Mitsch, W.J. (2001). Dynamics of mixtures of *Typha latifolia* and *Schoenoplectus tabernaemontani* in nutrient-enrichment wetland experiments. *American Midland Naturalist* 145: 309-324.

Tanner, C.C. (1995). Plants for constructed wetland treatment systems – A comparison of the growth and nutrient uptake of eight emergent species. Ecological Engineering 7: 59-83.

Tiwari, A., Rachlin, J.W. (2018). A review of road salt ecological impacts. *Northeastern Naturalist* 25(1): 123-142.

Tixier, G., Lafont, M., Grapentine, L., Rochfort, Q., Marsalek, J. (2011). Ecological risk assessment of urban stormwater ponds: Literature review and proposal of a new conceptual approach providing ecological quality goals and the associated bioassessment tools. *Ecological Indicators* 11: 1497-1506.

Van Buren, M.A., Watt, W.E., Marsalek, J. (1996). Removal of selected urban stormwater constituents by an on-stream pond. *Journal of Environmental Planning and Management* 40(1): 5-18.

Vincent, J., Kirkwood, A.E. (2014). Variability of water quality, metals and phytoplankton community structure in urban stormwater ponds along a vegetation gradient. *Urban Ecosystems* 17(3): 839-853.

Vymazal, J. (2011). Plants used in constructed wetlands with horizontal subsurface flow: A review. *Hydrobiologia* 674: 133-156.

Walaszek, M., Bois, P., Laurent, J., Lenormand, E., Wanko, A. (2018). Micropollutants removal and storage efficiencies in urban stormwater constructed wetlands. *Science of the Total Environment* 645: 854-864.

Watt, W.E., Waters, D., McLean, R. (2001). Climate change and urban stormwater infrastructure in Canada: Context and case studies. *Toronto-Niagara Region Study Report and Working Paper Series*, Report 2003-1.

Weiss, J.D., Hondzo, M., Semmens, M. (2006). Stormwater detention ponds: Modeling heavy metal removal by plant species and sediments. Journal of Environmental Engineering 132(9): 1034-1042.

Williams, C.J., Frost, P.C., Xenopoulos, M.A. (2013). Beyond best management practices: Pelagic biogeochemical dynamics in urban stormwater ponds. *Ecological Applications* 23(6): 1384-1395.

Wingfield, R., Murphy, K.J., Gaywood, M. (2006). Assessing and predicting the success of Najas flexilis (Willd.) Rostk. & Schmidt, a rare European aquatic macrophyte in relation to lake environmental conditions. *Biology and Environmental Studies*: Journal Articles. Paper 1.

Wu, J.S., Holman, R.E., Dorney, J.R. (1996). Systematic evaluation of pollutant removal by urban wet detention ponds. *Journal of Environmental Engineering* 122(11): 983-988.

Yang, Y.Y., Lusk, M.G. (2018). Nutrients in urban stormwater runoff: Current state of the science and potential mitigation options. *Current Pollution Reports* 4(2): 112-127.

Zedler, J.B., Kercher, S. (2004). Causes and consequences of invasive plants in wetlands: Opportunities, opportunists, and outcomes. *Critical Reviews in Plant Sciences* 23(5): 431-452.

APPENDICES

Appendix A: Supplementary Material for Chapter 2

A.1 Figures

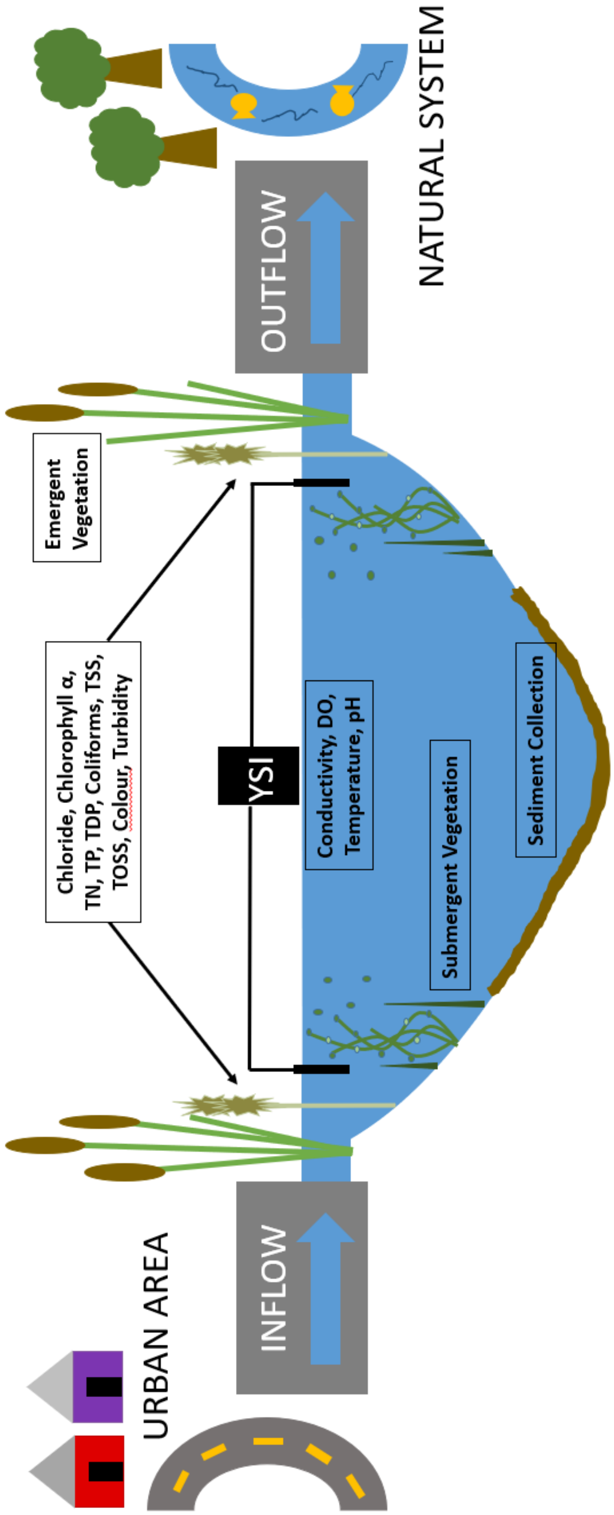

Figure A1. Concept map showing sampling locations for water conditions, water quality parameters, sediment collection, and vegetation sampling for SMP sites.

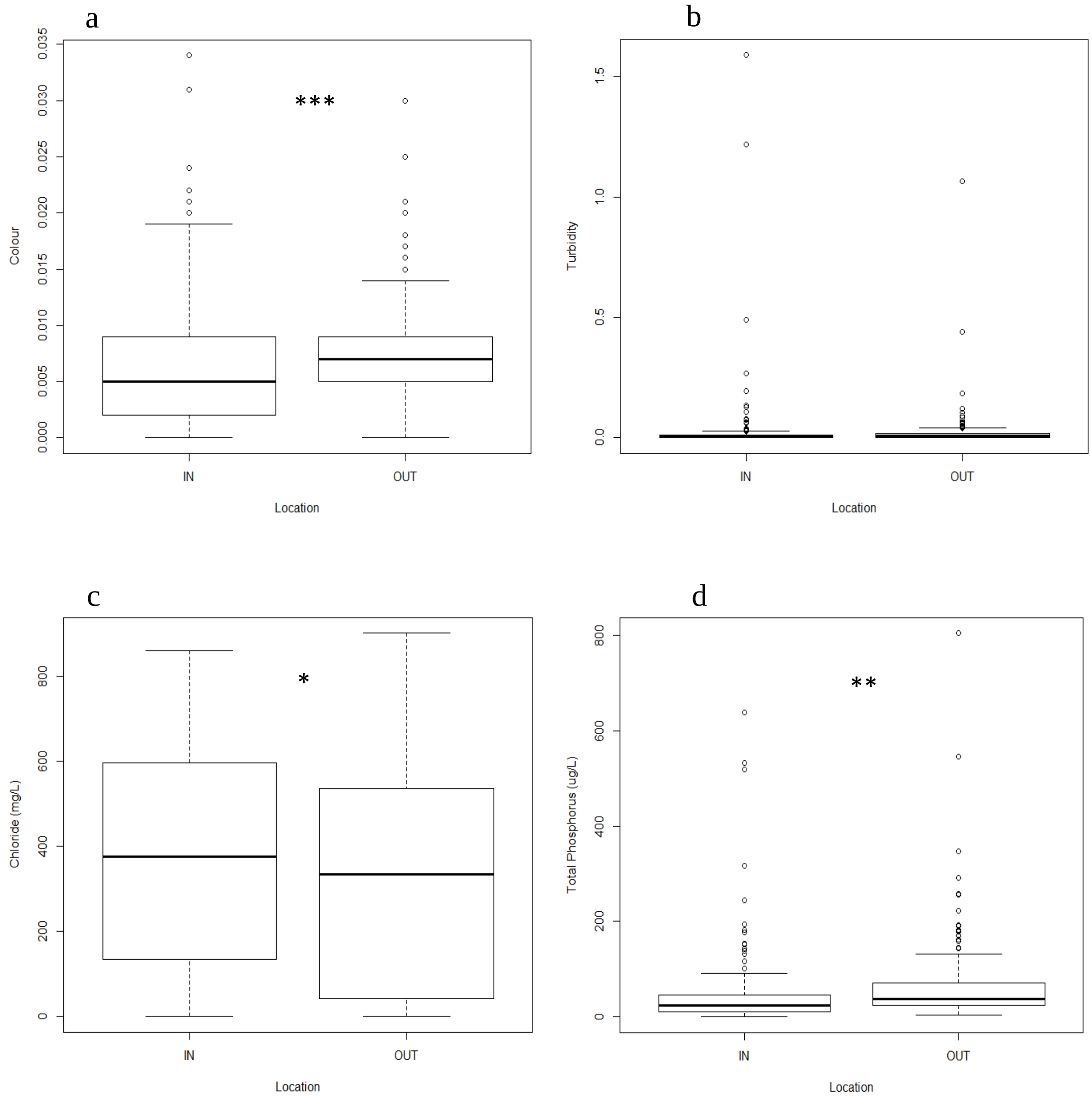

Figure A2. Colour (a), Turbidity (b), Chloride (c) and Total Phosphorus (d) for inflow and outflow locations all 15 ponds and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with p < 0.05 *, p < 0.01 **, p <0.001 ***.

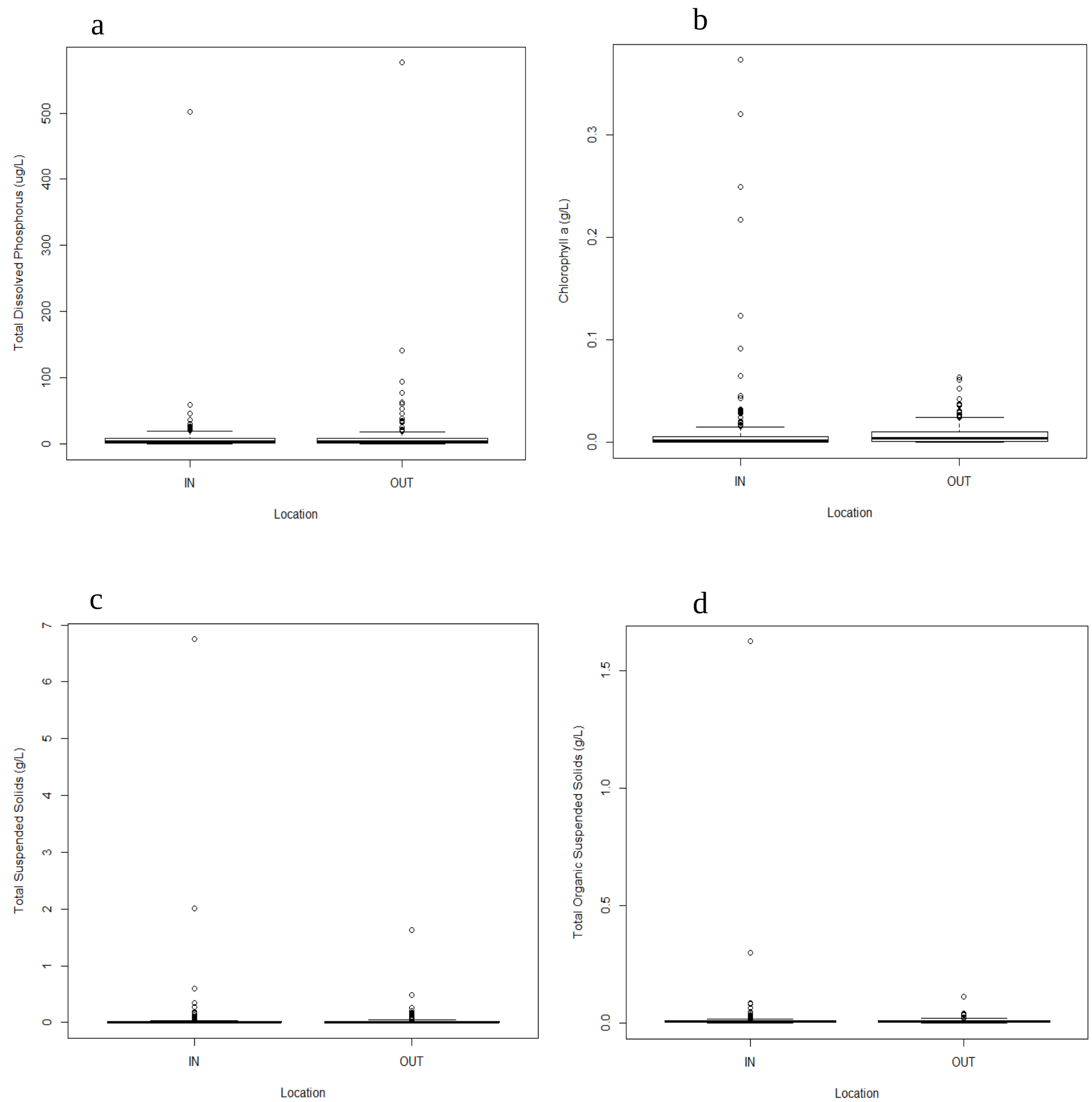

Figure A3. Total Dissolved Phosphorus (a), Chlorophyll α (b), Total Suspended Solids (c) and Total Organic Suspended Solids (d) for inflow and outflow locations all 15 ponds and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

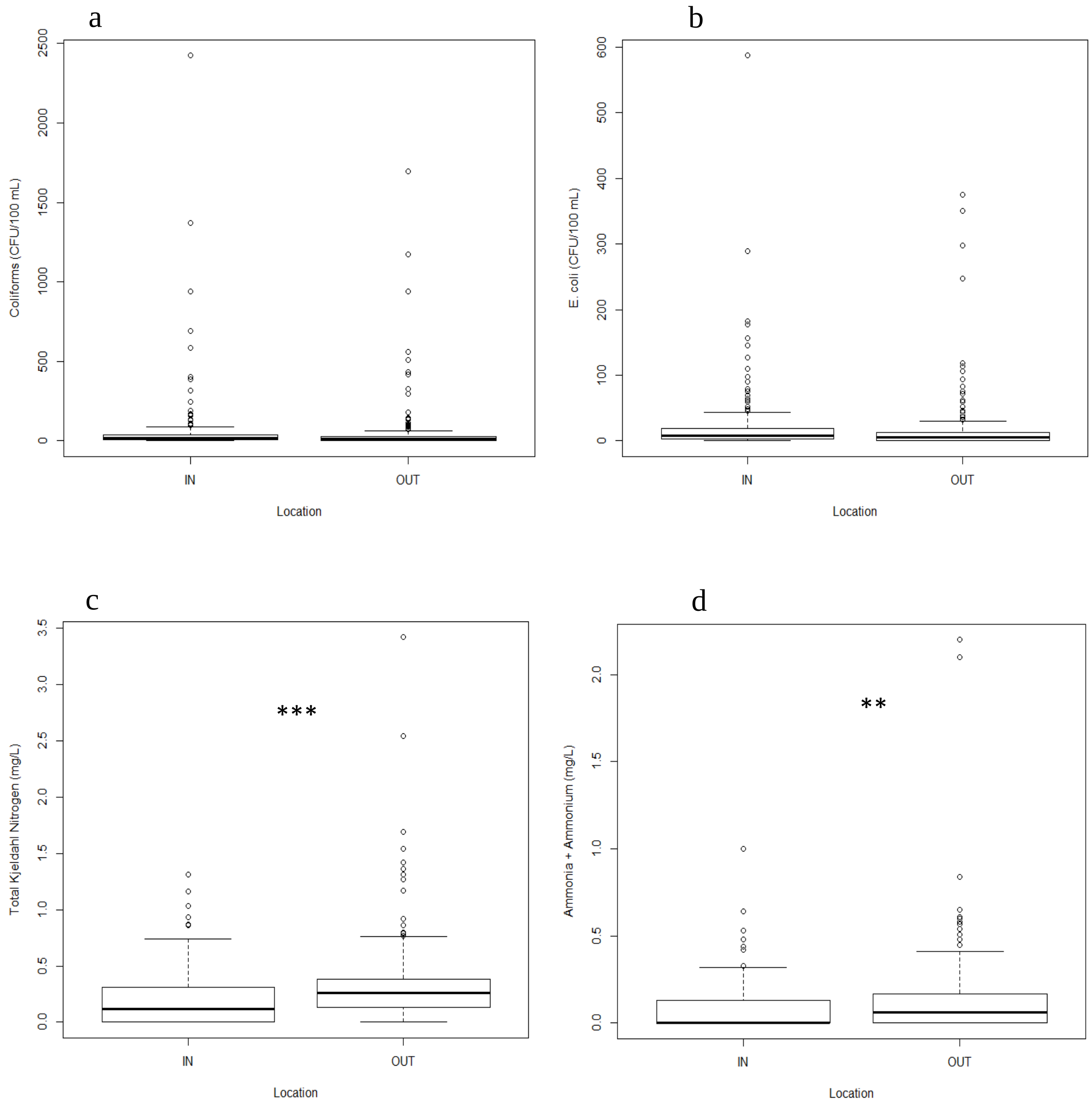

Figure A4. Coliforms (a), *E. coli* (b), Total Kjeldahl Nitrogen (c), and Ammonia + Ammonium (d) for inflow and outflow locations all 15 ponds and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

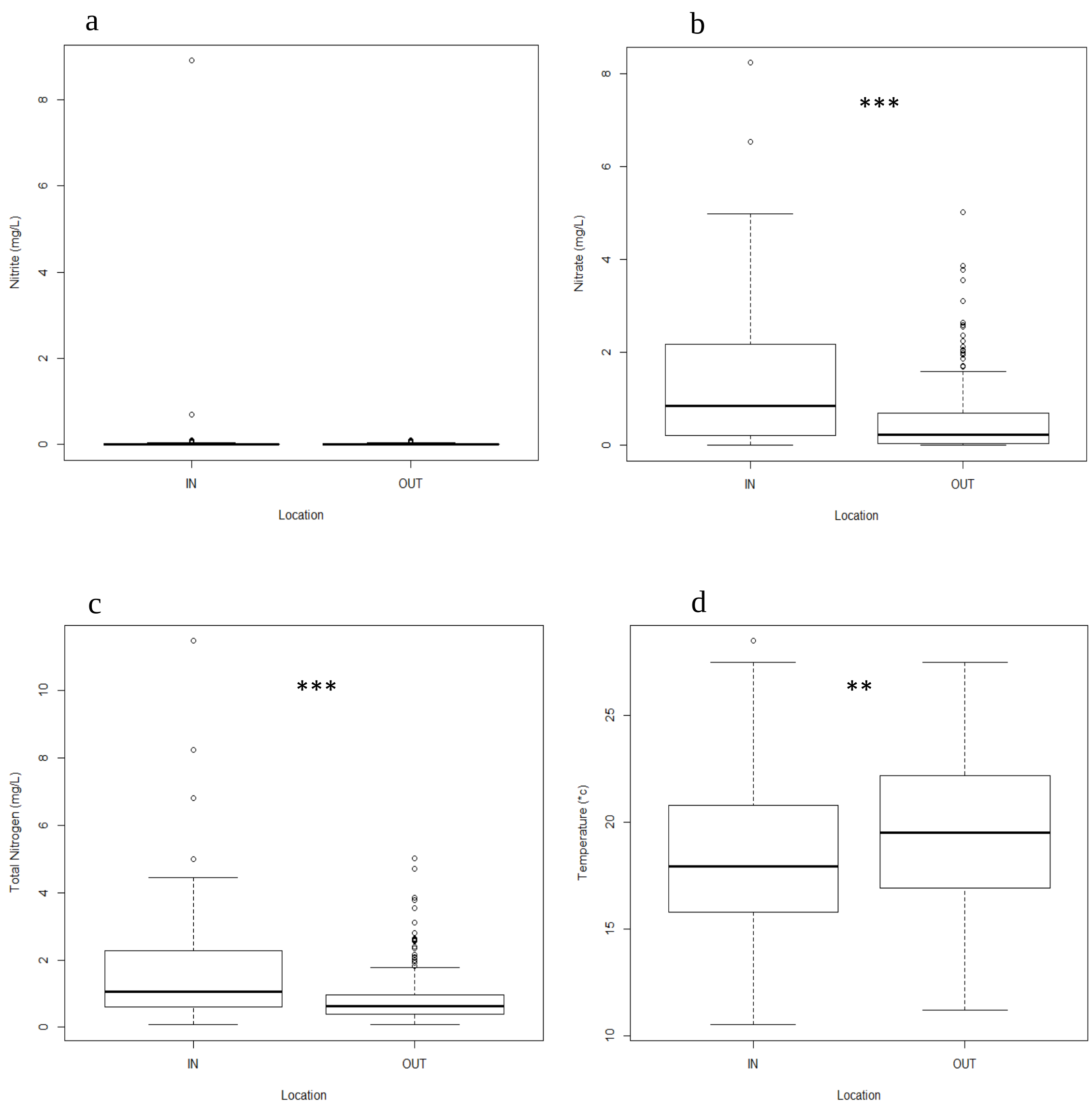

Figure A5. Nitrite (a), Nitrate (b), Total Nitrogen (c) and Temperature (d) for inflow and outflow locations all 15 ponds and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

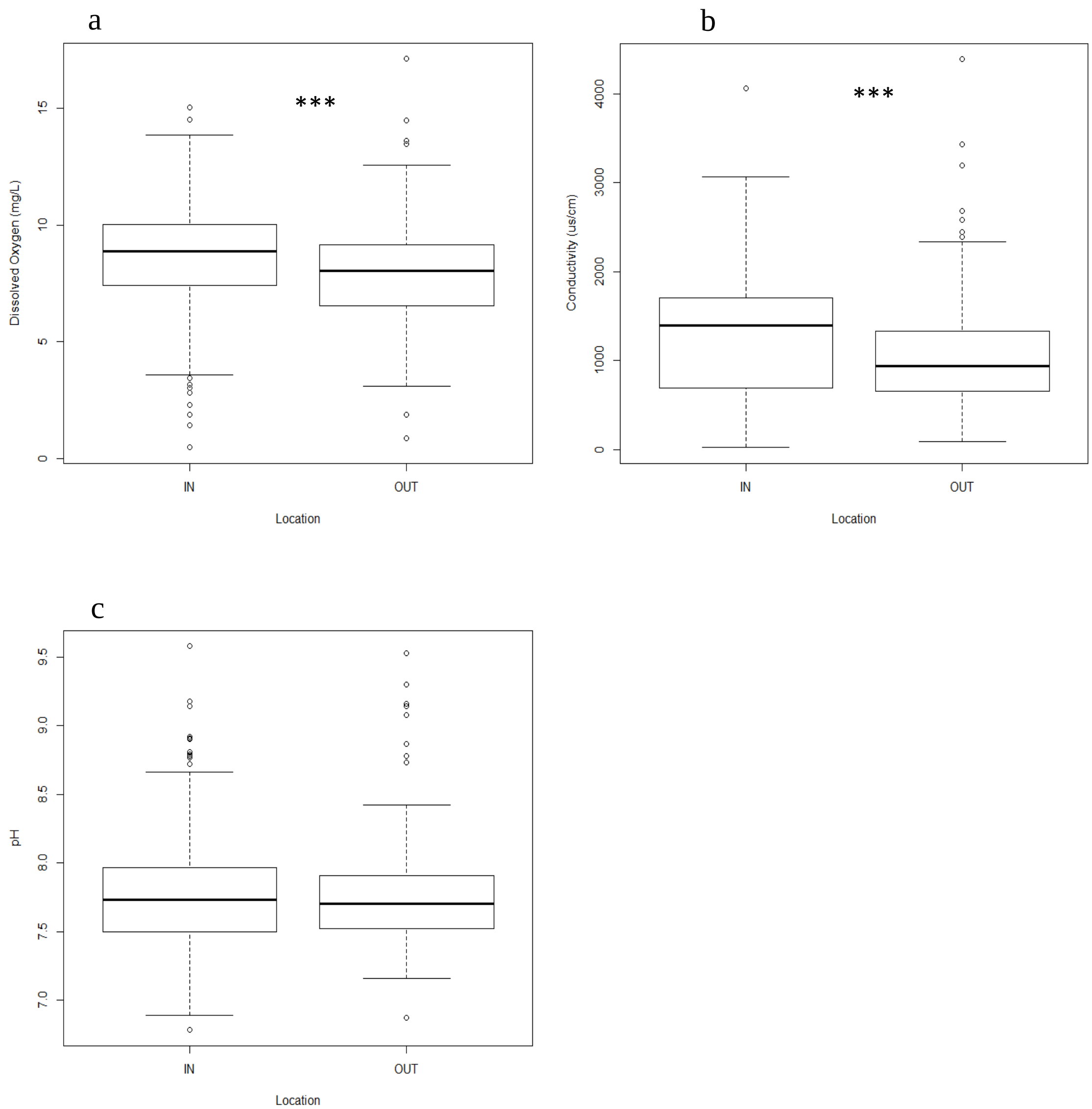

Figure A6. Dissolved Oxygen (a), Conductivity (b) and pH (c) for inflow and outflow locations all 15 ponds and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

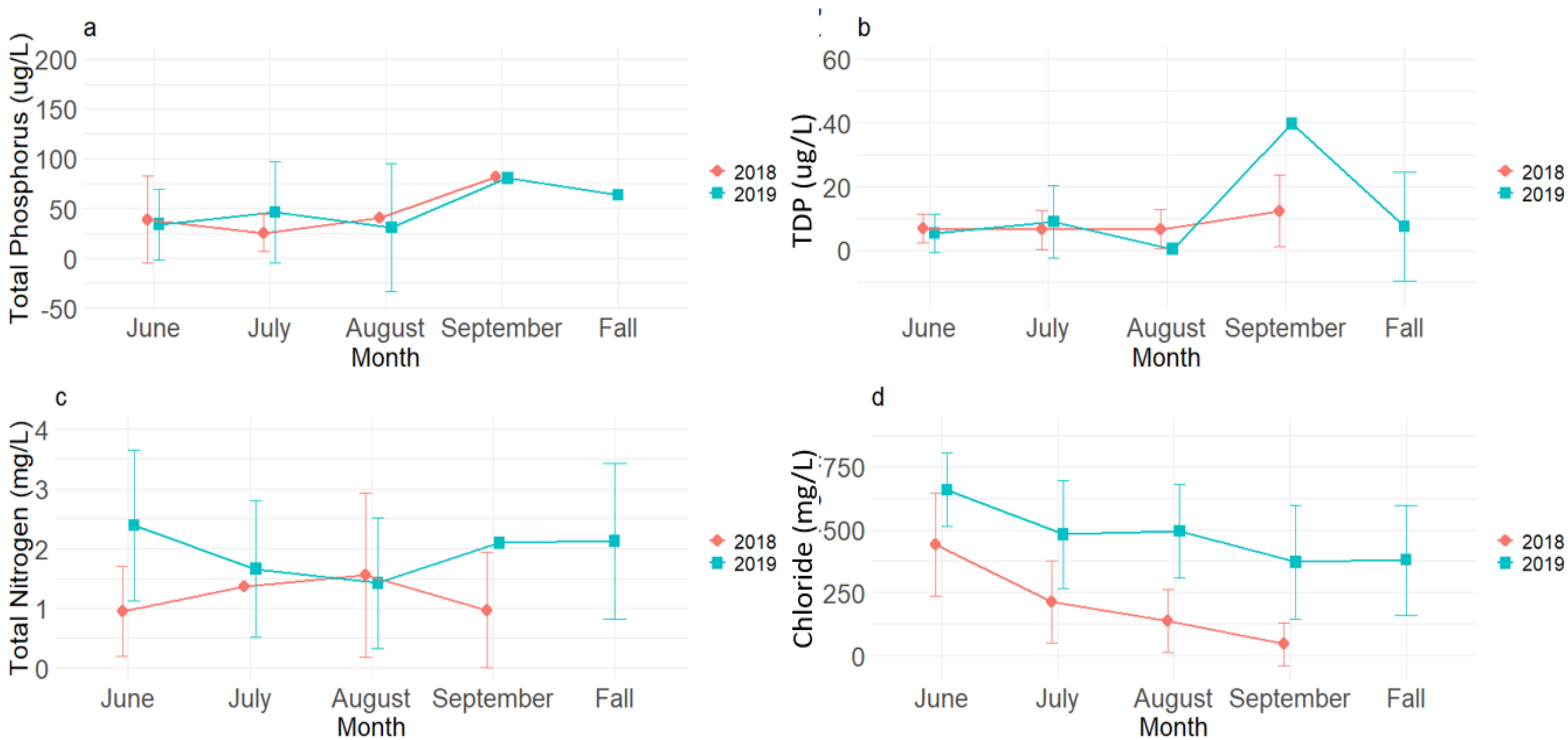

Figure A7. Average Total Phosphorus (a), Total Dissolved Phosphorus (b), Total Nitrogen (c) and Chloride (d) at the **inflow** location for all 15 ponds, illustrating changes in water quality across seasons. Fall includes two sampling dates in October and one date in November for 2019 only.

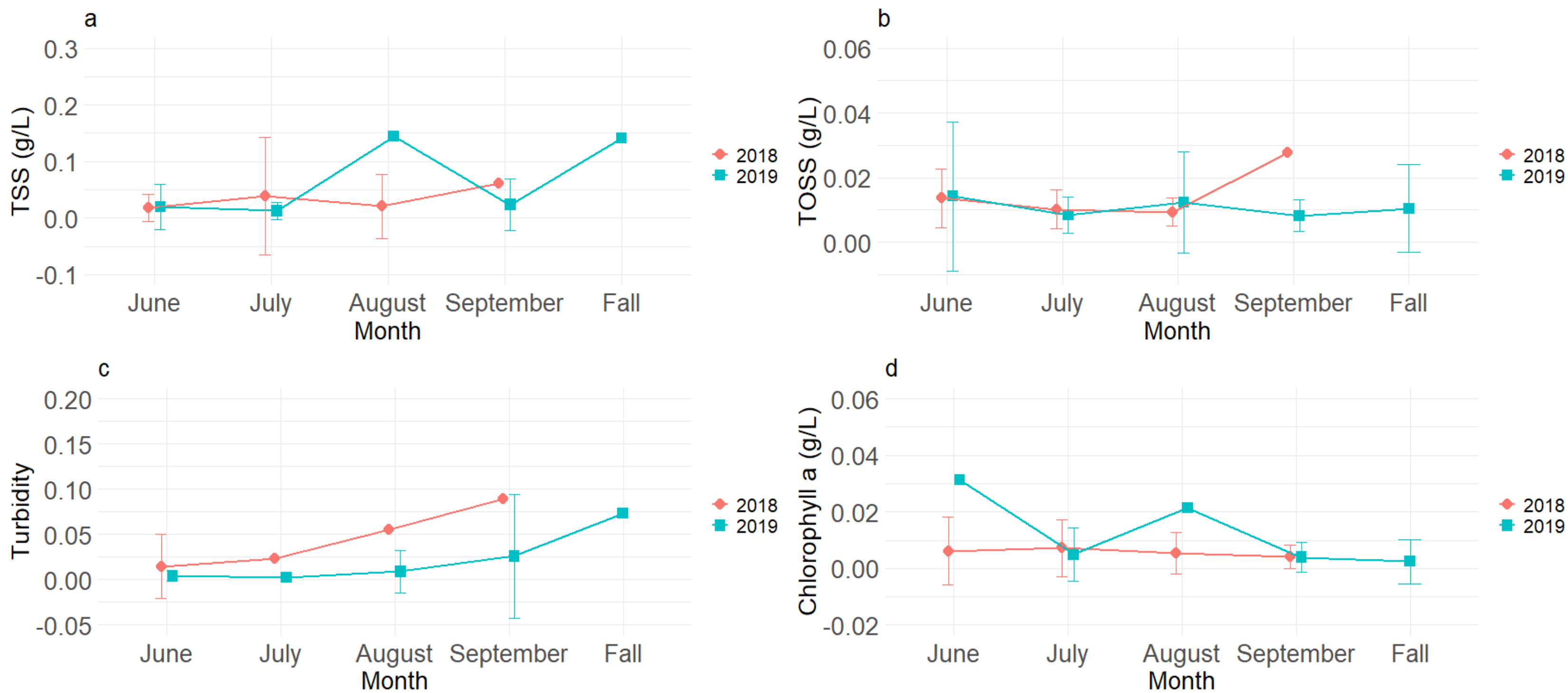

Figure A8. Average Total Suspended Solids (a), Total Organic Suspended Solids (b), Turbidity (c) and Chlorophyll α (d) at the **inflow** location for all 15 ponds, illustrating changes in water quality across seasons. Fall includes two sampling dates in October and one date in November for 2019 only.

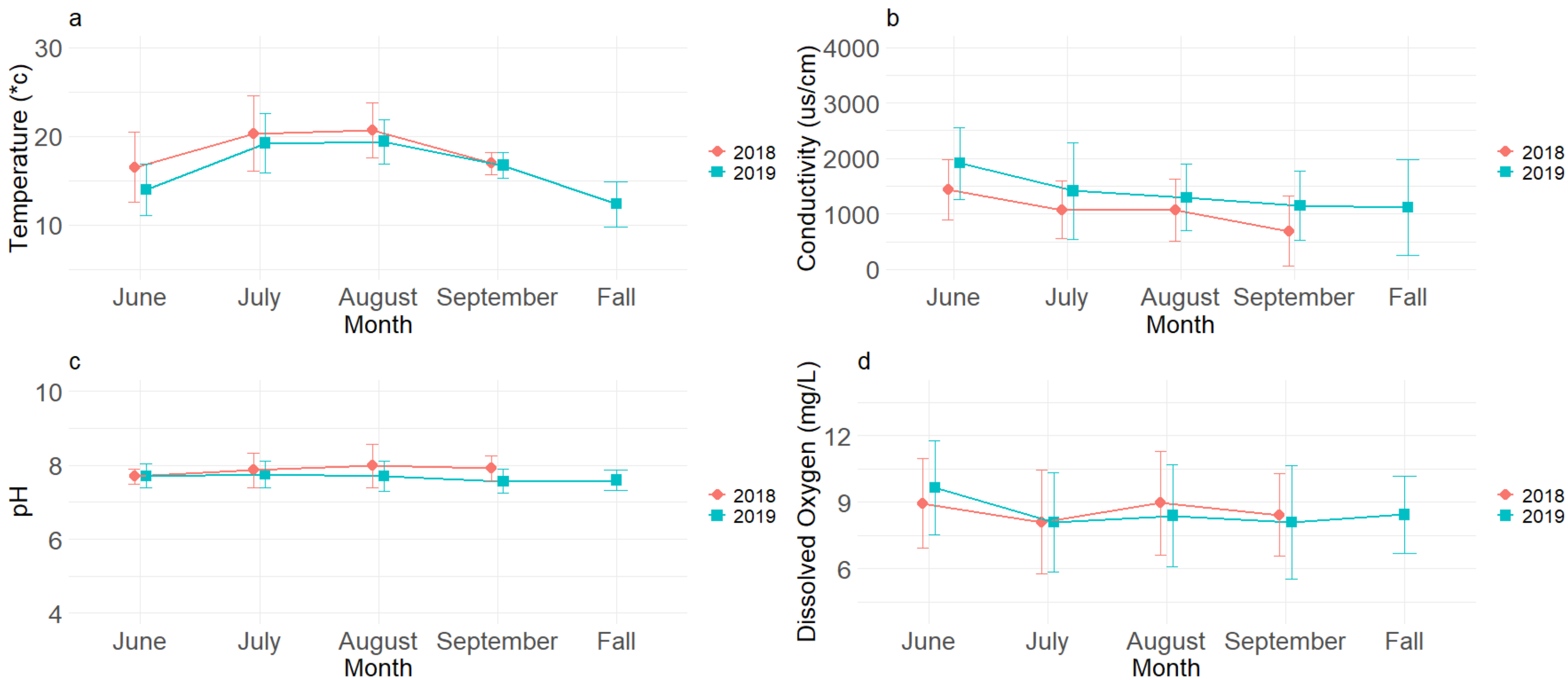

Figure A9. Temperature (a), Conductivity (b), pH (c) and Dissolved Oxygen (d) at the **inflow** location for all 15 ponds, illustrating changes in water quality across seasons. Fall includes two sampling dates in October and one date in November 2019 only.

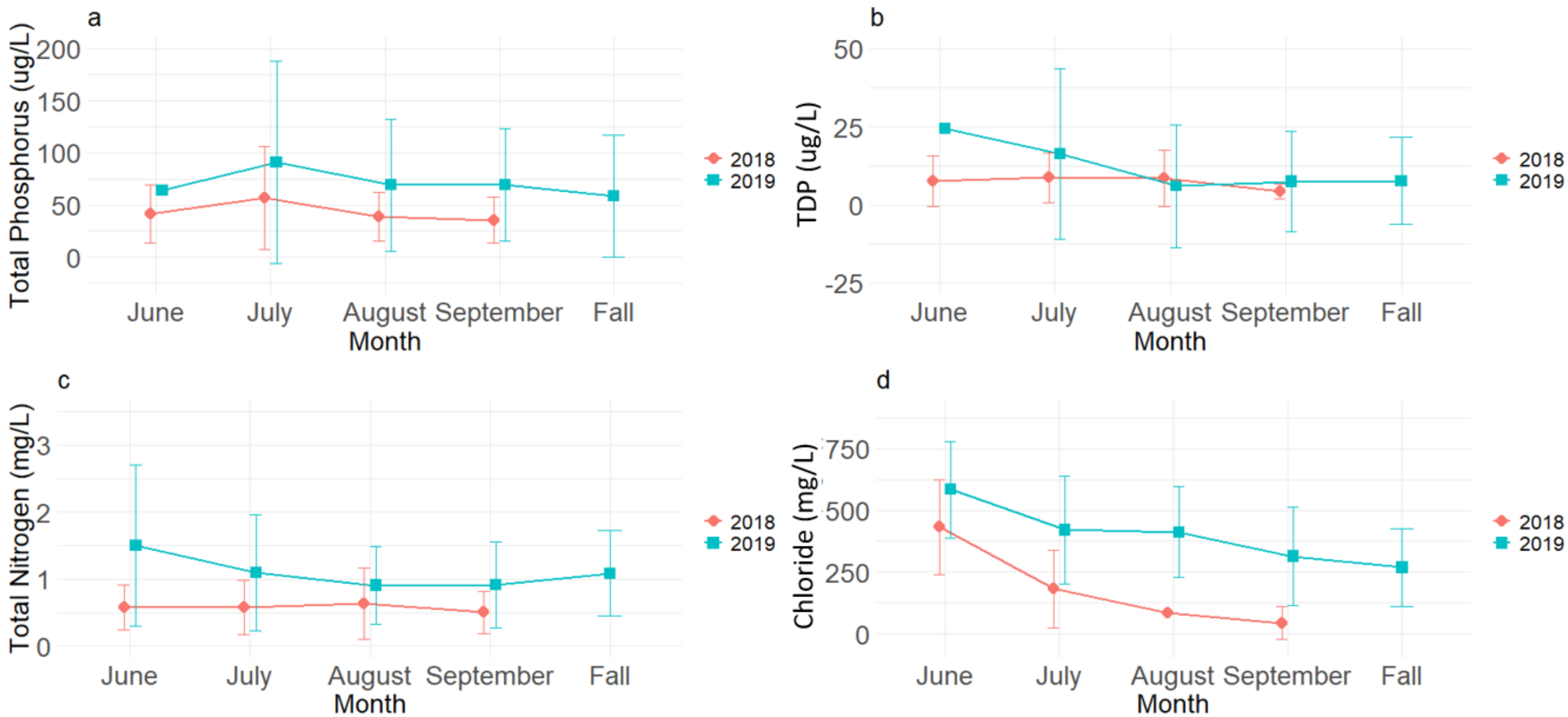

Figure A10. Average Total Phosphorus (a), Total Dissolved Phosphorus (b), Total Nitrogen (c) and Chloride (d) at the **outflow** location for all 15 ponds, illustrating changes in water quality across seasons. Fall includes two sampling dates in October and one date in November 2019 only.

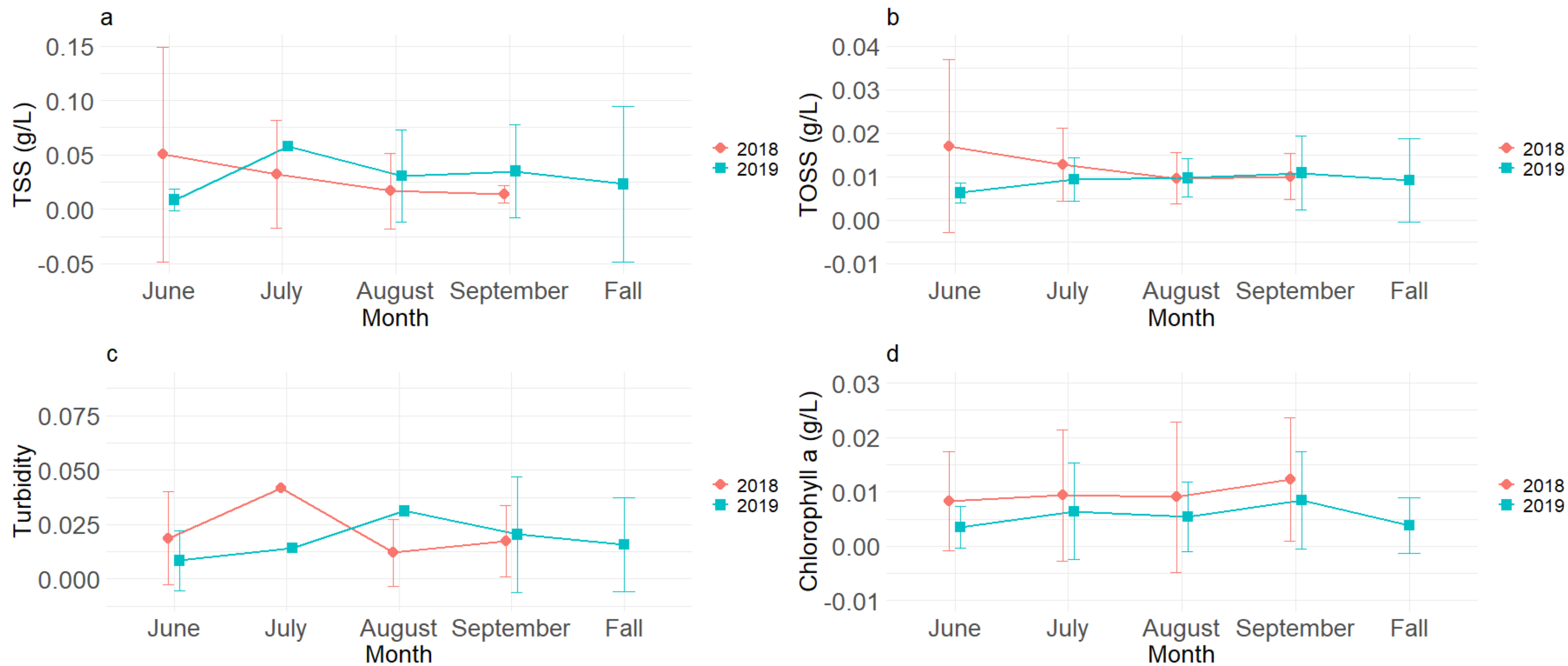

Figure A11. Average Total Suspended Solids (a), Total Organic Suspended Solids (b), Turbidity (c) and Chlorophyll α (d) at the **outflow** location for all 15 ponds, illustrating changes in water quality across seasons. Fall includes two sampling dates in October and one date in November 2019 only.

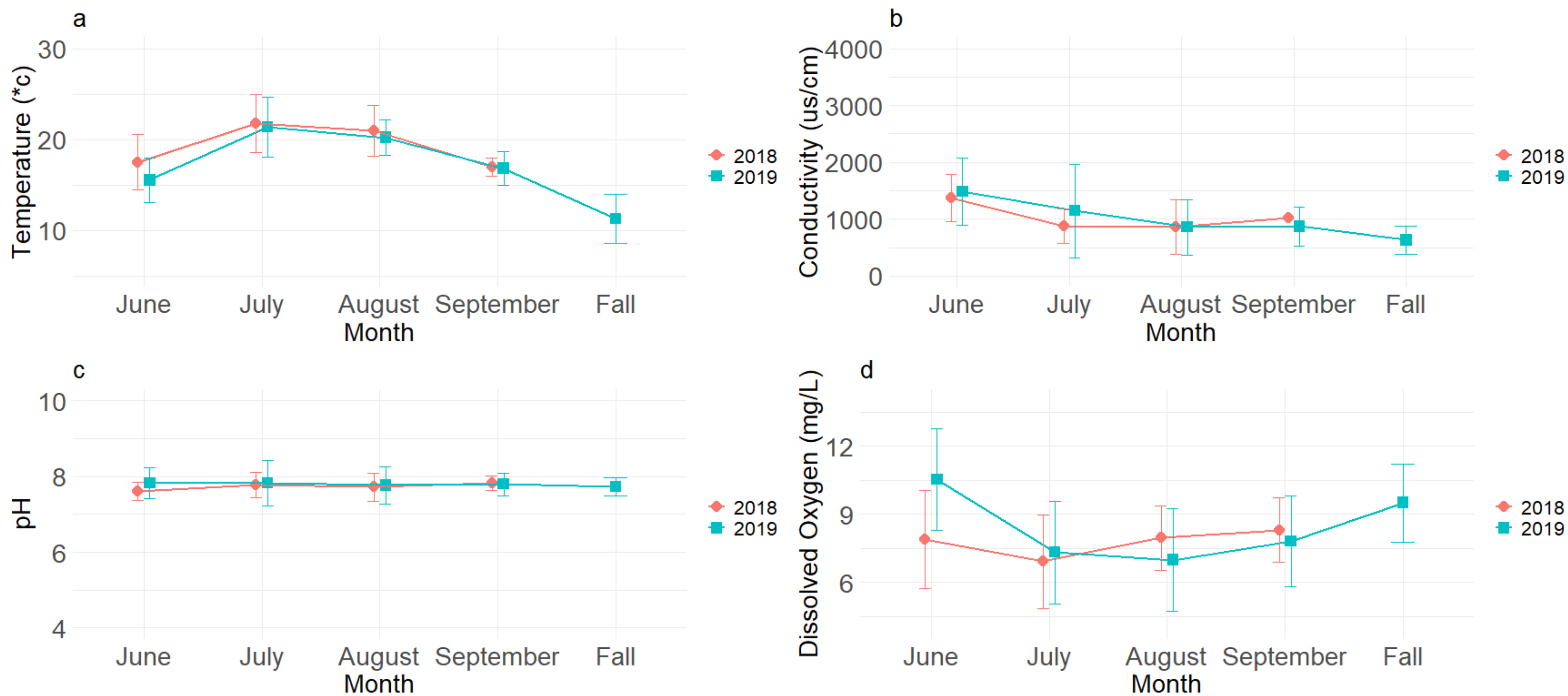

Figure A12. Temperature (a), Conductivity (b), pH (c) and Dissolved Oxygen (d) at the **outflow** location for all 15 ponds, illustrating changes in water quality across seasons. Fall includes two sampling dates in October and one date in November 2019 only.

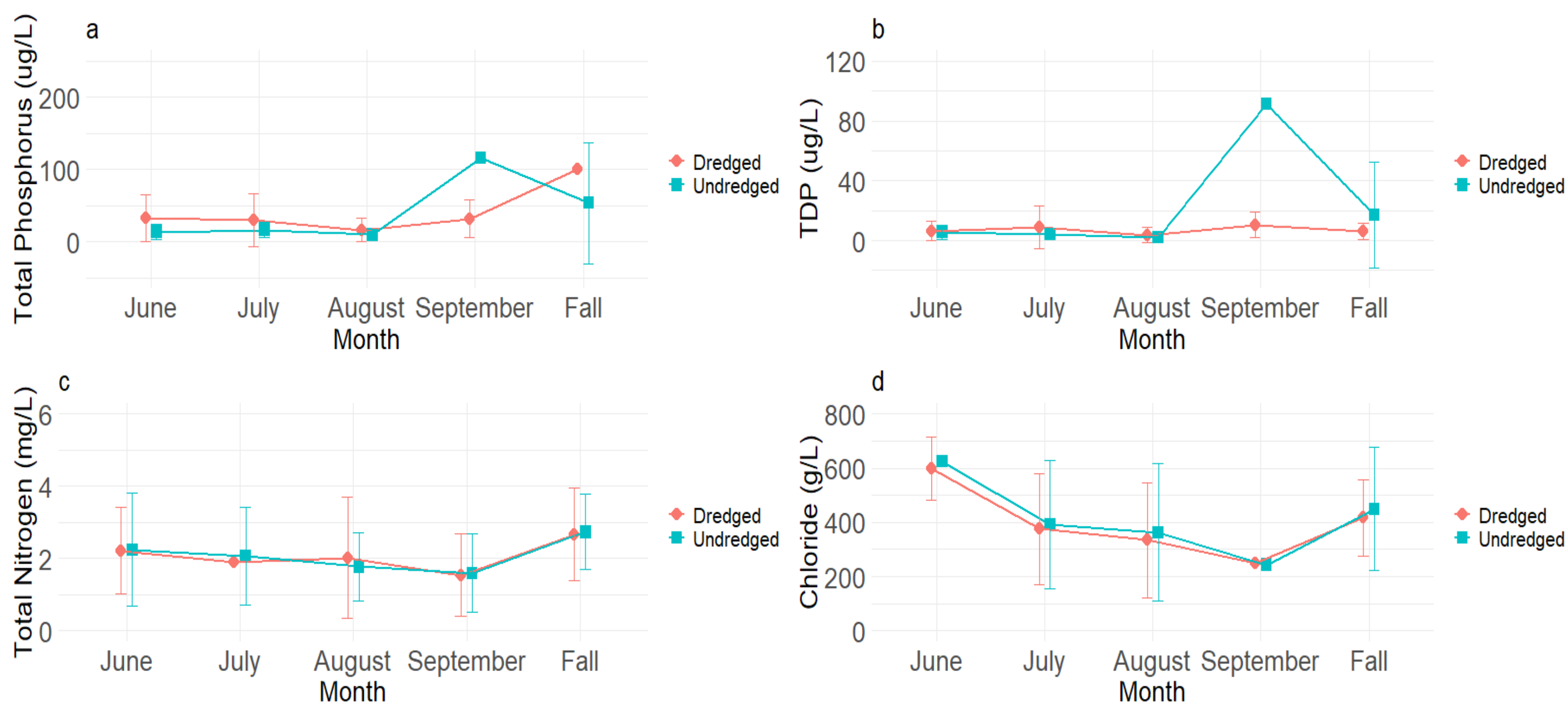

Figure A13. Total Phosphorus (a), Total Dissolved Phosphorus (b), Total Nitrogen (c) and Chloride (d) measured at **outflow** locations for three dredged and three undredged ponds, showing trends in seasonality across 2018 and 2019 combined. Fall includes two sampling dates in October and one date in November 2019 only.

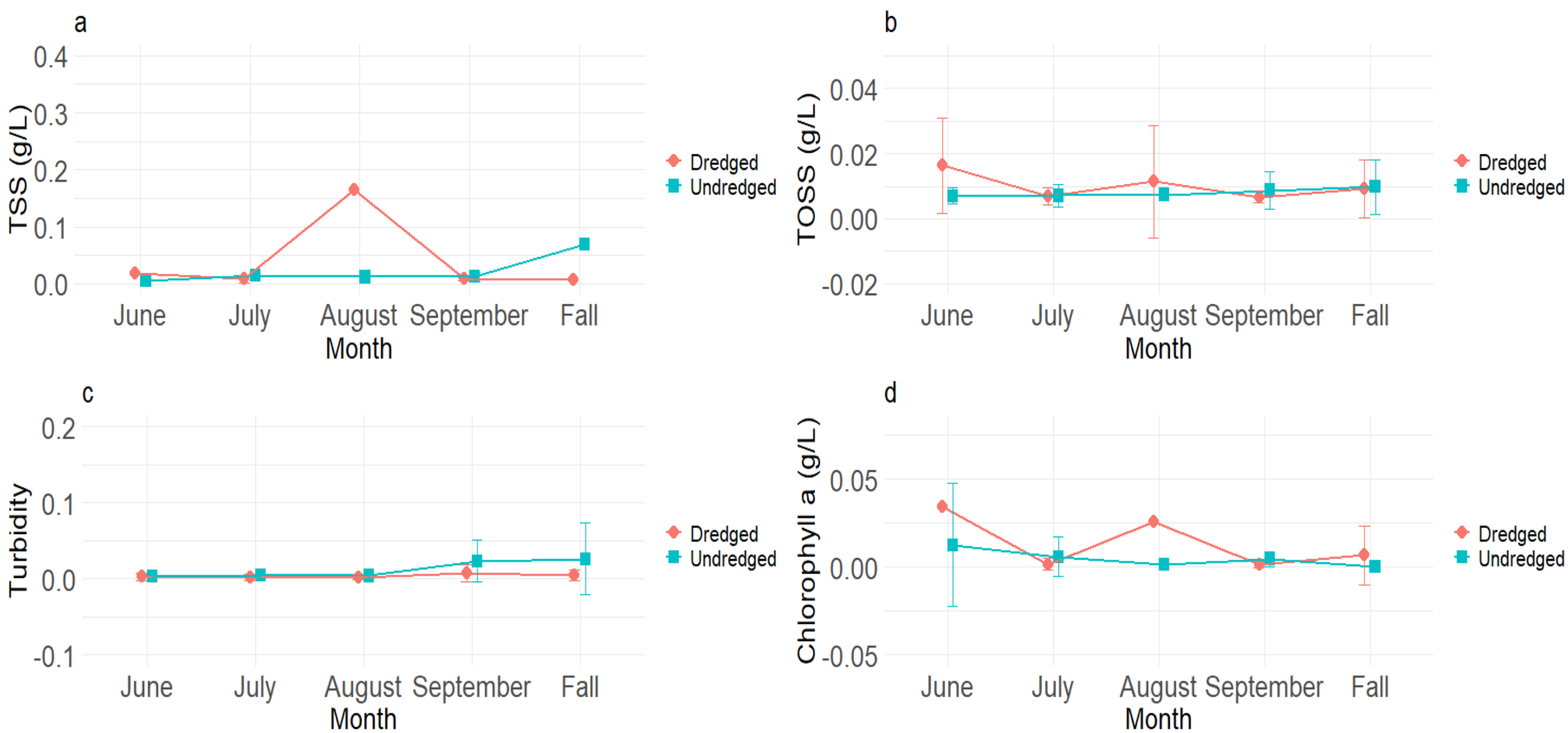

Figure A14. Total Suspended Solids (a), Total Organic Suspended Solids (b), Turbidity (c) and Chlorophyll α (d) measured at **outflow** locations for three dredged and three undredged ponds, showing trends in seasonality across 2018 and 2019 combined. Fall includes two sampling dates in October and one date in November 2019 only.

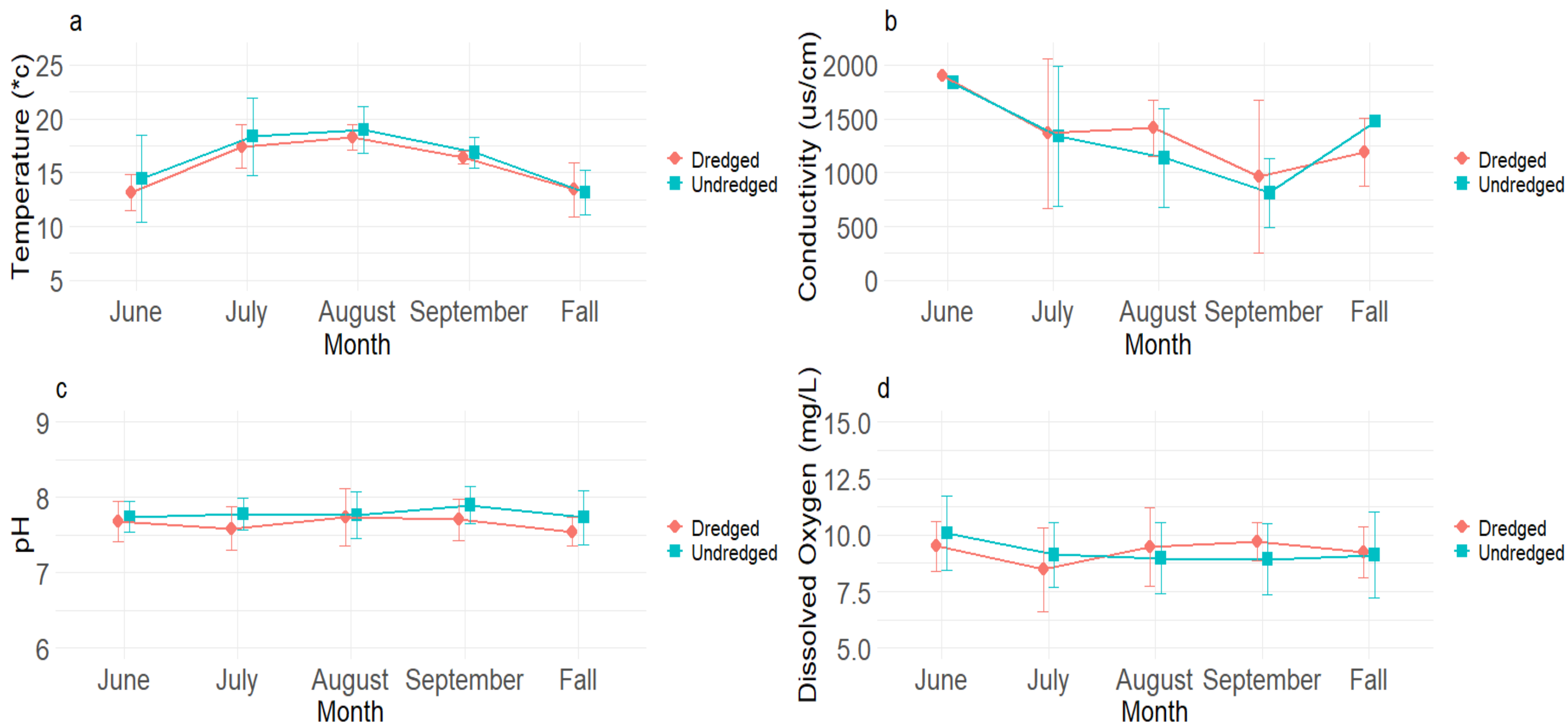

Figure A15. Temperature (a), Conductivity (b), pH (c) and Dissolved Oxygen (d) measured at **outflow** locations for three dredged and three undredged ponds, showing trends in seasonality across 2018 and 2019 combined. Fall includes two sampling dates in October and one date in November 2019 only.

A.2 Tables

Table A1. Pond 1 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
* Colour (A @ 440 nm)	0.005 (0.004)	0	0.014	0.008 (0.007)	0.002	0.014
Turbidity (A @ 750 nm)	0.015 (0.03)	0	0.128	0.018 (0.01)	0.004	0.046
Chloride (mg/L)	542.68 (154.7)	269.11	846.73	484.07 (175.99)	100.73	767.88
Total Phosphorus (µg/L)	68.15 (82.36)	8.65	316.57	78.07 (122.72)	13.08	545.75
Total Dissolved Phosphorus (µg/L)	7.45 (8.44)	0	30.46	14.06 (33.66)	0	141.75
Chlorophyll α (g/L)	0.0238 (0.0543)	0.0004	0.2172	0.0078 (0.0047)	0.0007	0.0163
Total Suspended Solids (g/L)	0.0374 (0.0546)	0	0.1744	0.0252 (0.0178)	0.0066	0.0666
Total Organic Suspended Solids (g/L)	0.0174 (0.0218)	0.006	0.0832	0.0128 (0.0062)	0.0052	0.0254
Total Coliforms (CFU/100 mL)	144.06 (331.49)	0	1370	88.25 (280.84)	0	1174
Total E. coli (CFU/100 mL)	26.71 (46.58)	0	182	8.57 (14.67)	0	59
** Total Nitrogen (mg/L)	2.67 (2.51)	0.31	11.48	0.63 (0.57)	0.11	2.56
*** Temperature (ºc)	15.36 (3.38)	10.5	22.9	20.23 (3.14)	13.6	25
Dissolved Oxygen (mg/L)	8.71 (2.53)	0.46	11	6.38 (4.06)	0.87	17.11
** Conductivity (µs/cm)	1534.88 (275.58)	975	2118	1138.04 (457.11)	407.7	2337
pH	7.47 (0.19)	7.2	7.9	7.61 (0.32)	7.16	8.42

Table A2. Pond 2 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
* Colour (A @ 440 nm)	0.005 (0.003)	0	0.013	0.01 (0.006)	0.002	0.021
Turbidity (A @ 750 nm)	0.007 (0.007)	0	0.027	0.141 (0.26)	0.011	1.065
Chloride (mg/L)	392.44 (296.97)	0	858.75	408.73 (217.27)	0	714.42
** Total Phosphorus (µg/L)	14.09 (10.52)	0	33.2	79.87 (64.49)	22.6	258.63
Total Dissolved Phosphorus (µg/L)	3.28 (3.57)	0	11.93	3.01 (3.09)	0	10.86
** Chlorophyll α (g/L)	0.0036 (0.0035)	0	0.0117	0.0081 (0.0052)	0.0193	0.0122
** Total Suspended Solids (g/L)	0.0119 (0.0127)	0	0.0554	0.0623 (0.0547)	0.0122	0.1984
*** Total Organic Suspended Solids (g/L)	0.0072 (0.0028)	0	0.013	0.0124 (0.0035)	0.0066	0.0188
Total Coliforms (CFU/100 mL)	16.44 (20.83)	0	90	29.13 (44.86)	0	182
Total E. coli (CFU/100 mL)	4.57 (4.91)	0	16	19.5 (31.26)	0	119
Total Nitrogen (mg/L)	1.01 (0.92)	0.17	3.11	1.29 (0.53)	2.15	0.29
Temperature (°c)	19.45 (4.51)	10.6	26.4	18.53 (3.18)	23.3	13.1
Dissolved Oxygen (mg/L)	8.17 (1.84)	5.51	10.78	9.05 (1.06)	10.69	6.33
Conductivity (µs/cm)	1193.51 (708.1)	386.5	2418	1101.86 (366.51)	541.7	1843
* pH	7.74 (0.19)	7.35	8.12	7.88 (0.15)	7.6	8.1

Table A3. Pond 3 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.004 (0.004)	0.001	0.018	0.005 (0.002)	0.001	0.01
Turbidity (A @ 750 nm)	0.004 (0.005)	0	0.015	0.009 (0.01)	0	0.04
Chloride (mg/L)	475.25 (228.09)	0	824.01	400.48 (208.31)	0	690.37
* Total Phosphorus (µg/L)	18.57 (19.23)	3.61	89.43	48.09 (50.57)	12.76	222.44
Total Dissolved Phosphorus (µg/L)	5.9 (7.69)	0	31.17	3.91 (3.42)	0	11.75
Chlorophyll α (g/L)	0.0148 (0.0302)	0	0.1237	0.0037 (0.0028)	0	0.0102
Total Suspended Solids (g/L)	0.0121 (0.0083)	0.0024	0.03	0.0589 (0.0849)	0	0.2632
* Total Organic Suspended Solids (g/L)	0.0067 (0.0031)	0	0.011	0.0129 (0.0108)	0.0044	0.0404
Total Coliforms (CFU/100 mL)	31.56 (36.44)	0	132	19.13 (21.92)	0	76
Total E. coli (CFU/100 mL)	14.5 (22.42)	0	79	9 (12.38)	0	39
Total Nitrogen (mg/L)	2.49 (1.2)	0.48	4.16			
Temperature (ºc)	15.93 (2.12)	10.7	18.9	16.7 (2.61)	11.2	21.8
** Dissolved Oxygen (mg/L)	9.91 (1.12)	8.49	12.65	8.51 (1.46)	6.37	11.86
Conductivity (µs/cm)	1532.5 (582.87)	487	2603	1144.36 (463.53)	89.6	1937
pH	7.6 (0.12)	7.44	7.8	7.69 (0.16)	7.39	7.96

Table A4. Pond 4 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with p < 0.05 *, p < 0.01 **, p <0.001 ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.006 (0.004)	0.001	0.02	0.006 (0.001)	0.004	0.009
Turbidity (A @ 750 nm)	0.004 (0.008)	0	0.029	0.001 (0.002)	0	0.007
Chloride (mg/L)	460 (213.83)	0	824.01	431.5 (217.56)	0	779.91
Total Phosphorus (µg/L)	27.65 (23.15)	1.94	91.49	18.61 (7.96)	7.84	42.65
Total Dissolved Phosphorus (µg/L)	3.94 (3.99)	0	15.94	5.56 (4.73)	0	14.35
Chlorophyll α (g/L)	0.0016 (0.0016)	0	0.0051	0.0022 (0.0037)	0	0.0159
Total Suspended Solids (g/L)	0.1346 (0.4829)	0.0018	2.0048	0.1081 (0.3923)	0.0004	1.6272
Total Organic Suspended Solids (g/L)	0.0083 (0.0041)	0.005	0.0222	0.0062 (0.0014)	0	0.0088
Total Coliforms (CFU/100 mL)	74.19 (223.83)	0	938	10.5 (15.69)	0	52
Total E. coli (CFU/100 mL)	12.64 (20.08)	0	69	1.14 (1.88)	0	5
Total Nitrogen (mg/L)	1.21 (0.76)	0.45	3.54	0.79 (0.57)	0.3	2.08
Temperature (ºc)	16.14 (1.91)	11.1	18.4	NA	NA	NA
Dissolved Oxygen (mg/L)	9.25 (0.76)	7.56	10.56	NA	NA	NA
Conductivity (µs/cm)	1623.84 (633.31)	101.5	2641	NA	NA	NA
pH	7.7 (0.17)	7.28	7.95	NA	NA	NA

Table A5. Pond 5 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.008 (0.003)	0.004	0.017	0.006 (0.003)	0	0.01
** Turbidity (A @ 750 nm)	0.014 (0.007)	0	0.03	0.034 (0.025)	0	0.091
Chloride (mg/L)	505.55 (273.63)	0	845.39	522.13 (260.47)	17.28	900.18
Total Phosphorus (µg/L)	85.58 (59.86)	23.93	244.84	72.23 (30.93)	20.29	132.47
Total Dissolved Phosphorus (µg/L)	6.64 (8.36)	0	36.89	8.83 (10.2)	0	34.19
Chlorophyll α (g/L)	0.0239 (0.0154)	0.0012	0.0649	0.0203 (0.0187)	0	0.0634
Total Suspended Solids (g/L)	0.0217 (0.0105)	0.005	0.0408	0.0319 (0.0197)	0.006	0.063
Total Organic Suspended Solids (g/L)	0.0154 (0.0058)	0.005	0.0296	0.0135 (0.0057)	0	0.0254
Total Coliforms (CFU/100 mL)	30.31 (27.32)	3	106	60.25 (99.82)	3	418
Total E. coli (CFU/100 mL)	16.5 (17.69)	0	65	47.07 (87.92)	0	350
Total Nitrogen (mg/L)	0.64 (0.31)	0.05	1.4	1.06 (1.2)	0.15	3.86
Temperature (ºc)	20.29 (3.13)	14.2	25.5	19.03 (3.14)	14.9	24.7
*** Dissolved Oxygen (mg/L)	9.77 (2.45)	4.58	15.04	6.66 (1.94)	4.1	12.56
Conductivity (µs/cm)	1730.61 (938.67)	106.7	4061	1872.88 (810.02)	708	3434
*** pH	7.84 (0.19)	7.48	8.16	7.47 (0.2)	7.2	7.79

Table A6. Pond 6 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
** Colour (A @ 440 nm)	0.003 (0.003)	0	0.011	0.007 (0.002)	0.004	0.011
*** Turbidity (A @ 750 nm)	0.003 (0.005)	0	0.015	0.012 (0.007)	0.003	0.034
Chloride (mg/L)	437.46 (246.33)	0	757.19	456.01 (249.6)	0	783.92
*** Total Phosphorus (µg/L)	11.09 (12.54)	0	41.97	58.29 (20.95)	26.11	91.2
Total Dissolved Phosphorus (µg/L)	4.52 (5.84)	0	19.9	5.25 (3.95)	0	15.78
*** Chlorophyll α (g/L)	0.0021 (0.0036)	0	0.0149	0.0169 (0.0071)	0	0.0228
** Total Suspended Solids (g/L)	0.0064 (0.0021)	0.0014	0.0144	0.0135 (0.0071)	0	0.0228
*** Total Organic Suspended Solids (g/L)	0.0061 (0.0021)	0.0036	0.0116	0.0103 (0.0026)	0	0.0132
Total Coliforms (CFU/100 mL)	66.81 (164.71)	0	694	124.81 (406.37)	0	1696
Total E. coli (CFU/100 mL)	17.64 (31.52)	0	127	14 (22.42)	0	83
*** Total Nitrogen (mg/L)	1.81 (1.05)	0.07	3.44	0.57 (0.21)	0.074	0.95
* Temperature (ºc)	17.01 (2.44)	12.2	20.2	19.69 (3.12)	14	25.9
Dissolved Oxygen (mg/L)	9.52 (1.64)	6.77	12.13	8.91 (0.57)	8.11	9.98
Conductivity (µs/cm)	1585.39 (536.61)	74.3	2540	1249.63 (408.76)	660	2083
*** pH	7.67 (0.29)	6.78	8.21	7.92 (0.12)	7.77	8.13

Table A7. Pond 7 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.007 (0.002)	0	0.01	0.007 (0.002)	0.004	0.011
Turbidity (A @ 750 nm)	0.007 (0.004)	0.001	0.018	0.006 (0.005)	0.001	0.021
Chloride (mg/L)	268.84 (205.45)	0	590.14	280.24 (209.89)	0	604.39
Total Phosphorus (µg/L)	39.22 (15.51)	11.76	73.38	47.68 (17.55)	24.35	76.73
Total Dissolved Phosphorus (µg/L)	6.06 (5.65)	0	17.93	9.49 (13.32)	0	46.73
Chlorophyll α (g/L)	0.0076 (0.0055)	0.0027	0.0295	0.0094 (0.0095)	0.0027	0.0361
Total Suspended Solids (g/L)	0.0109 (0.0055)	0	0.0228	0.01124 (0.0059)	0.003	0.0286
Total Organic Suspended Solids (g/L)	0.0098 (0.0039)	0.0056	0.0232	0.0094 (0.0046)	0.003	0.0246
Total Coliforms (CFU/100 mL)	48.25 (90.13)	0	317	124.5 (250.87)	0	938
Total E. coli (CFU/100 mL)	37.57 (82.8)	0	289	9 (10.81)	0	39
Total Nitrogen (mg/L)	0.38 (0.16)	0.17	0.78	0.46 (0.16)	0.26	0.93
Temperature (°c)	21.93 (3.62)	15.6	27.1	21.16 (3.02)	16	26.2
*** Dissolved Oxygen (mg/L)	9.43 (1.11)	6.7	11.23	7.49 (0.94)	5.74	9.53
Conductivity (µs/cm)	684.7 (450.41)	53.7	1616	716.93 (432.52)	347.8	1820
*** pH	8.3 (0.35)	7.73	8.91	7.75 (0.3)	7.24	8.37

Table A8. Pond 8 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
* Colour (A @ 440 nm)	0.004 (0.006)	0	0.021	0.008 (0.002)	0.004	0.011
Turbidity (A @ 750 nm)	0.017 (0.029)	0	0.107	0.02 (0.021)	0.004	0.071
Chloride (mg/L)	405.69 (245.49)	0	757.19	301.1 (227.83)	0	674.33
* Total Phosphorus (µg/L)	29.08 (40.17)	1.76	153.56	64.99 (47.52)	21.65	191.76
Total Dissolved Phosphorus (µg/L)	3.45 (3.37)	0	11.02	4.42 (3.18)	0	13.08
Chlorophyll α (g/L)	0.0179 (0.0599)	0	0.2493	0.0101 (0.0068)	0.0017	0.0295
Total Suspended Solids (g/L)	0.0361 (0.0436)	0.0034	0.1404	0.0553 (0.1175)	0	0.487
Total Organic Suspended Solids (g/L)	0.0165 (0.0208)	0	0.0876	0.0185 (0.0258)	0	0.1124
Total Coliforms (CFU/100 mL)	32.75 (45.93)	0	166	28.25 (34.77)	0	141
Total E. coli (CFU/100 mL)	22 (39.99)	0	156	15.21 (29.12)	0	114
*** Total Nitrogen (mg/L)	1.71 (0.93)	0.37	3.16	0.5 (0.23)	0.2	1.02
** Temperature (ºc)	17.16 (3.53)	10.7	27.5	20.69 (2.87)	16	24.4
** Dissolved Oxygen (mg/L)	8.91 (1.18)	5.6	10.19	6.19 (2.84)	3.25	14.47
* Conductivity (µs/cm)	1421.34 (521.61)	62.5	2325	990.24 (474.47)	324.8	1975
pH	7.62 (0.33)	6.89	8.34	7.58 (0.19)	7.34	7.97

Table A9. Pond 9 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
*** Colour (A @ 440 nm)	0.002 (0.003)	0	0.013	0.009 (0.004)	0.004	0.018
Turbidity (A @ 750 nm)	0.009 (0.019)	0	0.078	0.02 (0.028)	0.001	0.084
** Chloride (mg/L)	403.47 (223.38)	64.93	755.85	172.5 (196.26)	0	552.72
Total Phosphorus (µg/L)	46.47 (122.65)	0.28	518.61	38.88 (42.2)	9.17	162.29
Total Dissolved Phosphorus (µg/L)	35.6 (120.31)	0	501.39	5.31 (4.65)	0	17.93
* Chlorophyll α (g/L)	0.0002 (0.0004)	0	0.0015	0.0019 (0.0029)	0	0.0122
Total Suspended Solids (g/L)	0.0113 (0.0179)	0	0.0752	0.0247 (0.0426)	0	0.1804
Total Organic Suspended Solids (g/L)	0.008 (0.0035)	0	0.0184	0.0086 (0.0037)	0	0.0192
Total Coliforms (CFU/100 mL)	30.5 (19.9)	3	83	27.38 (25.21)	0	106
Total E. coli (CFU/100 mL)	16.07 (14.83)	0	46	18.71 (24.06)	0	94
*** Total Nitrogen (mg/L)	2.42 (1.0)	0.66	4.07	0.55 (0.25)	0.2	1.05
** Temperature (ºc)	16.73 (2.51)	11.7	20.7	20.01 (2.59)	13.9	22.8
*** Dissolved Oxygen (mg/L)	9.81 (0.76)	8.53	11.38	6.82 (1.88)	4.17	11.43
*** Conductivity (µs/cm)	1299.79 (529.52)	438.7	2347	656.31 (244.86)	236.4	1096
*** pH	7.99 (0.2)	7.53	8.4	7.58 (0.24)	7.21	8.06

Table A10. Pond 10 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.007 (0.002)	0.003	0.012	0.006 (0.002)	0.003	0.009
Turbidity (A @ 750 nm)	0.012 (0.008)	0.005	0.038	0.016 (0.016)	0.001	0.071
Chloride (mg/L)	261.69 (215.83)	0	603.5	268.76 (201.04)	0	596.82
Total Phosphorus (µg/L)	35.7 (16.77)	13.4	81.02	120.14 (192.75)	28.24	803.75
Total Dissolved Phosphorus (µg/L)	6.73 (7.01)	0	24.19	41.44 (138.14)	0	576.11
Chlorophyll α (g/L)	0.0104 (0.0059)	0.0022	0.0242	0.0133 (0.0098)	0.0012	0.0371
Total Suspended Solids (g/L)	0.0137 (0.009)	0	0.0386	0.0161 (0.0129)	0.003	0.0574
Total Organic Suspended Solids (g/L)	0.0099 (0.0025)	0.0062	0.0142	0.0097 (0.003)	0.0042	0.015
Total Coliforms (CFU/100 mL)	32.44 (42.71)	0	161	25.25 (25.36)	0	87
Total E. coli (CFU/100 mL)	27.29 (41.4)	0	146	20.14 (25.36)	0	76
Total Nitrogen (mg/L)	0.66 (0.32)	0.19	1.41			
Temperature (ºc)	21.23 (3.08)	14.7	26.2	NA	NA	NA
Dissolved Oxygen (mg/L)	7.31 (2.15)	3.87	12.48	NA	NA	NA
Conductivity (µs/cm)	840.86 (376.6)	461	1793	NA	NA	NA
pH	7.76 (0.24)	7.35	8.22	NA	NA	NA

Table A11. Pond 11 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.007 (0.007)	0	0.024	0.01 (0.005)	0.002	0.025
Turbidity (A @ 750 nm)	0.002 (0.003)	0	0.013	0.004 (0.01)	0	0.043
Chloride (mg/L)	313.75 (188.43)	0	604.84	255.5 (214.48)	0	636.91
Total Phosphorus (µg/L)	43.32 (37.18)	11.02	151.75	47.19 (38.3)	9.27	145.75
Total Dissolved Phosphorus (µg/L)	12.69 (13.89)	0	59.2	13.06 (8.59)	0	35.93
Chlorophyll α (g/L)	0.0453 (0.1143)	0	0.3728	0.0028 (0.009)	0	0.0376
Total Suspended Solids (g/L)	0.019 (0.0264)	0	0.0762	0.0143 (0.0355)	0	0.1508
Total Organic Suspended Solids (g/L)	0.0169 (0.0181)	0	0.066	0.0079 (0.0078)	0	0.0368
Total Coliforms (CFU/100 mL)	68.81 (140.21)	0	587	70.75 (162.66)	0	559
Total E. coli (CFU/100 mL)	65 (149.95)	0	587	54.36 (116.39)	0	375
** Total Nitrogen (mg/L)	2.88 (2.08)	0.55	8.24	1.22 (0.68)	0.3	2.6
* Temperature (ºc)	16.29 (2.98)	10.9	22.5	18.64 (2.81)	13	23.1
* Dissolved Oxygen (mg/L)	8.72 (1.98)	4.94	11.48	7.25 (1.61)	3.41	9.66
Conductivity (µs/cm)	1177.84 (470.64)	21.44	1597	945.56 (500.48)	187.7	2184
pH	7.74 (0.38)	7.26	8.92	7.54 (0.3)	6.87	8.19

Table A12. Pond 12 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with p < 0.05 *, p < 0.01 **, p <0.001 ***.

	INFLOW			**OUTFLOW**		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
*** Colour (A @ 440 nm)	0.012 (0.003)	0.008	0.021	0.003 (0.002)	0	0.007
Turbidity (A @ 750 nm)	0.003 (0.002)	0	0.008	0.007 (0.007)	0	0.024
** Chloride (mg/L)	70.55 (110.31)	0	336.22	226.31 (123.96)	38.2	444.8
* Total Phosphorus (µg/L)	58.87 (36.69)	15.46	182.6	31.65 (26.56)	4.67	101.81
** Total Dissolved Phosphorus (µg/L)	11.89 (7.97)	0	25.94	3.62 (3.04)	0	11.17
Chlorophyll α (g/L)	0.0054 (0.0092)	0	0.03	0.0039 (0.0056)	0	0.02
Total Suspended Solids (g/L)	0.049 (0.1423)	0	0.5976	0.0138 (0.0141)	0	0.049
Total Organic Suspended Solids (g/L)	0.0107 (0.0093)	0	0.0342	0.0113 (0.0094)	0	0.0408
Total Coliforms (CFU/100 mL)	14 (16.1)	0	62	20.15 (28.74)	0	106
Total E. coli (CFU/100 mL)	9.93 (12.14)	0	43	14.55 (26.33)	0	94
* Total Nitrogen (mg/L)	0.85 (0.38)	0.38	1.63	0.57 (0.24)	0.31	1.04
Temperature (ºc)	22.04 (3.37)	14.5	27.5	18.63 (5.24)	11.4	26.1
* Dissolved Oxygen (mg/L)	7.35 (2.9)	3.14	13.11	9.64 (1.57)	7.73	13.48
Conductivity (µs/cm)	599.21 (595.83)	176	2215	911.3 (679.45)	206.8	2245
pH	8.3 (0.58)	7.39	9.58	8.2 (0.72)	7.55	9.53

Table A13. Pond 13 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.004 (0.006)	0	0.022	0.006 (0.005)	0.002	0.018
* Turbidity (A @ 750 nm)	0.003 (0.009)	0	0.036	0.01 (0.007)	0	0.022
Chloride (mg/L)	488.72 (192.24)	188.58	781.24	322.67 (257.81)	0	698.39
** Total Phosphorus (µg/L)	17.21 (16.16)	5.94	73.56	42.46 (21.96)	7.37	80.65
Total Dissolved Phosphorus (µg/L)	8.31 (5.82)	0	21.56	7.69 (4.95)	2.69	18.11
* Chlorophyll α (g/L)	0.0003 (0.0004)	0	0.001	0.0043 (0.0061)	0	0.0251
* Total Suspended Solids (g/L)	0.0061 (0.0054)	0.001	0.024	0.0361 (0.0515)	0.0014	0.2026
* Total Organic Suspended Solids (g/L)	0.0059 (0.0028)	0	0.0122	0.0128 (0.009)	0	0.0392
Total Coliforms (CFU/100 mL)	25.81 (22.02)	0	98	45.81 (73.14)	0	298
Total E. coli (CFU/100 mL)	19.64 (23.33)	0	98	31.36 (63.16)	0	247
** Total Nitrogen (mg/L)	1.16 (0.5)	0.56	2.11	0.67 (0.38)	0.23	1.64
Temperature (ºc)	16.83 (2.18)	12.2	19.5	18.04 (2.72)	11.4	22.3
* Dissolved Oxygen (mg/L)	8.23 (0.99)	6.42	10.57	9.02 (0.46)	8.39	9.71
*** Conductivity (µs/cm)	1943.88 (538.12)	820	3070	1113.02 (493.18)	487.3	2445
** pH	7.41 (0.16)	7.16	7.8	7.64 (0.21)	7.23	8.07

Table A14. Pond 14 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with p < 0.05 *, p < 0.01 **, p <0.001 ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
Colour (A @ 440 nm)	0.012 (0.009)	0.003	0.034	0.015 (0.007)	0.005	0.03
Turbidity (A @ 750 nm)	0.03 (0.063)	0.001	0.269	0.001 (0.002)	0	0.006
* Chloride (mg/L)	160.67 (221.9)	0	582.12	2.39 (9.25)	0	38.2
** Total Phosphorus (µg/L)	41.29 (46.48)	3.61	194.72	123.11 (80.89)	30.38	290.83
** Total Dissolved Phosphorus (µg/L)	8.76 (11.48)	0	45.93	36.91 (31.8)	0	94.84
Chlorophyll α (g/L)	0.0025 (0.0036)	0	0.0124	0.0045 (0.0083)	0	0.0261
Total Suspended Solids (g/L)	0.0242 (0.0429)	0.0008	0.1866	0.0073 (0.0036)	0.0018	0.0162
Total Organic Suspended Solids (g/L)	0.0084 (0.0034)	0.0004	0.0168	0.0059 (0.0033)	0	0.012
Total Coliforms (CFU/100 mL)	48.94 (92.38)	3	388	12.88 (16.03)	0	65
Total E. coli (CFU/100 mL)	28.64 (38.57)	3	127	5.86 (9.55)	0	33
Total Nitrogen (mg/L)	0.91 (0.45)	0.41	1.93	1.01 (0.42)	0.34	1.82
Temperature (ºc)	21.37 (3.91)	12.3	28.5	NA	NA	NA
Dissolved Oxygen (mg/L)	8.15 (2.82)	3.01	14.5	NA	NA	NA
Conductivity (µs/cm)	606.81 (468.52)	217.3	1755	NA	NA	NA
pH	8.24 (0.48)	7.37	9.18	NA	NA	NA

Table A15. Pond 15 water quality parameters for inflow and outflow locations and all sampling dates in 2018 and 2019 (except Fall 2019). Welch two sample t-test significant differences are labelled with $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.

	INFLOW			OUTFLOW		
Parameter	**Mean (SD)**	**Min.**	**Max.**	**Mean (SD)**	**Min.**	**Max.**
* Colour (A @ 440 nm)	0.004 (0.004)	0	0.013	0.007 (0.003)	0.003	0.015
Turbidity (A @ 750 nm)	0.236 (0.463)	0	1.59	0.011 (0.013)	0	0.037
Chloride (mg/L)	348.42 (191.6)	7.03	612.86	232.06 (195.2)	0	521.98
Total Phosphorus (µg/L)	93.52 (187.28)	0	637.21	29.98 (13.31)	15.93	62.47
Total Dissolved Phosphorus (µg/L)	3.16 (3.61)	0	12.92	4.02 (3.36)	0	11.02
Chlorophyll α (g/L)	0.0019 (0.0036)	0	0.0149	0.0032 (0.0037)	0	0.0149
Total Suspended Solids (g/L)	0.4831 (1.6211)	0.0016	6.75	0.015 (0.0186)	0.002	0.0774
Total Organic Suspended Solids (g/L)	0.1306 (0.3922)	0	1.625	0.0076 (0.0029)	0.002	0.0144
Total Coliforms (CFU/100 mL)	171.81 (581.99)	0	2424	22.69 (35.14)	0	136
Total E. coli (CFU/100 mL)	10.86 (18.47)	0	76	15.93 (28.37)	0	106
*** Total Nitrogen (mg/L)	2.51 (1.28)	0.69	4.99	0.49 (0.15)	0.3	0.75
Temperature (ºc)	17.62 (2.45)	13	21.5	22.69 (3.14)	16.1	27.5
** Dissolved Oxygen (mg/L)	5.58 (2.84)	1.43	10.09	8.54 (2.01)	5.63	13.59
* Conductivity (µs/cm)	1657.56 (596.48)	843	2725	904.08 (925.81)	390.9	4386
*** pH	7.51 (0.3)	6.96	8.06	8.38 (0.48)	7.54	9.16

Appendix B: Supplementary Material for Chapter 3

B.1 Figures

Figure B1. Sample image captured by drone on September 6, 2018 showing overhead view of Pond 3. Images collected by drone were used to estimate percent emergent plant coverage using pond measurements obtained from Google Earth.

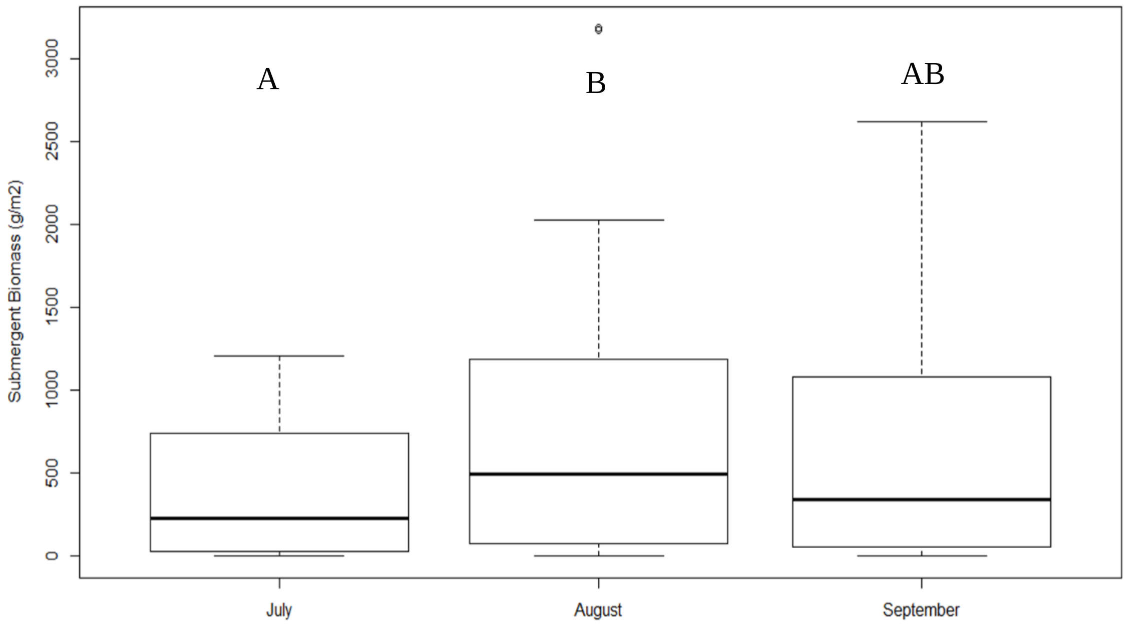

Figure B2. Total submergent biomass (g/m^2) by month for combined years (2018 and 2019). Welch two sample t-test where July (A) and August (B) are significantly different ($p < 0.05$).

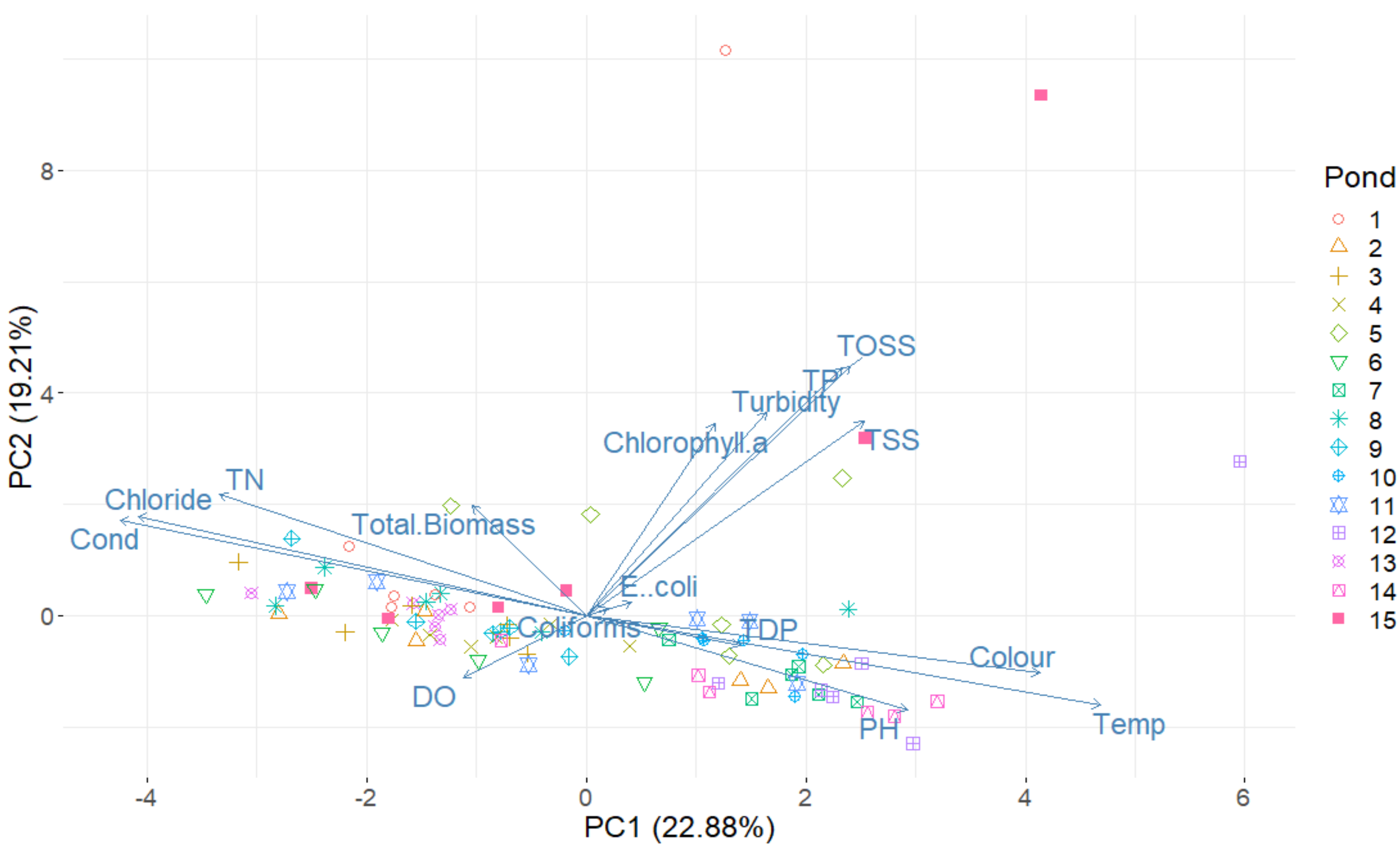

Figure B3. Principal component analysis for **inflow** water quality variables and submergent biomass collected in 2018 and 2019 dates when submergent vegetation was sampled. PC1 represents 19.21% of the variability and PC2 represents 22.88% of the variability.

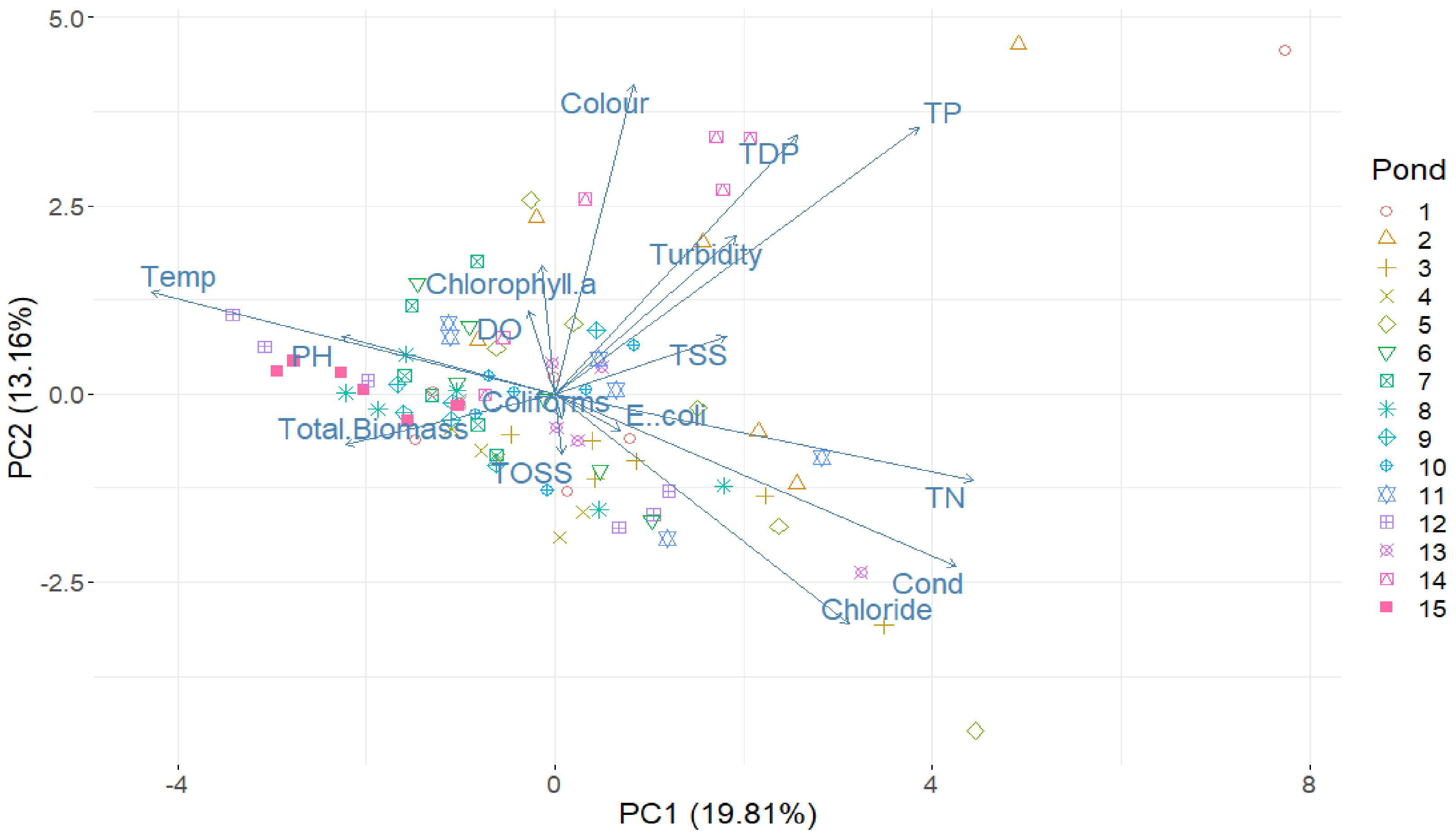

Figure B4. Principal component analysis for **outflow** water quality variables and submergent biomass collected in 2018 and 2019 dates when submergent vegetation was sampled. PC1 represents 19.81% of the variability and PC2 represents 13.16% of the variability.

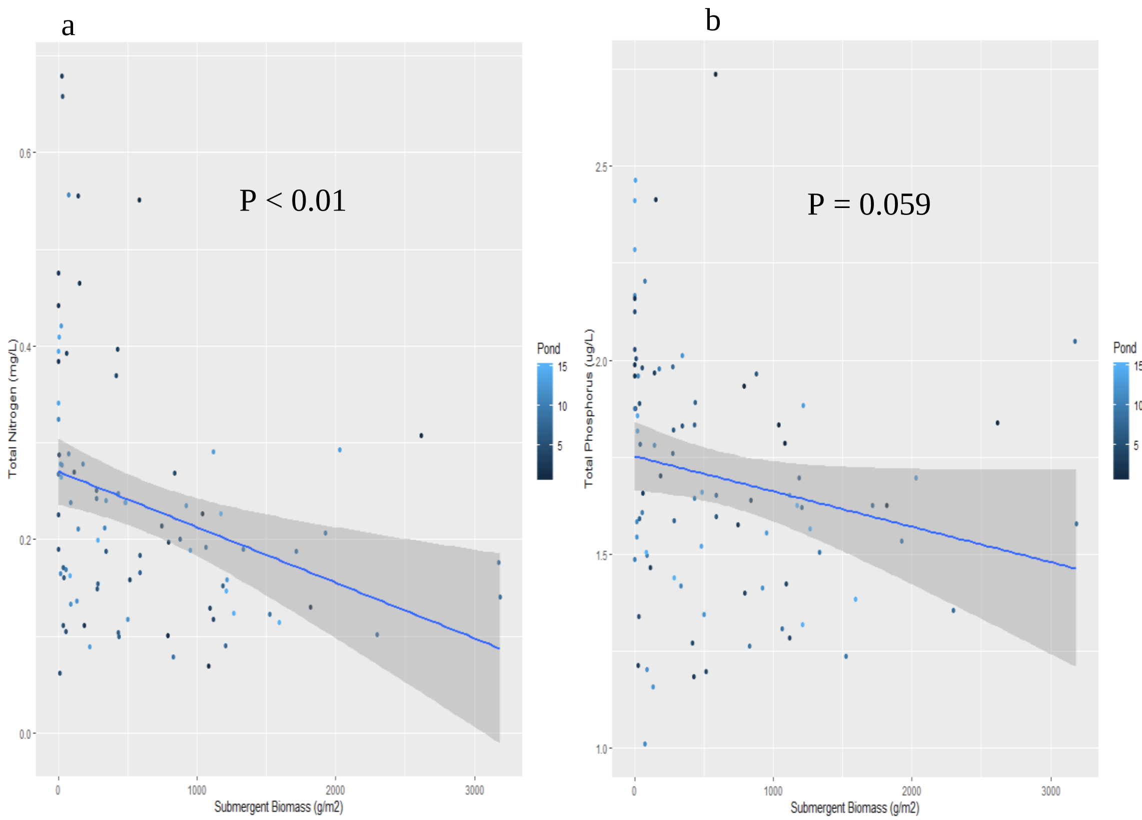

Figure B5. Pearson correlation analysis for submergent plant biomass and **outflow** total nitrogen (a) and total phosphorus (b) including dates when submergent vegetation was collected in 2018 and 2019.

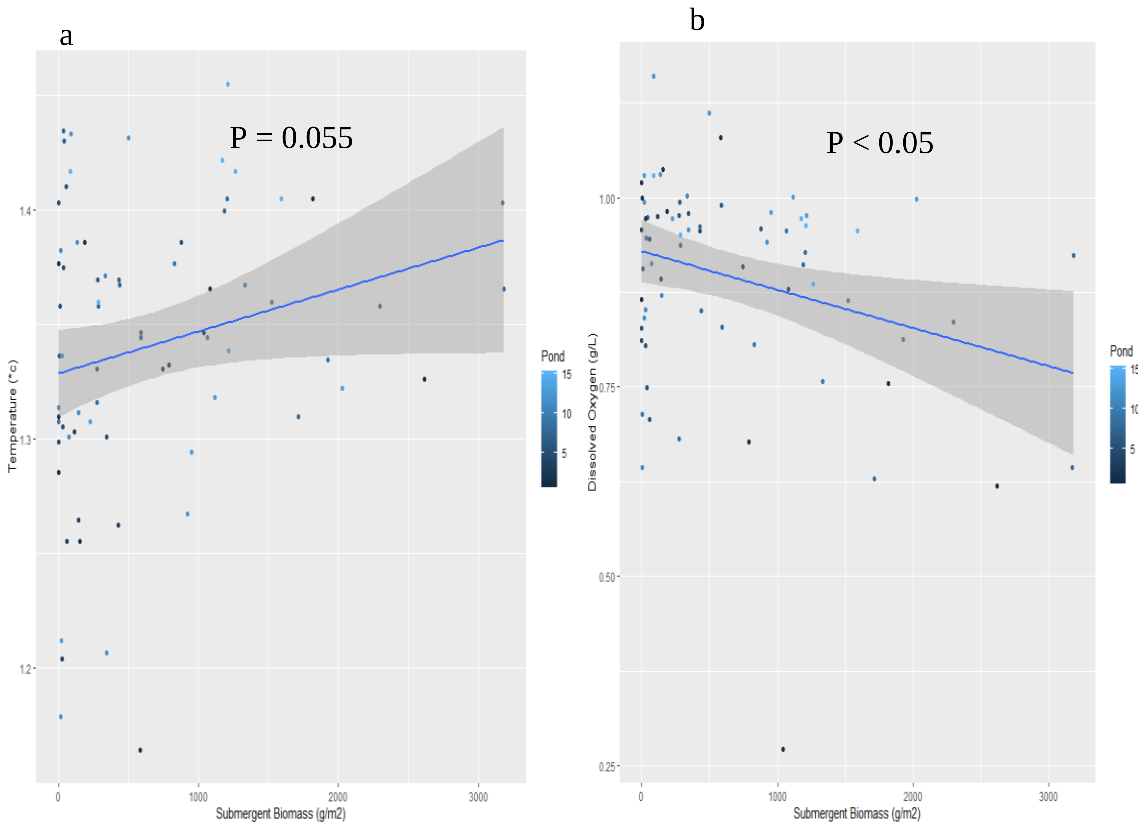

Figure B6. Pearson correlation analysis for submergent plant biomass and **outflow** temperature (a) and dissolved oxygen (b) including dates when submergent vegetation was collected in 2018 and 2019.

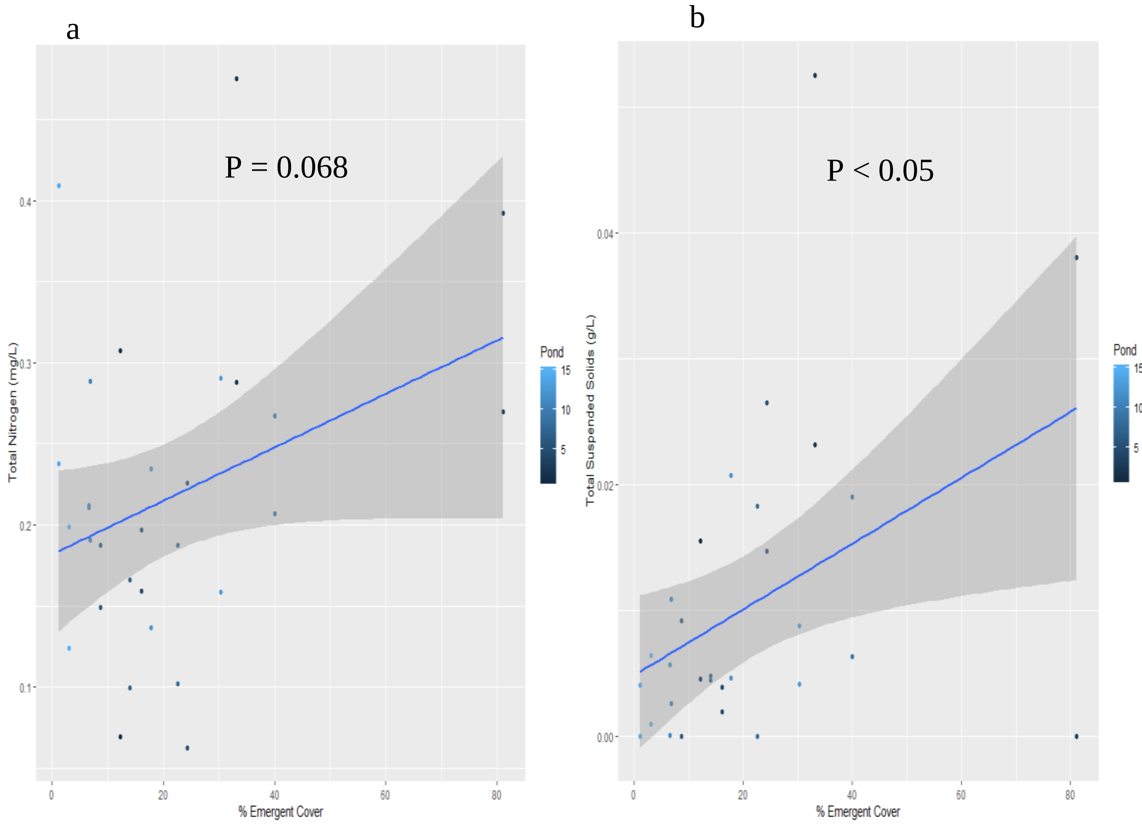

Figure B7. Pearson correlation analysis for emergent plant cover and **outflow** total nitrogen (a) and total suspended solids (b) including dates when emergent vegetation was sampled in August 2018 and 2019.

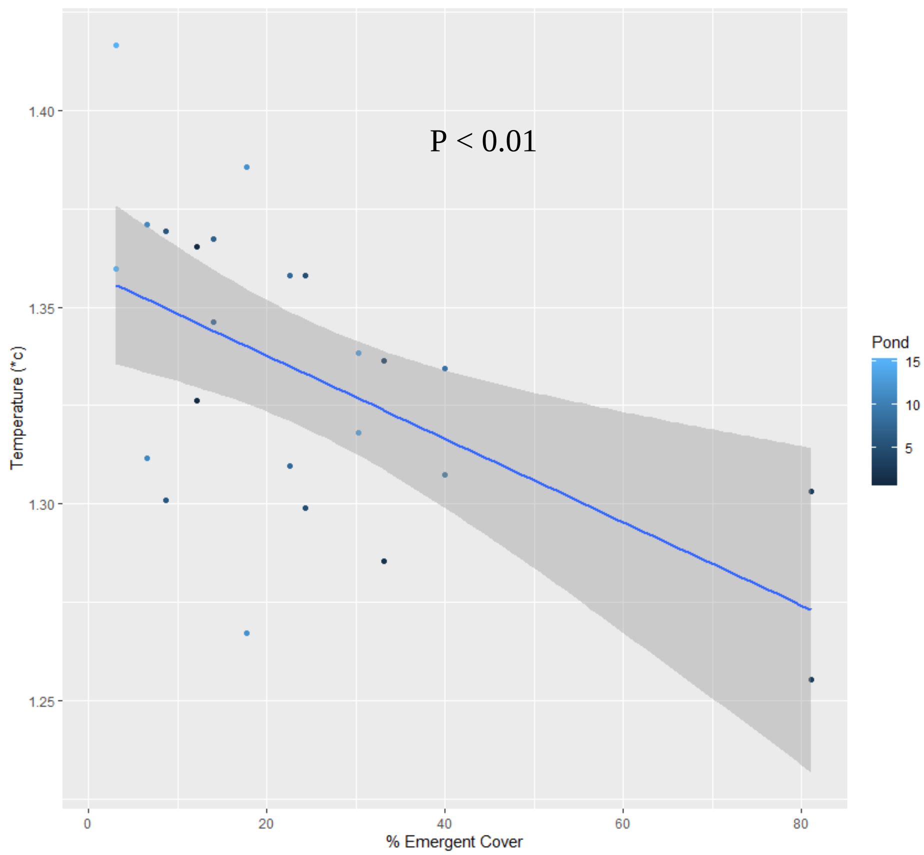

Figure B8. Pearson correlation analysis for emergent plant cover and **outflow** temperature including dates when emergent vegetation was sampled in August 2018 and 2019.

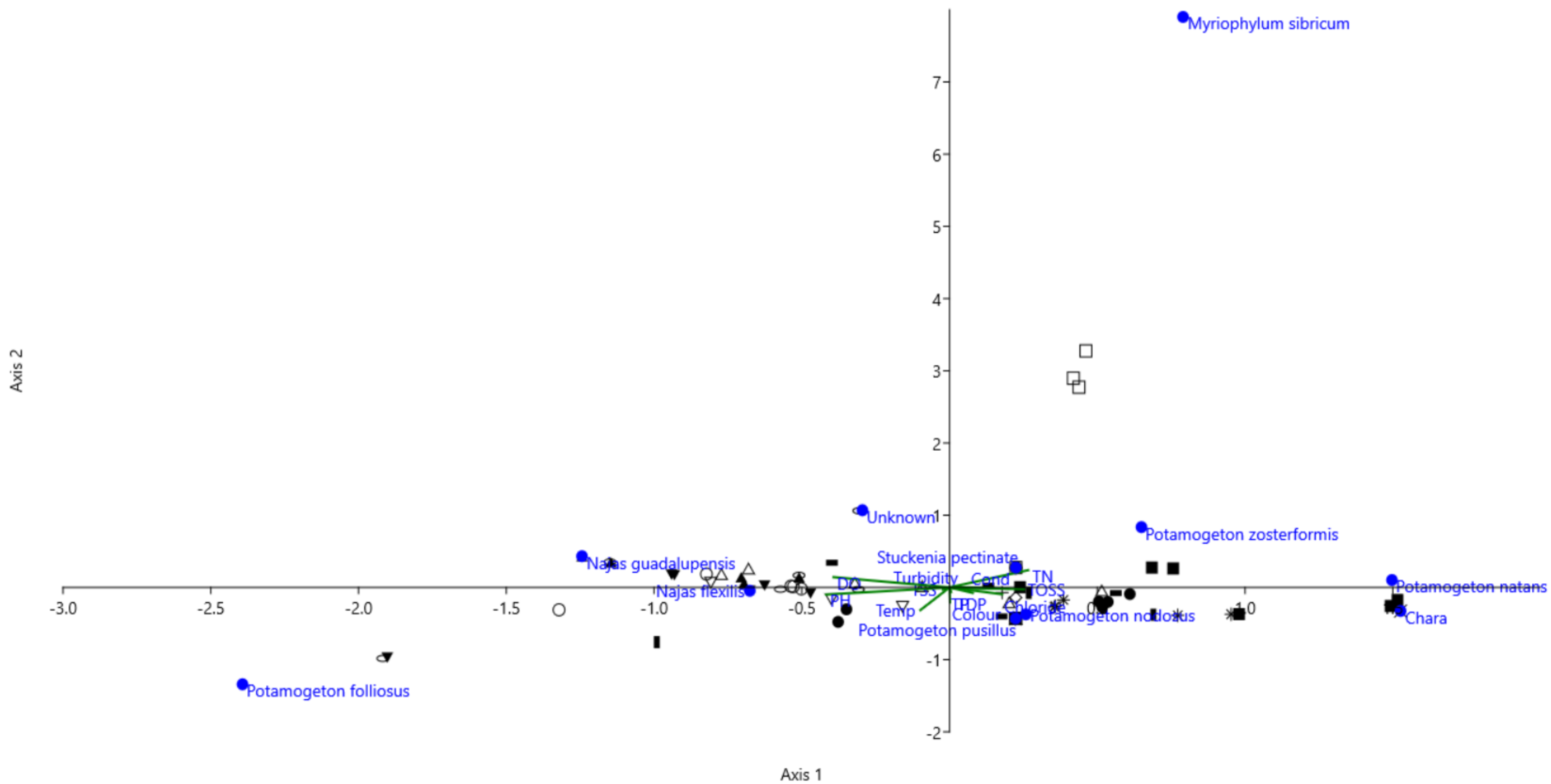

Figure B9. Canonical correspondence analysis for submergent plant biomass and **outflow** water quality parameters for 2018 and 2019. Axis 1 represents 33.08% of the variation and axis 2 represents 22.52% of the variation.

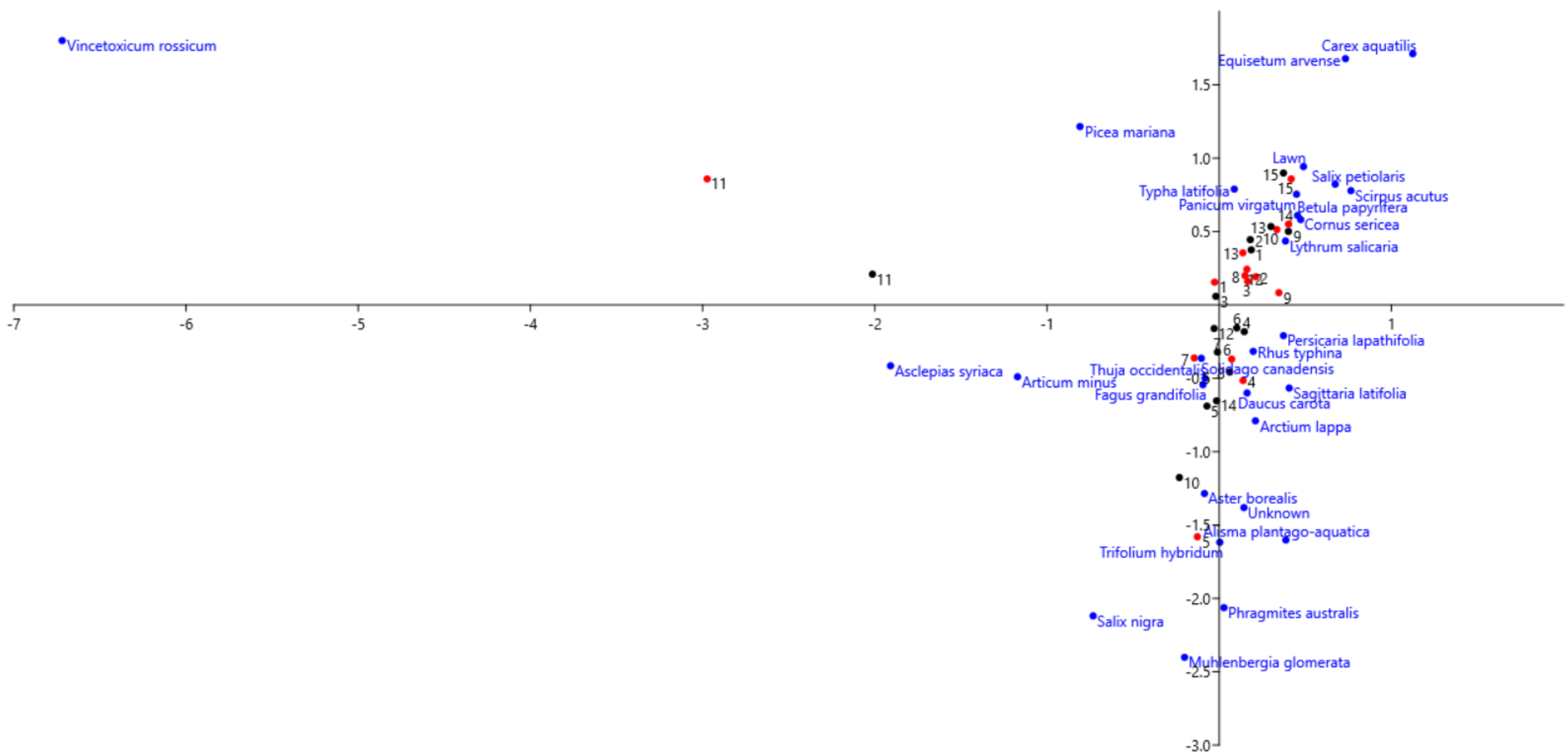

Figure B10. Correspondence analysis for all identified emergent plant species (aquatic and terrestrial) in 2018 (red) and 2019 (black) sorted by pond number. Axis 1 represents 15.14% of the total variation and axis 2 represents 13.59% of the total variation.

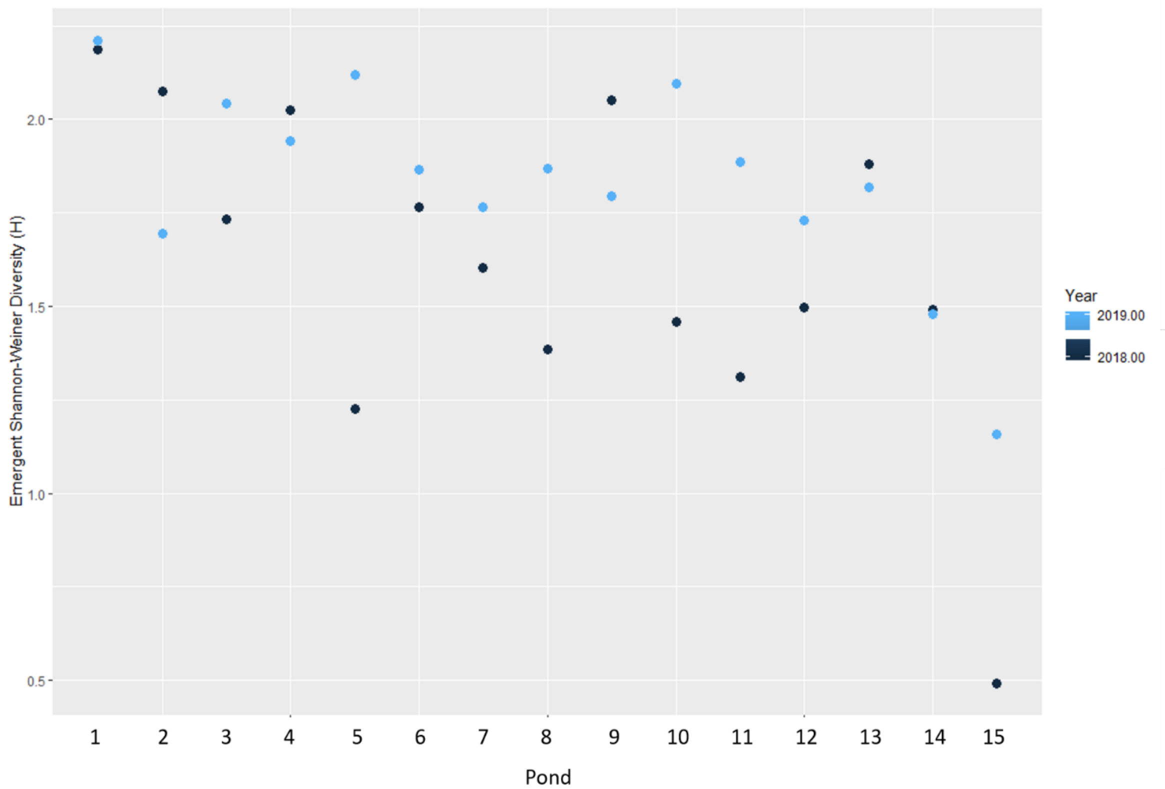

Figure B11. Emergent vegetation (aquatic and terrestrial) Shannon-Wiener Diversity index (H) for all 15 SMPs estimated from collection dates in August 2018 and 2019.

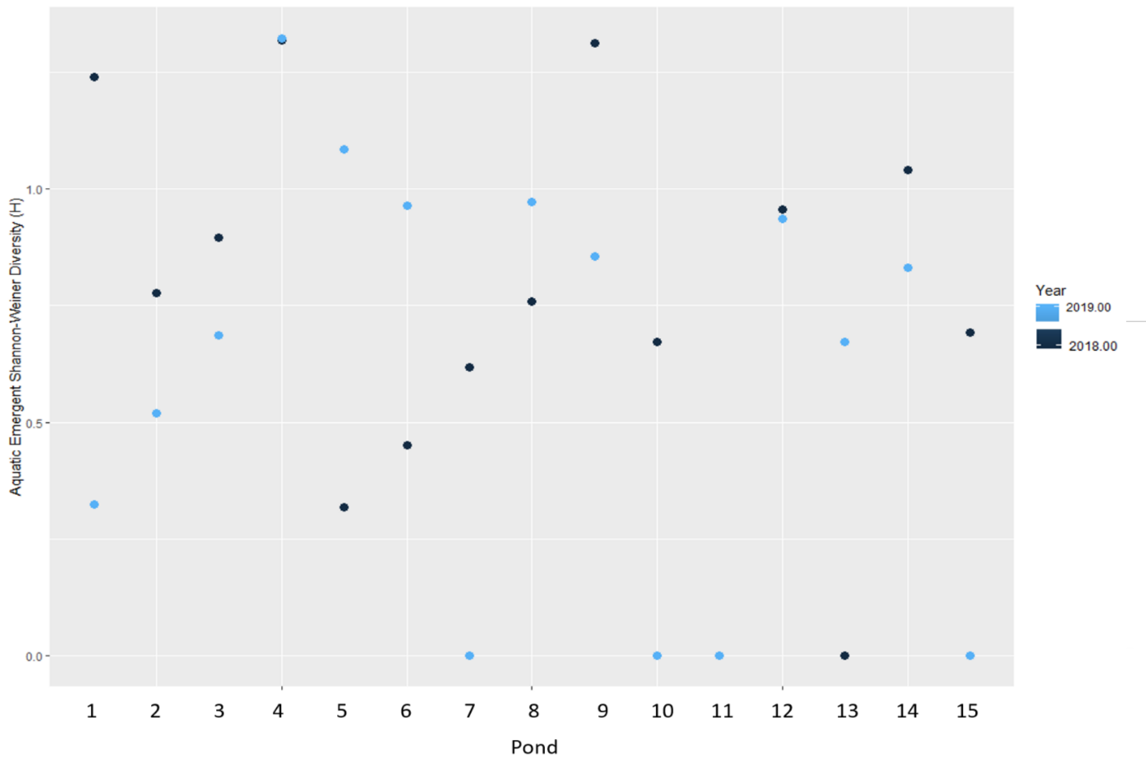

Figure B12. Emergent vegetation (aquatic only) Shannon-Wiener Diversity index (H) for all 15 SMPs estimated from collection dates in August 2018 and 2019.

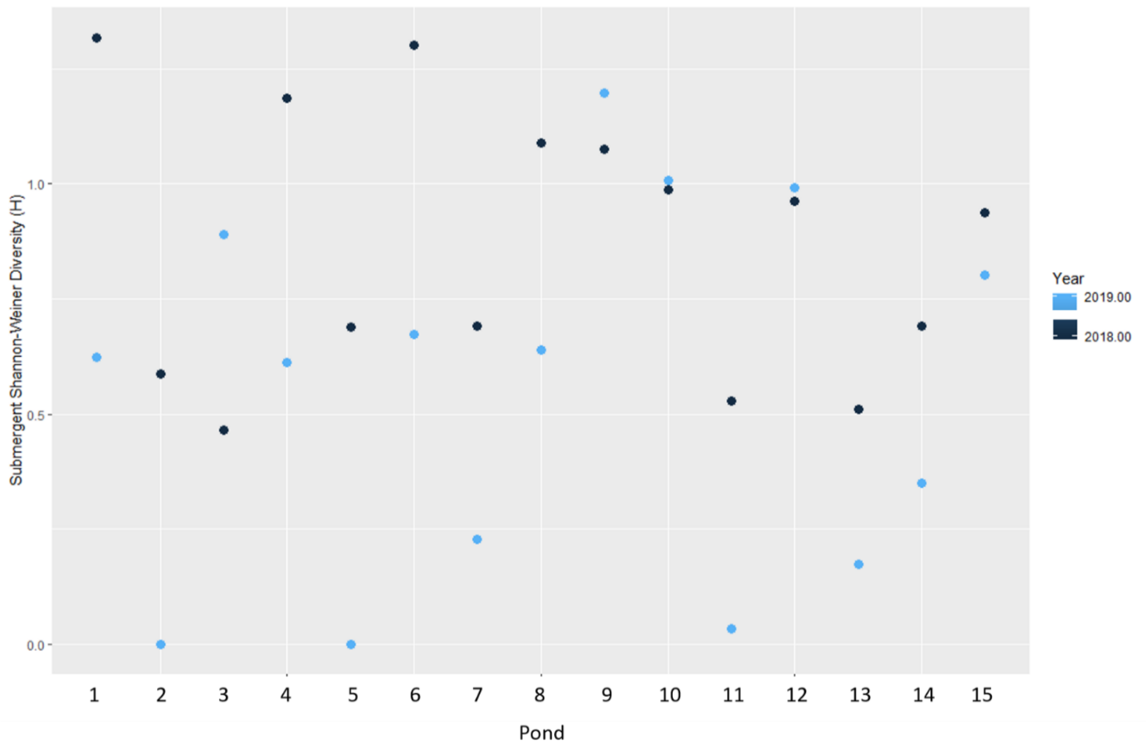

Figure B13. Submergent vegetation Shannon-Weiner Diversity index (H) for all 15 SMPs estimated from three collection dates in 2018 and 2019.

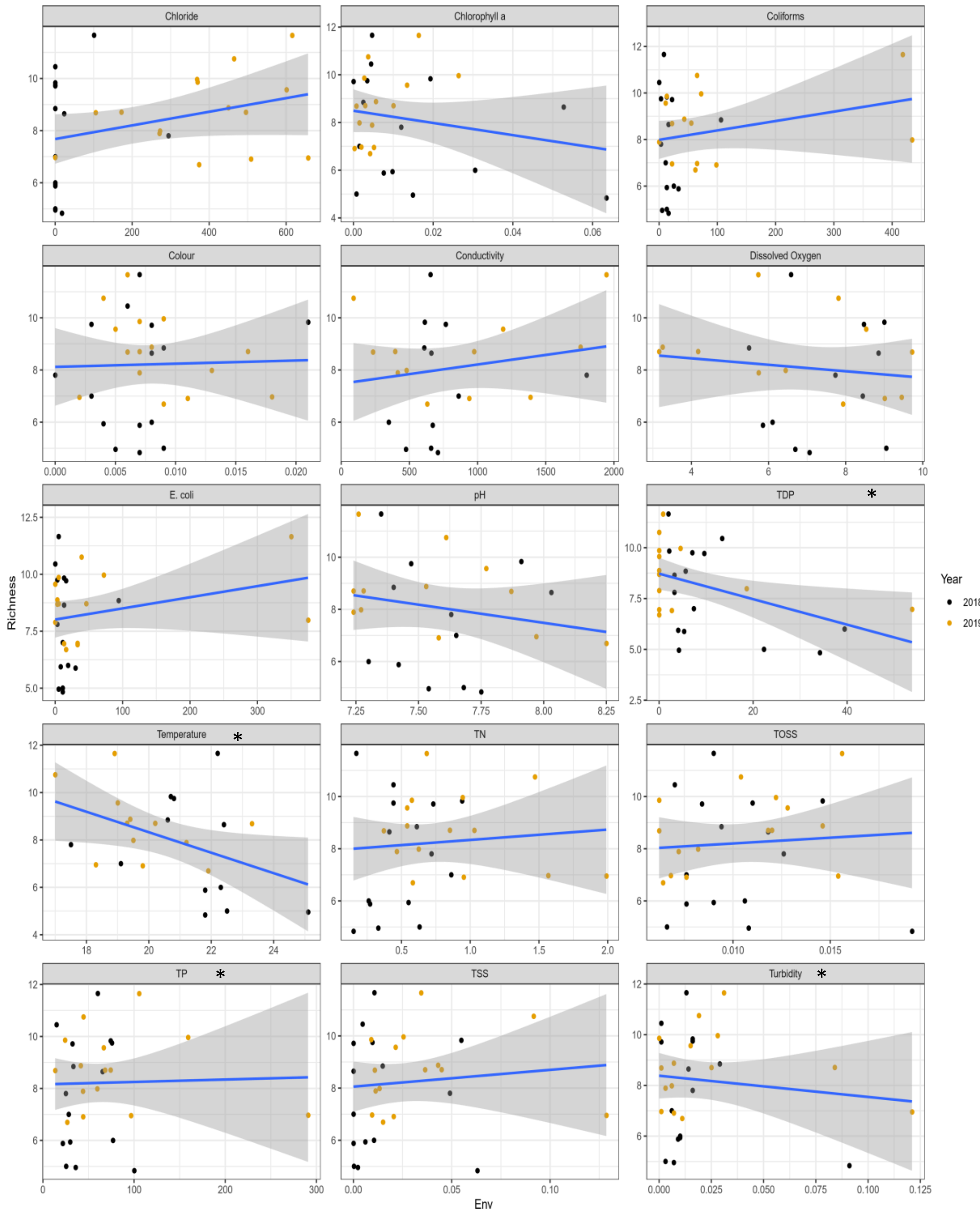

Figure B14. Species richness plots for emergent plant cover (aquatic and terrestrial) and **outflow** water quality parameters. Significant correlation analyses are denoted by $p < 0.05$ *, $p < 0.01$ **, $p < 0.001$ ***.