APPROACHING URBAN SUSTAINABILITY THROUGH RESTORATION ECOLOGY AND GREEN INFRASTRUCTURE

NATIVE PLANT PERFORMANCE ON A RIPARIAN BUFFER RESTORATION AND FEASIBILITY OF A CONSTRUCTED WETLAND AT AN URBAN PARKING LOT

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A Thesis Submitted to the School of Graduate Studies in

Partial Fulfillment of the Requirements for the Degree Master of Science

McMaster University

MASTER OF SCIENCE (2017)

Department of Biology, McMaster University

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TITLE: Native Plant Performance on a Riparian Buffer Restoration and Feasibility of a Constructed Wetland at an Urban Parking Lot

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NUMBER OF PAGES: xv, 126

ABSTRACT

Most cities are dominated by asphalt and concrete, which blocks the natural movement of rain water. Wetlands, riparian buffers, and roadsides are being lost or degraded in urban areas due to human development. Cities can be designed to benefit humans and nature by using techniques from green infrastructure and restoration ecology to improve urban sustainability. Parking lot M on McMaster University's west campus, constructed in 1968 on a former floodplain, directs the highly saline parking lot runoff into the adjacent Ancaster Creek. Natural groundwater sources along the surrounding hillslopes are directed into pipes under the parking lot and into the creek. A one-hectare riparian buffer restoration at lot M was used to assess the viability of depaving asphalt and establishing native plants through a vegetation study. Total native plant biomass was found to be similar to non-native plant biomass and was affected by road-salt salinity from the parking lot. Species richness per quadrat was higher for non-native plants, and greater for both non-native and native plants where less salt was present. Key hydrological fluxes were examined at the parking lot that could contribute to a proposed 0.6 hectare constructed wetland on the parking lot, known as “McMarsh.” Potential wetland water storage is in surplus year round, with an average storage of 265 mm/month. Successful restorations require maintenance following the establishment of native species. Management and maintenance of the restoration can help decrease non-native species. Engaging with the community through outreach and education on restoration projects is important for a successful restoration and increasing urban sustainability in cities.

ACKNOWLEDGEMENTS

I would like to thank my supervisor Susan Dudley for her guidance, wisdom, and encouragement during my masters. I am grateful for all the communication, field, and writing skills I have developed with her leadership. Thank you to Mike Waddington, who has helped with my research and served on my supervisory committee. Susan and Mike have both enabled me to explore research interests and outreach projects in areas that I am passionate about, which I do not take for granted.

A warm thank you to my field assistants over the past two field seasons; Sarah Thornley, Jessica Vu, Natalie Plociennik, and Noah Stegman. To all the Dudley Lab members, thank you for your friendship and assistance with navigating research and graduate school. I am grateful to Stefan Weber, who assisted me with plant identification numerous times in the field and lab.

To my wonderful parents and parents-in-law, thank you for your endless love and encouragement throughout my masters. A special thank you to Lori for her editing assistance in the final weeks of writing. I am thankful for the support from my friends near and far. I have the utmost gratitude to my wonderful husband Jesse. He has been my technical and emotional support throughout many years of school, and I am very lucky to have him.

Finally, I am grateful for research funding from the RBC Blue Water Initiative, made possible by a generous grant from the RBC Foundation.

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CHAPTER 1: INTRODUCTION

1.1 URBANIZATION

Urbanization has altered the natural landscape. By 2050, close to 70% of the world's population will live in urban areas (Giap et al., 2014). Historically, more people lived in rural areas, yet recent trends show people moving back to cities; Ontario’s urban population has risen from 14% to 86% from 1851 to 2011 (Statistics Canada, 2016). Larger city populations require more infrastructure. Asphalt, concrete, brick, and stone are impervious surfaces commonly used for roads, parking lots, and buildings (Littman, 2011). This imperviousness disrupts the hydrology and ecology of ecosystems (Arnold and Gibbons, 1996).

Natural infiltration of rain (storm) water is blocked by impervious surfaces (Han and Burian, 2009). Instead, water is routed into storm drains and as runoff, towards adjacent water bodies such as streams, rivers, creeks, or lakes (Bernot et al., 2011). Pre-development hydrology is disrupted through increased runoff volume and rate, along with limited evapotranspiration and interception (Palla and Gnecco, 2015). These hydrological changes increase the risk of urban flooding, which influences humans and ecosystems (Price and Vojinovic, 2008). Further, stormwater picks up salt, heavy metals, and other contaminants from paved surfaces that end up in adjacent streams and groundwater (Geronimo et al., 2014; Howard and Maier, 2005; Passeport et al. 2009).

Plant and animal habitat can be diminished or degraded where impervious surfaces exist, covering the soil and vegetation that were once present (Arnold and Gibbons, 1996;

Palla and Gnecco, 2015). Networks of roads and parking lots can fragment wildlife corridors, limiting migration and movement of animals (Gregory et al., 1991). Contaminants moving into water bodies have implications for aquatic flora and fauna habitat (Van Meter et al., 2011). Dark impervious surfaces, combined with loss of vegetation, can result in increased temperatures for ecosystems (Arnold and Gibbons, 1996).

1.2 DEGRADED ECOSYSTEMS

A wide variety of ecosystems experience degradation due to urbanization and land-use changes. Roadsides, riparian buffers, and wetlands will be introduced as they are the ecosystems of interest for this study.

1.2.1 ROADSIDE

As road networks expand to satisfy rising urbanization, roadsides have become an important ecosystem to restore (Forman and Alexander, 1998). Vegetation on roadsides can provide numerous ecosystem services such as erosion control and the filtering of contaminants in runoff (Haan et al., 2012; Karim and Malik, 2008). Roadside plant community success is based on several environmental gradients such as soil moisture content, bulk density, organic matter depth, and pH (Karim and Mallik, 2007). Roadside stresses for plants include compacted soil, contaminants, and competition (Haan et al., 2012). Two prominent areas of study on roadsides include the impact of de-icing salt and performance of native plants.

De-icing salt is used to melt ice and snow on roads (Mattson and Godfrey, 1994).

Road classification or amount of pavement is highly correlated with salinity found in nearby waterbodies, with higher salinity found where more road area exists (Daley et al., 2009; Mattson and Godfrey, 1994). Salt runs off roads and enters roadside habitat (Gibson and Carrington, 2008). Roadside plant germination can be inhibited by high concentrations of salt, as the saline soil causes water to leave plant cells due to the osmotic potential being higher in the plant (Gibson and Carrington, 2008). Phytoremediation, the degradation and stabilization of contaminants by plants, is possible on roadsides, which can delay or dilute runoff, but little is known about how to remove the salt (Gibson and Carrington, 2008). Halophytes are salt-tolerant plants that are able to maintain lower osmotic potential in their cells to enable water uptake from the soil (Flowers and Colmer, 2008). Salt is often compartmentalized into vacuoles so that the salt concentration in the cytoplasm remains low for halophytes (Flowers and Colmer, 2008).

Roads can act as seed corridors, encountering a variety of species through the vehicles and people that pass through (Richardson et al., 2007). Species composition on roadsides is also influenced by road type, age, maintenance, and traffic density (Rentch et al., 2005). Consequently, non-native and invasive plant species are successful on roadsides (Guido and Pillar, 2017). However, once established, native plants have the potential to minimize non-native plants (Quarles, 2003). Restoration work on roadsides needs to balance the priority of native plant establishment along with management of de-icing salt.

1.2.2 Riparian Buffer

A riparian zone (or buffer) is the area that connects a stream's aquatic environment to the adjacent terrestrial environment (Gregory et al., 1991; Renouf and Harding, 2015). Three zones generally characterize riparian buffers; Zone 1, closest to the stream, is mainly trees that shade the stream and stabilize the bank; Zone 2 is mainly shrubs that provide habitat for wildlife; and Zone 3 is mainly native grasses and forbs, which absorb water and contaminants (Gregory et al., 1991). The diverse topography and hydrology of a riparian buffer provide an ecological role that is beyond their area (Collins et al., 2013; Gonzales et al., 2017). Broadly, riparian buffers moderate climate, improve water quality, stabilize hydrological fluxes, and provide habitat (Soman, 2007; Mitch and Gosselink, 1993). In urbanized areas, riparian buffers are essential for processing contaminated stormwater that may enter streams from adjacent paved or developed areas (Gonzales et al., 2017).

Vegetation is a dynamic component of riparian buffers, reflective of the changing hydrological, topographical, and seasonal variability riparian buffers experience (Greggory et al., 1991). The dynamic nature of riparian vegetation provides several ecosystem services. Riparian vegetation improves water quality through increasing dissolved oxygen, while also decreasing turbidity and erosion (Collins et al., 2012; Dosskey et al., 2010). Habitat is provided for animals in terrestrial and aquatic ecosystems through riparian vegetation (Collins et al., 2013; Coelho et al., 2014). Phytoremediation is possible, whereby plants can remove (volatize), contain (store), or inactivate (transform) environmental pollutants such as heavy metals and nutrients (Bert et al., 2009; Dosskey et al., 2010).

1.2.3 WETLAND

Wetlands provide ecosystem services that benefit humans, flora, and fauna (Thoms, 2003; Zedler and Kercher, 2005). Persistently wet conditions create habitat and can enhance biodiversity for aquatic plant, animal, and other communities (Hansson et al., 2005; Zedler and Kercher, 2005). Riparian wetlands form an important ecotone (transition zone) between aquatic and terrestrial ecosystems (Thoms, 2003). In floodplain regions, wetlands increase the groundwater residence time, which influences the exchange of carbon and nutrients between the stream and floodplain and enhances ecosystem services (Thoms, 2003; Whigham et al., 2017; Yao et al., 2016). Moreover, surface runoff into streams and flooding into adjacent uplands is buffered by riparian wetlands (Johnston et al., 1997).

In the last thirty years, a lack of information on the economic and environmental value of wetlands has led to wetland loss and degradation (Turner, 1991). Human expansion through urbanization, industrialization, and agriculture has resulted in the pollution and conversion of many wetlands (Turner, 1991; Hansson et al., 2005). Direct impacts include drainage, filling, dams, levees, water diversions, and groundwater pumping (Zedler, 2000). Recently, recognizing the diverse ecosystem services wetlands provide has made wetland restoration a priority (Moreno-Mateos and Comin, 2010).

1.3 URBAN SUSTAINABILITY

Sustainability was first introduced through the concept of a population’s "ecological footprint" by Wackernagel and Rees (1996). Now, urban sustainability has become a field of its own for scientists, planners, engineers, and city residents. Urban sustainability is a conceptual framework for cities to balance environmental, cultural, societal, and economic demands (Shen et al., 2011). A sustainable city works to sustain its own existence while also supporting a long-term conservation of global and local ecosystems (McGranahan and Satterthwaite, 2003). The fields of Green Infrastructure and Restoration Ecology can be combined to retrofit existing city infrastructure and design new systems reflective of the native ecosystems once present.

1.4 GREEN INFRASTRUCTURE

Roadsides, riparian buffers, and wetlands need not be removed from the urban landscape with increasing infrastructure requirements. Instead, cities can be designed to benefit nature and humans. Green Infrastructure is an emerging field that seeks to design urban areas that allow for natural ecosystem function, a present-day toolkit for ecosystem restoration in cities (Benedict and McMahon, 2002). Originating in 1903, green infrastructure has evolved from the priority of linking parks to green spaces for the benefit of people and linking natural areas to benefit plants and animals (Benedict and McMahon, 2002). Concepts like “Design with Nature,” “Ecological Urbanism,” and “Landscape Urbanism” were coined in the mid to late 1900s and highlight the need to link city design to nature (Smith, 2015).

From managing stormwater and processing contaminants on-site to providing habitat and creating aesthetically and ecologically functional spaces, green infrastructure is a modern approach with which to address the world's increasing urbanization and aging infrastructure needs in a sustainable manner.

1.4.1 Grey to Green Infrastructure

Traditionally, stormwater has been managed through end-of-pipe solutions (Jarden et al., 2016). Known as "grey infrastructure," this management style uses engineered pipes, pumps, ditches, and retention ponds to capture stormwater (Jarden et al., 2016), resulting in ecological issues. First, where older sewer systems exist, sewage and storm water is often collected in a single pipe, called a “combined sewer.” This water is then sent to a waste-water treatment plant and is able to be treated with small amounts of rain, but with heavy rain the water can overflow into adjacent streams (Prosser et al., 2015). Where stormwater flows into a separate storm drain and does not go to the treatment plant, contamination and warming of the stormwater moving across paved surfaces can degrade aquatic habitat (Passeport et al. 2009).

In the past few decades, urban planners have begun to combat these environmental issues through the growing utilization of green infrastructure. Benedict and McMahon (2006) define green infrastructure as “an interconnected network of natural areas and other open spaces that conserves natural ecosystem values and functions and provides a wide array of benefits for people and wildlife.” In contrast to the standard grey infrastructure, green infrastructure works to "build with nature" and manage stormwater before it enters aquatic systems. Green infrastructure can range in shape, size, and scale from large riparian buffers, to rain gardens and rain barrels. Green infrastructure utilizes natural infiltration and treatment capabilities of soil and vegetation, which will be described further. The following sections outline the hydrological and ecological benefits of green infrastructure.

1.4.2 Hydrological Benefits of Green Infrastructure

Green infrastructure works to mimic the pre-development hydrology of a site through enabling storage, infiltration, and evapotranspiration processes to reduce peak stormflows and storm volume (Revitt et al., 2014 , Palla and Gnecco, 2015, Jarden et al., 2016). Increasing infiltration decreases stormwater quantities in cities, aquatic systems, and treatment plants (Revitt et al., 2014). Resulting stormwater reductions through green infrastructure systems is quantified by the maximum storm volume (rainfall amount) capable of being captured, for example, a bioretention cell may capture a 5 mm storm (Lewellyn et al., 2016). Urban areas can be retrofitted with suitable drainage, soil, and vegetation to perform these functions (Revitt et al., 2014, Prosser et al., 2015). The terms "Low Impact Development (LID)" and "Best Management Practice (BMP)" are synonymous with green infrastructure (Denich and Zaghal, 2014).

Few studies have been done on hydrological improvements of green infrastructure at the catchment scale, making it difficult to quantify how well the retrofits mimic pre-development hydrology (Jarden et al., 2016). However, it has been found that hydrologic performance increases linearly with increased impervious area reductions through green infrastructure retrofits (Palla and Gnecco, 2015).

On a smaller scale, green infrastructure design of biofiltration (bioswale) and bioretention systems have shown quantifiable site hydrologic benefits. For both systems, water moves off of adjacent paved surfaces into soil and plants and is either retained or filters through, often into a perforated pipe below (Denich et al., 2013). Runoff reduction will vary depending on the size of system and drainage area (Limouzin et al., 2011). Bioretention cells in North Carolina were found to reduce runoff by 14-18% (Passeport et al., 2009). Several catchment scale green infrastructure systems including rain gardens, bioretention, and barrels in Ohio resulted in a 40% reduction in total storm volumes, and 33% reduction in peak flow (Jarden et al., 2016). Bioretention cells at the University of Maryland allowed a 49-58% reduction in peak stormwater flow (Davis, 2008).

1.4.3 Ecological Benefits of Green Infrastructure

Green infrastructure improves ecosystem health through increasing habitat for native plant and animal species, critical in an urban area (Tzoulas et al, 2007; Gregory et al.,1991). Recreation areas in cities such as parks, gardens, or back alleys can become biodiversity hot spots providing beneficial ecological services (Benedict and McMahon, 2002). Urban gardens can increase local plant biodiversity and pollination (Borysiak et al., 2017). A riparian buffer, a less public form of green infrastructure, regulates creek temperatures and provides habitat for plants and animals (Gregory et al.,1991; Soman et al., 2007)

Phytoremediation can be a measurable ecological benefit of green infrastructure systems (Bert et al., 2009; Gregory et al., 1991; Sleegers, 2010). Since the 1970's, phytoremediation has been a cost-effective alternative to harsher remediation techniques (Smith, 2015). "Phytoremediation-by-design" works to plan site restoration through matching native plant species with local soil characteristics and contaminant degradation potential (Smith, 2015; Haan et al., 2012).

Successful phytoremediation has been demonstrated in many studies. Bioretention cells in North Carolina were able to reduce fertilizers (nitrogen and phosphorus) from runoff by 47-88% (Passeport et al., 2009). Cold climate bioretention cells in Ontario could remove heavy metals (copper, lead, and zinc) from runoff at a rate of 86-88% (Denich et al., 2013). Tree box filters in Korea were able to remove 95% of particulates and 70% of heavy metals from runoff (Geronimo et al., 2014). Finally, bioswales in a California parking lot reduced nutrient and pollutant loading by 95.4% (Xiao and McPherson, 2011).

1.4.4 Native Plant Use

Native North American plants are those that were present before European colonization and have co-evolved with local ecosystems (Ogden and Rejmanek, 2005). Native plants are favoured for restoration projects as they provide locally beneficial ecosystem services such as increasing habitat for native fauna and maintaining native plant diversity (Knops et al., 1995; Gibson and Carrington, 2008). Locally adapted native plants can also buffer areas against disturbance such as flood or drought (Mandel et al., 2016).

Using native plants in green infrastructure projects is a common recommendation in guidelines and articles. However, research and published journals seem to lack information on utilizing native plants, and instead focus on stormwater and contaminant reduction. Some areas of green infrastructure have favoured non-native plant selection, for example in green roof design; hardy non-native *Sedum* or *Phedimus* species have long been the plant choice (Mandel et al., 2016). However, researchers are beginning to explore the feasibility of utilizing native plants for green infrastructure projects where non-natives were previously used. As green infrastructure seeks to create systems in cities that provide local ecosystem services, the use and priority of native species is inevitable (Hostetler et al., 2011).

The field of restoration ecology, which seeks to recover and reclaim ecosystems that have been degraded or removed, widely recognizes benefits and importance of establishing native plant species (Giannini et al., 2017; Shackelford et al., 2013). Choosing a diverse selection of native species that are adapted to the local area allows the greatest potential for restoring or reclaiming the original ecosystem (Elmarsdottir et al., 2003; Giannini et al., 2017).

A review of 10 recent bioinfiltration and bioretention studies found that only half mentioned the use of native plants, while none focused on plant performance. Rather, these studies looked at stormwater contaminant and flow reduction. As such, collaborative research involving green infrastructure and restoration ecology could be highly beneficial in the creation of sustainable urban ecosystems. Policies and incentives that encourage green infrastructure implementation in combination with native plant could help merge these fields (Hostetler et al., 2011). In addition, interdisciplinary collaboration on research and projects has been shown to break down barriers and allow enhanced unity and collaboration (Lennon et al., 2006).

1.5 SOCIAL BENEFITS OF URBAN SUSTAINABILITY

One key way in which to advance urban sustainability is by involving people. As restoration often occurs where humans and nature co-exist (Standish et al, 2013), public engagement can become an imporant part of restoration work (Martin-Ortega et al., 2017). Further, no matter how much is known about how to restore a system, social and economic constraints may override restoration plans (Gonzales, 2017). Sleegers (2010) proposed a vision of green infrastructure where "landscapes of cleaning can become part of the urban infrastructure to create new neighbourhoods for research, education, working, and living remediation could become an artistic, aesthetically pleasing intervention with environmental value." Thus, working to identify and implement programs and opportunities for people to interact with nature and sustainable design will provide social and environmental benefits (Standish et al., 2013). Of importance is the ability of scientists and researchers to convey complex information about ecosystem restoration that is accessible to a wider public (Martin-Ortega et al., 2017). The more that communities can be involved in sustainability initiatives, the greater value and momentum these initiatives will have. Social benefits to communities through green infrastructure and ecosystem restorations in urban areas are numerous and diverse; livability, health, and aesthetic benefits will be highlighted.

Livability or quality of life and well-being (Giap et al., 2014), can be improved by green infrastructure in the places we live, work, and shop (Sleegers, 2010), through designs that reflect the needs of people and nature. There is an empirical link between environmental conditions in cities and walking or activity (Owen et al., 2004). Natural areas can reduce the impact of temperature, noise, and pollutants, improving the quality of life in the city (Saumel et al., 2016). Public health is connected to livability, which has been shown to potentially improve with increase in green infrastructure in cities (Smith, 2015; Tzoulas et al., 2007). Green space in cities may lead people to spend more time outdoors, which could lead to improved physical and mental health and potentially result in reduced public health costs to individuals and cities (Coutts and Hahn, 2015). Aesthetically, green infrastructure can become unique and meaningful landscapes for city residents (Sleegers, 2010) that may reflect cultural or personal values. A study looking at favourite places found that natural settings made up 50-60% of individuals' favourite places and were shown to regulate their feelings (Korpela and Hartig, 1996). The biophilia hypothesis, an inherent human need to affiliate with lifelike processes, is attainable in cities where green infrastructure is present (Kellert and Wilson, 1993). This innate biophilia could develop attitudes of ecological conservation among city residents (Kellert and Wilson, 1993).

1.6 THESIS OUTLINE

This thesis takes an interdisciplinary, cross-field approach to urban sustainability. Ecology and Hydrology disciplines are merged with the fields of green infrastructure and restoration ecology to assess the performance of one restoration and propose the feasibility of another.

Parking lot M on McMaster University's west campus was constructed on a floodplain in 1968 with outdated infrastructure and provides a living laboratory to implement and research green infrastructure projects. A one-hectare riparian buffer was created in 2014 (Figure 1.1). Bioretention cells and a constructed wetland in the south section of the parking lot are proposed as future projects (Figure 1.1).

The overall study objectives were to:

1. Assess performance of native plants along a newly constructed one-hectare meadow restoration.

2. Determine the feasibility of creating a constructed wetland on a section of the parking lot.

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1.8 FIGURES

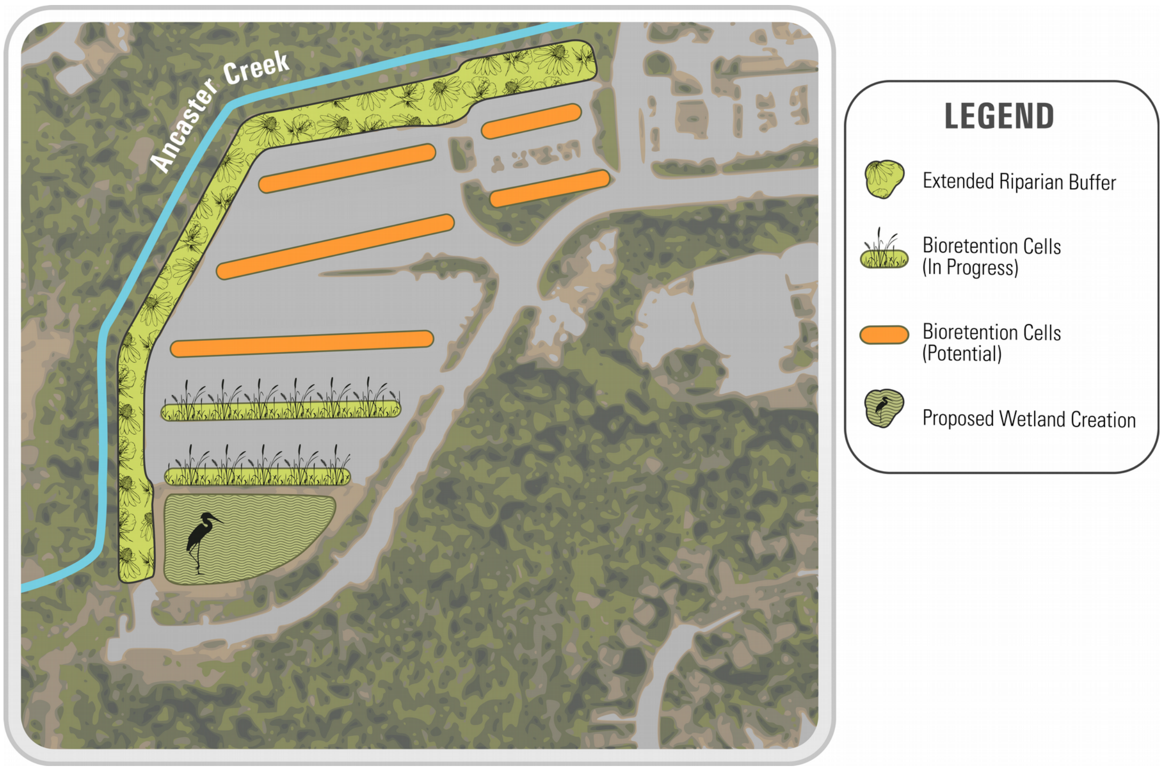

Figure 1.1: Green Infrastructure projects created, in progress, and proposed for parking lot M. Chapter two looks at native plant performance along the extended riparian buffer. Chapter three evaluates the hydrological feasibility of the proposed wetland.

CHAPTER 2: RIPARIAN PLANT PERFORMANCE ALONG A NEWLY CONSTRUCTED BUFFER ON A FORMER PARKING LOT

2.1 ABSTRACT

Establishing native species and re-establishing prior ecosystem function are the key goals in restoring ecosystems. Native plants increase plant diversity and provide habitat for native fauna. This study evaluates plant performance in relation to soil salinity on a two-year-old, one-hectare, newly restored extended riparian buffer for Ancaster Creek. The site is impacted by an adjacent parking lot where de-icing salt is applied in winter. A vegetation study was conducted along the riparian buffer harvesting all aboveground plants for 48 quadrats 0.5 m^2 in size. Salinity in the riparian buffer soil as a result of de-icing salts was highest on the edge of the riparian buffer due to stormwater runoff carrying salt, and snowpack melt. A 100 m section of the buffer was found to have the highest salinity values as it is adjacent to the most frequently used section of parking lot, which in turn is salted the most. A total of 1694 g of plant biomass was harvested, composed of 53% native plants and 47% non-native plants. Species richness was higher for non-native plants, with seeds available from the parking lot roadside or the original and degraded riparian buffer. Salt marsh sand spurrey (*Spergularia marina*), a coastal halophyte, volunteered on the riparian buffer, and was found in 79% of the edge quadrats. Sand spurrey had an increased biomass in response to buffer areas with higher salinity. Two problematic non-native plant species were found, phragmites (*Phragmites australis*) and queen anne's lace (*Daucus carota*). This study highlights some of the

common challenges associated with native plant restorations, and sheds light on the necessity for future management and maintenance.

2.2 INTRODUCTION

Establishing native plant species is an important component of a successful restoration (Elmarsdottir et al., 2003; Giannini et al., 2017; Shackelford et al, 2013; Trowbridge et al., 2017). Native North American plants are those that were present before European colonization (Ogden and Rejmanek, 2005). Native plants are favoured for restoration projects as they provide locally beneficial ecosystem services such as increasing habitat for native fauna and maintaining native plant diversity (Gibson and Carrington, 2008; Knops et al., 1995; Trowbridge, 2017). Potential adaptations to the local soil and climate allow native plants to persist in their ecosystems (Quarles, 2003). Riparian buffer and roadside restorations both allow the opportunity to prioritize native plants in planting plans; however, both provide challenges in establishing native plants.

Riparian buffers (zones), the land connecting streams (aquatic ecosystems) to adjacent land (terrestrial ecosystems), are critical for the exchange of water and chemicals between the two systems (Greggory et al., 1991; Dosskey et al., 2010). Where adjacent land to streams is impacted by human development, such as urbanization or agriculture, riparian buffers become important for conserving and protecting the aquatic ecosystem (Collins et al., 2011; Gonzales et al., 2017). Vegetation on riparian buffers moderate impacts from adjacent lands on the river or stream (Greggory et al., 1991). Riparian buffers also provide habitat for native plants and animals (Soman, 2007; Collins et al., 2013). When riparian buffers are removed or degraded, water quality and habitat benefits are lost or diminished (Dosskey et al., 2010; Gregory et al., 1991). Wider riparian buffers have been found to have increased plant taxonomic diversity than narrower buffers (Renouf and Harding, 2015).

Roads, highways, and parking lots often have adjacent areas of undeveloped land with potential for ecological restoration with native plants (Haan et al., 2012). Roadside vegetation can provide numerous ecosystem services such as erosion control, habitat, managing runoff, and filtering contaminants (Haan et al., 2012; Karim and Malik, 2008).

While studies have looked at restoring vegetation on roadsides (Gibson and Carrington, 2008; Karim and Mallik, 2008) or riparian buffers (Dosskey et al., 2010; Pattison et al., 2017; Renouf and Harding, 2015), few have focused specifically on the stresses to overall native plant establishment across a young restoration site. Native plants experience abiotic and biotic stresses on riparian buffer and roadside restoration sites.

Abiotic stresses to native plants on roadsides are contaminants that move in stormwater runoff through or over soil from the road to the roadside (Forman and Alexander, 1998). Key contaminants include de-icing salt, heavy metals, and fertilizers (Forman and Alexander, 1998). In urban areas, de-icing salt is a main concern, and roadside plants receive salt from the road through spray, blowing, or plowing of snow (Gibson and Carrington, 2008). Roadside salt can inhibit germination and growth of plants through inducing osmotic stress (Gibson and Carrington, 2008). Contaminants also alter soil quality through changes to soil pH, nutrient availability, and nutrient balance (Davison, 1971). Riparian buffer plants, depending on the location, may experience similar abiotic stresses.

Biotic stresses to native plants on roadsides and riparian buffers is the susceptibility to the introduction of non-native (alien) plant species (Forman and Alexander, 1998; Jodoin et al, 2007). Dynamic water levels of the stream next to a riparian buffer can act as a “conveyor belt” for non-native seeds to enter the buffer (Richardson et al., 2007). Vehicles act as efficient seed dispersal vectors for non-native plant seeds towards roadsides (Forman and Alexander, 1998; Spellerberg, 1998). Regular mowing on roadsides increases light and compacts soil, which further promotes non-native plant establishment (Berges et al., 2013). Non-native plants may compete with native plants, which could lead to a decrease in native plant biodiversity (Guido and Pillar, 2017). Some non-native plants are known to be invasive, meaning they are capable of forming large, dense, monospecific stands that exclude native plants for resources and space (Beerling and Perrins, 1993; Guido and Pillar, 2017). A lack of predators and adaptability in degraded environments allow invasive species to thrive (Guido and Pillar, 2017). Invasive species are one of the key causes of biodiversity loss in ecosystems (Guido and Pillar, 2017).

Outline of the Study

In this study, we measured native plant performance at a two-year-old, one-hectare (ha) meadow restoration that serves as an extended riparian buffer for Ancaster Creek. The riparian buffer is located between a 5.7 ha parking lot where de-icing salt is used, and an existing degraded 10 metre wide (0.5 ha) riparian buffer. With stresses to native plants on either side of the riparian buffer, this study provides the opportunity to research native plant performance in a highly stressed, urban environment. We characterized soil quality and plant performance along the 500 metre riparian buffer. Soil texture, fertility, and salinity were measured. Plant biomass and species richness were measured at 48 quadrats across the buffer. Soil salinity was used as an indicator of plant stress, and the resulting performance of species and communities (native vs. non-native) was assessed.

We asked:

1. How does road salting from the parking lot vary across the buffer?
2. How do native and non-native plants perform on the buffer?
    a) How did plants perform from the original seed mix?
    b) Which species (native and non-native) had the greatest biomass?
    c) How did species richness and biomass vary across position and distance?
3. How does salinity impact native and non-native plant growth on the buffer?
    a) Which species perform best under saline conditions?

2.3 METHODS

2.3.1 Study Site

Coldspring Valley Nature Sanctuary

A former nature sanctuary has become McMaster University's west campus (Figure 2.1, top), located in Hamilton, Ontario on the southwestern shore of Lake Ontario. West campus is positioned in the centre of an ecologically significant wildlife corridor, connecting the Niagara Escarpment in the south to Cootes Paradise Marsh and Lake Ontario in the north. The Niagara Escarpment is designated by UNESCO as a World

Biosphere Reserve and Cootes Paradise Marsh is designated by the City of Hamilton as an Environmentally Significant Area (ESA).

McMaster University's west campus, found at a topographically low area in relation to the main campus, was once a natural floodplain prior to the construction of parking lots. Four areas along the south and east hillslopes adjacent to the parking lot have groundwater springs that once fed wetlands and forests. The groundwater-fed wetland spilled into Ancaster Creek, running through the centre of the floodplain (Figure 2.1, bottom). From 1944-1963, the Royal Botanical Gardens (RBG) owned the land and the area was known as the "Coldspring Valley Nature Sanctuary" from 1958-1963. A series of trails were created in the sanctuary (Figure 2.2).

Ancaster Creek

Ancaster Creek is a rare cold water creek ecosystem, supporting numerous fish, reptile, amphibian, mammal, and plant species. Totalling 34 km in length, Ancaster Creek originates on the escarpment just south of Garner Road West (Figure 2.3, top). The creek runs down the escarpment as Sherman Falls, through McMaster Forest and the Dundas Valley, then under Osler street towards McMaster's west campus (Figure 2.3, top). Just north of Lot N and the baseball diamond on McMaster's west campus, the mouth of Ancaster Creek merges with Spencer Creek, which flows into Cootes Paradise Marsh and then into Lake Ontario (Figure 2.3, bottom).

Ancaster Creek's watershed is a 13.7 km^2 area with six catchments. Around 30% of the watershed remains undeveloped as forest, wetland, or meadow (Table 2.1). Wetlands historically comprised 8% of the watershed area, however, only 0.3% of wetlands remain

(HCA, 2008). Urbanization has largely impacted the watershed, with 40% of the area now residential, and 36% covered in impervious surfacing (Table 2.2, Figure 2.4).

Parking Lot M

In response to rising student populations in the 1960's, McMaster University purchased the Coldspring Valley Sanctuary from the RBG in 1963 (p.c. Randy Kay, RBG Minutes). The floodplain was filled in and paved in 1968 to create what is now parking lots M, N, O, and P (Figure 2.1, top). Parking lot M had 1662 parking spaces, which represented around 42% of the total parking on McMaster's campus (p.c. Randy Kay).

Ancaster Creek was diverted further west and channelized to create space for the parking lot (Figure 2.1, bottom). Around three metres of fill was placed on the wetland surface to create the parking lot. The initial design of lot M left a narrow 10 m wide riparian buffer next to Ancaster Creek. When lot M was created, groundwater springs from the surrounding hillslopes were routed into storm drains and under the parking lot to the creek in pipes. In addition, storm (rain) water on the parking lot surface moves into storm drains and into the creek as well. This storm water picks up contaminants on the surface, such as de-icing salt, oil, grease, or garbage, ending up in Ancaster Creek.

Several ecological issues were identified by the "Ancaster Creek Stewardship Action Plan", created in 2008 by The Hamilton Conservation Authority. The Lower Ancaster Creek catchment, where the creek runs adjacent to Lot M, was identified (Figure 2.3, bottom) by the Hamilton Conservation Authority (2008) as an area with "Stormwater Mismanagement." In addition, "Terrestrial Habitat Fragmentation & Lack of Riparian Buffers" was identified at 11 sites in all 6 catchments of the Ancaster Creek

Subwatershed.

A few years after the Stewardship Action Plan, Environment Canada (2013) released a document entitled "How much habitat is enough," in which a riparian buffer guideline required 30 metres of naturalized area next to a creek or waterbody adjacent to urban or developed lands. Additionally, in 2011, the peak demand for parking occupied only 69% of all parking spaces on campus (Urban Strategies, 2011). The combination of policy and underused parking provided an opportunity to sustainably retrofit parking lot M.

Riparian Buffer

McMaster University depaved one hectare of lot M on the west and north section adjacent to Ancaster Creek in the spring of 2014. The depaved section is 20 metres wide, and 500 metres long, which removed around 300 parking spaces (Figure 2.1, bottom). The area was dug to around 0.5 m depth and then filled with compacted fill from the McMaster stadium complex on campus. The depaved section, referred to going forward as the riparian buffer, was planted in the fall of 2014.

2.3.2 Seeding the Prairie

A seed mix was designed for the newly restored riparian buffer by St. Williams Nursery and Ecology Centre with native to Ontario meadow species. A meadow is a native (central) North American ecosystem, home to graminoids and forbs. Forbs are herbaceous, non-woody, and green plants that are not graminoids. Grasses, sedges, and rushes are all graminoids. The riparian buffer was seeded with 20 forb and 13 graminoid species, for a total of 33 species (Table 2.2). Three seed mixes were used for the dry and wet areas; and a turtle nest mix. Dry and wet mixes were used on corresponding areas on the buffer which tended to be dry or wet. Generally, areas close to the edge of the parking lot were wet, while moving toward the creek was dryer. The turtle mix was used on the perimeter of four sand turtle mounds, which were created in September 2014. The mounds are composed of coarse sand, 5 metres in diameter and 0.5 metres high. Tys Theymester from the Royal Botanical Gardens was consulted regarding turtle habitat design. The buffer was sown by hand, with seed mixtures divided into coarse and fine seed, ensuring an even distribution of seeds.

In September 2015, the riparian buffer section from 0 m to 100 m (Figure 2.4) was over seeded, due to the lack of plant growth in 2015. The lack of growth was potentially due to de-icing salt used frequently in the adjacent parking lot area. This area is generally the first to be used for parking at lot m, given the closest proximity to the bus shuttle, located adjacent to the Origin at 0 m (Figure 2.4). A seed mix was created by St. Williams Nursery and Ecology Centre of *Elymus virginicus* (virginia wildrye), *Elymus riparius* (riverbank wildrye), *Desmodium canadense* (showy ticktrefoil), and *Rudbekia hirta* (black-eyed susan). These species were chosen based on observing their success on the riparian buffer in summer 2015.

2.3.3 Establishing a Measurement Grid

A system to locate distance (buffer length location), and position (buffer width location) on the riparian buffer was established in June 2015, to reference measurements made for soil and plant studies. The origin of the grid is on the north-east section of the buffer, adjacent to the bus shuttle stop (Figure 2.4). This location was selected as the origin because the adjacent parking lot area has the greatest human and vehicle activity. The grid continues west and then south ending at 500 m (Figure 2.4). The cement curb was marked every 15 m beginning from the origin (0 m) with black nail polish indicating the distance in metres.

The position along the width of the buffer was also established in summer 2015. Three locations were chosen with reference to the parking lot curb. Edge, Middle, and Creek positions were 1 m, 10 m, and 13-19 m from the curb edge, respectively (Figure 2.5). The creek position was variable due to slight changes in buffer width. The creek positions for 10-115 m, 130-190 m, 205-340, and 355-475 m were 19 m, 17 m, 15 m, and 13 m respectively. On July 31, 2015, flags were added to each measurement location, that is, at 3 locations along the width (position), every 15 metres along the length (distance) of the buffer. This ensured ease, efficiency, and accuracy in locating measurement points, and collecting data.

2.3.4 Soil Monitoring

Wet Sensor

In July 2015, monitoring of the buffer soil began. A WET sensor was used to record measurements of Water content (%), Electrical Conductivity (mS/m), and Temperature (°C) of the buffer soil. A delta-T WET Sensor probe and chord was attached to an HH2 meter. The HH2 meter was set to mineral soil, which allows for a maximum of 60%

volumetric water content at saturation. Due to the soil being quite compacted and rocky, a small device made of 3 nails on a piece of wood in the same orientation as the probe, but slightly shorter and thinner to avoid tampering with the soil, was used to score holes into the soil prior to using the probe. This device also allowed checking whether the soil would be soft enough to insert the probe, which reduced wear on the probe. Next, the probe was inserted into the soil within a 20 cm diameter of the flag, and data was recorded in a notebook. If the soil was too dry or rocky at a given location to insert the probe, no measurement was taken. Some areas were always too rocky to insert the probe, while other areas only allowed for measurements when the soil was wetter and softer.

WET sensor monitoring began in July 2015 (July 2, 6, 16, and 21) and was carried out around every week until the end of August (August 8, 13, and 21). After seeing little changes in electrical conductivity for short time scales in 2015, monitoring frequency decreased to once a month in 2016 from April to November, on as similar of dates as possible (April 28, May 30, June 27, July 26, August 24, September 21, October 24, and November 24). WET measurements were preferable on days following rain events as the ground was softer, allowing for larger data sets, creating some variability for the monthly dates.

From July 2015 to September 2016, measurements were taken from 0 – 400 m. Upon realization that the buffer restoration was 500 m long an additional 5 locations were added from 400 – 475 m for WET measurements in October and November 2016.

Guelph Lab Samples

Soil samples were collected from the centre of the quadrat for the vegetation study

(Section 2.3.5) after all plants were removed. A soil core 2 cm diameter and 8 cm in depth was hammered into the ground using a sledge hammer. Samples were stored in the fridge in plastic bags.

In November 2016, samples were submitted to the Agriculture and Food Laboratory, run by the University of Guelph, located in Guelph, Ontario. Nine samples were tested for texture. Seven samples were tested for fertility, which measured organic matter (Walkley Black), soil pH , phosphorus (sodium bicarbonate), potassium, and magnesium (ammonium acetate). Several samples were tested for electrical conductivity, which was used to validate field WET sensor measurements. Texture and fertility values were used to assess buffer soil health.

2.3.5 VEGETATION STUDY

Field Work

From August 15-19 in 2016, a vegetation study was carried out along the riparian buffer from 0-400 m. Remaining plots at 430 m and 460 m were sampled on September 23, 2016. Vegetation was sampled at quadrats every 30 m distance along the buffer, starting at 10 metres (Yellow lines on Figure 2.4). Edge, middle and creek positions were sampled (Figure 2.5). Three positions at 16 distances were sampled along the buffer, for a total of 48 quadrats (16 distances x 3 positions = 48). Each sample location corresponded to a WET sensor measurement location.

A 50 cm^2 quadrat was used at each sample location, made out of PVC piping and copper edging for quick set up each day. The quadrat was laid out one metre to the right of the flag where soil monitoring was performed. This ensured that results could be compared to the soil data, while still ensuring the soil measurement area remained intact.

After laying out the quadrat, all plants found inside the quadrat were harvested using scissors, and separated by species. Only the aboveground portion of the plant was extracted. If the species could be identified in the field, the plant was put in a brown paper bag, and a 3-letter code was created for the species. However, if the species could not be identified, key characteristics were noted, and a sample of the plant was pressed for further identification in the lab.

Lab Work

Plants were dried for 24 hours in a drying oven at 100 °C. Following drying, plants were weighed for biomass measurements. Unknown species were identified using a combination of guidebooks, internet, and databases. Plant identification sessions were organized with a plant specialist for the hard to identify species. Species were then classified as being native or non-native using similar resources described above. For this study, native plants are defined as originating from eastern North America, while non-native plants are those that have been introduced following European colonization.

Data Analysis

Electrical conductivity data obtained from WET sensor measurements was plotted over the distance (0-500 m) of the buffer. Similar time segments were averaged for comparison; July-August in 2015 and 2016, April-June 2016, and September-November 2016. Yearly average electrical conductivity values for 2015 and 2016 were also plotted over the buffer distance.

Vegetation study plants were organized by origin (native and non-native) and lifeform (forb or graminoid). Summary tables of biomass and species richness of all species found in the vegetation study were prepared. Biomass and Species Richness was plotted by distance and position for plants by origin and lifeform. An Analysis of Covariance (ANCOVA) was performed on the potential effects of plant origin, lifeform, position, and distance on both plant biomass and species richness. Tukey's test was performed on biomass and species richness ANCOVA's.

Soil and plant data was combined in plots of biomass vs. electrical conductivity for origin and lifeform. An Analysis of Covariance (ANCOVA) was performed on the potential effects of plant origin, lifeform, and electrical conductivity on plant biomass. Finally, biomass of sand spurrey vs. electrical conductivity was plotted as this plant showed an increase in biomass with increasing electrical conductivity.

2.4 RESULTS

2.4.1 Riparian Buffer Soil Salinity

Pore water electrical conductivity is the conductance of ions in the water between soil grains. As de-icing salt or sodium chloride is the main source of ions (Na^+ and Cl^- ions) entering the buffer through snow movement and runoff, we will refer to salinity rather than conductivity throughout.

Distinct trends in salinity were seen along the riparian buffer distance, and across position (Figure 2.6). The edge had the highest salinity values, ranging from 81 mS/m in September-November 2016 at the 295 m distance, to 5032 mS/m in July-August 2016 at the 205 m distance (Figure 2.6). Edge salinity values above 1000 mS/m were seen consistently in July and August of 2015 and 2016 for 40, 70, 85, 100, 115, and 340 m (Figure 2.6). Salinity values were generally higher on the edge from 0-235 m, with an increase further along the distance only at 340 m (Figure 2.6). Spring (April-June 2016) had slightly lower salinity than summer for the edge (Figure 2.6). Fall (September-November 2016) had the lowest edge salinity.

The middle and creek positions had dryer soil, often with more gravel, making it harder to take measurements with the WET sensor (Figure 2.6). Thus, data for the middle and creek is sparse and only really complete for September to November 2016, when the soil was cool and damp (Figure 2.6). Middle salinity was much lower than the edge, ranging from 66 mS/m in September-November 2016 at 310 m, to 2435 mS/m in July/August 2016 at 100 m, an uncharacteristic reading in comparison to the rest of the data (Figure 2.6). The middle position at 100 m distance is actually closer to the parking lot edge than other middle points due to the shape of the buffer, which is extended (increased width) in that section. Salinity values are higher from 0-100m, with an increase at 340 m, similar to the edge (Figure 2.6). There does not appear to be a seasonal trend at the middle position (Figure 2.6).

Creek salinity was much lower than the edge, but similar to the middle, ranging from 67 mS/m in September-November 2016 at 280 m, to 1570 mS/m in July/August 2016 at 100 (Figure 2.6). Even fewer readings were possible at the creek with dryer soil than the middle due to being furthest from stormwater runoff from the parking lot (Figure 2.6). Similar again to the middle, salinity values are higher from 0-100m, and then level out to around or under 100 mS/m from 100-500 m (Figure 2.6). There does not appear to be a seasonal trend at the creek position (Figure 2.6).

On a yearly basis, salinity is greatest at the edge, followed by the middle, and then creek positions (Figure 2.7). From 0-100 m distance, the greatest salinity is seen for the edge, middle, and creek, with a few peaks further on for the edge at 190m and 205 m (Figure 2.7). The edge and middle both have a peak in salinity at 340 m (Figure 2.7).

Between years, 2015 and 2016 edge salinity surpass the other at various distances, with no real trend evident (Figure 2.7). Similarly for the middle and creek, salinity between 2015 and 2016 does not have a specific trend (Figure 2.7).

2.4.2 Native and Non-Native Plant Performance

A total of 1694 g of dried plant biomass was harvested from the vegetation study on the riparian buffer. There were 48 half-metre-squared quadrats sampled at 16 distances, every 30 m along the buffer distance (Figure 2.4) and three positions across the buffer (Figure 2.5). The total plant biomass in the vegetation study was composed of 53% native plants and 47% non-native plants (Figure 2.8). Forbs comprised 73% of the overall biomass, with 58% (705 g) of forbs being native species and 42% (517 g) non-native species (Figure 2.8). Graminoids comprised 27% of the biomass, with 41% (194 g) native species and 59% (278 g) non-native species (Figure 2.8).

A total of 705 g of biomass (Table 2.4) came from native forbs in the vegetation study that were either planted (6 species, 37 g biomass) or volunteered (4 species, 668 g biomass). Non-native forbs found in the vegetation study were not planted, but came from the seed bank or blew in for a total of 28 species and 517 g of biomass (Table 2.4). Native graminoids from the vegetation study were planted (2 species, 194 g biomass), and no volunteers were found (Table 2.4). Non-native graminoids from the vegetation study were not planted, but potentially came from the seed bank or blew in for a total of 9 species and 795 g of biomass (Table 2.4).

Original Seed Mix

Of the original 33 native species planted on the riparian buffer, 12 forb species (out of 20 planted) and 8 graminoid species (out of 13 planted) were found in the vegetation study (6 forbs, 2 graminoids), or observed (6 forbs, 6 graminoids) on the riparian buffer (Table 2.4). Native forbs that were planted and found in the vegetation study (6 species) totalled 37 g of biomass, and 3% of the overall study biomass (Table 2.4). Native graminoids that were planted and found in the vegetation study (2 species) totalled 194 g of biomass, and 11% of the overall study biomass (Table 2.4).

The greatest amount of biomass for the riparian buffer vegetation study came from volunteer native forbs at 668 g of biomass and non-native plants at 795 g of biomass, which made up 39% and 47% of the vegetation study biomass, respectively (Table 2.4).

Successful Species

Of the native forbs (705 g) that were planted and found in the vegetation study (Table 2.5), the highest biomass was found for common evening primrose (22 g, biomass rounded to nearest gram going forward), black-eyed susan (5 g), and lance leaf aster (5 g). The remaining three species were between 0-3 g of biomass (Table 2.5). Volunteer native forbs (Table 2.5) had two main contributors, salt marsh sand spurrey (416 g, 25% of overall study biomass) and goldenrod (211 g, 12% of overall study biomass). Seaside knotweed was one other volunteer native forb with high biomass of 37.85 g (Table 2.5).

Of the 28 species of non-native forbs (Table 2.6) found in the vegetation study, 9 species had biomass greater than 5 g. The 9 species were queen anne's lace (148 g), birdfoot trefoil (100 g), black medic (74 g), canada thistle (37 g), chamomile (37 g), common plantain (29 g), cow vetch (29 g), butter and eggs (20 g), and lance-leaved plantain (14 g). Of the remaining 19 species, 10 species were between 1-5 g and 9 species were between 0-1 g of biomass (Table 2.6).

Two native graminoid species were planted and found in the vegetation study (Table 2.7); virginia or riverbank wildrye was the dominant species with 187 g of biomass and poverty rush had 7 g of biomass.

Of the 9 species of non-native graminoids found in the vegetation study (Table 2.8), 5 species had biomass greater than 10 g. The 5 species (Table 2.8) are red fescue (163 g), barnyard grass (53 g), witchgrass (16 g), crabgrass (16 g), and perennial rye (13 g). The remaining 4 species were between 2-7 g of biomass (Table 2.8).

Species Richness and Biomass Along the Riparian Buffer

Native plant biomass per quadrat (Figure 2.9, Table 2.9) decreased with increasing distance along the riparian buffer (From 0-500 m) at the edge and middle positions, but increased at the creek position. Non-native plant biomass per quadrat (Figure 2.9) increased with increasing distance along the riparian buffer at the edge and middle positions, and also slightly increased at the creek position. Forb biomass (1222 g) was greater than graminoid biomass (472 g) in the vegetation study ($p<0.0001$). The edge biomass per quadrat for the vegetation study was greater than the middle biomass per quadrat ($p=0.002$). The p values in this section all come from Analysis of Covariance (ANCOVA), with output values found in Tables 2.9, 2.10, and 2.11.

Native forbs had greater biomass (Figure 2.9) than native graminoids ($p=0.0005$), and similarly non-native forbs had greater biomass than non-native graminoids ($p=0.005$). Native forb biomass per quadrat at the edge position was greater than forb and graminoid biomass per quadrat at all positions except native forb and graminoid biomass per quadrat at the creek position ($p<0.05$).

Species Richness (Figure 2.10, Table 2.10), the number of species per quadrat found in the vegetation study, increased with increasing distance along the riparian buffer (From 0-500 m) at all three positions ($p<0.0001$). Non-native plant species richness (Figure 2.10, Table 2.10) was greater than native plant species richness ($p<0.0001$). Forb species richness (10 native, 28 non-native) was greater than graminoid species richness (2 native, 9 non-native) ($p<0.0001$). The edge species richness per quadrat for the vegetation study was greater than the middle species richness per quadrat ($p=0.0152$).

Non-native forb species richness per quadrat (Figure 2.10) was greater than native forb and graminoid ($p<0.0001$), and non-native graminoid species richness per quadrat ($p<0.0001$). Native forb species richness per quadrat was greater than native graminoids ($p<0.0001$). Non-native forb species richness per quadrat (Figure 2.10) at the creek position was greater than all native and non-native graminoid and forb species richness per quadrat at all positions, except for edge non-native forbs ($p<0.05$). Non-native forb species richness per quadrat (Figure 2.10) at the edge position was greater than all native and non-native graminoid and forb species richess per quadratat all positions, except for non-native edge graminoid, non-native middle forb, and non-native creek forb species richness per quadrat ($p<0.05$). Non-native graminoid species richness per quadrat (Figure 2.10) at the edge position was greater than native graminoid species richness per quadrat at the edge, middle, and creek ($p<0.05$). Similarly, non-native forb species richness per quadrat (Figure 2.10) at the middle position was greater than native graminoid species richness per quadrat at all three positions ($p<0.05$). Finally, native forb species richness per quadrat (Figure 2.10) at the edge position was greater than edge and middle native graminoid species richness per quadrat ($p<0.05$).

2.4.3 Salinity Impact On Plant Growth

Plant biomass per quadrat in the vegetation study (Figure 2.11, Table 2.11) changed based on varying salinity ($p=0.0331$). Native plant biomass per quadrat in the vegetation study increased with increasing salinity (Figure 2.11, Table 2.11) , while non-native plant biomass per quadrat decreased with increasing salinity ($p=0.0055$). Native and non-native forbs also exhibited this trend, and a spread of measurements was found across most conductivities (Figure 2.11).

Of the 10 native forb species found in the riparian buffer vegetation study (Table 2.5), only salt marsh sand spurrey (*Spergularia marina*) responded to increasing salinity with increasing biomass per quadrat (Figure 2.12). Salt marsh sand spurrey was found in 19% of the vegetation study plots, with 416 g of biomass (Table 2.5), representing 60% of the native forb biomass, and 25% of the overall vegetation study biomass.

2.5 DISCUSSION

The riparian buffer vegetation study found an even biomass of native and non-native plants, but a high species richness of non-native plants. Salinity and introduction of non-native plants were the greatest stresses on the restoration. Road salting practices determined areas of high and low salinity, with greater salinity on the edge of the buffer. Most plants did poorly in high salinity areas, except one volunteer forb, sand spurrey (*Spergularia marina*). Non-native plants were able to enter the buffer through the parking lot on one side, or Ancaster Creek and the original riparian buffer on the other side. Two non-native species present in the vegetation study pose an additional stress on the restoration as they can out-compete native plants for resources.

2.5.1 Salt Movement and Buffer Design determine Buffer Salinity

As parking lot M is large and requires a shuttle bus or ten minute walk to get to campus, the majority of people tend to park in spots closest to the shuttle, or walking access. With the shuttle location close to the 0 m distance on the buffer (Figure 2.4), the most frequently used parking area is directly south of 0-100 m along the riparian buffer. This same parking area is the main pedestrian corridor for people parking in that section, or parking further west. As such, in winter, de-icing salt is likely applied to this parking section in the greatest quantity. Snow is likely shovelled towards medians in the winter, but also onto the newly extended riparian buffer. The combination of snow pack melt and the movement of salt across the parking lot with rain likely allows salt to enter the riparian buffer and subsequently the soil and plants (Forman and Alexander, 1998; Gibson and Carrington, 2008; Van Meter et al., 2011).

The edge salinity is greater than the middle and creek, which likely reflects the movement of salt in stormwater (rainfall) runoff to the edge most readily (Cunningham et al., 2008; Gibson and Carrington, 2008). Lower curbs along the riparian buffer edge allow stormwater to move into the riparian buffer at most distances. The accumulated snowpack melting on the edge also likely contributes to the high edge salinity. From the 0-100 m distance, increased salinity is seen for the edge, middle and creek positions, which is potentially a result of the increased de-icing salt application along the most used section of parking. The 100 m distance salinity is consistently high across the 3 positions, which is likely due to the successful runoff swale at that location, which drains well from the parking lot edge, and is generally a wetter area. The highest salinity seen at the edge at 205 m reflects another swale location, where water is directed into large rocks (“rip rap”) at a topographic low on the riparian buffer edge. After large rain events, water is often seen ponded around 205 m, which could allow salt to concentrate along the edge. At the 50 m distance, the drop in salinity is likely due to the higher curb on the edge at that distance, blocking most stormwater runoff that could move salt from the parking lot.

Seasonally, salinity in summer (July-August) tends to be highest, spring (April-June) is moderate, and fall (September-November) is lowest. Increased moisture at the edge position due to stormwater runoff allowed for consistent measurements and defined trends in comparison to middle or creek positions, which were often dryer than the edge. The increase in summer salinity along the edge is likely a response to the spring snow melt and stormwater movement from the parking lot bringing in salt. Decreased salinity in the fall could be a response to the dilution of the soil pore water from rainfall events. Of concern is the potential for salt to leach belowground and accumulate year to year (Scott, 1980), which could have detrimental impacts on Ancaster Creek's aquatic ecosystem.

2.5.2 Plant Performance on the Riparian Buffer

The even mixture of 53% native and 47% non-native plant biomass found in the vegetation study is expected, as high contributions of non-natives to restorations is typical in comparison to untouched ecosystems (Cunningham et al., 2008; Gibson and Carrington, 2008; Hillhouse, 2011). Non-native plant species are able to move into restoration sites through wind, water, or animal dispersal from nearby areas, or be a part of the site seed bank (Richardson et al., 2007; Forman and Alexander, 1998). At a parking lot, humans and vehicles act as additional vectors of dispersal for non-native plants (Spellerberg, 1998).

From the original seed mix of 33 native species, 20 species (60%) were found in the vegetation study or observed on the riparian buffer. Restorations typically do not see full recruitment from seed mixes, due to limitations involving amount of seeding, site conditions, climate, or seed quality (Barak et al., 2017; Hillhouse and Zedler, 2011; Pywell et al., 2003). The lot M riparian buffer was initially seeded in fall 2014, with only one more seeding of a few species on the section from 0 – 100 m distance on the buffer. The riparian buffer restoration is young, and the vegetation study was performed in the second growing season, which is an early successional stage, where some species may be late to germinate (Berges et al., 2013). However, roadside plant communities can be highly disturbed and remain in early successional stages (Rentch et al., 2005). Additional constraints for germination on the riparian buffer was the low rainfall in spring 2015 (March-May), and summer 2016 (May-September) (See Chapter 3, Figure 3.4).

Soil conditions on the riparian buffer were not favourable for seedling development (Table 2.3). Soil texture (Table 2.3) ranged from "loam" to "gravel loam," with extremely low organic matter (0.5-3.6%), creating a fairly inhospitable environment for seedling germination (Hillhouse and Zedler, 2011). As meadow plants often prefer sandy soils, the high silt and clay in the buffer soil, combined with high salinity, would have made germination difficult for the native species (Davison, 1971; Haan et al, 2012). Further, belowground components of meadows are important, as the aboveground is frequently removed from grazers or fire (Johnson and Matchett, 2001).

Native Plant Performance

While native forbs that were planted comprised only 3% of the overall study biomass, volunteer native forbs comprised 668 g or 39% of the study biomass. Volunteer native plants benefit restorations by increasing native plant diversity and could be able to locally adapt (Lascoux et al., 2016) to disturbed environments. For example, a native meadow plant species may exhibit local adaptation to salinity. Salt marsh sand spurrey (*Spergularia marina*), goldenrod (*Solidago sp.)*, and seaside knotweed (*Polygonum glaucum*) were all key volunteers found in the vegetation study.

Native graminoids that were planted contributed to 11% of the overall study biomass. Virginia or riverbank wildrye (*Elymus virginicus* or *riparius*) was the key species found across 27% of the plots in the vegetation study.

Non-Native Plant Performance

Non-native plant biomass amount found in the vegetation study was similar to native plants. Non-native forbs (28 species, 517 g) had higher species richness and biomass than non-native graminoids (9 species, 278 g). This reflects the availability of seeds able to enter the ecosystem from elsewhere (Richardson et al., 2007). The two closest areas where non-native seeds come from are the adjacent roadside on one side, or adjacent existing riparian buffer for Ancaster Creek (Rentch et al., 2005; Spellerberg, 1998**).**

Of particular concern for native plant restorations is the entry of non-native plants that are invasive (Guido and Pillar, 2017; Ogden and Rejmanek, 2005). Invasive species are one of the biggest threats to plant biodiversity loss as they alter ecosystem function and extinguish native plant species (Guido and Pillar, 2017). The highly adaptable nature of invasive species allows quick reproduction in disturbed areas with the benefit of few predators, leading to out-competing native species for food and habitat (Guido and Pillar, 2017). Two non-native plants of concern were found in the vegetation study, queen anne's lace and phragmites (European common weed). Queen anne's lace (*Daucus carota*), considered a noxious weed in Ontario, had the highest biomass (148 g, 29% of non-native forb biomass) of non-native forbs in the vegetation study, found in 54% of the plots (Table 2.6). Queen anne's lace grows a vertical taproot that absorbs water and nutrients in the surrounding soil, removing those resources from native plants (Rentch et al., 2005). A less prevalent invasive species found in vegetation study was phragmites (*Phragmites* *australis*), with only 6.86 g of biomass found (Table 2.6). However, once phragmites establishes, it creates a dense rhizomous root network, establishing monospecific stands which can quickly compete with native plants for space and resources (Asaeda and Karunaratne, 2000). Phragmites can spread rapidly in degraded areas as it is tolerant of water level changes, salinity, and open areas (Jodoin et al., 2007; Richburg et al., 2001). Roadsides and riparian zones provide optimal phragmites habitat. There are several areas along the original riparian buffer with dense sections of phragmites that could spread rapidly into the extended riparian buffer. Cootes Paradise Marsh, downstream from Ancaster Creek, has been impacted by phragmites since around 1970, and its establishment is leading to the loss of habitat for native species like native cattails (*Typha latifolia*) (Wei and Chow-Fraser, 2005). While queen anne's lace can be removed without much difficulty by hand, phragmites is strong and difficult to fully remove by hand, even with shovels.

2.5.3 Buffer Location Influence on Plant Performance

Plant Biomass

Native forb biomass per quadrat was greatest at the edge position, and near the origin (0 m), decreasing towards the end (500 m) of the riparian buffer. In contrast, non-native forb biomass per quadrat increased with increasing distance. Biomass can act as an indicator of competition occurring between native and non-native plants (Evan et al., 1999; Roberts et al. 2010), that is, where native plants do well, non-native plants do not, and vice versa. There may be a resource benefiting native forbs over non-native forbs at lower distances on the buffer, which will be described later in the discussion. Native forb biomass per quadrat at the edge was greater ($p<0.05$) than native forb and graminoid biomass per quadrat at all positions except native forbs at the creek position. The edge success is likely due to the access to stormwater from the parking lot at this position, allowing improved growth for the plants. Middle and creek positions are further from the parking lot edge, with less access to water. Native forb biomass per quadrat at the middle follows similar trends to the edge, but the creek flips with native biomass per quadrat increasing with distance and non-native biomass decreasing. Native and non-native graminoid biomass per quadrat did not vary across the buffer distance as seen in forbs (Figure 9).

Plant Species Richness

Species richness was higher for non-native (37 species) plants than native (12 species) plants ($p<0.05$) found in the vegetation study. Forb species richness (10 native, 28 non-native) was greater than graminoid species richness (2 native, 9 non-native) ($p<0.05$), which likely helped non-native forbs have higher overall biomass than graminoids. As the restoration is young, and adjacent areas are degraded, the abundance of non-native plant species is typical (Hillhouse and Zedler, 2011). Non-native forbs in the creek and edge position were greater than the native and non-native forbs in the middle ($p<0.05$). This is potentially another reflection of the introduction of non-native species from either end of the restoration, the parking lot roadside and the riparian buffer.

Along the buffer distance, species richness per quadrat increased for non-native species. Lower salinity at further distances allowed for a more diverse establishment.

Further, the lower distances from 0-100 m are south facing, which would dry out the soil faster, potentially making germination and seed establishment difficult.

Salinity Impacts on Plant Growth

With increasing distance, species richness per quadrat increased for all forbs and graminoids. This is likely due to the decrease in salinity with increasing distance, that is, higher plant diversity is possible where lower salt is found. Salt can inhibit germination of new seedlings or induce osmotic stress on more mature plants, causing water loss and plant desiccation (Gibson and Carrington, 2008). However, halophytes are plants that can tolerate salinity through an osmotic adjustment to low external water potential and a controlled uptake and compartmentalization of Na^+ and Cl^- (Flowers and Colmer, 2008).

Increasing salinity resulted in increased growth (biomass per quadrat) for native forbs in the vegetation study (Figure 2.11). While increased biomass per quadrat may appear to indicate plant success, the number of species able to tolerate the high salinity was low. Only one native forb displayed an increasing biomass per quadrat with increasing salinity (Figure 2.12), salt marsh sand spurrey (*Spergularia marina*). Sand spurrey can explain the overall trend as it comprises 416 g or 59% of the native forbs found in the vegetation study. The buffer edge, where the highest salinity exits, is where sand spurrey grew best, found at 79%, 43%, and 7% of the edge, middle, and creek quadrats in the vegetation study. As a halophyte, sand spurrey is capable of absorbing and storing salt, indicating the potential to use sand spurrey in phytoremediation of saline soils. Cheeseman et al. (1984) found that sand spurrey grown in saltwater was two times larger than in freshwater, indicating that sand spurrey growth is improved with salt.

Originally a native of coastal regions in Britain, Europe, and North America (Cheeseman et al., 1984), finding sand-spurrey in Ontario indicates a potential range shift to inland, freshwater sites. Shifting ranges of coastal halophytes could become more common with the increased use of de-icing salts across North America. Another coastal species volunteering as a native forb in the study was seaside knotweed (*Polygonum glaucum*), with a total of 37.85 g found in the study. Range shifts of native plants due to changing environmental conditions highlights the difficulty of native plant classification (Shakelford et al., 2013).

2.5.4 Native Plant Restoration Challenges and Recommendations

This study explores some of the challenges for native plant restorations, especially on sites with nearby degradation. Soil type, seeding amount, species selection, invasions, climate, and animal interactions are some of the necessary considerations when restoring land (Barak et al., 2017; Davison, 1971; Haan et al, 2012; Hillhouse and Zedler, 2011). An immediate concern for a new restoration is the establishment of non-native invasive species. If left unmanaged, invasive plants could spread and out-compete native species rapidly. Further, controlling invasives becomes more complicated at a landscape level than the site level (Ogden and Rejmanek. 2005). Finally, it is important to fully consider desired function of a restoration, as restoration's are rarely found to be functionally equivalent to reference sites (Barak et al., 2017). Thus, a restoration is more often a reclamation, which reclaims the site to a function reflective of the original site, but rarely equivalent, which is what restoration implies (Barak et al., 2017).

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2.7 TABLES

Table 2.1: Natural Areas and Land Use of the Ancaster Creek watershed. Derived from the Hamilton Conservation Authority's Stewardship Action Plan for the Ancaster Creek watershed in 2008.

NATURAL AREAS		
Ecosystem	*Area (km^2)*	*Watershed Area (%)*
Forest	3.9	28.5
Wetland	0.04	0.3
Meadow	0.2	1.6
Total	**4.14**	**30.4**
LAND USE		
Land Use	*Area (km^2)*	*Watershed Area (%)*
Agricultural	2.2	16
Commercial	0.3	2
Industrial	0.04	0.2
Institutional	1	7
Open Space	2.3	16.7
Residential	5.6	40.1
Transportation	1.86	13.5
Utility	0.4	2.9
Impervious Surfacing	4.92	36
Total	**13.7**	**100**

Table 2.2: Native plants seeded in 2014 on the riparian buffer. 33 species; 20 forbs and 13 graminoids. Mixes for dry and wet areas, along with the edges of the turtle mounds.

Seed Mix	Scientific Name	Common Name	Lifeform
DRY	*Rudbekia hirta*	black-eyed susan	forb
	Rudbekia laciniata	tall coneflower	forb
	Desmodium canadense	showy ticktrefoil	forb
	Oenothera biennis	common evening primrose	forb
	Monarda fistulosa	wild bergamot	forb
	Verbena hastata	swamp verbena	forb
	Penstemon digitalis	foxglove beardtongue	forb
	Lobelia siphilitica	great blue lobelia	forb
	Euthamia graminifolia	grass-leaved goldenrod	forb
	Symphiotrichum novae-angliae	new england aster	forb
	Symphiotrichum lanceolatum	lance-leaf aster	forb
	Pycnanthemum virginianum	virginia mountain mint	forb
	Elymus virginicus	virginia wildrye	graminoid
	Elymus riparius	riverbank wildrye	graminoid
	Elymus hystrix	bottlebrush	graminoid
	Bromus cilliatus	fringed brome	graminoid
	Scirpus atrovirens	green bulrush	graminoid
	Juncus tenuis	poverty rush	graminoid
	Carex vulpinoidea	fox sedge	graminoid
	Glyceria striata	fowl mannagrass	graminoid
WET	*Mimulus ringens*	monkey flower	forb
	Symphiotrichum puniceus	swamp aster	forb
	Asclepias incarnata	swamp milkweed	forb
	Lobelia cardinalis	water lobelia	forb
	Carex bebbii	bebb's sedge	graminoid
	Carex hydrericena	porcupine sedge	graminoid
	Carex retrorsa	retrorse sedge	graminoid
	Schoenoplectus tabernamontani	softstem bulrush	graminoid
	Juncus effuses	soft rush	graminoid
TURTLE	*Asclepias tuberosa*	butterfly weed	forb
	Penstemon hirsutus	hairy beardtongue	forb
	Lobelia inflata	indian tobacco	forb
	Solidago nemoralis	gray goldenrod	forb
	Juncus tenuis	poverty rush	graminoid

Table 2.3: Soil texture and fertility at several distances (From 0 to 500 m) and positions (Edge, Middle, and Creek) along the riparian buffer. Samples were taken at the centre of the vegetation study quadrat with a core (2cm diameter and 8cm depth). The percentage of gravel, sand, silt, and clay determines the texture classification for each soil sample.

SOIL TEXTURE						
Distance	Position	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Texture
40	Edge	22.4	42.7	40.9	16.4	Gravelly Loam
100		8.7	42.4	40.9	16.7	Loam
280		15.6	49.0	39.3	11.7	Loam
160	Middle	11.3	38.9	42.1	19.0	Loam
340		35.2	48.1	37.9	14.0	Gravelly Loam
460		36.5	45.0	40.0	15.0	Gravelly Loam
70	Creek	6.0	21.2	55.0	23.8	Silt Loam
190		14.5	41.5	39.6	19.0	Loam
400		7.5	43.1	37.4	19.5	Loam
SOIL FERTILITY						
Distance	Position	Phosphorus (mg/L)	Magnesium (mg/L)	Potassium (mg/L)	Organic Matter (% dry)	pH
190	Edge	65	150	100	3.2	7.3
10	Middle	21	96	67	0.5	7.5
100		36	240	74	2.1	7.4
370		20	150	39	1.6	8.0
430		54	180	130	3.3	7.5
160	Creek	32	190	97	2.3	7.7
340		29	220	120	3.6	7.7

Table 2.4: Summary of the number of species and biomass found (or planted and not found) for all plants on the riparian buffer. Native and non-native plants are further divided by lifeform (forb or graminoid) and then the treatment. Non-native plants on the riparian buffer but not in the vegetation study were not investigated as thoroughly as the native plants that were planted and then either observed, or not observed along the riparian buffer. All biomass values rounded to the nearest gram.

NATIVE PLANTS			
Lifeform	Treatment	Number of Species	Biomass (g)
Forb	Planted and Not Observed	8	
	Planted and Observed	6	
	Planted and in Vegetation Study	6	37
	Volunteers in Vegetation Study	4	668
	Vegetation Study Native Forbs	*10*	*705*
Graminoid	Planted and Not Observed	4	
	Planted and Observed	6	
	Planted and in Vegetation Study	2	194
	Volunteers in Vegetation Study	0	
	Vegetation Study Native Graminoids	*2*	*194*
	Vegetation Study Native Plants	**12**	**899**
NON-NATIVE PLANTS			
Forb	Not Planted and In Vegetation Study	28	517
Graminoid	Not Planted and In Vegetation Study	9	278
	Vegetation Study Non-Native Plants	**37**	**795**

Table 2.5: Native forbs found in the riparian buffer vegetation study. Biomass in grams and the percentage of plots the species was found in (Out of 48 plots).

NATIVE FORBS				
Location	Scientific Name	Common Name	Biomass (g)	% Plots
Planted & In Vegetation Study (6)	*Oenothera biennis*	common evening primrose	21.98	6
	Rudbekia hirta	black-eyed susan	5.02	6
	Symphiotrichum lanceolatum	lance-leaf aster	4.59	6
	Euthamia graminifolia	grass-leaved goldenrod	2.77	4
	Solidago nemoralis	gray goldenrod	1.25	2
	Aslepias tuberosa	butterfly milkweed	0.89	4
		Total Biomass (rounded to nearest g)	**37**	
Volunteers in Vegetation Study (4)	*Spergularia marina*	salt marsh sand-spurrey	416.10	19
	Solidago (c.f. canadensis)	goldenrod sp.	210.57	40
	Polygonum glaucum	seaside knotweed	37.85	31
	Symphiotrichum ericoides	heath aster	3.83	4
		Total Biomass (rounded to nearest g)	**668**	
Planted & Not Observed on Buffer (8)	*Rudbekia laciniata*	tall coneflower		
	Desmodium canadense	showy ticktrefoil		
	Penstemon digitalis	foxglove beardtongue		
	Lobelia siphilitica	great blue lobelia		
	Lobelia cardinalis	water lobelia		
	Lobelia inflata	indian tobacco		
	Pycanthemum virginianum	virginia mountain mint		
	Mimulus ringens	monkey flower		
Planted & Observed on Buffer (6)	*Monarda fistulosa*	wild bergamot		
	Verbena hastata	swamp verbena		
	Symphiotrichum novae-angilae	new england aster		
	Asclepias incarnata	swamp milkweed		
	Symphiotrichum puniceus	swamp aster		
	Penstemon hirsutus	hairy beardtongue		

Table 2.6: Non-Native forbs (28 species) found in the riparian buffer vegetation study. Biomass in grams and the percentage of plots the species was found in (Out of 48 plots).

NON - NATIVE FORBS			
Scientific Name	Common Name	Biomass (g)	% Plots
Daucus carota	queen anne's lace	147.95	54
Lotus corniculatus	birdfoot trefoil	100.05	13
Medicago lupulina	black medic	73.62	54
Cirsium arvense	canada thistle	37.3	4
Anthemis sp.	chamomile sp.	37.06	23
Plantago major	common plantain	29.36	23
Vicia cracca	cow vetch	29.11	2
Linaria vulgaris	butter and eggs	19.77	27
Plantago lanceolata	lance-leaved plantain	13.61	10
Euphorbia maculata	milk purslane	4.39	6
Dipsacus sp.	teasel sp.	3.61	4
Polygonum cespitosum	lady's thumb	3.51	2
Latuca serriola	prickly lettuce	3.37	8
Chenopodium sp.	goosefoot sp.	2.82	6
Taraxacum officinale	dandelion	2.43	17
Bidens frondosa	beggar ticks	2.23	2
Erigeron canadensis	horseweed	1.94	6
Erigeron sp.	fleabane sp.	1.34	4
Articum sp.	burdock sp.	1.11	2
Hieracium sp.	hawkweed sp.	0.99	4
Veronica persica	persian speedwell	0.39	2
Trifolium pratense	red clover	0.3	2
Melampyrum lineare	cowwheat	0.22	2
Sonchus arvensis	field sow thistle	0.11	2
Lepidium sp.	wild peppergrass	0.1	6
Artemesia vulgaris	common mugwort	0.1	2
Ulmus pumila	siberian elm (tree)	0.09	2
Anagallis arvensis	scarlet pimpernel	0.06	2
Total Biomass (rounded to nearest g)		**517**	

Table 2.7: Native graminoids found in the riparian buffer vegetation study. Biomass in grams and the percentage of plots the species was found in (Out of 48 plots).

NATIVE GRAMINOIDS				
Location	Scientific Name	Common Name	Biomass (g)	% Plots
Planted & In Vegetation Study (2)	*Elymus virginicus* or *riparius*	virginia or riverbank wildrye	186.83	27
	Juncus tenuis	poverty rush	6.92	2
	Total Biomass (rounded to nearest g)		**194**	
Planted & Not Observed (4)	*Glyceria striata*	fowl mannagrass		
	Scirpus atrovirens	green bulrush		
	Carex bebbii	bebb's sedge		
	Carex retrorsa	retrorse sedge		
Planted & Observed (6)	*Carex hystericena*	porcupine sedge		
	Juncus effusus	soft rush		
	Elymus hystrix	bottlebrush grass		
	Schoenoplectus tabernaemontani	softstem bulrush		
	Bromus cilliatus	fringed brome		
	Carex vulpinoidea	fox sedge		

Table 2.8: Non-Native graminoids found in the riparian buffer vegetation study. Biomass in grams and the percentage of plots the species was found in (Out of 48 plots).

NON - NATIVE GRAMINOIDS			
Scientific Name	Common Name	Biomass (g)	% Plots
Festuca rubra	red fescue	162.56	42
Echinocloa sp.	barnyard grass	53.28	40
Panicum capillare	witchgrass	16.25	29
Digitaria sp.	crabgrass	15.74	15
Lolium perenne	perennial rye	12.51	4
Phragmites australis	european common weed	6.86	2
Bromus inermis	smooth brome	4.93	15
Cyperus rotundus	nut sedge	3.43	13
Setaria sp.	foxtail	2.61	2
	Total Biomass (rounded to nearest g)	**278**	

Table 2.9: Analysis of Covariance (ANCOVA) of the effect of plant origin, lifeform, position, and distance on plant biomass (dry weight) per quadrat from the riparian buffer vegetation study. Origin is native or non-native. Lifeform is forb or graminoid. Position is edge, middle, or creek across the width of the lot m riparian buffer. Distance is 0-500 metres along the length of the lot m riparian buffer.

	Plant Biomass		
Source	d. f.	F-ratio	P-value
Origin	1	0.33	0.5646
Lifeform *	1	17.48	**<0.0001**
Position *	2	6.25	**0.0024**
Distance	1	0.84	0.3594
Origin x Lifeform	1	2.31	0.1307
Origin x Position	2	0.29	0.7511
Lifeform x Position	2	2.79	0.0641
Origin x Distance *	1	9.83	**0.0020**
Lifeform x Distance	1	0.51	0.4763
Position x Distance	2	0.22	0.7996
Origin x Lifeform x Position	2	1.34	0.2649
Origin x Lifeform x Distance *	1	7.83	**0.0057**
Origin x Position x Distance *	2	4.05	**0.0192**
Lifeform x Position x Distance	2	0.18	0.8332
Origin x Lifeform x Position x Distance *	2	6.86	**0.0014**

Note: P Values < 0.05 are bolded, and corresponding source has a star (*)

Table 2.10: Analysis of Covariance (ANCOVA) of the effect of plant origin, lifeform, position, and distance on plant species richness (number of species) per quadrat from the riparian buffer vegetation study. Origin is native or non-native. Lifeform is forb or graminoid. Position is edge, middle, or creek across the width of the lot m riparian buffer. Distance is 0-500 metres along the length of the lot m riparian buffer.

	Species Richness		
Source	d. f.	F-ratio	P-value
Origin *	1	66.17	**<0.0001**
Lifeform *	1	51.41	**<0.0001**
Position *	2	4.30	**0.0152**
Distance *	1	19.58	**<0.0001**
Origin x Lifeform	1	0.30	0.5883
Origin x Position	2	1.45	0.2378
Lifeform x Position	2	3.01	0.0520
Origin x Distance *	1	9.80	**0.0021**
Lifeform x Distance	1	2.78	0.0975
Position x Distance	2	0.26	0.7720
Origin x Lifeform x Position	2	2.69	0.0708
Origin x Lifeform x Distance *	1	4.85	**0.0290**
Origin x Position x Distance	2	0.58	0.5636
Lifeform x Position x Distance	2	0.04	0.9644
Origin x Lifeform x Position x Distance	2	0.21	0.8094

Note: P Values < 0.05 are bolded, and corresponding source has a star (*)

Table 2.11: Analysis of Covariance (ANCOVA) of the effect of plant origin, lifeform, and electrical conductivity on plant biomass (dry weight) per quadrat from the riparian buffer vegetation study. Origin is native and non-native. Lifeform is forb or graminoid. Position is edge, middle, or creek. Electrical conductivity values are taken from the average of 2016 WET sensor measurements.

	Plant Biomass		
Source	d. f.	F-ratio	P-value
Origin	1	0.33	0.5674
Lifeform *	1	17.22	**<0.0001**
Conductivity *	1	4.61	**0.0331**
Origin x Lifeform	1	2.27	0.1335
Origin x Conductivity *	1	7.88	**0.0055**
Lifeform x Conductivity	1	3.64	0.0580
Origin x Lifeform x Conductivity *	1	27.38	**<0.0001**

Note: P Values < 0.05 are bolded, and corresponding source has a star (*)

2.8 FIGURES

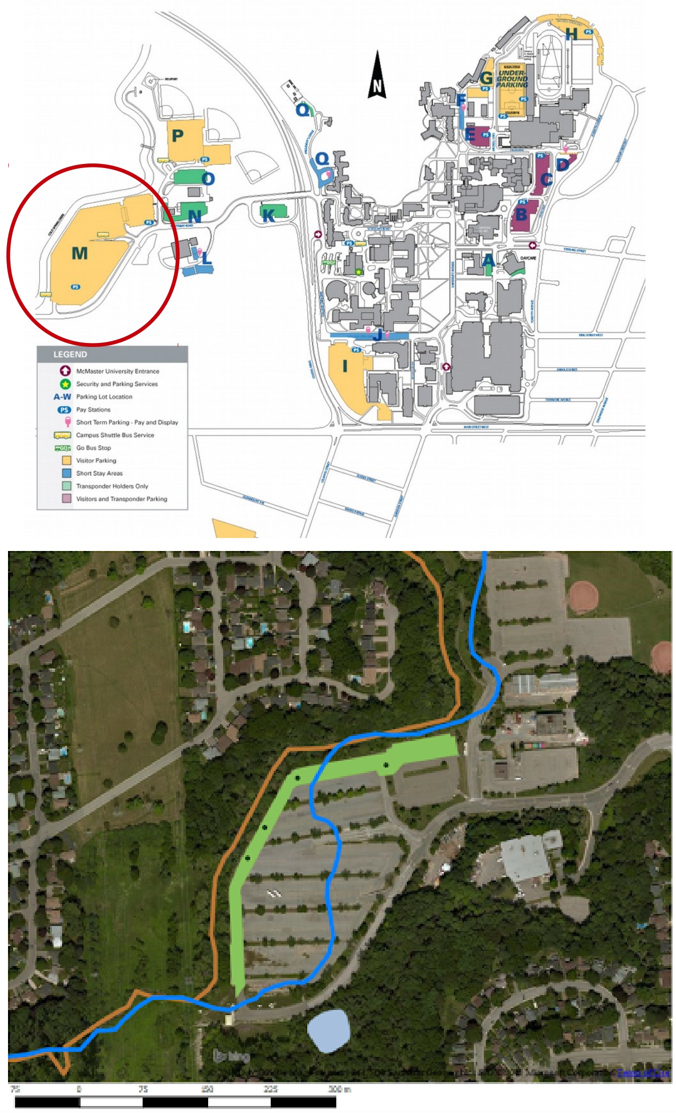

Figure 2.1: Top: McMaster University campus with parking lot M on the west campus highlighted with a red circle. Bottom: Ancaster Creek flow path at parking lot M before construction (Blue line - 1965) and present day (Orange line – 2017). Riparian buffer addition in 2014 is indicated in green, with four dark green circles representing turtle habitat. Blue circle in south is the location of a historic pond.

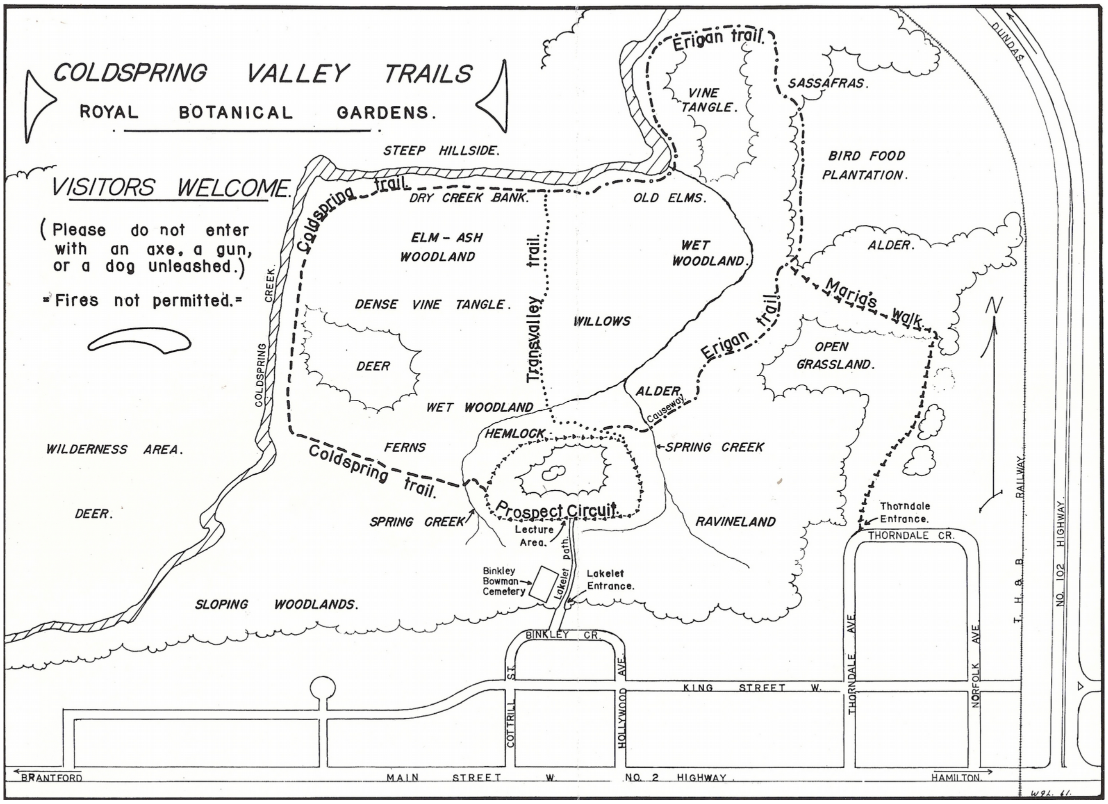

Figure 2.2: Historic trail map of the Coldspring Valley Nature Sanctuary from the Royal Botanical Gardens, who protected the area from 1958-63. Coldspring (Ancaster) Creek ran through what is now parking lot m, constructed in 1968.

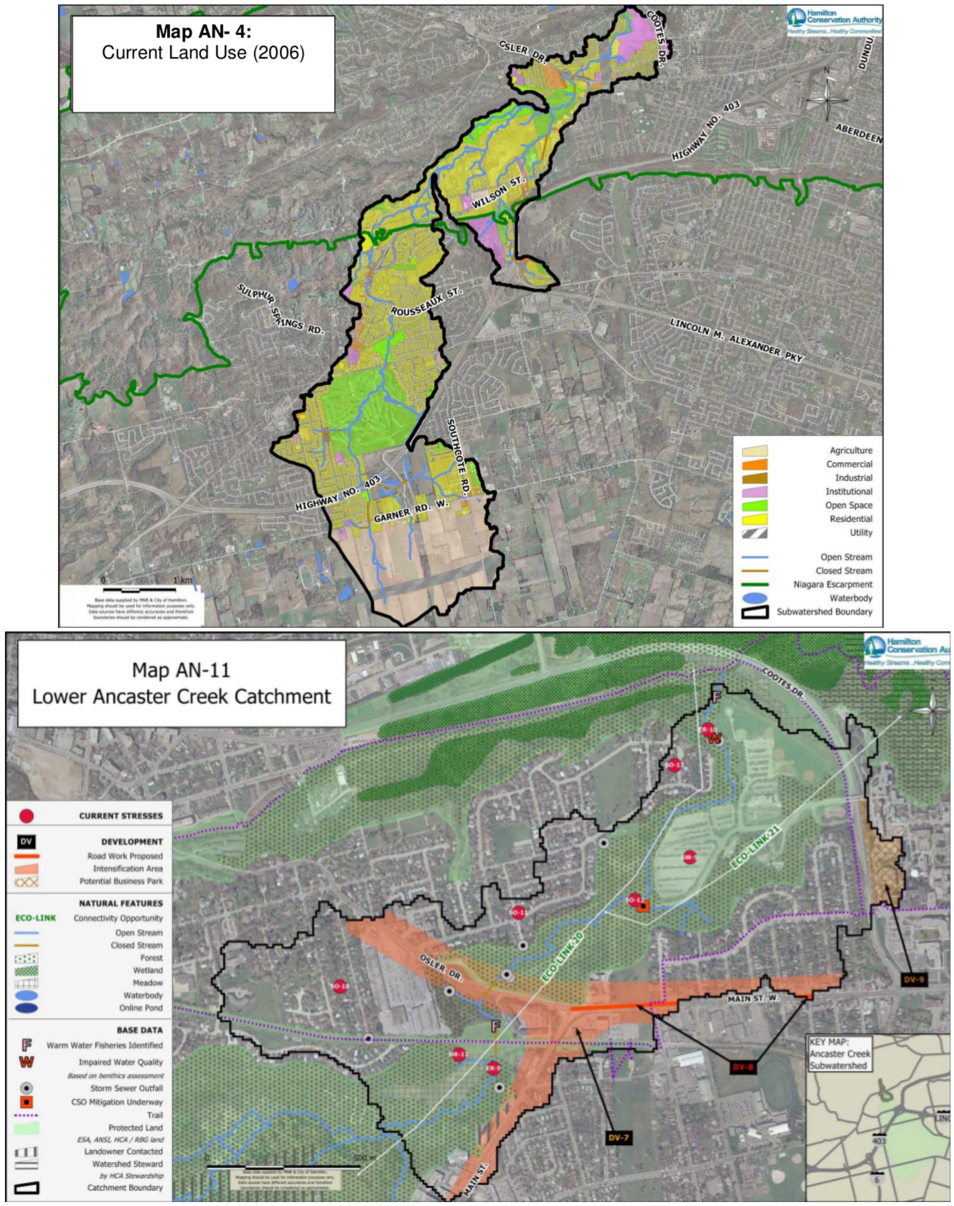

Figure 2.3: Top: Land use for the Ancaster Creek watershed from 2006. Bottom: Lower Ancaster Creek Catchment displaying constructed and natural features. A red circle on lot M indicates a current stormwater stress. Derived from the Hamilton Conservation Authority's Stewardship Action Plan for the Ancaster Creek watershed in 2008.

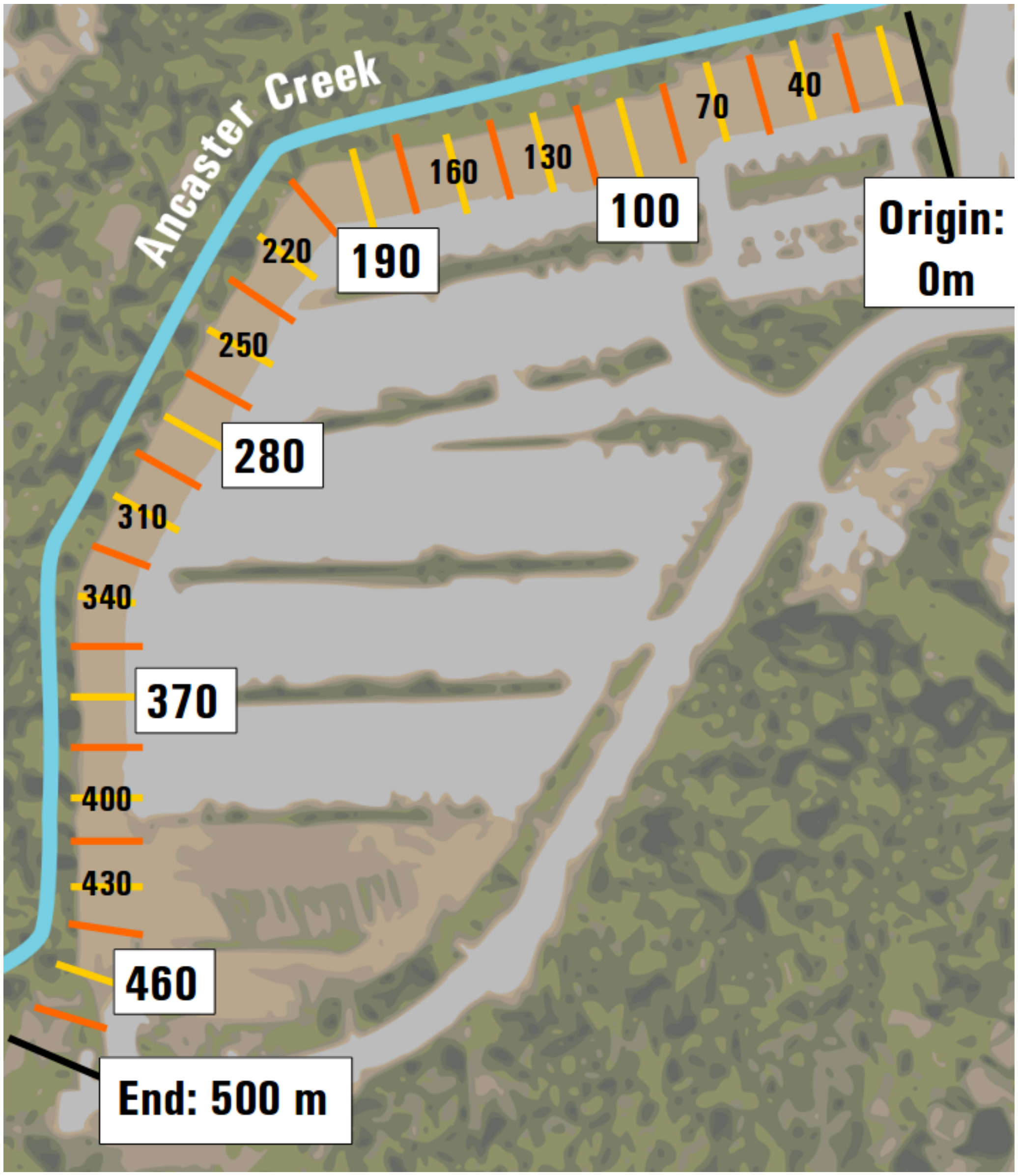

Figure 2.4: Distances used for soil and vegetation field studies on the riparian buffer, in 15 m increments. WET sensor (soil) measurements were taken every 15 m beginning at 10 m (orange and yellow lines). Vegetation study quadrats were sampled every 30 m beginning at 10 m (yellow lines), for a total of 16 locations and 48 plots.

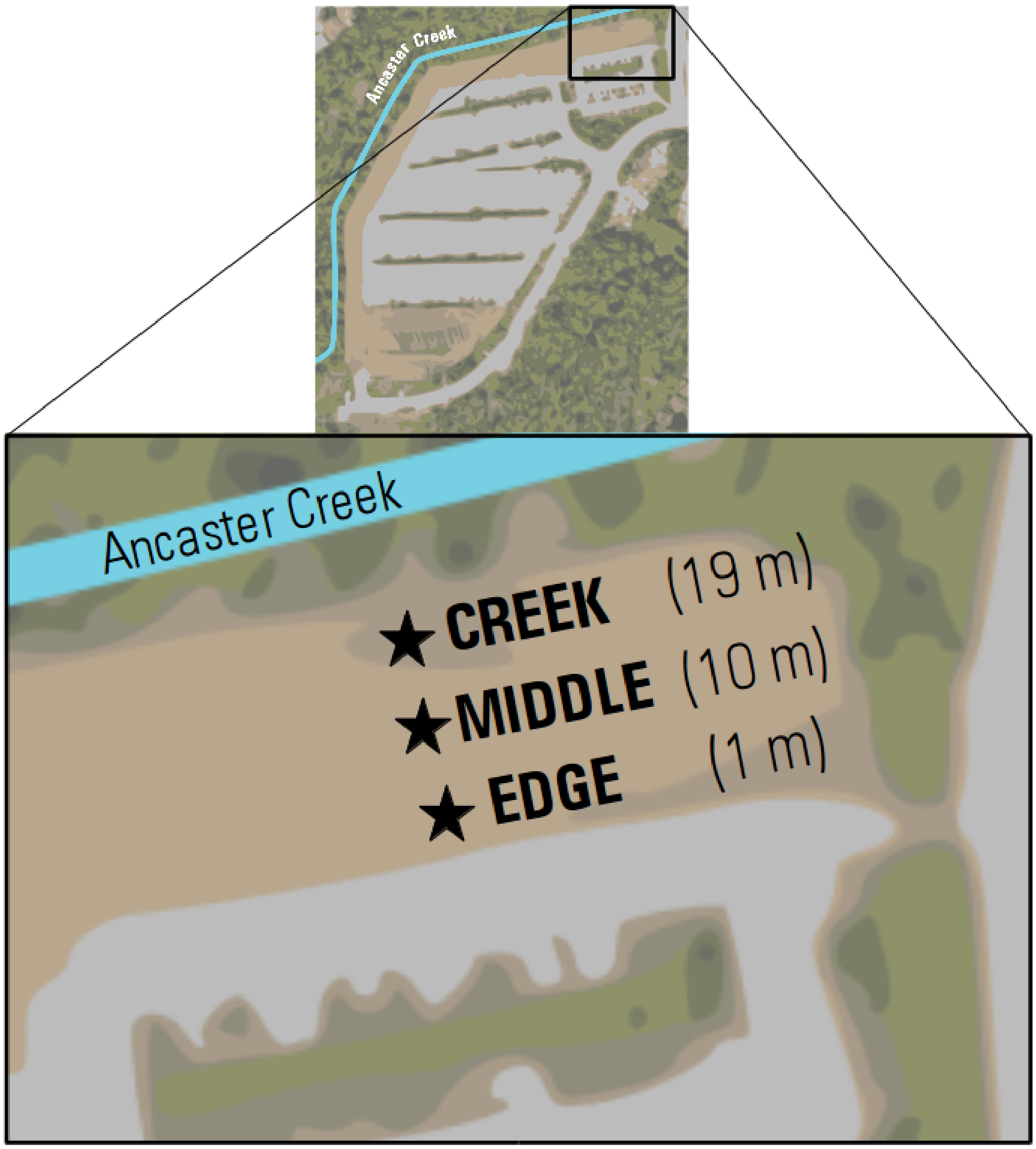

Figure 2.5: A section of the lot m riparian buffer is expanded to show positions along the width of the buffer. Positions include the edge (1 m from parking lot edge), middle (10 m), and creek (19 m). Position was used for soil and vegetation measurements.

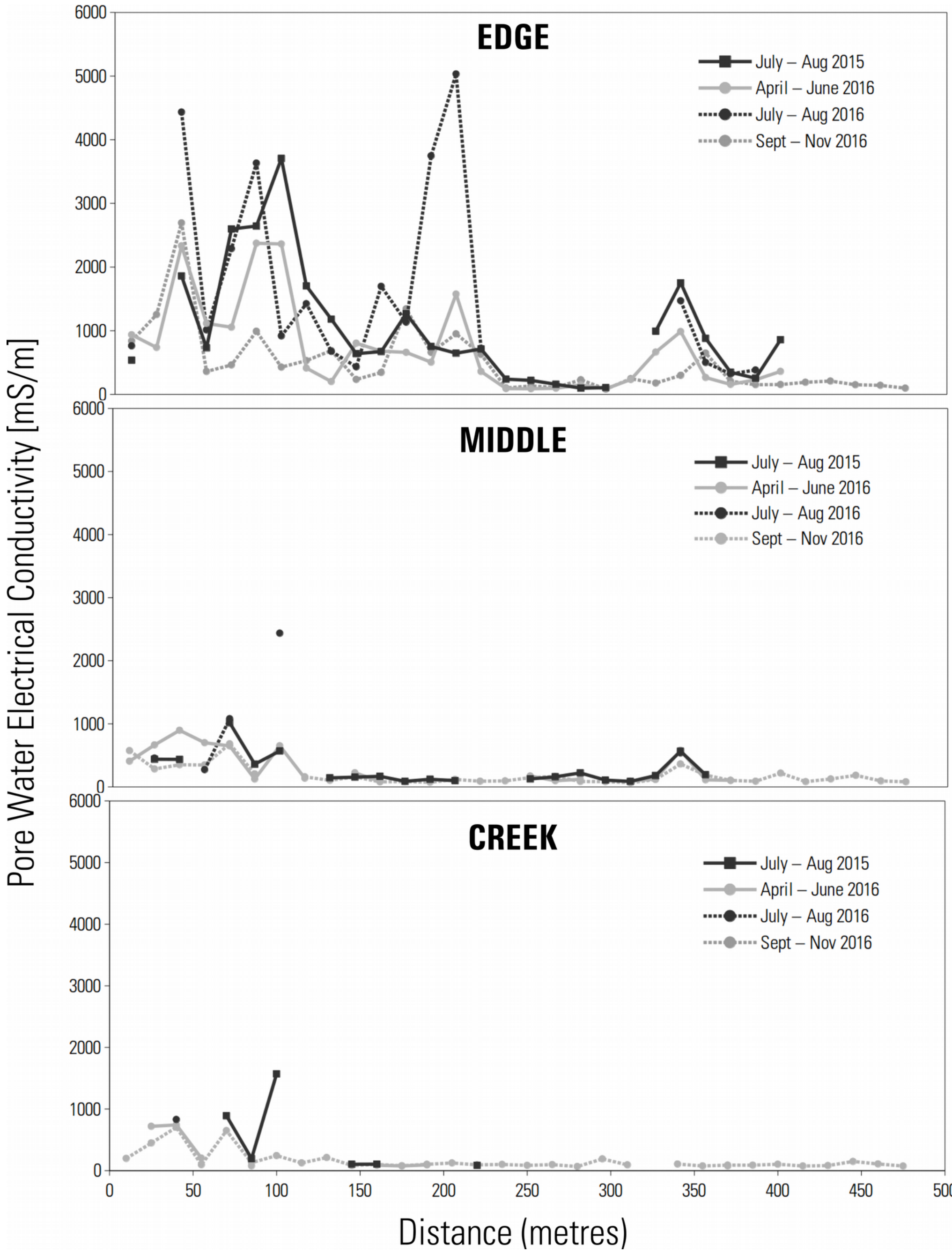

Figure 2.6: Pore Water Electrical Conductivity at the edge, middle, and creek positions measured every 15m beginning at 10 m along the 0-500 m distance of the riparian buffer averaged for months in 2015 and 2016. Data is most continuous for edge as the soil was softest there, with dryer soil moving towards the middle and creek making it more difficult to insert the WET sensor probe. Conductivity values come from discrete measurements along the buffer and can be identified by points on the graph.

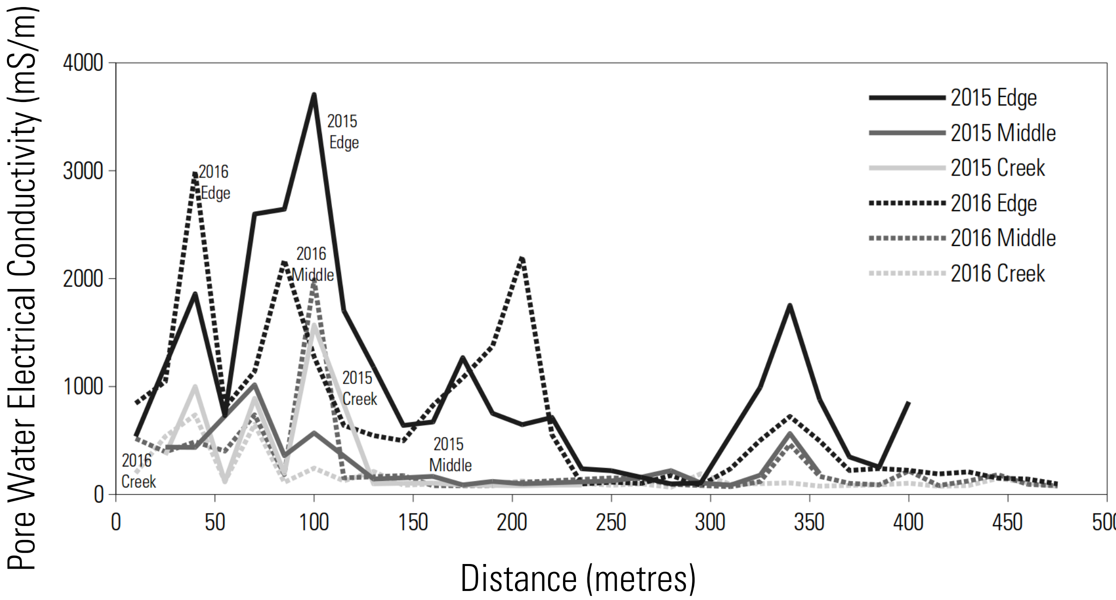

Figure 2.7: Yearly average pore water electrical conductivity at the edge, middle, and creek positions measured every 15m beginning at 10 m along the 0-500 m distance of the riparian buffer. 2015 measurements, collected in July and August only, are indicated by solid lines, while 2016 measurements, collected from April to November only, are indicated by dotted lines. Average conductivity values come from discrete measurements from 2015-16 and can be identified at every peak or valley on the graph.

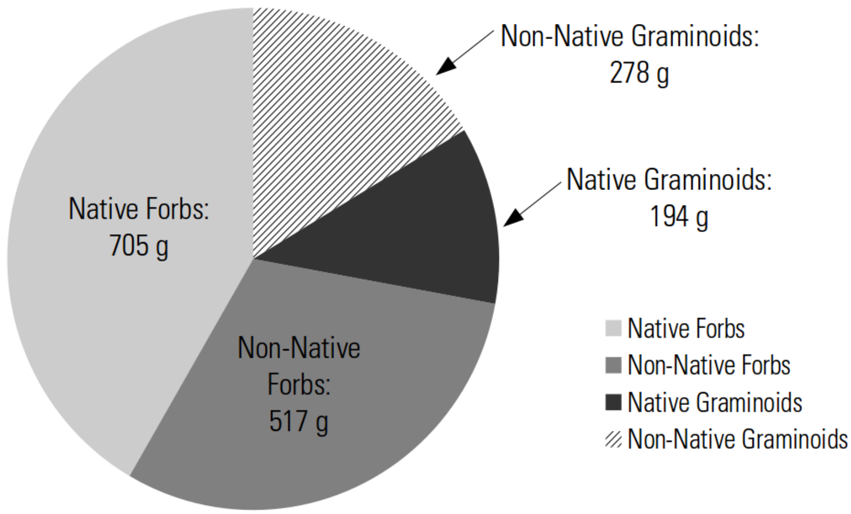

Figure 2.8: Vegetation biomass proportions for the riparian buffer vegetation study. Labelled by origin (Native or Non-Native) and lifeform (Forb or Graminoid).

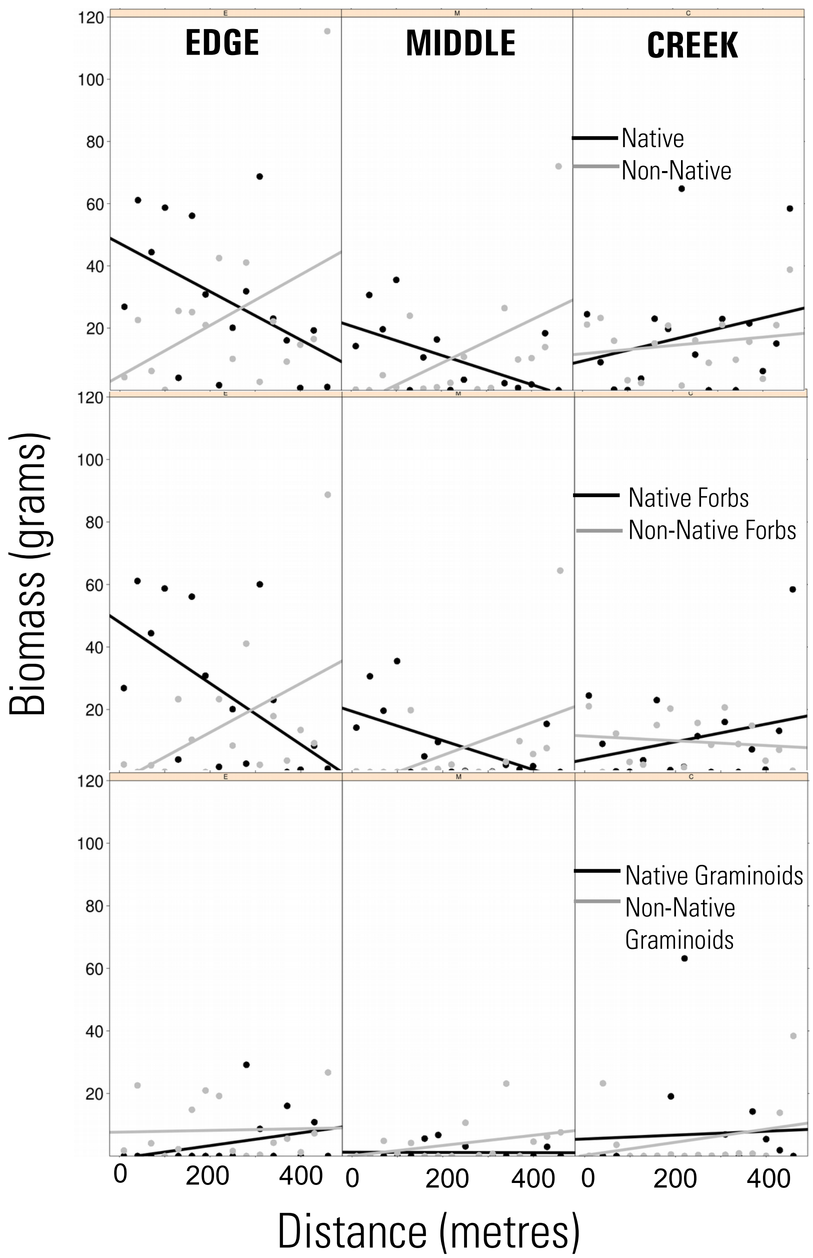

Figure 2.9: Vegetation Biomass per quadrat for native and non-native plants found in the riparian buffer vegetation study along the buffer distance (0-500 m) and at three positions across the buffer (edge, middle, and creek). Total biomass per quadrat (Top graph), Forb biomass per quadrat (Middle graph), and Graminoid biomass per quadrat (Bottom graph).

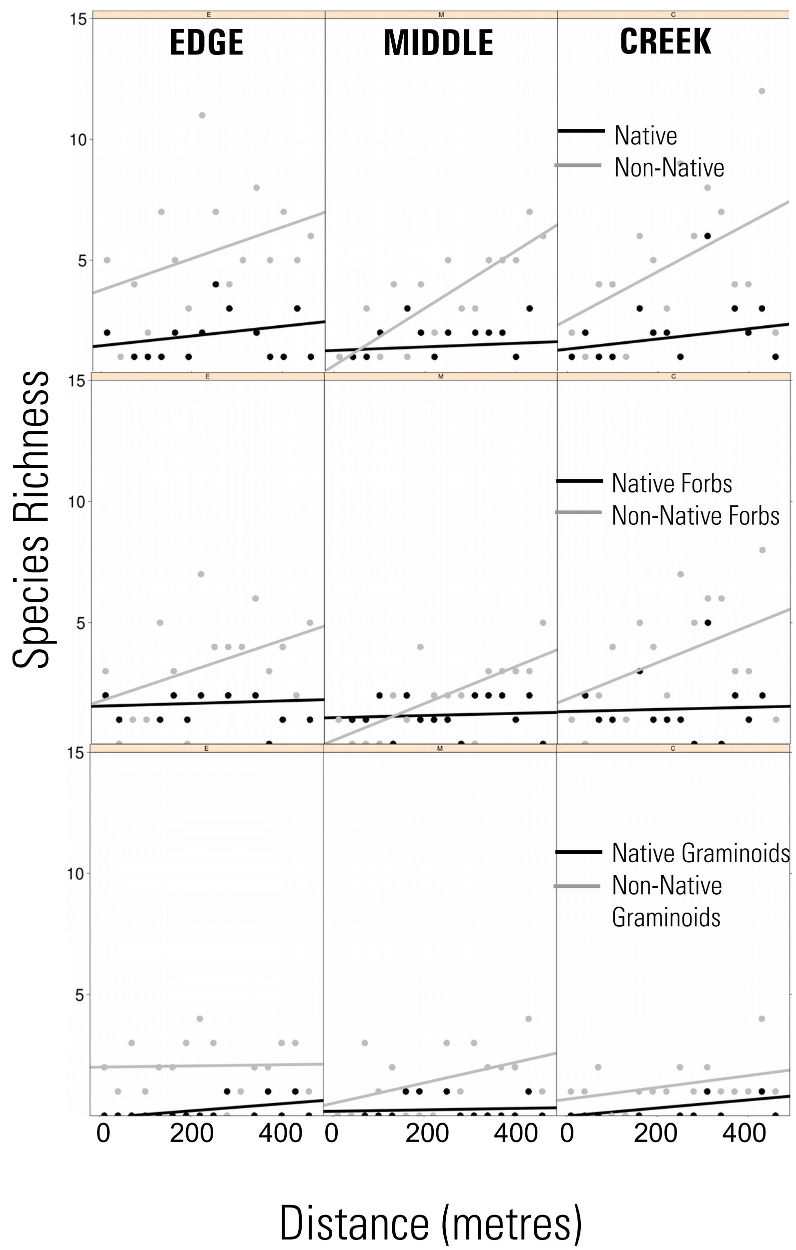

Figure 2.10: Species Richness (number of species) per quadrat for native and non-native plants found in the riparian buffer vegetation study along the buffer distance (0-500 m) and at three positions across the buffer (edge, middle, and creek). Total species richness per quadrat(Top graph), Forb species richness per quadrat (Middle graph), and Graminoid species richness per quadrat (Bottom graph).

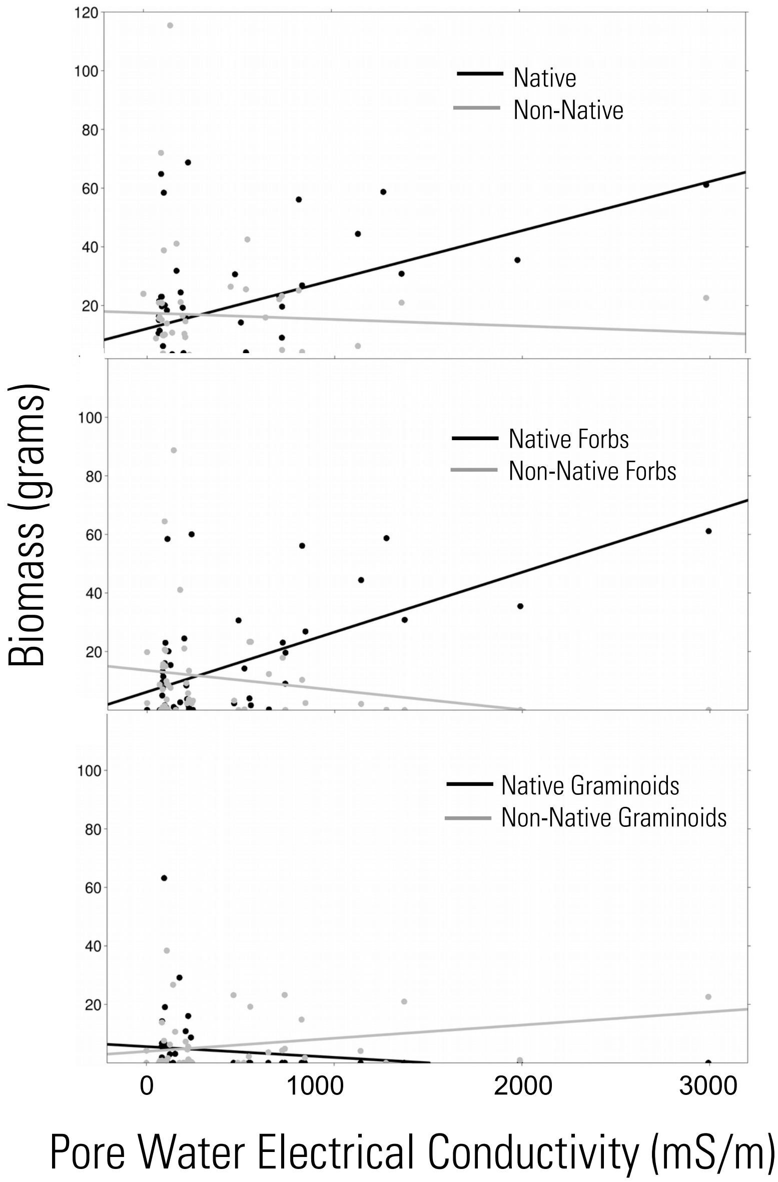

Figure 2.11: Biomass per quadrat for native and non-native plants found in the riparian buffer vegetation study in response to pore water electrical conductivity of the soil, inferred as salinity. Total biomass per quadrat (Top graph), Forb biomass per quadrat (Middle graph), and Graminoid biomass per quadrat (Bottom graph). Electrical Conductivity values are taken from the average from 2016 WET sensor (soil) measurements.

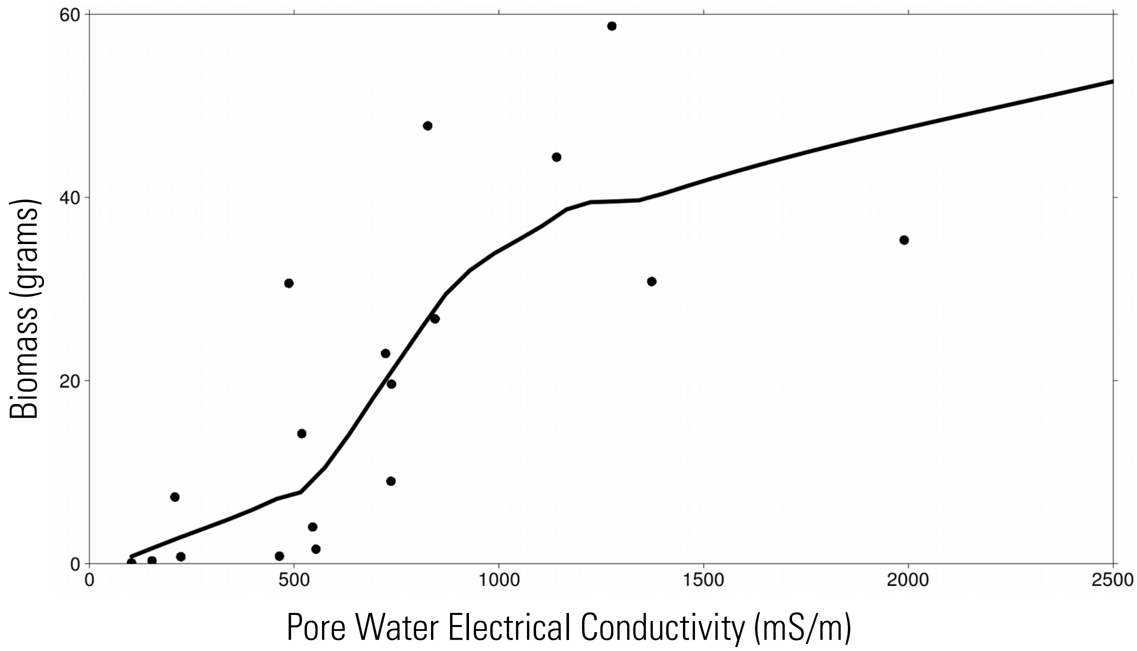

Figure 2.12: Sand Spurrey (*Spergularia marina*) biomass per quadrat found in the riparian buffer vegetation study, with increasing pore water electrical conductivity. Electrical conductivity values are taken from the average from 2016 WET sensor (soil) measurements.

CHAPTER 3: EVALUATING POTENTIAL WATER SOURCES FOR A RECLAIMED URBAN WETLAND

3.1 ABSTRACT

Wetlands exist because of an excess of water, and are dominated by water movement in an out of the ecosystem. Key hydrological fluxes were examined at parking lot M on the west campus of McMaster University in Hamilton, Ontario, that could contribute to a proposed 0.6 hectare constructed wetland on the parking lot, known as "McMarsh." Rainfall and groundwater flow was measured, and evapotranspiration was estimated using historic temperatures. Hillslope spring groundwater, a locally fed source, was the largest input to the potential wetland at 250 mm/month. Precipitation was a small input of 67 mm/month. Evapotranspiration was estimated to be 50 mm/month. Wetland water storage was found to be in surplus year round, with storage of 265 mm/month. An additional source of groundwater may be available of up to 764 mm/month, which is around three times the known hillslope spring groundwater rate. The potential wetland connection to Ancaster Creek may provide ecosystem benefits such as improved water quality and habitat creation. This study proves the feasibility of sustaining a 0.6 ha groundwater-fed constructed wetland in the south section of parking lot M.

3.2 INTRODUCTION

Water availability is the primary determinant of whether or not wetlands form at a particular position on the landscape (Cole et al., 1997; Winter, 1999). Wetlands are defined as land that is saturated with poor drainage, retaining water long enough to promote aquatic processes (NWWG, 1997). Water sources for wetlands include groundwater, precipitation (surface water), or both surface and groundwater (Brinson, 1993). Precipitation is water that falls as rain and/or snow. Groundwater is water moving below the surface, often fed by an aquifer, and can exist as local flow, where recharge and discharge areas are close together, or regional flow, where recharge and discharge areas are separated by one or more topographic changes (Winter, 1999). Local groundwater flow is more affected by seasonal variations, such as temperature or precipitation changes (Winter, 1999).

Marshes, swamps, and fens receive water from local or regional groundwater and precipitation (NWWG, 1997). Less common are bogs, which receive water from precipitation only, and are limited by flat terrain and/or fine textured soil such as clay (NWWG, 1997; Winter, 1999). Wetlands can be further classified by vegetation and subsurface geology (NWWG, 1997).

When wetlands are changed to suit development, their hydrology is fundamentally altered (Acreman and Miller, 2007). Therefore, restoring wetlands to their original function and state is most desirable to maximize ecosystem services (Moreno-Mateos and Comin, 2010). While new projects can accommodate natural hydrologic fluxes, retrofitting existing structures is more difficult, because of site limitations and degradation (Zedler, 2000). In this case, reclaiming the habitat to a similar or equivalent function is often desired (Bradshaw, 1997). Therefore, wetland reclamation priorities must first focus on determining how the hydrology of a particular site has changed, and then aim to re-establish key fluxes of water and solutes that existed prior to disturbance (Bradshaw, 1997; Zedler, 2000).

A water-balance approach that approximates all inputs and outputs of water into a wetland (Figure 3.1) is commonly employed to identify critical hydrological fluxes (Arnell, 1999; Ayub et al., 2010; Mogavero et al., 2009). Inputs to the wetland can include groundwater, precipitation, and surface water runoff (Figure 3.1). Outputs from the wetland can include groundwater, evapotranspiration, drainage, and surface water runoff (Figure 3.1). Evapotranspiration is the movement of water from surfaces (evaporation) and plants (transpiration) combined. Drainage is one potential disturbance that adds a new outflow from the wetland (Figure 3.1). Thus, when evaluating wetland reclamation measures, identifying key hydrological fluxes in the area are critical. Whether or not each natural flux is important in a given wetland depends on its position on the landscape, especially in the case of groundwater (Bedford, 1999). The magnitude of these water fluxes also determines the water quality in a particular wetland (Winter, 1999). For example, wetlands that receive most of their water inputs from groundwater often exhibit moderated thermal regimes (*e.g.* Lowry et al., 2007) and biogeochemistry reflective of groundwater sources (*e.g.* Devito and Hill, 1999).

Outline of the Study

A pilot project has recently been initiated at McMaster University in Hamilton, Ontario, where a portion of parking lot M will potentially be converted into a constructed wetland. However, because the magnitude and timing of hydrological fluxes are currently unknown at the site, the purpose of this study was to quantify hydrological fluxes at the parking lot. Herein, a water balance approach was employed to provide guidance on reclamation techniques that should be employed to utilize existing water flows and establish adequate hydrological conditions for the proposed wetland.

We measured and estimated the dominant hydrological fluxes on a monthly basis for the proposed constructed wetland at parking lot M. This study assumes that the proposed wetland is an impermeable, closed system underground with no groundwater or surface water outputs, and with no surface water inputs.

First, cumulative rainfall was measured through five rain gauges across the parking lot. Second, groundwater flow was measured at four surface spring sites on the surrounding east and south hillslopes. Third, drainage flow rates into the creek through parking lot storm drain outflows was measured. Evapotranspiration was estimated using comparable values for the region.

We asked the following questions:

1) How does each hydrological flux vary throughout the year, and which flux dominates?

2) How would the key hydrological fluxes contribute to a 0.6 ha constructed wetland?

3) How would wetland water storage vary on a monthly basis?

4) Would the wetland stay wet year round?

3.3 METHODS

3.3.1 STUDY SITE

Parking lot M is located on McMaster University's west campus and was built on a 5.7 ha former floodplain (Figure 3.2), where wetlands and forest once existed. Natural groundwater springs on the south and east hillslopes surrounding the parking lot sustained the floodplain ecosystem. Ancaster Creek, a 34 km long stream, ran through the centre of what is now lot M, meeting up with Spencer Creek 500 m north before entering Cootes Paradise Marsh, a large 250 ha urban wetland (Thomasen and Chow-Fraser, 2012).

In 1968, the floodplain was drained and the creek was shifted in order to make the area suitable for parking lot construction. Ancaster Creek was channelized and shifted west (Figure 3.2). Construction fill was added, and surface and ground water were redirected to the creek. The water balance shifted from one dominated by natural fluxes of precipitation, evapotranspiration, and groundwater (Figure 3.3: Past) to one dominated by unnatural movement through drainage and diversion, which lowered the water table and decreased the water storage below ground (Figure 3.3: Present). Evapotranspiration was decreased through the addition of asphalt for the parking lot surface. Groundwater inflow, primarily occurring via seepage (springs) from the surrounding south and east hillslopes, as well as surface runoff on the parking lot, was drained (Figure 3.3: Present). Hillslope spring water enters storm drains at the base of the slopes, and precipitation moves into storm drains on the parking lot surface. Both ground and surface water move into pipes below the parking lot directly into Ancaster Creek through outflow pipes (Figure 3.3:

Present). Rain water on the parking lot surface likely picks up contaminants such as oil, grease, and road salt before entering the storm drains, depicted with red arrows (Figure 3.3: Present). In contrast, the proposed reclamation (Figure 3.3: Proposed) would block drainage features and, potentially, divert groundwater into a constructed wetland from groundwater springs on the adjacent hillslopes. Natural fluxes would be restored for the 0.6 ha section of the parking lot; groundwater directed towards the wetland, increased evapotranspiration, and decreased output to the creek.

Refer to Chapter two, section 2.3.1 for a full description of the study site including ecological and historical background.

3.3.2 Rain Measurements

Five Tru-Chek rain gauges were installed across the parking lot (Figure 3.2). Gauges were mounted one metre above ground on wood (2 inch wide by 4 foot long) with nails on either side of the rain gauge. The first four gauges (R1-R4) were placed on the lower parking lot level, while R5 was at a higher elevation on the south service road (Figure 3.2). Around two centimetres of mineral oil were added to each gauge to prevent evaporation. The gauge was read by identifying the bottom of the meniscus of the rain below the mineral oil at eye level. Cumulative rainfall measurements were taken at irregular intervals (daily to weekly) from June 2015 to July 2017. Gauges were dumped and new mineral oil was added when the gauge was full.

Data Analysis

Cumulative measurements from the five rain gauges were averaged to create one value for each measurement date. Monthly cumulative precipitation values were compared to historical precipitation values from 1981 to 2010 found online (Current Results, 2017). Data collected at lot M only occurred from May or June to December, when temperatures were above 0 ºC. Data for months with missing data, or when snow occurred, was obtained from Environment Canada's historic data website at the Royal Botanical Gardens (RBG) Station. Snowfall was not recorded at the RBG site, therefore a simple distinction of months with only rainfall, or months that had snowfall was made. The monthly precipitation values from 2015-2017 were averaged and used as an input for the wetland water balance.

3.3.3 Evapotranspiration

Evapotranspiration for the proposed wetland was estimated using historical climate data and the Thornthwaite method for potential evapotranspiration (PET), as follows (Thornthwaite, 1948):

$$PET = 16\left(\frac{L}{12}\right)\left(\frac{N}{30}\right)\left(\frac{10\,T_a}{I}\right)^{\alpha} \qquad \text{Equation 1}$$

Where PET is the estimated potential evapotranspiration (mm/month), T_α is the average daily temperature for the specific month, N is the number of days in the month, and L is the average day length (hours) of the month.

$$\alpha = (6.75 \times 10^{-7})I^3 - (7.71 \times 10^{-5})I^2 + (1.792 \times 10^{-2})I + 0.49239 \quad \text{Equation 2}$$

$$I = \sum_{i=1}^{12} \left(\frac{T_{ai}}{5}\right)^{1.514} \quad \text{Equation 3}$$

Equation 2 is a constant, and I is the heat index for the full year.

Average monthly air temperatures from 1985 to 2015 were obtained online from the Hamilton Airport station (Time and Date, 2017). Day lengths from 2016 were also found online from the Hamilton Airport (Time and Date, 2017).

The constructed wetland will likely have a considerable area of ponded water with emergent cattails and other vegetation. Under these conditions, the actual evapotranspiration can be similar to undisturbed marshes in southern Ontario. Therefore, the actual evapotranspiration can be estimated as being close or equal to potential evapotranspiration (Price, 1994). Therefore, the potential evapotranspiration values estimated for the site will become the actual estimated evapotranspiration values.

3.3.4 Groundwater Measurements

Weir Locations

Flow rates were measured at the hilllslope groundwater springs and parking lot storm drain outflows using 90° V-Notch weirs. Four hillslope and four parking lot weirs were installed at parking Lot M (Figure 3.2). The four hillslope weirs (Figure 3.2: H1-4) were constructed at all locations where groundwater comes to the surface as a spring directly surrounding the parking lot. On the hillslopes, weirs capture upstream water and channelize it behind the weir in order to measure the flow rate at the v. At the parking lot outflows, weirs capture all water entering the creek from the underground pipe.

The parking lot outflow weirs (Figure 3.2: P1-4) were constructed at outflows from the parking lot, which capture surface water on the parking lot, and hillslope groundwater that enters storm drains. Two historic outflow locations were not measured, found between P3 and P4; one is a small broken pipe with a slow trickle of water, and the other is an outflow pipe infilled with sediment and vegetation.

Weir Design

Weirs were constructed with plywood and metal flashing to create the “V.” The overall size of the weir was created by surveying the site and estimating the length and width required. The V was cut out of plywood using a jigsaw. Two pieces of metal flashing were bent and merged at the bottom of the V, then attached to the wood with adhesive and staples on top. Each weir was reinforced with two pieces of rebar, driven into the surrounding soil. To mitigate erosion, a vinyl barrier was attached to the front of the weir.

During weir installation on the hillslope, water was blocked or diverted a few metres upstream, allowing a dry area to dig a one foot deep trench the width of the weir. Where thick roots were encountered, hand sawing was employed to remove roots and create the trench. Next, the weir was placed into the trench, and surrounding sediment was shovelled into the trench and mounded around the left and right sides of the weir. All weirs were installed between June and July 2015 (Table 3.1).

Weir installation for creek locations proved more challenging. Outflow pipes from the parking lot discharged onto narrow cement pads with raised banks on either side, making diversion impossible. To compensate for this, water was blocked with sand bags during weir installation and, as water impounded, weirs were quickly installed. Following the initial installation of P1, P2, and P4 in June 2015, no weir proved to be functional, and could not stop the water flow. Therefore, a second design was created and then installed in July 2015. Design modifications included: 1) larger weir size, 2) attaching vinyl to the upstream section of the weir with adhesive and staples, and 3) more adhesive used on the base and front of the weir to prevent leakage. Weirs were attached to the cement pad with construction adhesive. A large section of vinyl was attached to the back of the weir that also covered the upstream side of the weir on the cement pad bottom and the surrounding sediment bank. Sediment was piled on the vinyl to create a seal, and construction adhesive was applied to the bottom front edge of the weir to prevent leakage. Even after these retrofits, these four weirs did not appear to be working well, having various leakage issues. In particular, P4, with the highest flow of around 1L/s, inadequately functioned and water moved underneath the weir. Thus, for 2015, only hillslope weirs were monitored (Table 3.1).

In 2016, after settlement and vegetation growth, weirs P2 and P4 stabilized and a tight seal formed. Monitoring of all four parking lot weirs began on September 1, 2016 (Table 3.1), with weirs P2 and P4 being the most reliable. P3 still commonly exhibited leakage issues and P1 was often dry or back filled with sediment. Thus, the data set for parking lot outflow rates is much more sparse than the hillslope data set.

Weir Sampling

Flow rates were manually measured by positioning a three-litre plastic bucket under the weir V. The maximum amount of water that could be captured was timed, for three trials. For this study, seasonal changes and overall flow rates were of interest, so manual measurements sufficed. Measurements were weekly to bi-weekly, taken when there was no rain occurring to ensure the capture of groundwater alone, capturing the seasonal groundwater base flow. Measurements were not taken during precipitation events, however, latent precipitation from a recent event may have been flowing on the hillslopes and through the parking lot outflow during measurements. During each manual sampling, leaves and accumulated sediment were cleared upstream of the weirs using a shovel. A one metre channel behind the weir and two metres downstream were cleared throughout the study period. If sediment buildup was obstructing the flow, measurements were not taken.

A Heron Instruments Inc. Conductivity Plus Groundwater Monitoring Meter was used to measure conductivity and temperature at the weirs. On October 12, 2016, conductivity and temperature measurements began. The probe was inserted into three positions directly upstream of the weir: to the left, centre, and right of the V. The probe was left in the water until the readings equilibrated, which took 30 seconds to 2 minutes. During times where most water was frozen upstream of the weir, and only a small channel of water was flowing at the V, the probe was inserted directly into the outflow. The conductivity metre is only able to measure when water is flowing and above 0 ºC. Conductivity and temperature measurements were carried out from October 12 to

December 21 in 2016, and from January 13 to May 3 in 2017, with dates corresponding to (but not on all) days when flow rates were measured.

Data Analysis

Daily groundwater flow rates from the hillslope and parking lot weirs were expressed as an average of the three trials taken for each measurement date. The overall time series of flow rates from 2015-2017 was plotted on a three panel plot of cumulative precipitation, hillslope groundwater flow rates, and parking lot outflow rates. Monthly Hillslope flow data from 2015-2017 were separated, totalled, and averaged to determine groundwater input potential for the constructed wetland. Hillslope spring groundwater flow rates were converted into depths for the 0.6 ha constructed wetland.

Daily Conductivity and Temperature data were also averaged over three trials measured on each date and expressed as a time series from 2016-2017. Conductivity and Temperature data were graphed for all eight weirs over time.

3.4 RESULTS

3.4.1 PRECIPITATION

Monthly precipitation at parking lot M from 2015 to 2017 showed considerable variability between years (Table 3.2, Figure 3.4). Total precipitation in 2015 (760.5 mm) was reflective of historic amounts (647 mm), while 2016 saw around half historic precipitation (279.98 mm, Table 3.2). Spring of 2015 (March - May) began dry and then reflected historic amounts and spring 2016 was wetter to start and then decreased to around half of historic amounts in April and May (Table 3.2, Figure 3.4). In contrast, Spring 2017 was a very wet spring, with total spring precipitation of 387.45 mm, around half of annual historical amounts (Table 3.2). Summer rainfall (June – August) in 2015 was similar to historic values, but extremely dry in 2016 (Table 3.2). Fall (September - December) precipitation in 2015 was typical of historic values for September and October, but was quite low for November and December (Table 3.2). Fall precipitation in 2016 was low in September and November (Table 3.2).

3.4.2 GROUNDWATER

Hillslope Groundwater Flow

Hillslope groundwater (GW) spring flow rates (Figure 3.5, middle) found at four weirs from June 2015 to July 2017 varied from the lowest rate of 0.02 L/s at weir H3 in August 2015 to the highest rate of 0.74 L/s at weir H2 in May 2017. Total average GW spring flow rates for the four weirs for 2015, 2016, and 2017 were 0.5, 0.6, and 0.9 L/s respectively. It is important to note that these three values capture different time periods, with, for example, a greater summer length in 2016 (Figure 3.5, middle).

Similar GW flow rates are found at weir H1, H2, and H4, ranging from around 0.1 to 0.4 L/s, not including the three sharp peaks in 2017 (Figure 3.5, middle). GW flow at H3 is consistently low, from 0.02 to 0.15 L/s (Figure 3.5, middle). Sharp increases in flow rates tend to match with increases in cumulative precipitation (Figure 3.5, top and middle).

Seasonally, 2016 (the largest data set) shows greater flow rates in the spring (April-May), lower flow rates in summer (June-August), and greater flow rates similar to spring in the fall (September – December) (Figure 3.5, middle). This trend is mirrored in 2015 for the summer to fall increase, despite the drastic difference in rainfall amounts between 2015 and 2016, with 2016 receiving around half of the summer (June – August) rainfall as 2015 (Figure 3.4 and 3.5, middle). Spring (April and May) 2017 GW flow rates were quite high, reflecting snow melt and high amounts of precipitation (Figure 3.4 and 3.5, middle). Winter 2017 (January to March) had greater GW flow rates than fall 2016 (Figure 3.5, middle).

Parking Lot Outflow

Parking lot weir outflow (OF) rates were considerably higher than Hillslope GW flow rates from June 2015 to July 2017, ranging from the lowest rate of 0.005 L/s at weir P1 in December 2016 to the highest rate at weir P4 of 1.9 L/s in February 2017 (Figure 3.5, bottom). Both P1 and P3 have similar OF rates ranging from 0.005 to 0.5 L/s (Figure 3.5, bottom). P2 OF rates ranged from 0.06 to 0.9 L/s, and the fastest OF rates occur at P4, which ranges from 1.04 to 1.09 L/s (Figure 3. 5, bottom).

Seasonally, Fall 2016 (September to December) parking lot OF rates are slightly lower than Winter 2017 (January to March) and Spring 2017 (April to June) (Figure 3.5, bottom). Total average parking lot OF rates for the four weirs for 2016 and 2017 were 1.6 and 2.2 L/s respectively; which come from different time periods (Figure 3.5, bottom).

Parking lot OF and Hillslope GW spring flow rates are quite different. Parking lot OF rates are around two to five times greater than the hillslope GW flow rates, indicating that the parking lot weirs are capturing more than just the groundwater springs from the hillslopes.

Conductivity and Temperature

Temperature of water behind the hillslope and parking lot weirs reflected seasonal variability from measurements taken October 2016 to May 2017 (Figure 3.6, top). From October 2016 to February 2017, water temperature decreased overall from a maximum of 17 ºC at P2 to a minimum of 0 ºC at H1 (Figure 3.6, top). The parking lot weirs were around 2-3 ºC warmer than the hillslope weirs (Figure 3.6, top). Halfway through February 2017, both the hillslope and parking lot weir temperatures increase, climbing to a maximum of 16 ºC at H2, and a minimum of 10 ºC at P2 (Figure 3.6, top). Interestingly, in late February the hillslope weir temperature becomes 3-5 ºC higher than the parking lot weir temperature.

Electrical conductivity of the water behind the hillslope and parking lot weirs reflected some seasonal variability from October to May 2017 (Figure 3.6, bottom). Hillslope weir conductivity remained fairly constant over the measurement period, ranging from 100 to 400 mS/m (Figure 3.6, bottom). In contrast, parking lot weir conductivity ranged from 100 to 3800 mS/m from October 2016 to May 2017 (Figure 3.6, bottom). On December 21, 2016, P3 rose to 1640 mS/m, and then peaked on February 1, 2017, at 3800 mS/m (Figure 3.6, bottom). Other sharp increases were at P2 in January and February 2017.

3.4.3 MONTHLY WATER BALANCE

Measured and estimated key hydrological inputs and outputs for a 0.6 ha constructed wetland showed seasonal trends (Figure 3.7, top). Hillslope spring GW is the largest water input for the potential wetland, ranging from 140 mm in June to 390 mm in March (Figure 3.7, top). Seasonally, hillslope GW input is greatest in the winter (January to March) and April to May in the spring. June through to August has lower GW amounts, increasing in the fall (September to December) (Figure 3.7, top).

Precipitation is a far smaller input to the potential wetland than hillslope GW, ranging from 38 mm in November to 92 mm in April (Figure 3.7, top). Hillslope GW input is three to four times greater than precipitation input (Figure 3.7, top). Precipitation follows the seasonal trend described earlier, with greater amounts occurring in spring and fall, and lower amounts in summer (Figure 3.7, top).

Evapotranspiration does not occur from December to March, when the average monthly temperature is below 0 ºC (Figure 3.7, top). As temperatures warm up in April and average monthly temperature surpasses 0 ºC, evapotranspiration begins, and increases until July to a maximum of 135 mm (Figure 3.7, top). In August, as temperatures begin to decrease, so does evapotranspiration, to a minimum amount in November of 13 mm (Figure 3.7, top). Evapotranspiration makes the biggest impact on storage in the hot summer (June to August) when groundwater and precipitation inputs are low (Figure 3.7, top).

Monthly wetland water storage predicted for the constructed wetland demonstrates a seasonal pattern (Figure 3.7, top). Groundwater inputs from the hillslope are the drivers of storage throughout the year, while evapotranspiration dampens storage potential in the summer (Figure 3.7, top). Winter and the beginning of spring (January to May) see the greatest storage potential, with an average storage of 380 mm (Figure 3.7, top). Summer

(June to August) has the lowest storage, an average of 97 mm (Figure 3.7, top). Finally, fall (September to December) has moderate storage, between spring/winter and summer, at an average of 239 mm (Figure 3.7, top).

The overall yearly wetland water storage is significant, at 3190 mm (Figure 3.7, bottom). Annual precipitation (810 mm) and evapotranspiration (610 mm) are similar, effectively cancelling out, leaving a surplus of the yearly hillslope GW spring input of 2989 mm over the year (Figure 3.7, bottom). These estimates are conservative, as they do not account for the additional groundwater present in the system measured at the parking lot outflow weirs (Figure 3.5), and do not take into account surface water movement.

Monthly water balance expressed proportionally (Figure 3.8) further demonstrates the contributing inputs and outputs each month. December to March have similar proportions of groundwater, precipitation, and storage (Figure 3.8). April, May, October, and November exhibit similar proportions with dominant groundwater input around three times the precipitation, and moderate evapotranspiration (Figure 3.8). June to September show a decreased proportion of groundwater input in relation to the increase in evapotranspiration in hot months, which is one to two times more than the proportion of contributing precipitation (Figure 3.8).

3.5 DISCUSSION

There is enough hillslope groundwater to result in a water surplus throughout the year for the proposed 0.6 ha constructed wetland. There are other potential groundwater sources that are around three times the groundwater being measured. This is likely due to the larger discharge area into the creek than what was measured at the four groundwater sources on the hillslopes. Wetland design must include topographic features that will allow for water to exit during months of excess in fall and winter, alongside the ability to retain water during the summer months of low water availability.

3.5.1 Trends in Precipitation and Groundwater

Hillslope spring groundwater flow and parking lot outflow at parking lot M exhibited rapid responses to rainfall events. Spring and fall bring increased precipitation and groundwater flow, while summer exhibits lower precipitation and in turn groundwater flow. Discharge from parking lot outflows were of greater magnitude than the hillslope springs. This was likely due to the larger area being drained by storm drains (construction maps were not available to calculate the area being drained). Storm drains on the hillslopes were several metres downstream of the hillslope weirs, which is likely a contributing area to the parking lot outflows. Additionally, the lack of evapotranspiration on the parking lot surface and direct routing of precipitation into the storm drains are other likely factors for increased parking lot outflow.

Electrical conductivities exhibited seasonal variability, most notably during the winter months. Water quality differences between the groundwater springs and parking lot outflows was most evident in late winter, when electrical conductivities were up to nine times higher than groundwater seepage on the hillslope. This is likely attributable to the use of road salt and its rapid movement during precipitation events through storm drains on the parking lot surface. The weir with the greatest conductivity (P3) had the second slowest flow rates (after P2), while P4 with the lowest conductivity had the highest flow rates. These patterns demonstrate the concentration of salt being higher in smaller volumes of water than large volumes of water. Rerouting this water through the riparian buffer, bioretention cells, or constructed wetlands would likely mitigate road salt impacts by attenuating its release (Denich et al., 2013). Plants can be selected based on their salt uptake and tolerance capacity (Karim and Mallik, 2008; Singh and Stasolla, 2016).

Temperatures of weir water showed strong seasonal variability. In winter (January/February), temperatures stay low, ranging from 0-6 ºC, then increase in mid-February. The water temperature increased more rapidly for the hillslope groundwater than the parking lot outflows. This is likely due to the lower heat capacity of soil than cement, meaning the soil temperature is able to warm up faster than the cement, thus increasing the water temperature faster on the hillslope than the water from outflows that moves through cement pipes. As temperatures begin to cool in October, however, the lower heat capacity of soil cools the groundwater faster than the cement pipes can cool the outflow water, causing the 3-5 ºC higher temperatures of outflow water throughout the cooling period. Due to the annual variability of hillslope groundwater temperatures and quick warming and cooling, the water can be described as originating from local flow (Winter, 1999). Local groundwater flow systems often recharge at an upland, and discharge into topographically lower lakes or wetlands (Winter, 1999). In contrast, regional groundwater flow is often associated with larger topographic differences between recharge and discharge areas, which can allow for more moderated seasonal thermal regimes (Winter, 1999).

3.5.2 Water Balance Considerations for Reclaimed Wetland Design

Under consideration is the reclamation of a 0.6 ha portion of lot M to be converted into a wetland (Figure 3.2). This wetland would likely resemble a marsh, with water inputs from precipitation and groundwater. To aid in the evaluation of water sourcing and management for this proposed wetland reclamation, a hypothetical assessment of water balance components is conducted in this section. Presumably, reclamation will block the storm drains at the base of the surrounding east and south hillslopes, resulting in a water balance dominated by groundwater spring inflows/outflows, precipitation, evapotranspiration, and surface outflows rather than fluxes through human-made drainage features.

Wetland Inputs

The largest input to the wetland would be groundwater (250 $mm/month_{average}$), followed by precipitation (67 $mm/month_{average}$). By utilizing all hillslope groundwater at the four springs, the wetland could store 265 mm/month on average, or up to 3190 mm/year (Figure 3.7). However, the groundwater input could change based on the manner in which the springs are connected to the wetland from the surrounding hillslopes. Groundwater flow rates at weir H1, H2, and H4 (Figure 3.3) are similar, ranging from 0.1 to 0.4 L/s, while H3 has lower rates of 0.02 to 0.15 L/s. (Figure 3.5, middle). H4 may be difficult to connect to the wetland as it is currently located behind McMaster's Facility Services compound (Figure 3.2).

The additional source of groundwater measured by parking lot outflows was not factored into the estimated water balance, but it could be utilized for the constructed wetland. This water amounted to an average total for 2016 and 2017 of 1.6 and 2.2 L/s respectively (Note the short and unmatched time scale represented by the data from September 2016 to July 2017). Taking an average of the two years, around 1.9 L/s or 764 mm/month is potentially available to the wetland. This represents around three times the measured hillslope groundwater that is known to be available. Further investigation into storm drain locations and drainage areas is required to determine where the additional groundwater originates, and whether the diversion to a wetland is feasible. Measured hillslope groundwater spring flow rates would be sufficient to sustain a wetland with positive storage year round. However, additional water would be more important during summer months where storage is low, from June to August, where only 100 to 150 mm of storage was predicted for the potential wetland (Figure 3.7, top). Backup groundwater sources would be especially important in dry years during extended periods without precipitation. Decreased water levels could have negative repercussions for wetland habitat and nutrient cycling (Causanarano, 2009; Miao, 2013).

Weir Measurement Error

Water flow rate accuracy may be impacted by microtopography or other variables surrounding the weirs. Groundwater measurement error is estimated to be up to 25 percent. However, even at the upper bounds of error, during the driest summer months, groundwater input would still be positive, as shown in Figure 3.7. Therefore, the feasibility of the wetland is not impacted.

Wetland Outputs

Once constructed, the primary water-loss mechanism in the reclaimed wetland will likely be evapotranspiration (50 mm/month$_{average}$). Given that the wetland is likely to support ponded water in some areas adjacent to Ancaster Creek, its hydrologic regime may be most similar to that of undisturbed marshes in southern Ontario, with actual evapotranspiration estimated as being close or equal to potential evapotranspiration (Price, 1994). Long-term evapotranspiration changes will vary based on temperature and vegetative cover (Lafleur, 1990). Depending on whether the wetland is connected to Ancaster Creek, an increase in evapotranspiration could cause water to flow more readily from the creek to the wetland, altering the chemistry of the wetland (Price, 1994). However, output from evapotranspiration will rarely exceed precipitation and groundwater inflows to the wetland, thus the wetland should be in a water surplus in most years.

From the predicted values for the 0.6 ha constructed wetland, the yearly total input of precipitation and groundwater (~3700 mm/year) is six times the evapotranspiration (610 mm/year). As such, outflows from the wetland will likely need to occur in order to ensure that the wetland does not store too much water and become a disconnected, open water body. For this reason, wetland design should incorporate features that remove excess water and function similarly to mechanisms in natural ecosystems, such as internal channels or sills. Of particular importance is ensuring that these incorporated features do not result in erosion. This can be facilitated by incorporating microtopography within the wetland as well as ensuring that there is not a steep slope/grade within the wetland.

Drainage should flow generally from the hillslope towards Ancaster Creek. Desired surface and groundwater flow rates out of the wetland, which were not measured in this study, should be taken into consideration for slope grading and creation of channels. Design should simulate various water balance storage outcomes for different outflow features, with the aim of storage surplus in the wetland, especially during summer months.

3.5.3 Subsurface material of constructed Wetland

Subsurface Soil

Characterizing subsurface materials is important before designing and constructing a wetland on a former parking lot. The consulting firm Terraprobe drilled three boreholes in the south section of parking lot M on October 13, 2016 (Figure 3.9, top), recording soil and groundwater observations and measurements to a 9 m depth below the surface (Figure 3.19, bottom). The parking lot subsurface layers include asphalt (50 mm), coarse (sand and gravel) fill (45 cm), and fine (silty clay) fill (0.9 m, 1.6 m, and 3.2 m at BH1, BH2, and BH3 respectively) (Terraprobe, 2017). Organic silt is found below the asphalt and fill, and is presumably the pre-1968 floodplain surface (Terraprobe, 2017). Trace roots and shells were found in the soft and grey organic silt, with a texture of 0% gravel, 4% sand, 76% silt, and 20% clay (sampled at 3.3 m depth in BH2) (Terraprobe, 2017). The organic silt permeability was estimated to be around 10^{-5} to 10^{-6} cm/second, a low rate likely reflective of the 48 years of compaction of the parking lot above (Terraprobe, 2017).

Soil Quality

Till samples from each borehole were compared to the Ministry of Environment and Climate Change (MOECC) standards to find out if the soil was suitable for reuse, and whether it should be considered waste. Two Boreholes were found to exceed the MOECC standards. Borehole 3's sample exceeded the standard for electrical conductivity (57 mS/m) at 63.5 mS/m, and Borehole 2's sample exceeded the standard for sodium adsorption ratio (2.40) at 3.380 (Terraprobe, 2017). High electrical conductivities likely are due to road salt leaching into cracks and the medians of the parking lot into the fill. Parking lot fill passed MOECC standards that designate the soil as waste; however, reuse may be unfavourable due to the high electrical conductivity and compressible nature of the soil (Terraprobe, 2017).

Water Table

The water table level is characterized by a sharp decrease in depth moving east from Ancaster Creek (Figure 3.9, bottom). The water table was found at 3m, 7.5m, and greater than 9m for BH1, 2, and 3 respectively (Figure 3.9; Terraprobe, 2017). Due to the unknown water table location for BH3, the water table line is dotted from BH2 to BH3 (Figure 3.9; Terraprobe, 2017). However, as boreholes were drilled and observations made within the same day, sufficient groundwater recharge may not have been possible, showing a much lower water table than the true position (Figure 3.9, bottom). However, it is likely that disconnecting hillslope groundwater from the former floodplain and re-routing into storm drains caused some amount of water table draw down.

3.5.4 Ancaster Creek ecosystem benefits

The wetland could be connected to Ancaster Creek, which runs adjacent to the parking lot, to provide ecosystem services for the creek. While the wetland does not need additional water input, the connection may provide other ecological benefits. Wetland outflow, predominantly groundwater, would provide a flux of dissolved ions and organic matter into Ancaster Creek (Klove et al., 2011). Slowing down hillslope groundwater increases the residence time for geochemical processes to occur in the wetland before entering the creek (Klove et al., 2011). Connecting the constructed wetland to the creek would allow movement of water and fish from the creek to the wetland, potentially creating wetland spawning habitat for species such as salmon or pike (Gray et al., 2002; Klove et al., 2011). Habitat may be made available to amphibians, reptiles, insects, invertebrates, and birds (Klove et al., 2011; Lehtinen, Galatowitsch, and Tester, 1999). Groundwater quality entering the creek may improve as it will be be filtered through marsh vegetation and by aquatic species (Moreno-Mateos and Comin, 2010). Other ecosystem benefits could include strengthening biodiversity and improving soil quality (Moreno-Mateos and Comin, 2010).

3.5.5 Climate change implications

Climatic changes could greatly influence hydrological fluxes in a wetland ecosystem (Erwin, 2009; Mortsch, 1998; Short et al., 2016). Increased air temperatures can cause increased precipitation; increased precipitation as rainfall instead of snow; reduced snow pack size and duration; and increased evapotranspiration rates (Mortsch, 1998). Great Lake water levels may also fluctuate in response to climate changes, in turn impacting wetland hydrological fluxes (Mortsch, 1998). Groundwater dominant wetlands are less impacted by climatic fluctuations due to the consistent flow of groundwater from underground aquifers (Winter, 2000; Short et al., 2016). However, local groundwater-fed wetlands, such as the proposed constructed wetland at lot M, could be impacted by climate changes as local flow is more influenced by temperature variability (Winter, 2000). Climate change impacts are variable for different landscape levels and habitat types, highlighting the need for tailored restoration priorities for each specific ecosystem (Erwin, 2009).

3.5.6 Further Research

This study proves the feasibility of constructing a wetland with the available water sources at lot M. Further research to compliment these findings are recommended to evaluate the:

1) Ecology and geochemistry of the area (Klove et al., 2011).

2) Parking lot outflow drainage areas.

3) Mechanics of connecting hillslope groundwater to a constructed wetland.

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3.7 TABLES

Table 3.1: Weir Installation and Measurement Dates from 2015-2017. H1 to H4 are the hillslope weirs, and P1 to P4 are the parking lot outflow weirs. Weir installation dates all occurred in 2015. Flow was measured for hillslope weirs from 2015-2017, while parking lot outflow weirs were only measured from 2016-2017.

<table>
<tr><th rowspan="2">Weir</th><th rowspan="2">Installation Date (First/Second)</th><th colspan="3">Flow Measured</th></tr>
<tr><th>2015</th><th>2016</th><th>2017</th></tr>
<tr><td>H1</td><td>June 19</td><td rowspan="4">June 19 – Dec 16</td><td rowspan="4">March 30 – Dec 21</td><td rowspan="8">Jan 13 – July 6</td></tr>
<tr><td>H2</td><td>June 26</td></tr>
<tr><td>H3</td><td>June 26</td></tr>
<tr><td>H4</td><td>July 2</td></tr>
<tr><td>P1</td><td>June 3/July 25</td><td rowspan="4">Not Working</td><td rowspan="4">Sept 1 – Dec 21</td></tr>
<tr><td>P2</td><td>June 3/July 29</td></tr>
<tr><td>P3</td><td>July 29</td></tr>
<tr><td>P4</td><td>June 3/July 28</td></tr>
</table>

Table 3.2: Precipitation values for parking Lot M from 2015 – 2017. Historical precipitation is an average from 1981-2010 from the Current Results website. Bolded values were obtained from historical data from an Environment Canada station at the Royal Botanical Gardens. Values in italics indicate months where snowfall occurred. Measurement time interval is the period when cumulative rainfall was measured at parking lot M, using five rain gauges spaced across the parking lot.

Month	**Historical Precipitation (mm)**	**Cumulative Precipitation (mm)**			**Measurement Time Interval**		
		2015	2016	2017	2015	2016	2017
January	57	***51.5***	***38***	***69.4***			
February	57	***36***	***81.5***	***71.7***			
March	64	***21.5***	***145.5***	***87.3***			
April	73	***85.5***	***36.3***	***154.9***			
May	85	**53.7**	47.6	145.25		April 20 – May 30	May 3 – June 8
June	73	118.4	19	103	June 1 – June 29	May 30 – June 29	June 8 - July 4
July	83	55.6	24.3	82.5	June 29 – July 31	June 29 – July 28	July 4 – July 27
August	90	56.9	36.9		July 31 – Aug 28	July 29 - Aug 31	
September	81	81.2	32.5		Aug 28 – Sept 23	Sept 1 – Sept 21	
October	72	109.2	77		Sept 17 – Oct 30	Sept 21- Oct26	
November	91	**34.1**	42.8		Oct 30 – Nov 13	Oct 26 – Nov 17	
December	72	***56.9***	***73***		Nov 13 – Dec 16		
TOTAL	647	760.5	279.98				

Table 3.3: Monthly hydrological fluxes for the proposed 0.6 ha constructed wetland at parking lot M, rounded to the nearest millimetre. Groundwater values are from the total average flow rates from the parking lot m hillslope springs from 2015-2017. Precipitation includes the lot M and RBG Station Data averaged from 2015-2017. Evapotranspiration values estimated from the Thornthwaite method using 1985-2015 air temperature and 2016 day length data from the Hamilton Airport. Storage is the addition of groundwater and precipitation, minus evapotranspiration.

Month	+ **Groundwater** (mm)	+ **Precipitation** (mm)	- **Evapotranspiration** (mm)	= **Storage** (mm)
January	362	53	0	415
February	218	63	0	281
March	391	85	0	476
April	314	92	35	371
May	394	82	77	400
June	141	80	112	109
July	170	54	135	90
August	163	47	119	91
September	182	57	80	158
October	242	93	39	296
November	229	38	13	254
December	182	65	0	247
Yearly Total	**2989**	**810**	**610**	**3190**

3.8 FIGURES

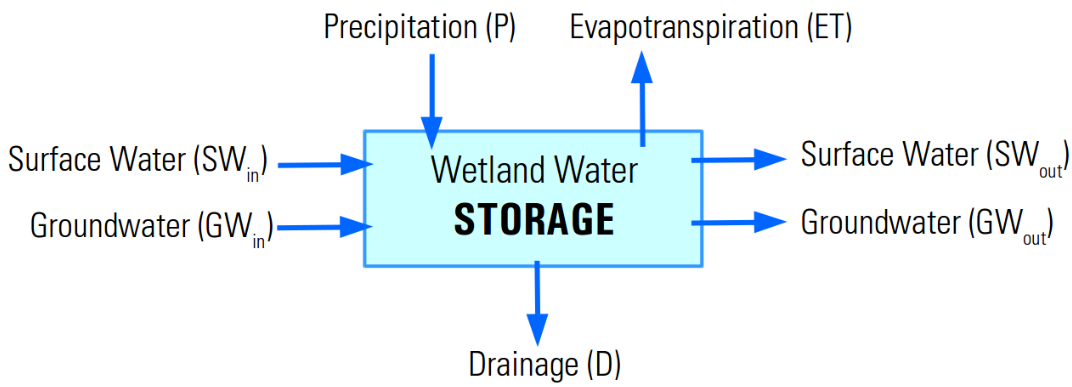


Figure 3.1: Water balance of a typical wetland displaying key inputs and outputs of the system. Inputs include surface water, groundwater, and precipitation. Outputs include evapotranspiration, surface water, groundwater, and drainage.

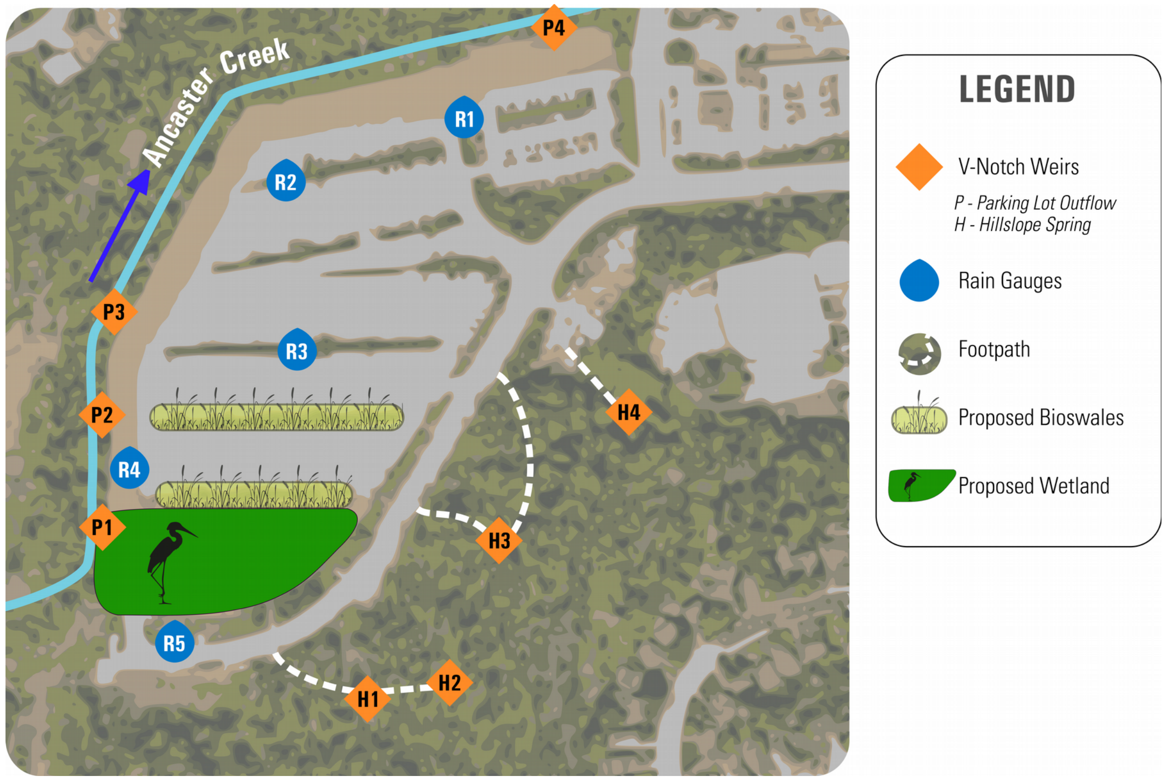


Figure 3.2: Parking lot M Map with hydrology instrumentation highlighted. Weirs and rain gauges were installed in June and July 2015. Four hillslope weirs are on the south and east hillslope of the parking lot. Four parking lot outflow weirs are found along Ancaster Creek on the west and north side of the parking lot. The proposed constructed wetland is shown as a green shape with a heron on top at the south section of the parking lot.

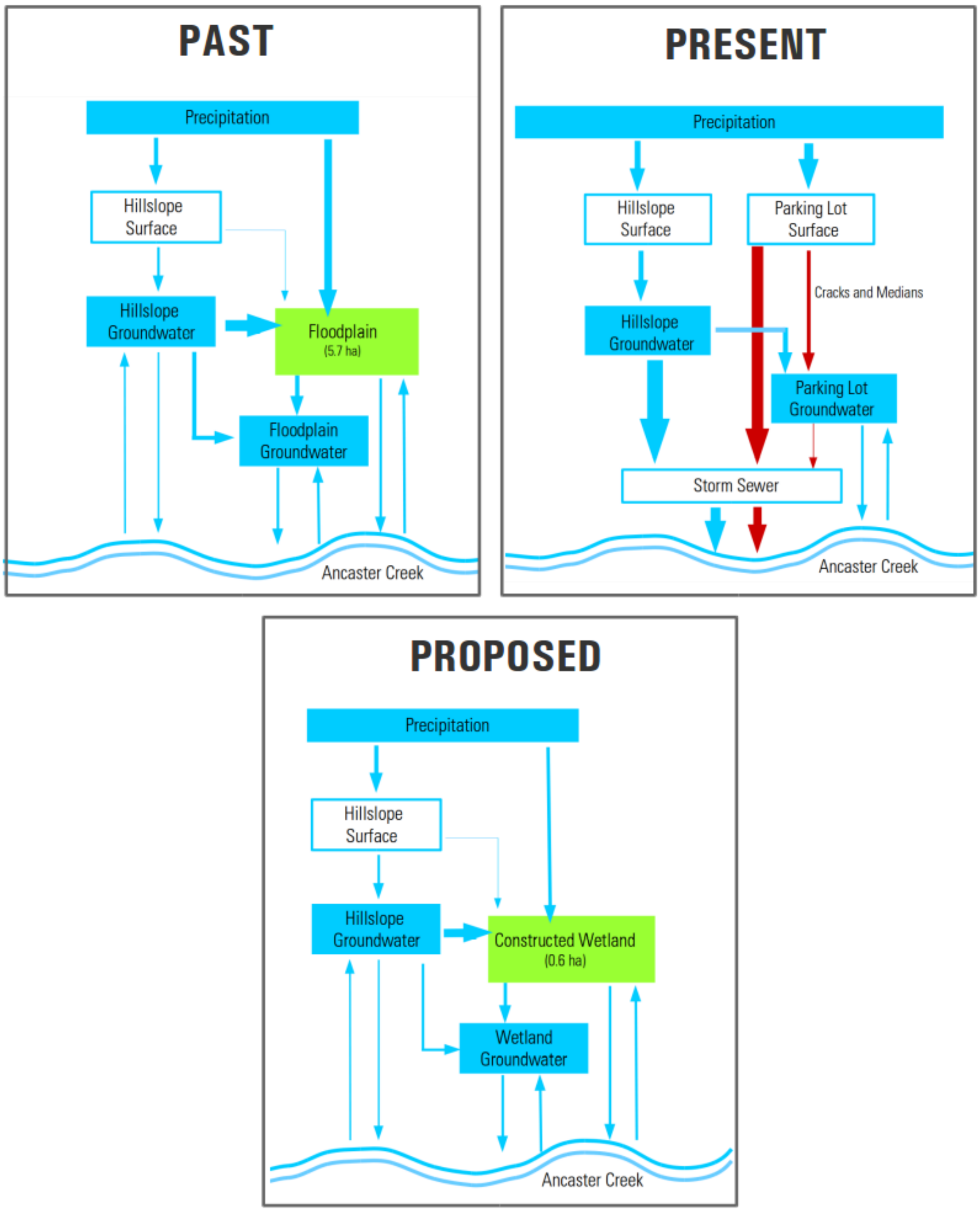


Figure 3.3: Past, Present, and Proposed Hydrology at parking lot M. Past occurs before 1968. Present hydrology is 1968 onward; red arrows indicate movement of stormwater with contaminants. Proposed hydrology includes a 0.6 ha constructed wetland. The remaining 5.1 ha of parking lot surface is not depicted in proposed diagram. Additionally, precipitation into Ancaster Creek and evapotranspiration, is not illustrated.

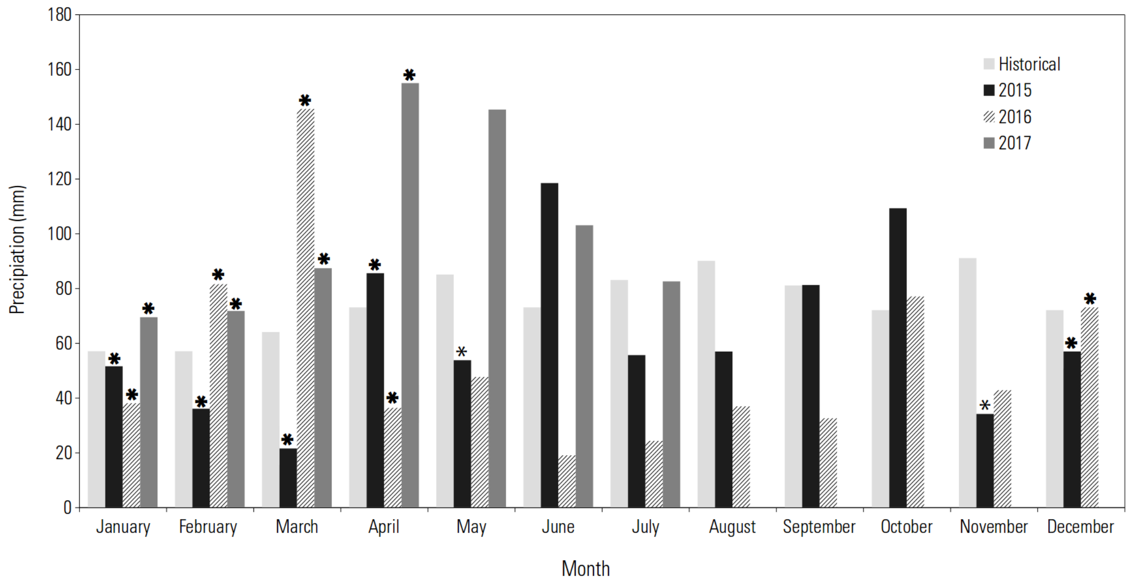


Figure 3.4: Monthly Precipitation at parking lot M. Historical Precipitation is averaged from 1981-2010, and obtained from the Current Results website. All data collected from 2015-2017 was rainfall from rain gauges at parking lot M except for bars with stars on top. Starred data comes from Environment Canada's historic data record from the Royal Botanical Gardens station. Bolded stars denote months where snow occurred, while unbolded stars denote months where only rainfall occurred.

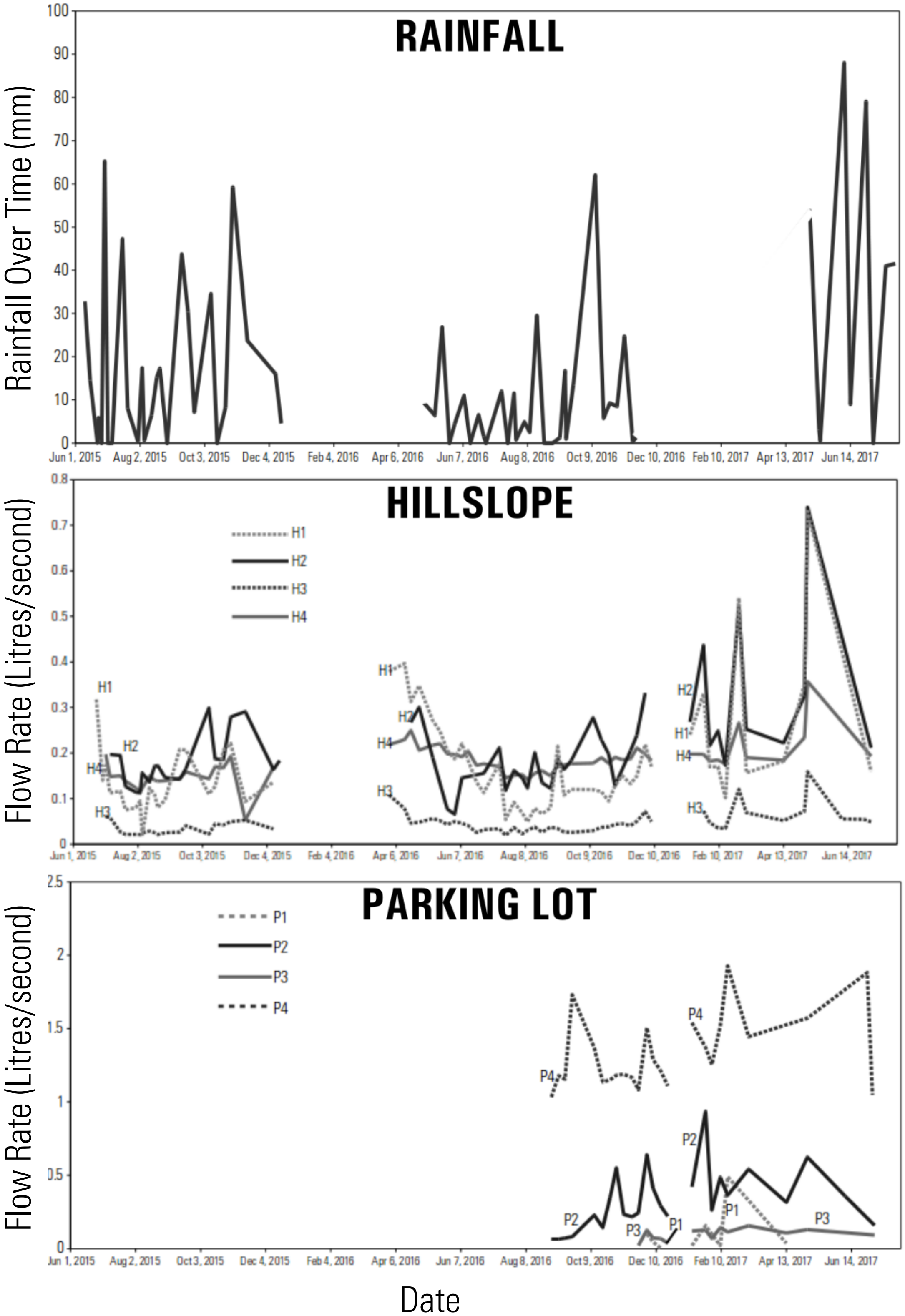
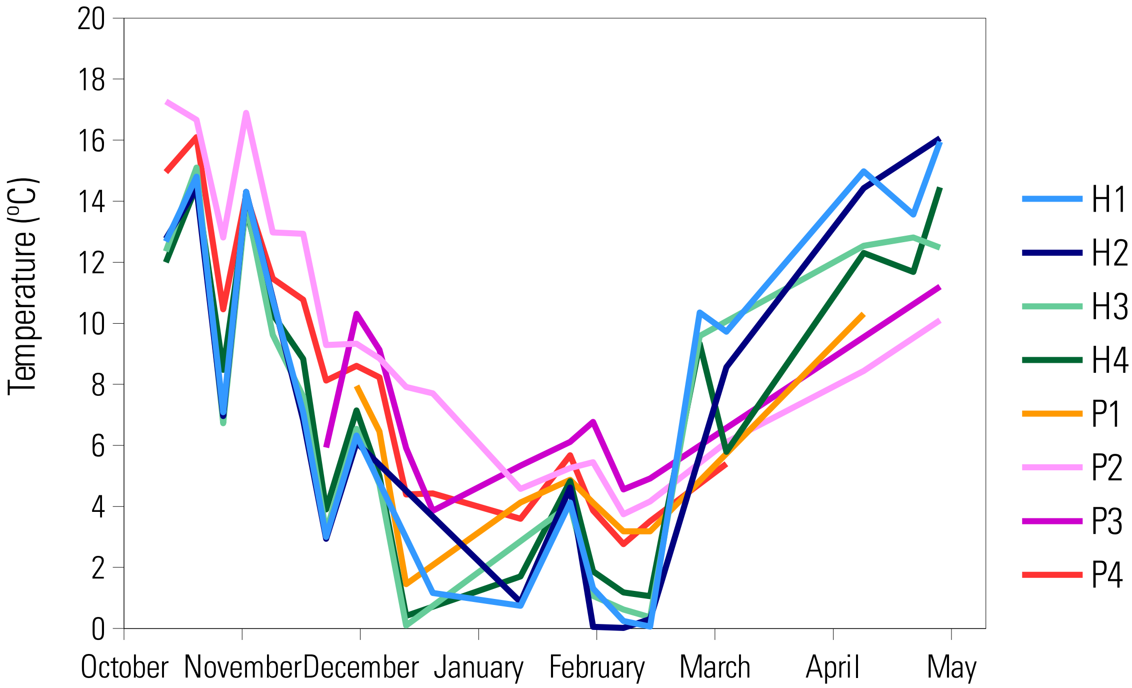

RAINFALL
Rainfall Over Time (mm)
HILLSLOPE
H1
H2
H3
H4
Flow Rate (Litres/second)
PARKING LOT
P1
P2
P3
P4
Flow Rate (Litres/second)
Jun 1, 2015
Aug 2, 2015
Oct 3, 2015
Dec 4, 2015
Feb 4, 2016
Apr 6, 2016
Jun 7, 2016
Aug 8, 2016
Oct 9, 2016
Dec 10, 2016
Feb 10, 2017
Apr 13, 2017
Jun 14, 2017
Date

Figure 3.5: Rainfall and Flow Rates from June 2015 to June 2017. Top: Rainfall over time from 2015-2017 collected from rain gauges at lot M. Rainfall measurements accounted for all rain since the last measurement. Middle: Hillslope groundwater spring flow rates from 2015-2017 collected from four weirs; H1, H2, H3, and H4. Bottom: Parking lot outflow rates from May 2015 to July 2017 collected from four weirs along Ancaster Creek; P1, P2, P3, and P4. Note change in scale magnitude from middle to bottom graph. The four weirs are denoted by solid or dotted lines indicated in the top left legends, and also labelled on the graph itself. All rainfall and flow rate measurements were discrete ranging from weekly to bi-weekly and can be identified at every peak or valley on the graph.


Temperature (°C)
20
18
16
14
12
10
8
6
4
2
0
October
November
December
January
February
March
April
May
H1
H2
H3
H4
P1
P2
P3
P4

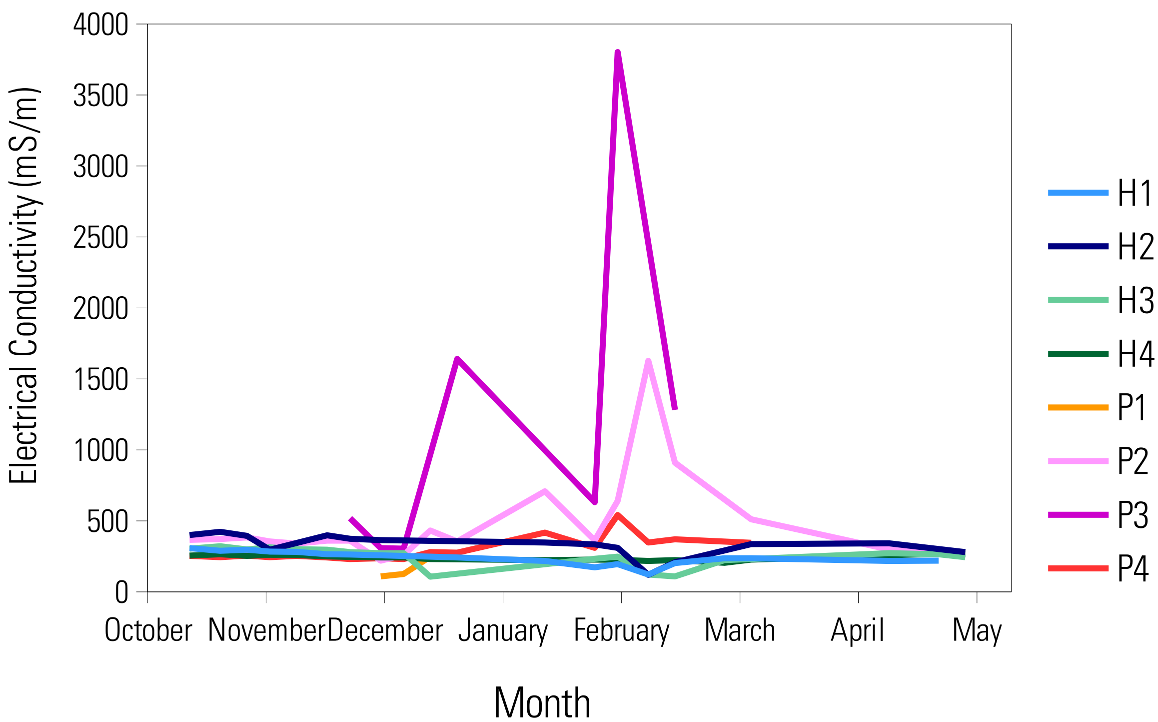
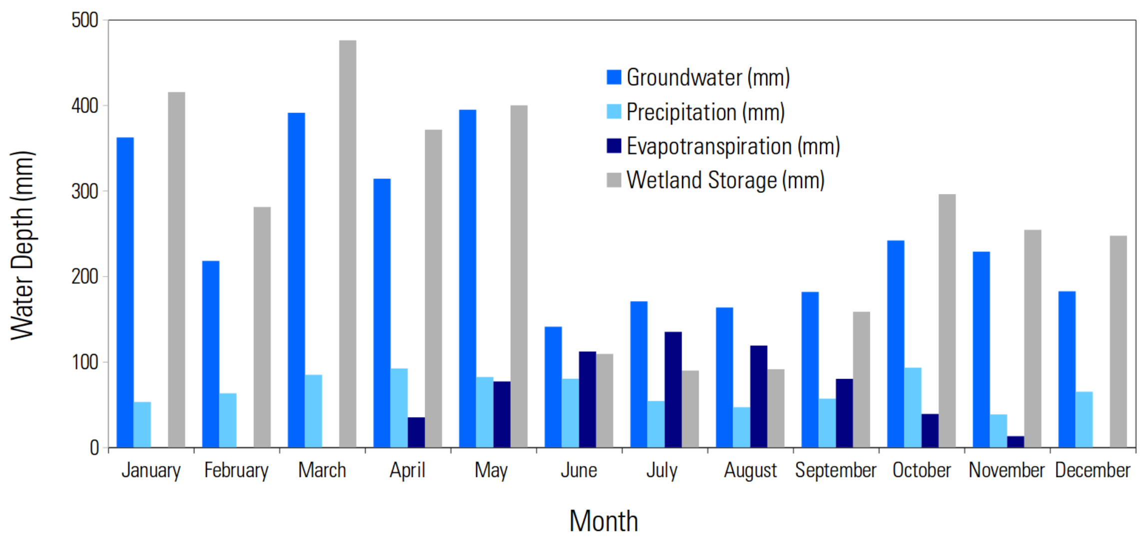

Electrical Conductivity (mS/m)
4000
3500
3000
2500
2000
1500
1000
500
0
October
November
December
January
February
March
April
May
H1
H2
H3
H4
P1
P2
P3
P4
Month

Figure 3.6: Temperature and Conductivity of water directly behind weirs from October 2016 to May 2017. Top: Water Temperature at hillslope spring and parking lot outflow weirs. Hillslope lines in shades of blue and green. Parking lot outflow lines in shades of red and orange. Bottom: Electrical Conductivity for hillslope spring and parking lot outflow weirs. All temperature and conductivity measurements were discrete ranging from weekly to bi-weekly and can be identified at every peak or valley on the graph.



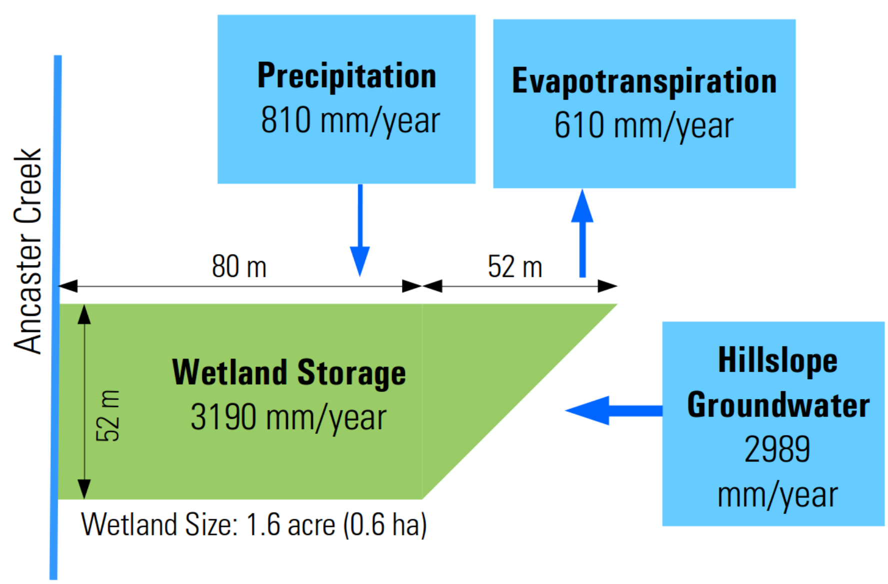


Figure 3.7: Inputs and outputs for a potential constructed wetland. Top: Monthly measurements and estimates of inputs and outputs. Bottom: Diagram of yearly measurements. Values rounded to the nearest millimetre. Groundwater values are from the total average flow rates from the hillslope springs from 2015-2017. Precipitation includes the lot M and RBG station data averaged from 2015-2017. Evapotranspiration values estimated using the Thornthwaite method and 1985-2015 air temperature and 2016 day length values from the Hamilton Airport. Storage is the addition of groundwater and precipitation, minus evapotranspiration.

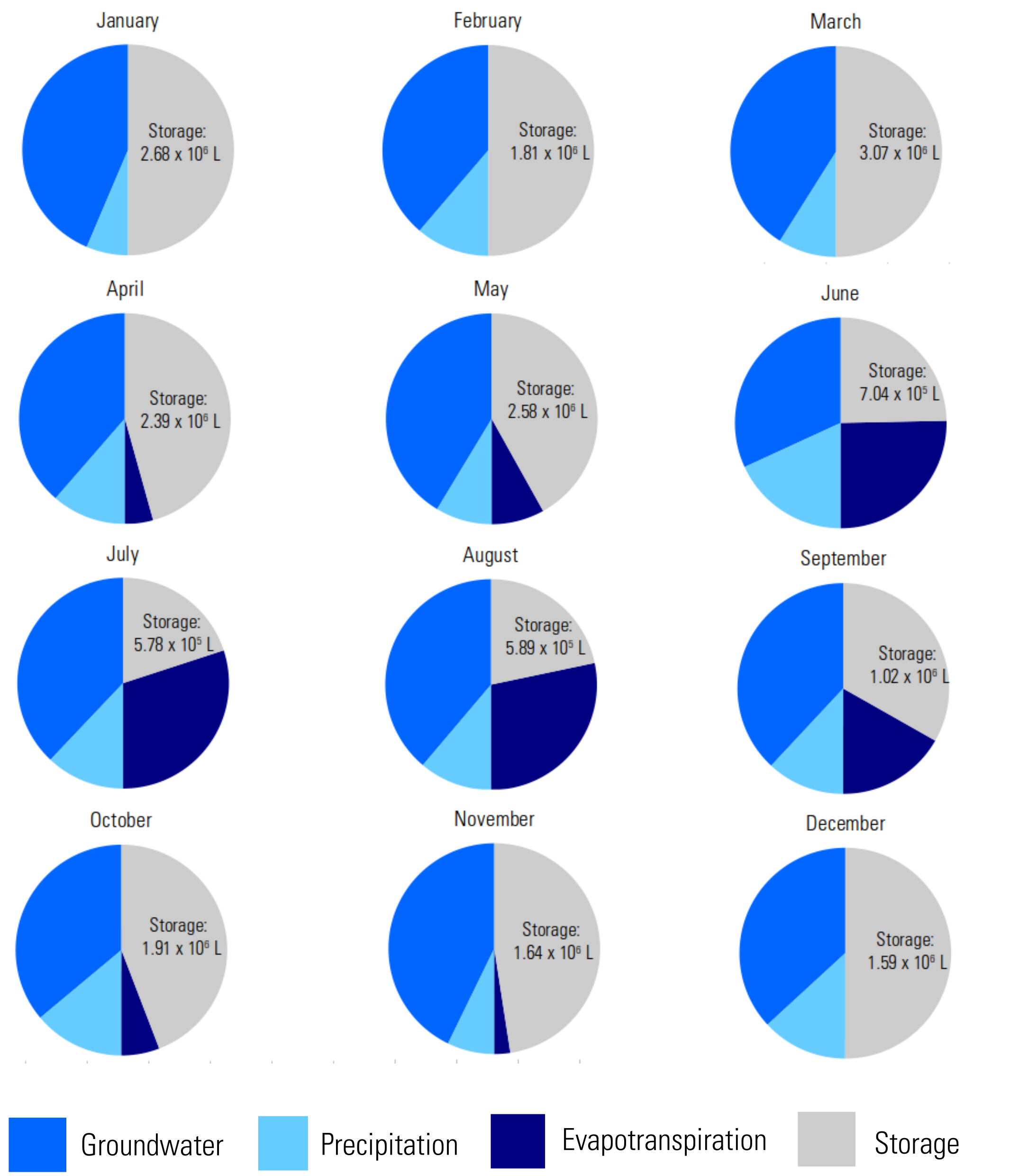


Figure 3.8: Monthly water balance components expressed proportionally for the potential constructed wetland. Storage volume in litres highlighted in the grey section. Groundwater values are from the total average flow rates from the hillslope springs from 2015-2017. Precipitation includes the lot M and RBG station data averaged from 2015-2017. Evapotranspiration values estimated using the Thornthwaite method and 1985-2015 air temperature and 2016 day length values from the Hamilton Airport. Storage is the addition of groundwater and precipitation, minus evapotranspiration.

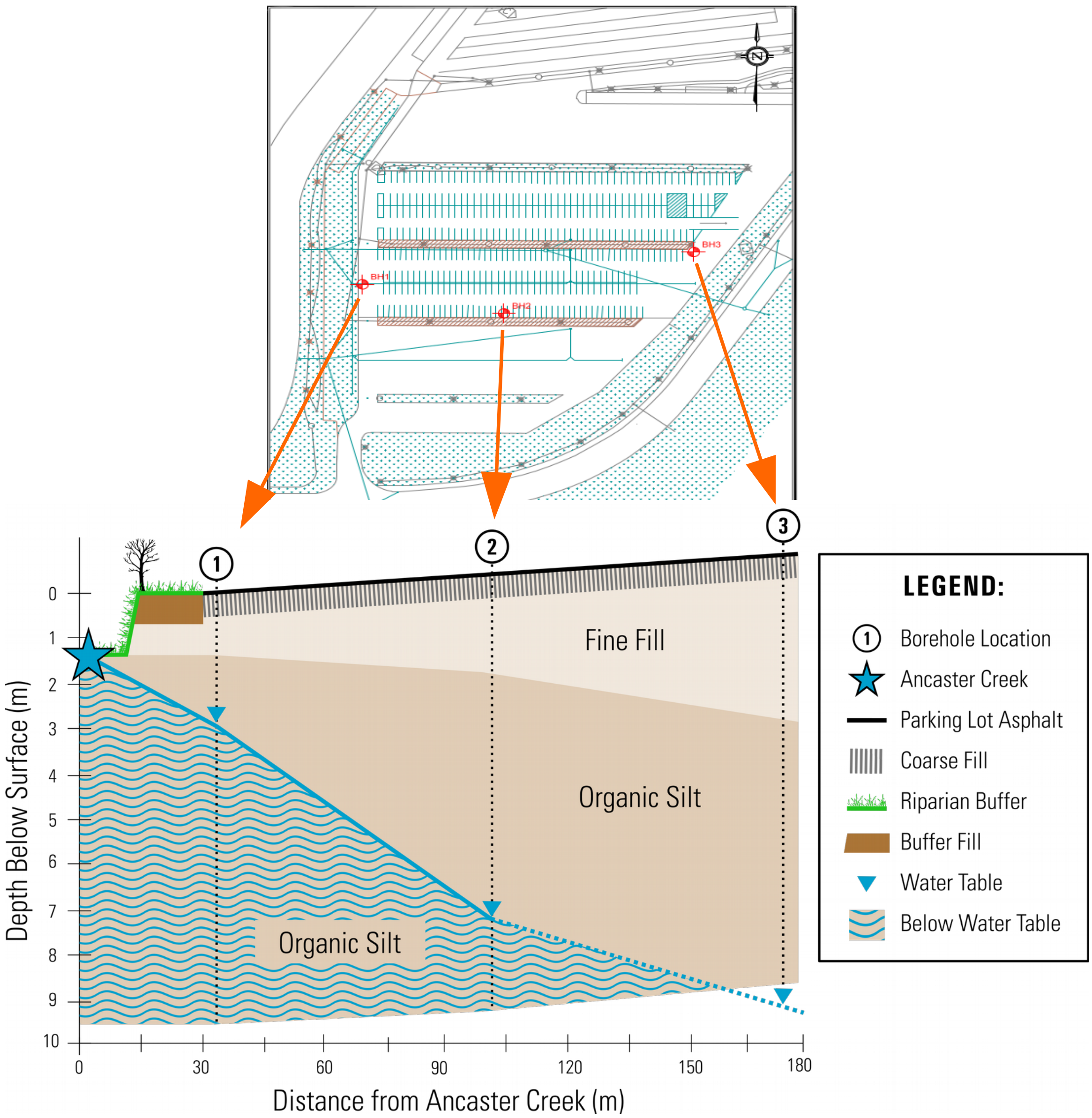


Figure 3.9: Subsurface hydrogeology of parking lot M. Top: Boreholes were dug in three locations by Terraprobe in the south section of lot M on October 13, 2016 for a proposed bioretention cell project. Bottom: Each borehole's vertical profile to a 9 metre depth is characterized. Horizontally, the diagram begins at Ancaster Creek and moves to around 180 metres away at Borehole 3. Vertical and horizontal scales differ. Solid blue line indicates known water table depth and dotted blue line indicates potential water table depth.

CHAPTER 4: CONCLUSION

This thesis has outlined the plant performance at a young restoration site and determined the feasibility for constructing an urban wetland. Chapter two evaluated plant performance in relation to soil salinity on a two-year-old, one-hectare, restored riparian buffer. A similar biomass of native and non-native plants was found, with greater species richness in non-native plants. Road salinity resulted in decreased performance (biomass) for most plants, except one native halophyte, sand spurrey (*Spergularia marina*). Introduction of non-native and invasive species poses a stress on the restoration that could become worse. Chapter three evaluated the feasibility of constructing a 0.6 hectare wetland on the parking lot using naturally occurring groundwater sources on the east and south hillslopes. The estimated water balance implies an excess of available water year round, proving that a wetland could be constructed and sustained naturally at lot M.

These two studies demonstrate how one restoration has performed and provide a rationale for the feasibility of creating another. Successful restorations need to move beyond this first step of establishing native species and ecosystem function (Shackelford et al., 2013). First, management and maintenance are required to enable a successful restoration. Second, engaging and involving the public in the process is imperative, especially in urban restoration projects.

Restoration management and maintenance can involve diverse activities such as controlling invasive species and re-planting certain areas (Nillson et al., 2016). As each site will have its own challenges, active and adaptive management is required (Trowbridge et al., 2017). Climate variability impact is a restoration constraint that is hard to predict and plan for (Harris et al., 2006; Trowbridge et al., 2017). For a meadow restoration, climate variability could allow non-native and invasive species to persist and spread at a faster rate (Richardson et al., 2007). For the wetland reclamation, increased air temperatures could result in increased precipitation, which could disrupt the wetland water balance (Mortsch, 1998). Ancaster Creek and Lake Ontario water levels could fluctuate in response to climate changes, which could impact wetland fluxes (Mortsch, 1998) and the movement of seeds into nearby restorations (Richardson et al., 2007). Understanding how restoration sites respond to management treatments can further the restoration ecology field and the progress of successful native habitat restoration (Clark et al., 2012).

Ecosystem restoration is a human approach that can assess and repair ecosystems that are perceived to be degraded or disturbed, making human values and involvement essential in restoration work (Shackelford et al, 2013; Standish et al, 2013). Restoration management work can be accomplished by scientists and land managers working alongside the community. Scientists should be able to convey complex information about ecosystem restoration in a way that is accessible and interesting for a wider public audience (Martin-Ortega et al., 2017). Communicating the benefits of natural spaces to people is a suitable starting point. Natural spaces in urban areas have been shown to improve quality of life through noise, temperature, and pollutant reduction (Saumel et al., 2016); physical activity opportunities (Owen et al., 2004); and reduced health costs (Coutts and Hahn, 2015). Beyond quantifiable advantages, natural settings can become aesthetic, personal, and cultural places for people (Korpela and Hartig, 1996). Once people understand the benefits natural spaces provide, the next step is to communicate the benefits that restoration projects have for local ecosystems.

Restoring ecosystems in urban areas requires merging the fields of Restoration Ecology and Green Infrastructure. At an urban parking lot site, this study found that road salting and non-native species affected the health of a meadow restoration, and that a wetland could be constructed using natural groundwater sources available. While understanding and evaluating the science of restoring ecosystems is important, it is critical that people are engaged and involved in the process. Collaboration and communication between scientists, engineers, biologists, planners, and the community are important for advancing urban sustainability through ecosystem restoration. Urban areas have the potential to move from networks of asphalt and concrete to networks of bioretention cells, permeable pavement, reclaimed wetlands, urban gardens, restored riparian buffers, green roofs, and parks; all of these benefitting both humans and nature.

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