ESTABLISHING A PROCESS FOR A WETLAND VEGETATION REHABILITATION AND MANAGEMENT PROGRAM FOCUSED ON REED CANARYGRASS: A PARKLAND MEWS CASE STUDY

by

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ABSTRACT

Wetland value is threatened by invasive plant species such as Reed Canarygrass (*Phalaris arundinacea*). Hence the research objectives of this project were to determine if reed canarygrass abundance has an effect on plant species diversity and assess the effectiveness of novel treatments on reed canarygrass control in a constructed wetland. Four treatments (mowing, herbicide, mowing plus herbicide, and a control) followed by broadcast seeding were applied to regulate growth of reed canarygrass. Principal components analysis, biodiversity measures, and ANOVA were used to identify community composition, quantify biodiversity values and identify treatment differences respectively. Results indicated differences in species composition between east and west blocks of the study site, reed canarygrass abundance appears to keep plant species diversity low, indigenous species were rare, and reed canarygrass was resistant to treatments.The results of this study are not surprising considering there is little evidence that treatments for reed canarygrass control are effective.

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CHAPTER 1

INTRODUCTION

1.1 Background

There are few places where the landscape has not been altered at some level by anthropogenic activity. Globalization, human lifestyles, population growth, technology, industrial development, agriculture, resource extraction, and waste disposal are in and of themselves considered to be anthropogenic activities that have contributed to the excessive exploitation of nature. Urban environments have been dramatically altered by anthropogenic activity while rural environments, although less so have not been immune (Freedman, 2007).

1.2 Importance of Study/Issue Statement

The term wetland includes a broad range of areas that consist either of saturated soils or areas that retain water at some point in a calendar year. Regardless, the interactions that occur within wetland systems are as complex as any aquatic or terrestrial ecosystem (Campbell and Ogden, 1999). Unfortunately, wetlands are often viewed by humans as an unsightly and unimportant landscape feature and many wetlands are now drained, altered or influenced by anthropogenic activity (Moore, 2008). As a result international efforts have been made to conserve or restore wetlands.

An International Treaty, the Ramsar Convention on Wetlands came into effect in 1975. The convention now has 159 signatories including Canada. The convention's mission is "the conservation and wise use of all wetlands through local and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world"

(The Ramsar Convention, 1971). Canada is not immune to wetland loss where the threat to wetland loss still exists as a result of agriculture, anthropogenic influence on water levels, industrial activity, and urban sprawl (Rubec and Hanson, 2009).

Canada contains approximately 25% of the world's wetlands covering a total area of 127 million hectares, the equivalent of 14% of Canada's total land area (Natural Resources Canada, 2009). Agriculture has accounted for 85% of Canada's wetland area decline since European settlement. This is the equivalent of 20 million hectares with the prairies experiencing a 71% decline in wetland area (Natural Resources Canada, 2009). Greater than 80% of wetland decline in urban areas is accounted for through urban sprawl and agricultural growth. The result is that only 0.2% of Canada's wetlands are situated within a 40 km radius of the 23 largest urban centers in the country which contain 55% of Canada's population. Furthermore, Manitoba wetlands cover 41% of the provincial land mass and account for 18% of Canadian wetlands (Natural Resources Canada, 2009). Nonetheless, there exists no level of government in Canada with the authority to create or mitigate an approach to retaining wetlands across all jurisdictions (Rubec and Hanson, 2009).

Wetlands play a three part role in the hydrologic cycle. They filter water prior to it seeping into the groundwater, recharge aquifers or flow into streams, and through resident plants transpire water back into the atmosphere and through evaporation. Additionally, wetlands act to mitigate floods by acting as storage basins for runoff, assisting in the trapping and settling of runoff sediment, filtering and transforming chemical contaminants and nutrients thus improving water quality, mitigate the effects of erosion and flushes of runoff, and trap sediment while also providing intrinsic value (Holechek et al., 2000; Libby et al., 2004). Prairie wetlands, consisting of both prairie potholes and massive lacustrine systems are some of the world's most productive ecosystems (Perry and Vanderklein, 1996; Murkin et al., 2000).

Consequently, wetlands are the ecotones responsible for linking aquatic and terrestrial ecosystems (Turner et al., 1995; Perry and Vanderklein, 1996; Holechek et al., 2000; Novotny, 2003; Libby et al., 2004) and thus wetland biological, physical, and chemical processes are highly integrated. Their location in the landscape is dictated by landform, geology, and hydrology with natural wetlands occurring where water balance and physiography act to support the retention of water. This usually occurs in low areas, locations where flooding occurs, or in the case of groundwater discharge, on uplands or slopes (Maxwell, et al., 1995). Wetlands influence the immediate ecosystem to a degree in which the soil, flora, and fauna differ from those nearby and are thus considered to be some the most diverse types of ecosystems (Holechek et al., 2000; Libby et al., 2004; Naiman et al., 2005; Moore, 2008). It is not uncommon for even small wetlands to play host to a wide range of vegetation types including sedges, rushes, woody shrubs, herbaceous grasses, mosses, and cattails. When combined with the aquatic environment, these provide habitat for an extensive and eclectic collection of aquatic and or terrestrial species including birds, snails, amphibians, reptiles, fish, and insects (Holechek et al., 2000; Libby et al., 2004). As a result, wetlands provide crucial ecological services not only to plants and animals, but to humans as well (Holechek et al., 2000; Libby et al., 2004). This includes plant species biodiversity which can be enhanced with the appropriate wetland vegetation (Hooper et al., 2005). However, wetlands and their associated vegetation are not only negatively affected by the obvious influences of man, but more subtle ones as well.

Critically, numerous perturbations including increased nutrient loading, sedimentation, and water level fluctuations as a result of anthropogenic activity significantly influence biological interactions in wetlands and the suite of vegetation of which they are comprised. As a result, the influence of invasive plant and animal species are becoming more prominent (Tori et al., 2002). In fact, O'Connell (2002) identifies the introduction of alien species as one of the five most influential anthropogenic influences on wetland modification. The suite of species found in wetlands is most commonly dictated by hydrologic fluctuations (van der Valk, 2005; Aronson and Galatowitsch, 2008) and consequently "active management" is required to overcome the negative influences associated with invasive vegetation (Seabloom and van der Valk, 2003).

Those ecosystems in close proximity to urban centers, and likely most influenced by anthropogenic activity appear to possess a significant need for ecosystem retention, restoration, and management to maintain biodiversity and garner the greatest ecosystem services. It is suggested here that nature interpretive centers such as Parkland Mews, located in Winnipeg, Manitoba, can serve as a conduit to bridge the gap that has left a deep chasm between humans and nature and between urban and rural environments. Retaining and restoring ecosystems within and in close proximity to urban environments provides not only the benefit of retaining these ecosystems and their associated environmental goods and services, but also provides a convenient opportunity to expose, educate, and connect urban dwellers to the ecosystems on which they rely for those same goods and services. Yet, constructed wetlands are just that and don't necessarily mimic the traits and functions of natural wetlands.

1.3 Constructed Wetlands

Constructed wetlands, including those near urban centers are often developed in an attempt to mitigate anthropogenic influences on water quality and quantity and restore ecosystem functions (Boers et al., 2007; Ballantine et al., 2012). Yet urban wetlands typically possess lower species richness (Boers et al., 2007). Furthermore, the establishment and persistence of desirable native species in constructed wetlands is challenged by the prevalence and dominance of invasive species (Mulhouse and Galatowitsch, 2003).

Leck and Leck (2005) found that invasive species were dominant in constructed wetland soils and the vegetation community while Boers et al. (2007) and Garde et al. (2004) suggest that controlling invasive species followed by seeding with native species, initially increased diversity but did not mitigate re-invasion and consequently repeated management techniques (particularly hydrological and vegetation control) aimed at the control of invasive species and establishment of indigenous species is essential (Mulhouse and Galatowitsch, 2003; Garde et al., 2004; Boers et al., 2007, Hsu et al., 2007). Furthermore, tall emergent vegetation tends to exclude other species thus lowering plant diversity (Weisner and Thiere, 2010) as may be the case with reed canarygrass, a tall and dominant emergent species. Nonetheless management techniques aimed at invasive and introduced species control may prove to be futile.

Ballantine et al. (2012) suggest that the plant community composition and treatment effectiveness are most influenced by initial site conditions. This is further supported by Galatowitsch and van der Valk (1996) who found that propagule availability and environmental conditions are what influences wetland species establishment and survival and Hausman et al., (2007) who suggest that water table location is the factor most attributed to plant community composition. Furthermore, location in the landscape also acts as a pre-cursor to anticipated vegetation guilds in constructed wetlands. As such Alsfeld et al. (2010) suggest that proximity to forest edge has a significant influence on wetland vegetation diversity and richness, as does proximity to agricultural areas and drainage ditches (Luckeydoo, 2006), and urban centers **(**Matthews et al., 2009). As a result native species populations tend to be low while non-native and invasive species dominate. However, although species composition may be largely influenced by location, it should not eliminate the need to explore options aimed at managing invasive species and increasing plant diversity associated with constructed wetlands.

In the case of Parkland Mews, this involves the rehabilitation and management of vegetation within a constructed wetland ecosystem with a focus on the invasive species reed canarygrass (*Phalaris arundinacea*).

1.4 Research Objectives

Prolonged inundation, herbicide, mowing, and burning treatments have been demonstrated to be the most successful methods to control reed canarygrass (Weber, 2005; Kaufman and Kaufman, 2007). Few studies examine the effectiveness of broadcast seeding combined with a variety of treatment applications such as burning, herbicide application, and mowing to increase plant species diversity and for the control of reed canarygrass in constructed wetlands. Consequently, the objectives of this paper are;

1. To determine if reed canarygrass abundance has an effect on plant species diversity by;
    a. Revealing trends in species abundance according to location on site and treatment effect.
    b. Identifying if a select number of species tend to dominate the site through the use of dominance diversity curves, and measures of diversity, evenness and richness.
2. To assess the effectiveness of four treatments on reed canarygrass control where it has been intentionally seeded in a constructed wetland, by;
    a. Identifying tendencies in reed canarygrass abundance within treatments

b. Revealing if there are significant differences between treatments at the treatment level, the block level and the treatment and block level.

1.5 Wetland Vegetation Dynamics and Succession

The predictability or direction of wetland vegetation succession is largely influenced by climate and although climate has a major impact on the vegetation suite contained within a geographic region, wetland vegetation is further influenced by the cyclical nature of water-level changes particularly in prairie pothole and lacustrine wetlands (van der Valk and Davis, 1978; Moore, 2008). Water chemistry and the influence of topography on water movement and accumulation patterns further contribute to wetland succession processes. Still, wetland succession patterns are a function of two primary processes. Outside processes, referred to as allogenic processes include the contribution of materials, such as mineral and organic matter that are transported by upstream flows and deposited in the wetland from outside the wetland boundary (Moore, 2008). The nature of these materials varies with the type, shape and size of the basin in which the wetland is positioned. Autogenic, or internal processes such as vegetation slowing the flow of water and causing sedimentation or the contribution of detritus to the formation of organic matter also influence wetland succession patterns (Moore, 2008). Although these patterns can be complex and stochastic (Moore, 2008), wetland ecologists acknowledge that there are general wetland succession patterns.

Water level fluctuations have a significant impact on the primary and secondary production, vegetation mosaic, animal inhabitants, and nutrient cycling associated with wetlands. These fluctuations in water level are often identified as the wet-dry cycle which is integral to prairie wetland productivity (Murkin et al. 2000). Importantly, van der Valk and Davis (1978) identified four stages of wetland vegetation succession as a result of water level fluctuations under ‘normal’ conditions. These are classified as; the dry marsh stage when the wetland basin is void of water; regenerating stage when the marsh has re-flooded and emergent species are spreading vegetatively; the degenerating stage when emergent populations begin to decline; and the lake stage where high water levels largely eliminate emergent species;. This cycle is neither distinct nor consistent in its process and is subject to change as a result of stochastic events (van der Valk, 2000). Thus, as a result of water level fluctuations zonation patterns of vegetation tend to develop.

van der Valk et al. (1988), identified four primary means of by which wetland vegetation zonation patterns develop. One, “the differential distribution of seeds along the elevation gradient; two the differential recruitment of species along the elevation gradient, three, the differential survival of seedlings and adults along the elevation gradient during the drawdown, and four, the differential survival of adults after re-flooding along the elevation (no water depth) gradient.” Furthermore, numerous factors including; disease, herbivory, competition and extreme environmental conditions, can all act to influence both seedling and adult survival. Seed dispersal patterns have little to no effect on the evolution of vegetation zones in most prairie wetlands with fluctuating water levels, where for the most part all species tend to be evenly distributed across various elevation gradients (van der Valk, 2000). However, germination of various species along environmental gradients appears to be correlated with the ampleness of seeds at that elevation, combined with soil moisture, temperature and salinity. With regard to seedling survival during a drawdown, seedling mortality appears to be a function of decreasing soil moisture. Lastly, the re-flooding stage appears to have only a minor impact on emergent species distribution along the environmental gradient (van der Valk et al., 1988). Thus, it is suggested that because of the stochastic nature of environmental perturbations, that it is a collection of mechanisms that ultimately determines the zonation pattern associated with the development and formation of wetland vegetation zonation patterns. The result being that a particular species is not guaranteed to be found associated with a particular water depth gradient either within or between wetlands (van der Valk, 2000). Importantly, there are a number of wetland vegetation zones that can be broadly classified as aquatic, riparian and upland.

1.6 Wetland Vegetation Zones

Riparian zones, the focus area of this project, are located at the interface between aquatic and terrestrial ecosystems such as the fringes of streams, wetlands, rivers, and lakes (Holecheck, et al., 2000; Naiman et al., 1993). The Canadian Federal Department of Fisheries and Oceans refers to riparian areas as the “vital zone” as they are some of the most productive ecosystems on the Canadian Prairies Fisheries and Oceans (Government of Canada, nd). Riparian vegetation provides a number of ecological functions that include stabilization of banks via the plants root systems, flood mitigation by slowing the flow of water, creation and diversification of habitats, a source or organic matter from senescing vegetation, and filtering of contaminants by absorbing, storing and cycling nutrients (Leveque and Mounolou, 2003).

Resident vegetation in riparian zones is generally not drought tolerant but is tolerant of moist soils and temporary flooding (Naiman et al., 1993). Moreover, the dense growth associated with these zones provides a network of protective corridors in which wildlife reside, feed, and travel while also acting to slow runoff (Naiman et al., 1993; Holecheck, et al., 2000). Accordingly, there is an increasing propensity to restore the vegetation in these ecosystems (Naiman et al., 1993; Holecheck, et al., 2000).

1.7 Wetland Value and Plant Species Biodiversity

Of the listed endangered species in the United States, many are dependent on wetland resources for their survival with the consequences of the extirpation of a single species near impossible for ecologists to predict (Niering, 1998). In fact, in the United States it has been estimated that wetlands are critical to the survival of 46% of endangered species whose reduction is likely to have a subsequent negative impact on biodiversity (Boylan and Maclean, 1997) which is typically rich in wetland environments (Cunningham and Cunnigham, 2005). The most immeasurable value of wetlands exists in the diversity of flora and fauna that inhabit them and consequently their loss and degradation is a threat to retaining the biodiversity for which they are recognized (Moore, 2008). According to Raedeke (1989) nearly 70% of vertebrates use riparian corridors in some noteworthy way. Moreover, Naiman et al. (1993) found that greater than 80% of riparian zones in Europe and North America have been lost in the last 200 years. Nonetheless, biodiversity appears to be in peril in many wetland ecosystems largely as a result of anthropogenic activity. One such threat is the introduction and persistence of invasive species which Perry and Vanderkein (1993) identify as a significant biological change induced by anthropogenic activity. Thus, it would seem sensible to link the retention, restoration and management of these ecologically diverse ecosystems through vegetation biodiversity retention and enhancement.

1.8 Invasive Species

The International Union for Conservation of Nature; Invasive Species Specialist Group (2011) has identified that approximately one fifth of the world's plant species are threatened largely as a result of anthropogenic activity that is mostly related to the conversion of land for agricultural activities such as crop and livestock production (International Union for Conservation of Nature, 2011). Moreover, global anthropogenic activity and interaction that either purposefully introduces species to new environments with the intent of utilizing the species, or through unintentional introduction as the result of an activity such as transportation of goods (Kaufman and Kaufman, 2007) exacerbates the spread of invasive species (International Union for Conservation of Nature, 2011) posing a critical threat to the retention of biodiversity (Weber, 2005).

Although globally a number of species have been introduced to new regions most introduced or alien plant species do not become invasive (Elton, 1958; Bhowmik, 2005; Inderjit et al., 2005; Wetzel, 2005), but it is those that have been successful at expanding their ranges that are of greatest concern (Elton, 1958). The introduction of alien plant species to North America had its beginnings in colonial times, but the real influx occurred as a result of modern transportation, the globalization of trade, and immigration with most being introduced to Canada between 1800-1900 (Weber, 2003; Bhowmik, 2005; Kaufman and Kaufman, 2007; Canadian Food Inspection Agency, 2008).

The estimated cumulative cost of all invasive plant species to the Canadian economy is estimated at 2.2 billion annually and is expected to increase (Canadian Food Inspection Agency, 2008). This economic value is largely associated with the impact of invasive species on the agricultural sector. However this value would be substantially greater if environmental impacts were economically quantified. Indirect economic impacts include such things as a reduction in ecological goods and services which are more difficult to accurately quantify (Canadian Food Inspection Agency, 2008).

Species that overtake and dominate ecosystems and impact the economy, society, human health and biodiversity are of greatest concern, and thus categorized as invasive species (Hobbes and Huenneke, 1992; Woods, 1997; Weber, 2003; Schooler et al., 2006; Kaufmann, 2007; Canadian Food Inspection Agency, 2008). The result is that invasive species often thrive in their new environment in the absence of predators, diseases or limited resources thus overtaking and dominating indigenous species (Kaufman and Kaufman, 2007).

Invasive species have been implicated as a factor in the threatened status of 44 of Canada's species at risk (Canadian Food Inspection Agency, 2008). In hindsight, it is likely that biological invasions as a single act have had a greater influence on the world's biota than more familiar aspects of environmental change such as climate change as a result of rising CO_2 levels, and decreasing ozone layers (Bhowmik, 2005).

Yet in spite of regulations and the advent of the environmental movement of the 1970's, innumerable introduced species, including a host of vegetation species, have been and continue to be introduced to North America, both intentionally and accidentally (Myers and Bazely, 2003; Weber, 2003; Bhowmik, 2005; Wetzel, 2005; Kaufman and Kaufman, 2007; Canadian Food Inspection Agency, 2008). Nonetheless, most ecologists and natural resource managers recognize the potential threat that invasive species have on not only on the economy but biological diversity and conservation management, as their invasion interferes with the heterogeneous natural environment (Elton, 1958; Weber, 2003; Bhowmik, 2005; Wetzel, 2005).

Introduced wetland plant species, of which reed canarygrass may be one, are considered to be among the most dominant invaders of natural ecosystems. The term ‘may’ is used because of the difficulties associated with definitively identifying whether a species is native or exotic species (Lavoie and Dufresne, 2005). In these instances species are referred to as cryptogenic. Although it may be argued that reed canarygrass is a cryptogenic species (Lavoie and Dufresne, 2005) for the purposes of this study it is considered an invasive species.

1.9 Reed canarygrass

Believed to be first introduced to North America in 1830 for use as a hay and forage grass in New England, reed canarygrass (RCG) (*Phalaris arundinacea* L.) is a member of the Poaceae (Grass) family (Lavergne and Molofsky, 2007) and is classified as a rhizome geophyte (Ingrouille and Eddie, 2006). Although reed canarygrass appears to be native to some regions of North America, evidence strongly suggests that there were at least two, and possibly multiple, discrete introductions of strains indigenous to southern and central Europe (Lavergne and Molofsky, 2007). As a result, of these introductions, multiple recombinations have generated new genotypes in the North American populations. In fact, there exists as many as 210 genotypes in North America that results in a broad range of phenotypic differences, thus explaining the invasive nature of this species in North America and not in Europe (Lavergne and Molofsky, 2007). It is generally suggested that the invasive form of reed canarygrass is a hybrid of native populations and European species of an exotic nature (Galatowitsch et al., 1999). Specifically, reed canarygrass is an aggressive specimen that is not indigenous to Manitoba. It reproduces by rhizomes making it difficult to manage and possesses a hardiness zone tolerance of “0” making it tolerant of Manitoba winters (Lahring, 2003).

Reed canarygrass is a perennial which initiates growth very early in the spring (Hubbard and Nicholson, 1968; Kaufman and Kaufman, 2007) adding to its competitive nature. It produces sturdy stems, 0.3 to 1.8 m in height, from long underground stems (rhizomes) while the leaves are flat and up to 1.9cm wide with a horizontal orientation. This horizontal arrangement contributes to its ability to absorb more light and create shading for plants that lie adjacent to and underneath it (Kaufman and Kaufman, 2007). It habituates areas across both Canada and the United States largely in areas of wet soils void of standing water, making it a common resident of shallow marshes, wet meadows, wetlands, riparian habitats, and drainage ditches which does not survive inundation unless the stand is well established. It can also persist any upland area typically receiving more than 500mm of annual precipitation (Looman, 1983), and has been shown to inhibit the germination and establishment of *Lythrum salicaria* (Myers and Bazely, 2003).

Although a prolific seed producer, it spreads both via seed and through underground stems and can alter the hydrology of an area by trapping silt while outcompeting indigenous species through its dense monospecific colonization, particularly in situations of high soil or water nitrogen levels. Although there is no literature that currently demonstrates a relationship between phosphorus and reed canarygrass abundance, its affinity to nitrogen enrichment in wetlands is well documented (Wilson and Gerry, 1995; Green and Galatowitsch, 2002; Kercher and Zedler, 2004; Martina and von Ende, 2008). Moreover, it prolific seed production and extensive and resilient underground stems make its control very difficult (Kaufman and Kaufman, 2007) and thus the importance that is placed on establishing native species vegetation early in the restoration process (Aronson and Galatowitsch, 2008). Furthermore it emphasizes the urgency of identifying reed canarygrass control measures that are effective for wetland areas.

A number of approaches to reed canarygrass control have demonstrated some success. Repeated cutting is reported to eliminate reed canarygrass and prolonged water inundation, or herbicide applications have demonstrated effective management (Kaufman and Kaufman, 2007). Furthermore, the planting of native grasses can reduce the recolonization of *Phalaris arundinacea* while reed canarygrass seedlings are usually effectively controlled by non-selective herbicides such as glyphosate (Weber, 2003).

Additionally reed canarygrass possesses exceptional competitive abilities. As Tilman (1988) points out, as plant height increases so too does the plants ability to capture light and furthermore, plants with a more extensive root system have the capacity to obtain more nutrients, particularly those that are limiting. Following the growth of these two primary components, and the associated net energy produced, plant energy allocation is then often, but not always focused on reproduction. Furthermore, in resource rich environments those species which allocate energy to leaf production seem to thrive whereas in resource limited environments those that allocate more toward roots and stems are generally favored (Tilman, 1988). Above-ground enhancement by reed canarygrass in nutrient rich circumstances allows the plant to absorb more light, while in nutrient poor situations it increases below ground biomass seeking additional nutrients. Moreover, due to its clonal nature, reed canarygrass ramets also have the potential to establish in conditions of reduced light through subsidies from the parent plant, particularly if the parent plant resides in an un-shaded environment (Tilman, 1998; Maurer and Zedler, 2002).

In the case of Parkland Mews reed canarygrass was seeded as a means of bank stabilization and erosion control, but unfortunately was showing a strong prevalence in the riparian zone of the study site. Consequently, reed canarygrass control and management have been identified as the number one priority prior to successfully establishing a diverse cover of indigenous vegetation adjacent to the Parkland Mews wetland cells. Accordingly, this study will seek to identify control measures that increase native taxa populations within practical financial means, while considering environmental integrity, for example, the option of not using herbicides as a management tool.

1.10 A review of reed canarygrass management

In a project that took place on 37 restored prairie pothole wetlands in south-eastern South Dakota, southern Minnesota, and northern Iowa over the course of 19 years, Aronson and Galatowitsch (2008) report that the greatest diversity of wetland species (14.4 species/year) occurred during the first twelve years following re-flooding, but subsequently declined to 1.6 species/year. Water level fluctuations were the major limiting factor to maintaining species richness with seasonal and non-permanent wetlands being the most unstable in comparison to semi-permanent wetlands. Unfortunately it appears as though the long term success of such restorations is also in peril as a result of dominance by invasive species such as reed canarygrass (*Phalaris arundinacea)* and Common Cattail *(Typha angustifolia x glauca).*

Shaw (2000) indicates that in general, reed canarygrass inhibits the establishment of indigenous species, but that this situation is further magnified under a hydrological condition referred to as “bounce”. “Bounce” is defined as large and rapid changes in water levels. Unfortunately, this occurrence usually favors less desirable species such as reed canarygrass as many indigenous species do not respond favorably to rapid water level fluctuations. Hence, water level control is of particular importance especially in the vegetation establishment stage. Lastly, the intrusion of non-desirable species is often the result of seed dissemination from existing upstream sources.

Seabloom et al. (1998) reported that water depth and temperature both influenced the emergence of vegetative species from soil seed banks. They also found that the composition of vegetation communities and species distribution were different between natural and restored wetlands (Seabloom and van der Valk, 2003). The Minnesota Board of Soil and Water Resources in their guide to restoring and managing native wetland and upland vegetation support the importance of considering the hydrology and land use of the wetland site and local watershed. Furthermore, management of invasive species such as reed canarygrass is critical to wetlands providing the goods and services they were intended to (Jacobsen, 2006). Parkland Mews location, within a large agricultural district for example, may be susceptible to stochastic runoff events and pesticide and fertilizer runoff. This is a critical influence on wetland plant communities.

Seabloom and van der Valk (2003) discovered that overall species richness in restored wetlands was less than that of natural wetlands after five to seven years. Species absence was most often attributed to the lack of native perennials in restored wetlands. Furthermore, restored wetlands had less overall species diversity than natural wetlands. Yet, it is apparent that long term vegetation management prescriptions are required to enhance regional biodiversity. As a result, Aronson and Galatowitsch (2008) provide five key recommendations for wetland vegetation recovery, particularly in restored or constructed wetlands where reed canarygrass is more prevalent than natural wetlands. These include limiting site isolation from existing natural wetlands, restoration of semi-permanent wetlands, maintaining soil saturation across the entire basin, re-vegetation via planting particularly of native sedge meadow and wet prairie perennial species, and the control of invasive species such as reed canarygrass. Seabloom and van der Valk (2003) concur with the notion that management of restored wetlands is paramount to the successful establishment of desired species. Unfortunately, water level control is not available at the Parkland Mews site and is likely unavailable at many constructed or restored wetland sites. Accordingly, wetland managers require alternative solutions for the management of reed canarygrass populations in the absence of water level control.

Kim et al. (2006) over the course of two years found that shading by live willow stakes reduced reed canarygrass biomass by 45%. Planting of trees and dense shrubs to create shade has also shown success in Western Washington (Antieau, 1998). These results and recommendations should be considered when evaluating the height and biomass potential in the selection of native grass species in order to be more competitive with reed canarygrass where woody vegetation is not desired.

Antieau (1998) indicates that reed canarygrass is highly susceptible to damage through mowing while Miller et al. (2008) obtained 72% control of reed canarygrass when mowing to within 2.5 cm of the soil surface. This is a result of reducing reproductive capabilities and removing the plants growing point as each stem possesses sexual reproductive characteristics and annually elevates its growing point above the crown. Hence, mowing can be employed as a management tool by cutting reed canarygrass below four inches in early to mid-spring and just as the plants are coming into flower when carbohydrate reserves are low later in the season. Plants are in turn damaged due to stress, thus providing an opportunity for less competitive native species to thrive. Non-selective systemic herbicide applications have shown effectiveness particularly if applied at the correct time, usually at or immediately after flowering (usually late

July) when carbohydrate reserves are at their lowest. Mowing immediately prior to herbicide applications weakens the plant as well as providing for reduced plant biomass requiring spray coverage (Antieau, 1998). Miller et al. (2008) when looking at the effects of reed canarygrass on the re-establishment of native broadleaf trees found that spot treatments with glyphosate resulted in 89% and 98% control at two different sites and suggests that glyphosate spot treatments may be the most cost effective means of eradicating reed canarygrass.

Seeding density has also shown that it can affect the management of reed canarygrass. Adams and Galatowitsch (2008) determined that total native species biomass was greater at higher native seed treatment (15,000 seeds/m^2) than with a lower native seed treatment (3000 seeds/m^2) which are representative of rates used in restoration, while reducing reed canarygrass biomass. Still, the number of individual reed canarygrass plants was equal under both treatments. As well they concluded that native species could withstand greater densities of reed canarygrass seed when seeded at greater densities themselves. In this instance seeding took place on a new seedbed void of vegetation and no indication was given as to the method of seeding.

Foster and Wetzel (2005) hand seeded native species into burn, herbicide, or control treatment plots at a typical prairie restoration seeding rate of 2.5 g seed/m^2 and found that percent cover of the native taxa were highest in the year they were planted with herbaceous seedlings covering an average of 15% of the plot area. Furthermore, they found no reed canarygrass cover or biomass differences among the treatment groups in either of the two years of the study. In fact, they identified numerous reed canarygrass plants re-sprouting from clumps in the soil in both burn and herbicide treatment plots. Furthermore, they indicate that fire is a tool to be used for litter removal as opposed to below ground biomass control as did Adams and Galatowitsch (2006) who suggest that burning did little to control biomass but did reduce the reed canarygrass seed bank. Foster and Wetzel (2005) also indicate that one herbicide treatment in late April, although reducing reed canarygrass biomass provided an insufficient window of opportunity for native species to establish. This finding coincides with that of Adams and Galatowitsch (2006) who found that late August and late September glyphosate treatments were more effective than spring treatments as glyphosate translocation to root structures was more effective at these times of year. Moreover, they found that in plots initially consisting of 75-100% reed canarygrass that were treated with burning or herbicide and then seeded by hand, that non reed canarygrass biomass establishment was greatest where reed canarygrass biomass was less than $50g/m^2$.

Wilcox et al. (2007) evaluated the success of burning and herbicide treatments combined with sowing native grass species in a wetland environment where water levels were not controlled. In this instance all 63m x 27m macroplots received two of each the burning and herbicide treatments over the course of two growing seasons to free the area of vegetation prior to seeding. Their results indicate that both spring and fall seeding of native species did not improve the establishment of native species over two years, but that spring seeding did increase the number of species. Furthermore, herbicide treatments combined with burning was the only treatment that reduced reed canarygrass populations. Critically however, native seeds were broadcast as opposed to the preferred drill seeding method (Wark et al, n.d.) and presumably (although it is not clear), seeded areas received no further establishment attention following broadcasting. For example, rolling of the seeded areas would increase seed to soil contact and thus the potential for germination while limiting predation.

Importantly, Aronson and Galatowitsch (2008) emphasize the significance of initiating control measures early in the restoration process for the management of reed canarygrass. This should be followed by planting of native species in order for vegetation guilds to resemble those of natural wetlands, thus enhancing biodiversity.

CHAPTER 2

SITE BACKGROUND

2.1 Introduction

Parkland Mews is a falconry and bird of prey education center (Parkland Mews, 2006 [1]) located approximately three miles south of the perimeter highway surrounding Winnipeg, in the transitional grassland ecoregion (Scott, 1995). Founded in 1994 by its Executive Director Robert Wheeldon, the intent of Parkland Mews is to create a conservation and education facility that focuses on the recovery of Peregrine Falcons (*Falco peregrinus*) (Parkland Mews, 2006 [2]). Consequently, the main focus of Parkland Mews is to raise and rear Peregrines in captivity for successful release into anthropogenic altered landscapes (Parkland Mews, 2006 [1]).

Peregrine Falcons prefer open landscapes that include grasslands and marshes and they nest on embankments in close proximity to wetlands inhabited by waterfowl and shorebirds which they often catch in flight for consumption (Manitoba Conservation, 2009). The Parkland Mews site is ideally located and designed for the rearing of both eyasses and immature birds as well as the seasonal habitation of returning migratory adults. A hacking shelter is located adjacent to and overlooking wetland cell one and acts not only as a nesting site for mature adults, but also places them in close proximity to the constructed wetland cells (Figure 2.1) which the predatory Peregrines are keen to hunt over. Hence, in order to support the Peregrines, these wetland cells must attract and host both an avian population and the trophic levels required for them to survive.

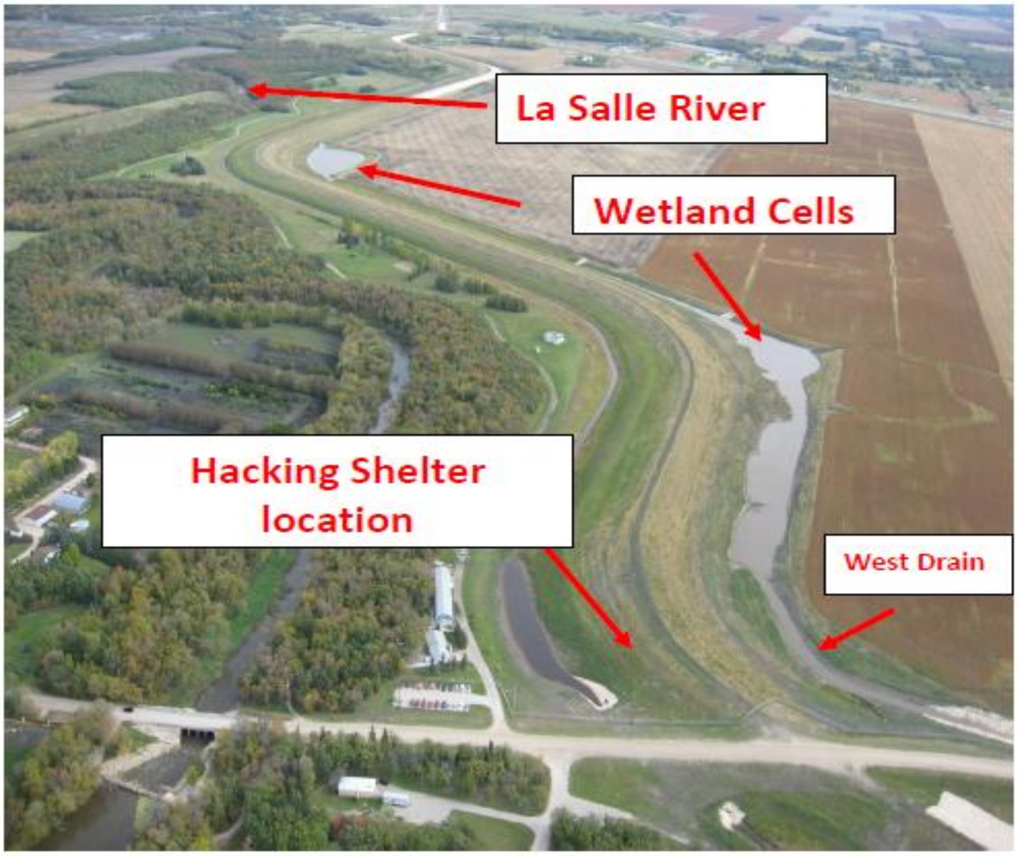

Figure 2.1 - Aerial view of Parkland Mews site (Robert Wheeldon, 2010)

2.2 Site History

The great flood of 1950 experienced by the City of Winnipeg precipitated the actions of then provincial minority leader, Duff Roblin, to initiate and pursue a vision for a channel that would divert water around the City of Winnipeg in severe flood events. It is estimated that since first being used in 1969 that the floodway has saved $10 billion in flood damages (Manitoba Floodway Authority[1], 2008). Still, the spring of 1997 presented new challenges for the residents of southern Manitoba and the City of Winnipeg.

The spring of 1997 delivered what has been coined “The Flood of the Century” which found southern Manitoba inundated with unprecedented high water levels that pushed the floodway to its capacity and required the hasty construction of the “Z” or “Brunkild Dike” to prevent water from entering the city from the south and west (Figure 2.2). Although the floodway and the Z dike, now referred to as the West Dike, ultimately protected the city of Winnipeg itself, both urban and rural residents of southern Manitoba, as well as some politicians, demanded increased flood protection for subsequent flood events. This expansion would include improvements to the West Dike, which is responsible for containing Red River flood waters from the southwest and preventing flow from entering into the La Salle River watershed (Figure 2.2) (TetrES Consultants Inc/Inter-Group Consultants Ltd., 2004; Manitoba Floodway Authority [2], 2008; Manitoba Floodway Authority [3], 2008) Accordingly, a crown corporation, “The Manitoba Floodway Authority” was created to oversee the floodway expansion project.

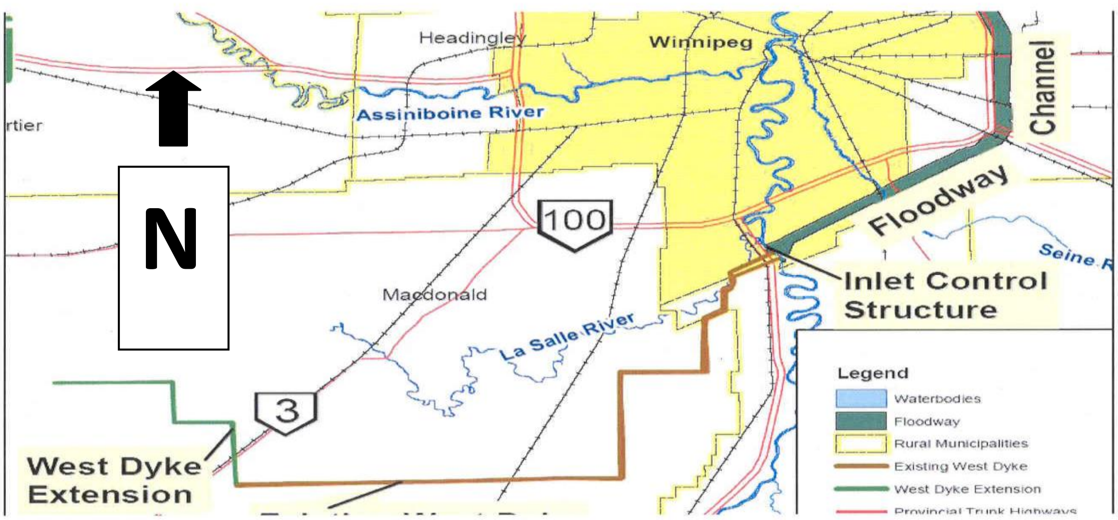

Figure 2.2- Floodway and West dike locations (Manitoba Floodway Authority, 2009)

The Floodway Authority's mandate to expand the floodway proper also included improvements to the West Dike (a compacted earthen embankment) which extends 70 km. from south of Winnipeg at the floodway's inlet control structure in a north-easterly direction until it connects with high ground at the west side of the Red River. These improvements consisted of;

- raising the dike (to 1:700 year flood event levels) to allow for more freeboard,
- grading of the west drain itself,
- culvert installation,
- extension of the dike to higher ground to the west
- and the installation of limestone rip rap for protection against wind erosion (1:10 year wind event),

This was all to be completed with minimal land acquisition and utility relocation (TetrES Consultants Inc/Inter-Group Consultants Ltd., 2004; Manitoba Floodway Authority, 2008[3]).

The location of the west dike situates it immediately south of and adjacent to the Parkland Mews location and immediately north of the west channel (Figure 2.3). It is within the confines of this water works control structure and as part of the channel itself that the Parkland Mews cells were constructed (Figure 2.3).

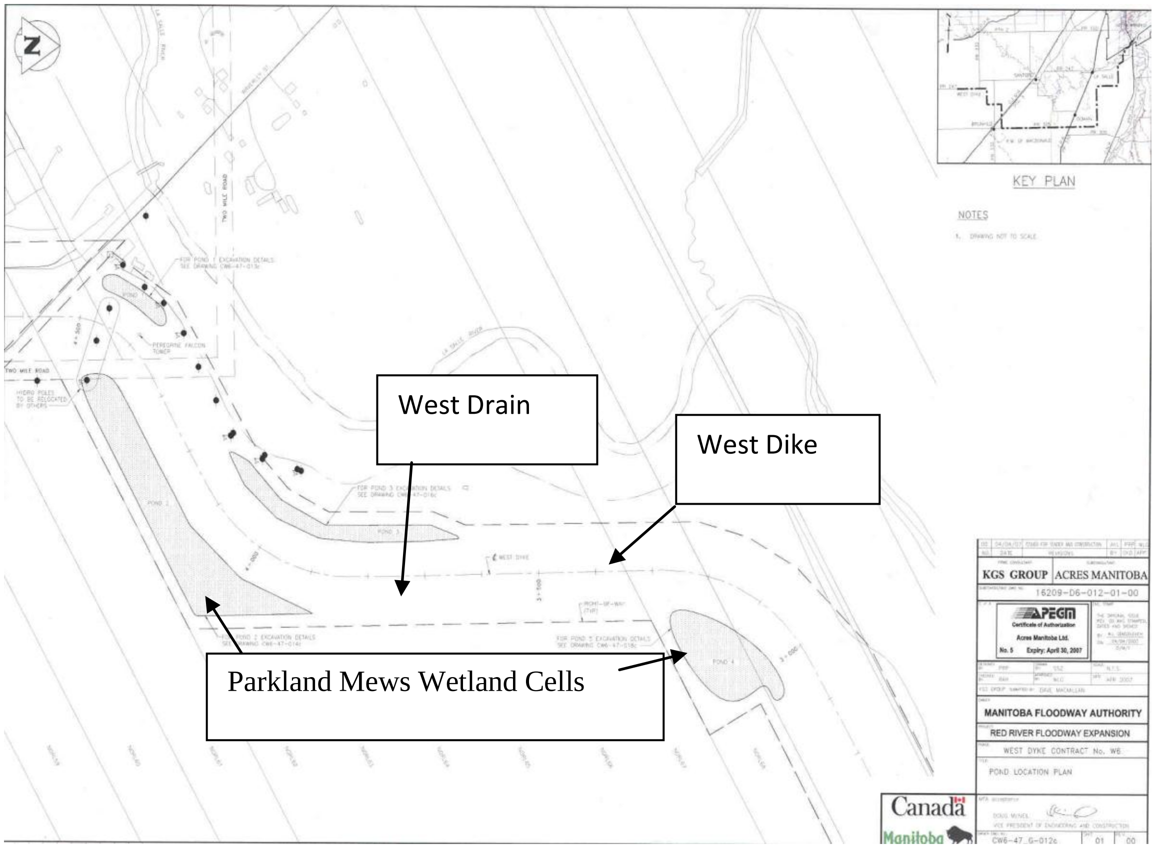

Figure 2.3- Wetland cell locations within West Drain (Manitoba Floodway Authority, 2009)

2.3 Parkland Mews Wetland Cells

The constructed wetland cells located at Parkland Mews were created by the Manitoba Floodway Authority at the request of Robert Wheeldon, Manager of Parkland Mews. They were designed and constructed to provide habitat over which the Peregrine Falcons could train and hunt. The fact that the cells are anthropogenically generated creates a challenge in identifying and classifying them as a specific type of wetland. However, the cells are integrated within a drainage canal (Figure 2.4) responsible for carrying runoff from the local watershed to the La Salle River (Figure 2.4). The hydrologic intent of the cells is to create a series of basins capable of slowing the flow of water within the canal, while still retaining adequate water volumes in the cells.

Fluctuating water levels will allow for the establishment of a wetland environment. High water levels are generated by runoff from the upstream watershed area (Robert Wheeldon, personal communication, 2010). The levels retained in each of the cells when flow ceases is dictated by the gradient established by the floodway authority in order to discharge flow toward the Red River. Minimum ditch gradients contained in the pre-design for the drain itself were 0.03% down-stream fall and 0.03% cross-fall (TetrES Consultants Inc/Inter-Group Consultants Ltd., 2004). Accordingly, these constructed wetlands are intended to provide the ecological benefits afforded by a natural wetland system and become a cog not only within the human altered landscape but within the Parkland Mews ecosystem.

The constructed wetland cells are integrated within a drainage canal (Figure 2.4) responsible for carrying runoff from the local watershed to the Red River via the La Salle River (Figure 2.4).

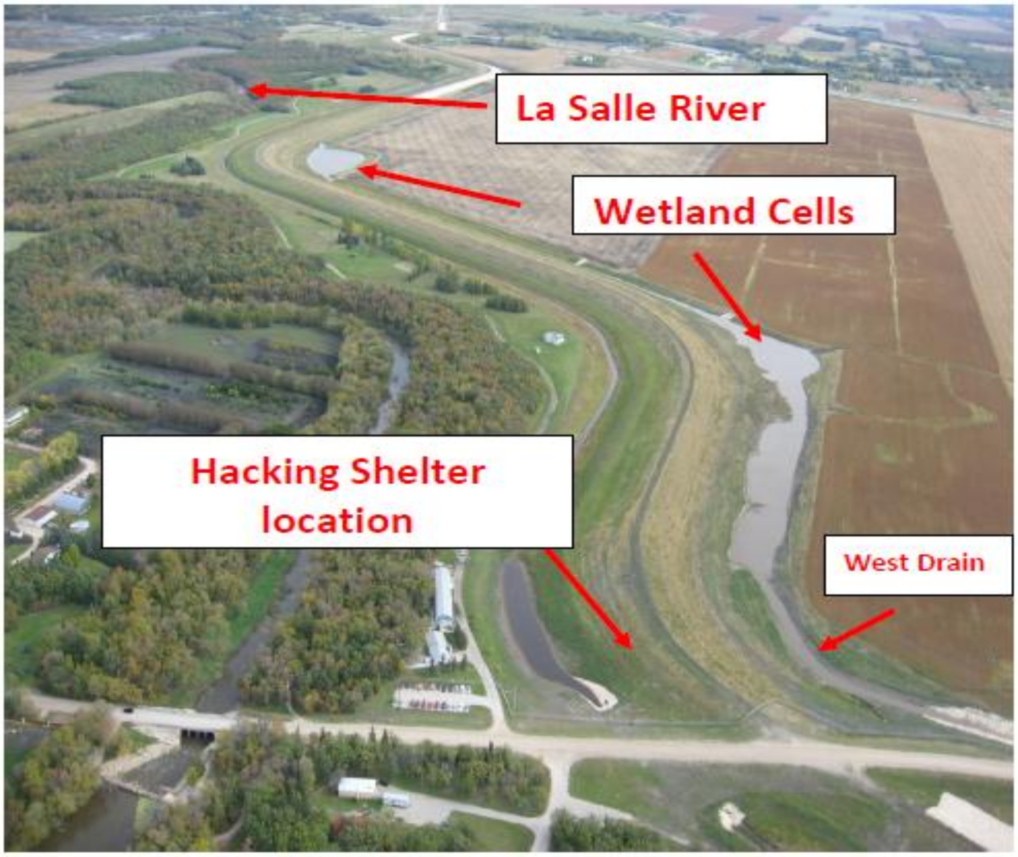

Figure 2.4 – Surface water pattern at Parkland Mews (Robert Wheeldon, 2010) Water moves through the system of wetland originating at the west drain location moving through the wetland cells and ultimately into the La Salle River.

The upstream drainage canal at Parkland Mews and the constructed wetlands cells themselves are ideal habitat for reed canarygrass and thus possess a large abundance of reed canarygrass. Consequently, in addition to being seeded on the study site, the upstream portion of the drain is also a likely reed canarygrass seed source. This drain also supplies a considerable volume of runoff to the Parkland Mews wetland cells. This water coupled with direct agricultural runoff immediately adjacent to the wetlands generates stochastic water levels, and provides assumed high nutrient and sediment loads to the west drain both of which are Parkland Mews wetland cells primary source of water. Furthermore, being that the contiguous drain feeds the Parkland Mews constructed wetlands from upstream (Figure 2.5), is inundated with reed canarygrass, it is assumed to be a primary source of reed canarygrass seed. Additionally, is the fact that that the Manitoba Floodway Authority, the body responsible for the management of the drain in which the Parkland Mews constructed wetland cells are located, included reed canarygrass as 10% of its base seed mix (Appendix 1) which was planted immediately above the wet meadow mix (Appendix 2) at the Parkland Mews site (Figure 2.6) when the site was seeded in the fall of 2008. Unfortunately preliminary vegetation inventories of September, 2009 indicate a wide variety of species including reed canarygrass on the Parkland Mews site (Appendix 3- see Royer and Dickinson, 1999; Vance et al., 1999; Kershaw, 2003- for species descriptions).

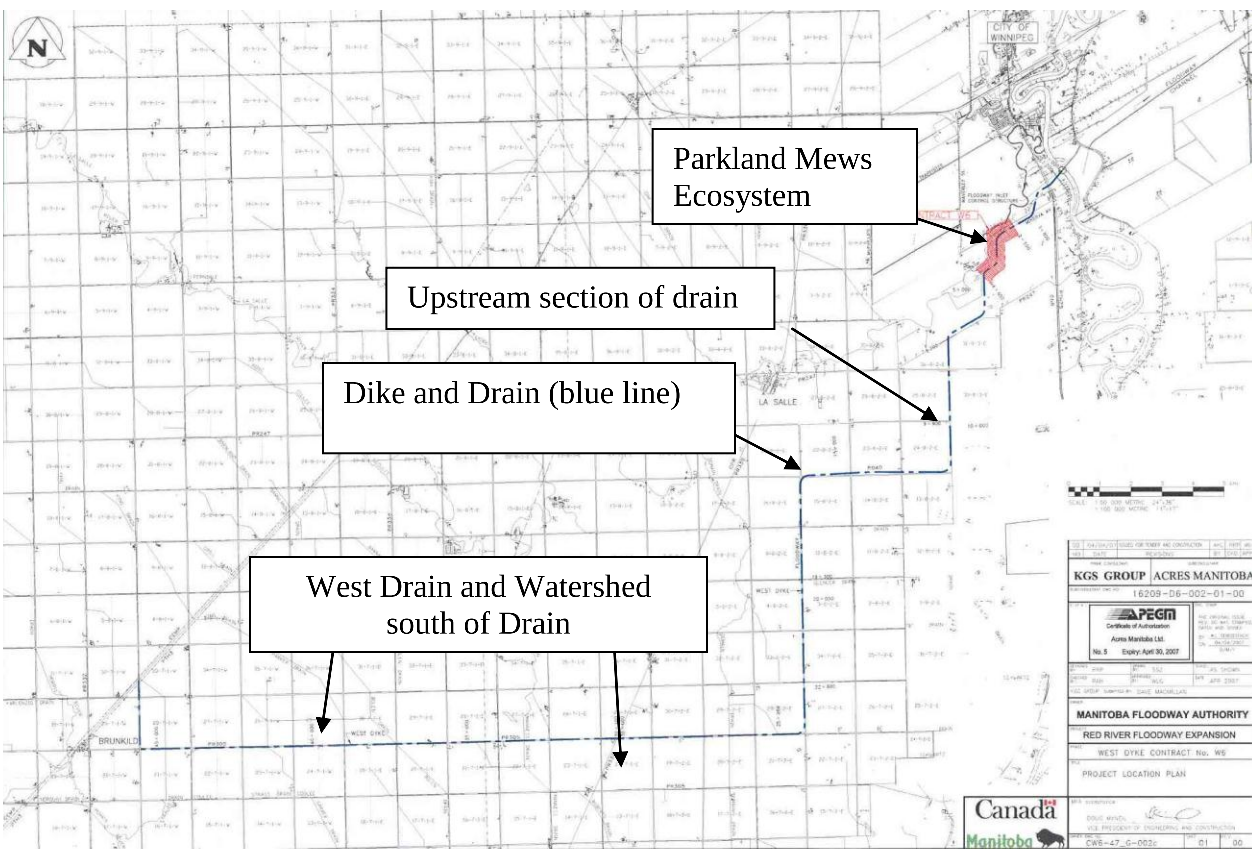

Figure 2.5 - West Dike/Drain Watershed (Manitoba Floodway Authority, 2009)

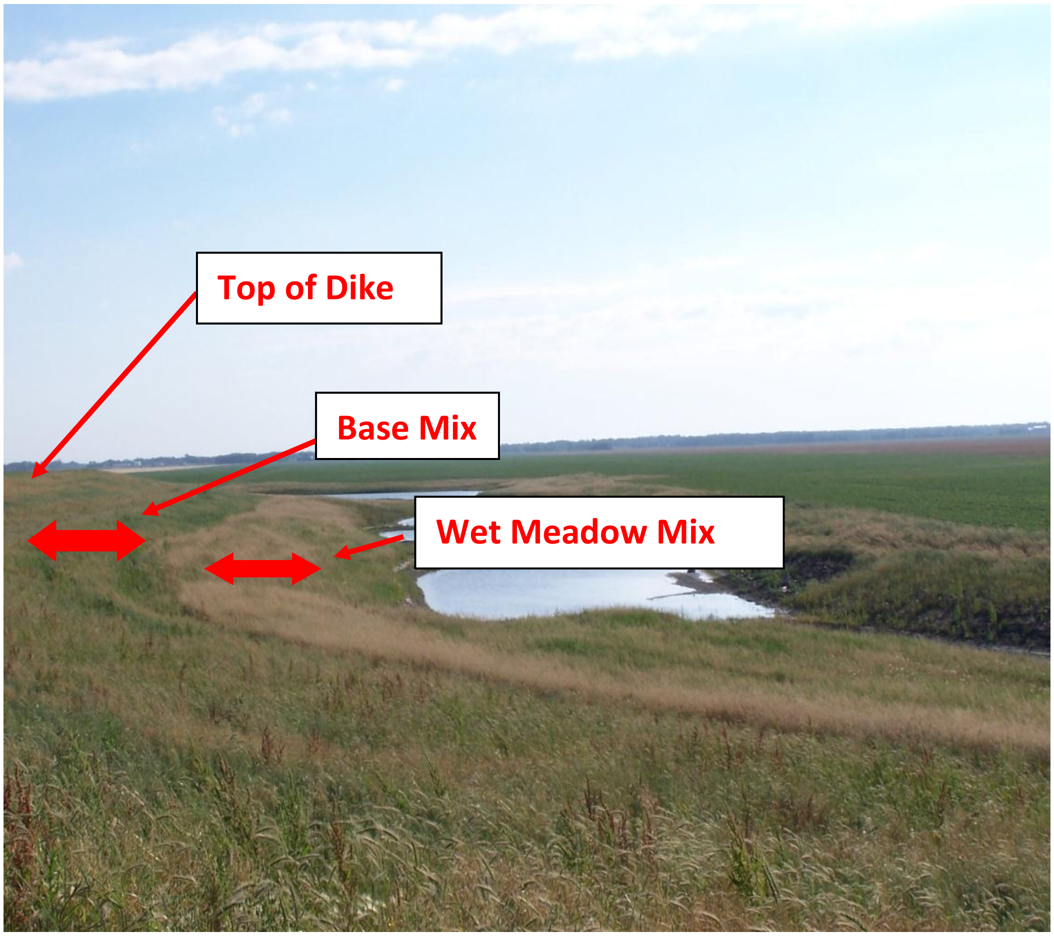

Figure 2.6 - Parkland Mews Seeding zones (Robert Wheeldon, 2010)

2.4 Biophysical Assessment

The intent of a biophysical assessment is to identify and describe both the biotic and abiotic factors contributing to the interactions that influence how a particular ecosystem or site has or is influenced by such factors. Such an assessment provides the background to analyze how these factors might guide the evolution of that ecosystem or site. In the case of Parkland Mews, this preliminary assessment includes identifying abiotic factors such as the ecozone, climate, geology and soils, the watershed, and vegetation as biotic and abiotic influences.

2.41 Ecozone and Ecoregion

Parkland Mews is located in an ecozone identified by Natural Resources Canada (2009) as the Prairies and is more explicitly located in the Lake Manitoba Plain Ecoregion. Lake Manitoba Plain Ecoregion encompasses an area that begins at the border between the United States and Canada and extends northwest to Dauphin Lake (Environment Canada, 2005). The western portion of southern Manitoba is contained within the geological province of the interior platform (Natural Resources Canada, 2004) which is known to commonly contain wetlands (Natural Resources Canada, 2009). Furthermore, it is part of the Northern Tallgrass Prairie which has seen a significant decline in wetland habitat as a result of intensive agricultural drainage network installation following European settlement. Consequently, 90% of wetlands have been lost from this region (Lant et al., 1995). Climate is the major factor influencing ecozone boundaries.

2.42 Climate

Climate at the Parkland Mews location is continental, with cold winters and hot summers (Freedman, 2006). Situated in the Lake Manitoba Plain Ecoregion positions it in one of the most humid and warm regions of the Prairie ecozone (Environment Canada, 2005) due to its close proximity to large lakes located in south and central Manitoba. More specifically Environment Canada data for Winnipeg Richardson International Airport located at 49^{o} 55.000’ N latitude and 97^{o} 14.000’ W at an elevation of 238.70 meters indicates mean daily air temperatures for the months of January, April, July, and October of -17.8, 4, 19.5, 5.3 degrees Celsius respectively, and total precipitation falling as rain at 415.6 mm., that as snow at 110.6 cm., for a total annual precipitation amount equivalent to 513.7 mm (Environment

Canada, 2010). The median start date of continuous snow cover occurs between November 16- November 30 with the median end date of continuous snow cover ranging from April 1- April 15 (Natural Resources Canada[1], 2007). Long term climate has had an effect on the geological make-up of the area as well.

2.43 Vegetation Composition

The composition of vegetation communities within the tall-grass prairie are influenced not only by soil type but relief as well. As such, Scott (1995) has identified categories of sites related to drainage and the typical suite of vegetation associated with each. Better drained sites are largely dominated by the tall grass species *Agropyron gerardii* (big bluestem) and *Schizachyrium scoparium* (little bluestem) amongst a host of other monocot species and approximately a dozen common forbs such as *Solidago canadensis* (tall goldenrod) and *Anemone canadensis* (Canada anemone).

Moderately drained sites consisting of Humic Gleysol soils have greater moisture content and thus favor species such as *Panicum virginatum* (switchgrass) *and Elymus canadensis* (Canada wild rye) but are likely to include species from the better drained classification as well. Depressions or sloughs that contain Humic Gleysol soils categorize the third classification. These areas are often dominated by a suite of species that consists of *Spartina pectinata* (slough or cordgrass) and *Eleocharis palustris* (spike rush) and a few others in non-saline conditions, while in saline environments *Spartina gracilis* (alkali-cord-grass) and *Puccinellia nuttalliana* (nuttall's alkali grass) are common (Scott, 1995).

Well drained uplands, the fourth classification, tend to experience moisture stress later in the growing season and thus lend themselves to dominant grasses such as *Stipa comata* (needle and thread grass), *Bouteloua curtipendula* (side oats gramma) and forbs such as

Solidago missouriensis (low goldenrod) and *Psoralea agrophylla* (silverleaf psoralea). Lastly, the fifth classification, Riverine Gallery Forest, is made up of broadleaf deciduous forests that line lowland waterways such as the Red, Assiniboine, and La Salle Rivers. A number of dominant species prevail in these areas namely deciduous species such as *Ulmus americana* (American white elm) *Fraxiunus pennsylvanica* (green ash), and *Acer negundo* (Manitoba maple) (Scott, 1995).

2.44 Geology and Soils

The western portion of southern Manitoba is contained within the geological province of the interior platform (Natural Resources Canada, 2004). This area consists mainly of sedimentary rocks of the Paleozoic era as a result of the consolidation of loose sediment that has accumulated in lake beds (Natural Resources Canada, 2003). The Red River Plain, a sub-region of the interior plain, is part of a central lowland basin located in southern Manitoba. This sub-region consists of clay basins, flood plains, and river levees which are positioned in the Lake Agassiz basin in an area located below the 850 foot contour of southern Manitoba. The only interruption to the massive expanses of flat flood plain are a number of natural river and stream channels and the extensive drainage systems that have been employed to drain wet meadow areas (Natural Resources Canada, 2003). The area is dominated by fine grained glaciolacustrine surficial materials (Natural Resources Canada, 2003) with soils consisting of alluvial deposits left by modern stream water and lacustrine clay that was deposited over limestone bedrock by glacial lake waters. These vary in depth from as little as a few feet to 60 feet in depth (Ehrlich, et al. 1953; Environment Canada, 2005;Agriculture and Agri-Food Canada, 2008).

Soils at the Parkland Mews location are classified in general as Black Chernozems (Scott, 1995) and of the Red River association. They have been highly influenced by excessive moisture at some point in their development with greater than half of the soils of this association demonstrating the impact of hydromorphism. Not surprisingly clay is the dominant textural category. Prior to drainage improvements in the region, the combination of clay soils and a landscape largely devoid of relief, combined to create expansive marshes and wet meadows that dominated the environment. However, the Red River association soils are identified as being of good to very good productivity when supplied with the provision of adequate drainage and thus improved surface drainage has been undertaken in much of this region (Ehrlich, et al., 1953) at the expense of naturally occurring wetlands and meadows.

2.45 Soil Testing

Soil samples were collected and analyzed to provide background and insight into existing soil conditions. Composite soil samples (Brady and Weil, 2004) were extracted on April 27, 2010 with a 5.08cm diameter auger to a 10.16cm depth with block's one and two (Sample1) and block 3 (Sample 2) each generating one testable sample with blocks 4-6 (Sample 3) generating just one testable sample as they are situated adjacent to each other. Final sample submissions were a homogenous blend of a composite of 18 samples from each block. Two samples were taken at the end of each block and 7 samples taken along the lower environmental gradient and 7 from the higher environmental gradient were blended prior to submission. As per lab recommendations, samples were placed in soil sample bags and kept in a covered cooler complete with cooler packs until they were submitted later the same day. Soil sample analysis was performed by ALS Labs (2010) in Winnipeg.

Soil texture was determined by hand texture while alkalinity, bicarbonate, and thus pH values, were all determined through the 1:2 saturated past extract method (ALS Labs, 2010). Organic Matter levels, were measured by loss on ignition (LOI) at 375 degrees celcius. Analyzing values for Sodium Adsorption Ratio (SAR) was determined by SAR and cations in a saturated soil while pH values, were determined with a 1:2 soil to water agricultural method. All macro and micro nutrients were measured as available nutrients (ALS Labs, 2010).

Mean and range values for all parameters tested for across the three sampling locations on the Parkland Mews site are summarized in table 5.1. Parameters measured for related to plant growth included soil texture identified as clay across all blocks, pH which averaged 8.08, organic matter which averaged 5.4% across the three blocks (Table 2.1) Moreover, the sodium adsorption ratio (SAR) mean of samples one, two and three was 1.23, while the electrical conductivity (EC) mean was 0.848 ds m-1. Results of from all sample locations can be found in Appendix 4.

Table 2.1- Mean and Range Values for all soil sample locations.

TEST DESCRIPTION	MEAN RESULTS	RANGE RESULTS	DETECTABLE LIMITS	UNITS
Texture	Clay	na	na	na
pH	8.08	7.80-8.41	0.10	pH
Conductivity	0.848	0.624-1.09	0.010	ds m-1
Organic Matter	5.4	3.6-6.8	1.0	%
Phosphate, available	7.4	3.9-11	1.0	ppm
Potassium, available	267	231-305	2.0	ppm
Sulfate, available	246.67	139-387	2.0	ppm
Copper, available	1.43	1.2-1.6	0.020	ppm
Boron, available	0.43	0.4-0.5	0.10	ppm
Nitrate, available	0.9	0.5-1.2	0.40	ppm
Chloride, available	55.17	44.7-63.5	0.20	ppm
Alkalinity, Total (as CaC03)	75.67	68-82	10	mg/L
Bicarbonate (HCO3)	275.7	82.8-99.5	2.0	mg/L
Calcium (Ca)-Total	225.33	158-297	1.0	mg/L
Magnesium(Mg)- Total	138	108-169	1.0	mg/L
Sodium (Na) Total	95.03	68.5-120	0.10	mg/L
Sodium Adsorption Ratio	1.23	0.94-1.38	0.10	SAR

Source: ALS Labs Analytical Group, Winnipeg, 2010

2.46 Water shed

The entire province of Manitoba is contained within the massive Hudson Bay watershed that accounts for the capture of 30% of Canadian runoff (Natural Resources Canada, 2006 [1]). However, although the City of Winnipeg and the Parkland Mews location are both contained within this watershed, they are hydrologically classified as part of the Western Plains Hydrogeological Region. The southern areas of the region are semiarid and thus dry-land farming prevails during the short but warm growing season prior to the onset of long, cold winters. Remaining indigenous vegetation in the region consists of aspen parkland, short, mixed and tall grass prairies. These rely on the 5-25% (Natural Resources Canada , 2009) of the area that consists of wetlands that are crucial to groundwater recharge (Natural Resources Canada, 2006 [2]) and as breeding and staging grounds for half of North America's migratory birds (Natural Resources Canada , 2009). More specifically however, in a 1953 soils reconnaissance survey Ehrlich et al. (1953) only categorized 1.97% of the land in the Rural Municipality of Ritchot as marsh and wasteland. Still, the Parkland Mews wetland cells are significantly affected hydraulically by both so called natural flood events and anthropogenic activity.

The portion of the Parkland Mews operation consisting of a home for the site manager, as well as breeding and interpretive centers is located in the La Salle River watershed. This portion of the site and the wetland cells themselves are separated by the west dike which creates a barrier to surface water flow which was once part of the La Salle River watershed, from moving toward the La Salle River. This ultimately positions the west drain and the associated wetland cells in the Red River watershed (Figure 2.7).

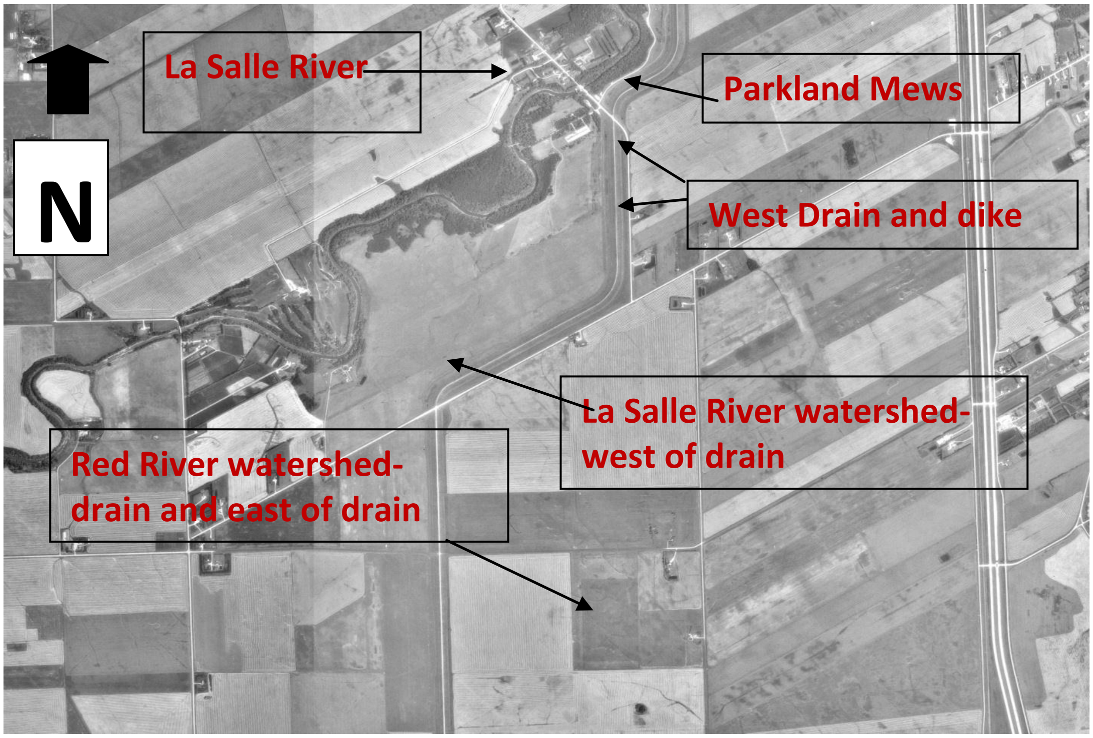

Figure 2.7- Parkland Mews location in relation to La Salle and Red River Watersheds (La Salle/Redboine Conservation District- personal communication October, 2009)

The Red River watershed encompasses a land area of approximately 290,000 km^2 with only 16 % of this area found within Canada while the remainder is within the United States. The flatness of the terrain immediately surrounding the Manitoba portion of the Red River Valley (Natural Resources Canada[2], 2007) means that flooding is an all too frequent reality in southern Manitoba. Parkland Mews is located in the Rural Municipality of Ritchot which claims to have been affected by high-magnitude floods on seven separate occasions over the course of the past 350 years. These events took place in 1762, 1747, 1826, 1852, 1950, 1979, and 1997. Furthermore, the municipality has been subject to numerous minor floods during this same time span (Rural Municipality of Ritchot, 2009). Although the Parkland Mews wetland cells were not in existence during these events, the area is certainly susceptible to the effects of such events in the future. Not unlike all surface water sources including wetlands, the Parkland Mews wetland cells obtain a portion of their water through natural precipitation in the form of rain and snow. However, the fact that the Parkland Mews cells are located in the west drain means that a large portion of the water volume received is a result of the runoff water from agricultural land. Most of this runoff enters the west drain from the watershed to the west of Parkland Mews, in addition to smaller volumes associated with the cultivated land immediately adjacent to and south of the cells themselves (Figure 2.7). Accordingly, the cells and the water contained in them are highly influenced by anthropogenic activities including land drainage and the drain itself which impact on both the volume and quality of water received and retained by the cells. As a result, vegetation surrounding the cells is also influenced.

2.47 General Water sampling

Water samples were obtained from four locations on the Parkland Mews site. These locations coincided with a number of inflow and outflow locations in the chain of wetland cells that make up the constructed wetland system. The intent of choosing these sampling locations was to identify over the course of the study whether or not the movement of water from wetland to wetland has any observable impact on water quality, and to obtain baseline water quality data. Samples were taken from adjacent to the specific sampling location by securing the plastic bottle to the end of a 2m pole and extending the pole out over the desired sampling location and then submersing the bottle to a depth of approximately 15cm. Samples were collected in one liter plastic containers complete with a screw on cap supplied by ALS Labs, Winnipeg. Sample bottles were rinsed three times with the water being sampled prior to the sample being taken. Once the sample was secured in the bottle, the bottle was wiped dry, marked with an identification code, placed in a sealed plastic bag, and then stored in a cooler while subsequent samples were secured. Once the four samples were collected they were immediately delivered (within two hours of the first sample taken) to ALS Labs, Winnipeg for testing of requested parameters (United States Environmental Protection Agency, 1994). Initial water samples were extracted October 7, 2010 at four locations at the Parkland Mews site and delivered to ALS labs, Winnipeg within two hours of being extracted. All samples were tested for conductivity, total phosphorus, turbidity, pH, alkalinity, nitrogen, sodium adsorption ratio, and total metals.

Water samples were collected and tested on six occasions at four different locations within the site beginning with the first sample on October 7, 2010 and followed by 5 subsequent samples in 2011 (April 7, May 16, June 16, July 12, July 29). All samples were tested for conductivity, total phosphorus, turbidity, pH, alkalinity, nitrogen, sodium adsorption ratio, and total calcium, magnesium and sodium. Mean and range sample results aggregated for all sample dates at all locations are identified in Table 2.2. Importantly, for eutrophication average total phosphorus levels were 0.2258 mg/L (258 ug/L$^{-1)}$. Results from all sample dates can be found in Appendix 5.

2.48 Pesticide Water Sampling

On June 27 and July 29, 2011 water samples were also tested for organochlorine pesticides, phenoxyacid herbicides, glyphosate and aminomethylphosphonic acid (AMPA) a major metabolite in glyphosate. Results for all samples indicated no detectable levels of either the organochlorine pesticides or phenoxyacid herbicides. Low levels of glyphosate and AMPA were detected at each sample date. Mean values of each were 7.84 ug/l and 3.99 ug/l respectively across all sample locations on site for the June 27 sampling date. On July 29 low levels of glyphosate were detected at each sample site and AMPA was detected at two of four sample sites generating mean values of 6.13 ug/l and 0.82 ug/l respectively across all sample locations on site.

Table 2.2- Mean and Range Values for all water sample dates and locations.

TEST DESCRIPTION	MEAN RESULTS	RANGE RESULTS	DETECTABLE LIMITS	UNITS
Conductivity	466.33	206-847	0.40	umhos/cm
Phosphorus, total	0.258	.100-.396	0.0020	mg/L
Turbidity	22.50	8.8-62.68	0.10	NTU
pH	8.66	7.94-9.27	0.10	pH units
Alkalinity, Total (as CaC03)	120.96	70.47-9.27	1.0	mg/L
Bicarbonate (HCO3)	111.80	56-163.75	2.0	mg/L
Carbonate (CO3)	17.46	<0.60-53.7	0.60	mg/L
Hydroxide (OH)	<0.40	na	0.40	mg/L
Nitrite-N	0.033	<0.050-0.134	0.050	mg/L
Nitrate-N	0.292	<0.050-0.242	0.050	mg/L
Nitrate and Nitrite as N	0.326	<0.071-0.197	0.071	mg/L
Total Kjeldalh Nitrogen	2.47	.98-3.51	0.20	mg/L
Total Nitrogen (Calculated)	2.88	1.37-3.75	0.20	mg/L
Sodium Adsorption Ratio	0.475	.2-.76	0.030	mg/L
Calcium (Ca)-Total	61.96	18.07-157.4	0.20	mg/L
Magnesium (Mg)-Total	25.21	8.95-41.2	0.050	mg/L
Sodium (Na) Total	16.76	4.27-31.93	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2010-2011

Differences between west and east blocks are summarized in Table 2.3 with slope differences both within blocks and the difference in elevation between the blocks and water level demonstrating the potential for hydrologic differences between the two sites.

Table 2.3 Differences between West and East blocks

Location	Soil Texture	Soil pH	Alkalinity mg/l	Soil Conductivity ds m-1	Sodium Adsorption Ratio %	Bare ground mean % May 2010/August 2010	Slope %	Difference in Elevation between block location and water level in meters
West Blocks	Clay	7.92	79.5	0.83	1.16	45.6/4.12	1.61	0.832086
East Blocks	Clay	8.41	68	0.624	1.36	56.41/19.37	1	0.7443546

2.49 Vegetation

Located in The Lake Manitoba Plain Ecoregion, Parkland Mews consists of a diversity of species as it forms an intermediary zone between the Aspen Parkland of the southwest and Boreal Forest to the north (Natural Resources Canada, 2009; United States Department of Agriculture, 2009) placing it in Canada's transitional grassland ecoclimate, but more traditionally in the tall-grass prairie region of Canada where the climate is semi-arid to semi-humid (Scott, 1995). Grasslands and stands of bur oak are found in drier areas while shrubs and trembling aspen can be found on sites receiving greater moisture. The indigenous vegetation of the Red River plain was made up of tall grass prairie and meadow which persisted in various areas in relationship to the natural drainage, with ferns and other herbaceous vegetation dominating those areas subjected to occasional flooding (Ehrlich, 1953). Vegetation in the region must be capable of withstanding a plant hardiness zone identified as 2b where cold winter temperatures can drop to between -40.0 to -42.7 degrees celsius (Natural Resources Canada, 2010; United States Department of Agriculture, 2009).

Nonetheless, some of the indigenous flora and fauna of the region are endangered largely as a result of the conversion of much of the landscape to agriculture (Freedman, 2006) as many existing wetlands have been drained or ploughed under to accommodate agricultural activity. Consequently, the retention of indigenous species has become more critical as has the retention and restoration of wetland areas including the control and management of invasive species such as reed canarygrass. See Classification of natural ponds and lakes in the glaciated prairie region by Stewart and Kantrud (1971) for potential wetland species likely to be found in the area.

CHAPTER 3

MATERIALS AND METHODS

3.1 Preliminary vegetation inventory results

Vegetation inventories, obtained along a series of transects, of September 29, 2009, approximately one year after seeding, revealed that some of the species seeded by the Manitoba Floodway Authority had successfully established and although none were quantified, it was visually apparent that reed canarygrass had established a dominant foothold on the site. This would be expected considering that the Manitoba Floodway authority included reed canarygrass as part of its seed mix and that the portion of the west drain upstream from the Parkland Mews location is assumed to be a source reed canarygrass seed to the Parkland Mews site. Furthermore, it is widely acknowledged that reed canarygrass proliferates in disturbed environments (Kercher and Zedler, 2004; Lavergne and Molofsky, 2004; Adams and Galatowitsch, 2008) such as the new wetland site and adjacent dike at Parkland Mews, and that invasion control is necessary following initial site clearing (Maurer et al. 2003; Adams and Galatowitsch, 2008), which was not practiced during the establishment phase at Parkland Mews. Consequently, establishing a research design focused on quantifying plant species biodiversity and treatment effectiveness for the control of reed canarygrass was established.

3.2 Research Design

The sampling technique used was a randomized block design. The purpose of this approach was three-fold. One, to group blocks into areas of fairly consistent environmental conditions (i.e. the riparian zone) and elevation and thus hydrologic gradient where reed canarygrass was most prevalent. Reed canarygrass prevalence on site tends to be situated in a 10 m band (Figure 3.1) parallel to the water, approximately 5 linear meters from what was normal water level in 2009-2010. Two, block locations were selected to encompass these bands while at the same time providing for an adequate block size to accommodate the treatment plots. Plot size was based on the initial intent of using a drill seeder on some plots to enable the comparison of drill seeding and broadcast seeding following mowing and herbicide treatments. Thirdly, the randomized block design is best suited to account for environmental heterogeneity in patchy environments such as those encountered in situations where reed canarygrass has yet to gain a foothold (Potvin, 1993). This allowed for analysis of objectives 1 and 2 in the areas where reed canarygrass was most abundant.

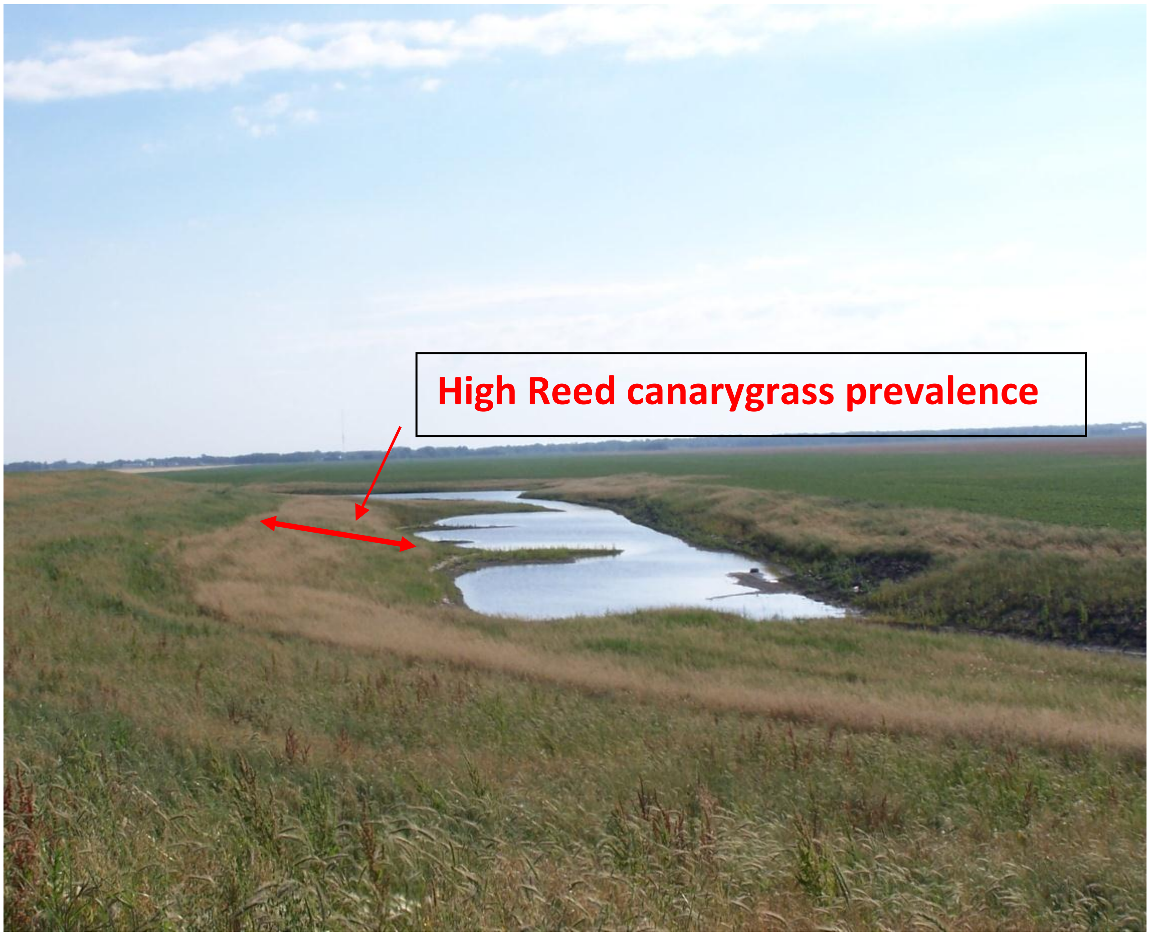

Figure 3.1- Reed canarygrass prevalence in 10 m band parallel to the water, approximately 5 linear meters from 2009-2010 normal water level (Robert Wheeldon, 2010)

Treatments (Table 3.1) applied to plots within blocks were randomly assigned using a random numbers table.

Table 3.1- Plot Treatments

Control (C)
Spring Mowing x Broadcast (SM x B)
Spring Herbicide x Broadcast (SH x B)
Spring Mowing x Herbicide x Broadcast (SM x H x B)
Total Treatments plot per Block = 7
Total Sub-samples/Block = 21
Total Sub-samples for six blocks = 126

Once block locations were identified in the spring (May 2010), individual plot locations (Figure 3.2) within blocks were identified using a tape measure and setting permanent stakes on May 4, 2010. Plot sizes were 3m wide x 6m long (Figures 3.2 and 3.3) and were originally intended to accommodate the width of the drill seeder. Corners of experimental plots were identified for each treatment plot within each block and the plot assigned a numerical identifier (Figure 3.4). Between each plot and between plots and the outer edge of the block, half meter buffers were identified and herbicide vegetation control employed to minimize the potential of encroachment of vegetation or treatments from adjacent plots. Following confirmation of plot locations, in-ground markers were placed at the four corners of each plot to ensure the exact location of the plots for the duration of the study. Last, GPS coordinates and elevations of each block and plots within each block were identified using a hand-held GPS unit and survey grade elevation equipment.

Figure 3.2- Block II example with treatment plots (Block: ; 1 treatment plot of 7 within the block) (Wiseman, 2011).

A complete randomized block, split plot, 4 (treatments in each block) x 6 (individual blocks) factorial design equating to a total of 42 experimental plots was employed for the purposes of field research and consequently a total of seven treatments including a control were assigned to each block using a random numbers table. Six blocks were used to increase the statistical degrees of freedom and block locations were identified by locating the blocks in zones of prominent reed canarygrass infestation most of which generally traversed parallel to and starting approximately 3 to 6 linear meters above normal water level and usually in a band approximately 7 m wide. Each treatment plot was subdivided into nine potential sub-sample locations (ie. 2.12, 2.22- 2.33 Figure 3.3) from which three (one from each potential hydrologic gradient within treatment plots) were randomly selected (i.e. 2.12, 2.23, 2.31- Figure 3.4) for measurements of percent cover change. This provided an objective approach to identifying where vegetation analysis would occur within each plot. Sub-sample locations selected from each hydrologic gradient within a treatment plot (Figures 3.4 and 3.5) were identified using a random numbers table (McGrew and Monroe, 2000) for use in vegetation and statistical analysis. Random selection was insured by having an independent individual blindly identify a starting value on a random numbers table. Beginning with that value the random numbers table was then employed to identify three sub-samples within each treatment that were used for measurement. This generated a total of 126 sub-samples for the purposes of vegetation and statistical analysis.

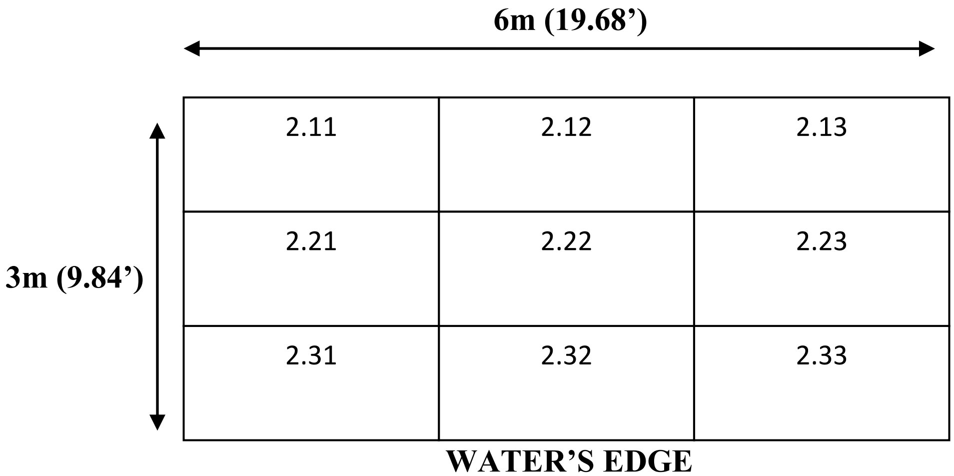

Figure 3.3- Block II Example of Sub-division of a treatment plot to sub-sample level (not to scale) As an example the identifier 2.12 refers to block 2; hydrologic gradient 1; sample 1.

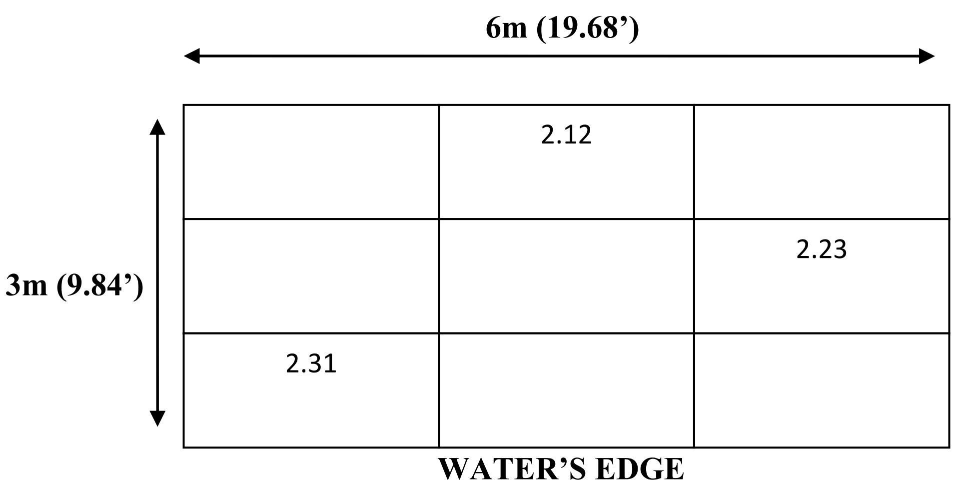

Figure 3.4 Example of three sub-samples (one from each hydrologic gradient) randomly selected in a treatment plot (not to scale) for vegetation measurement purposes throughout the duration of the study.

Figure 3.5- Randomly selected for vegetation measurement sub-sample locations- Block 2 example. There are three sub-samples per plot●(one per each hydraulic gradient) that were selected for vegetation measurement throughout the duration of the study (Wiseman, 2011).

3.3 Block/Plot Preparation

An early spring in 2010 allowed block locations to be selected by March 19, 2010 well in advance of the plot treatment dates. Block locations were identified on site and were measured and marked to the dimensions of 50 x 6 meters and then cleared of existing vegetation. This was done even though mowing would be a component of four treatments per block thus creating a split plot design. Not removing as much litter as possible would cause a “hair pinning” (crimping) of the debris into the soil during the drill seeding process causing the seed to be surrounded by litter and thus excluding the seed from being in contact with the soil subsequently jeopardizing germination of the seed (Glenn Coblin, Personal Communication, 2010). Following identification of block locations, blocks were cleared of most vegetation on April 16 and 18, 2010, to simulate mowing, to an approximate height of 2.5 cm., using a nylon string trimmer. Each block was then hand raked using a metal fan leaf rake to remove as much above ground biomass litter as possible. The string trimmer was then used a second time to remove standing vegetation that was still uncut or longer than the approximate 2.5 cm. This was then followed by a second raking on all blocks. Blocks 1-3 had considerably denser above ground biomass than blocks 4-6 and thus received a third raking on April 23, 2010 to create as much surface consistency as possible across all blocks.

3.31 Herbicide Treatment

Roundup Transorb HC liquid herbicide, active ingredient Glyphosate, guaranteed 540 grams/liter (Monsanto, 2010) was used for herbicide treatment. Roundup Transorb HC is a non-selective, systemic herbicide that inhibits amino acid production, was applied to four plots per block and to establish 0.5 m non-vegetated buffers between plots and 1.0 m buffers around all plots to minimize the influence of surrounding vegetation on treatment results. It has a short length of activity in the soil (Kaufman, and Kaufman, 2007) making it a preferred herbicide for use around wetlands.

Herbicide treatments were made on May 26, 2010 and were applied with a Shurflo backpack sprayer complete with an electric pump and flat fan Teejet nozzle that facilitated consistent solution output. Application rates were in accordance with label recommendations at a ratio of 30 ml Roundup Transorb HC to 4 l of water, equivalent to .67 l of herbicide per .40 hectare (Monsanto, 2010). Buffers adjacent to and between plots were maintained with Roundup Transorb HC spot sprayed at a rate of 80 ml/5 liters water on July 11, 2010 and June 1, 2011.

3.32 Mowing Treatment

On June 14, 2010, all plots receiving a mowing treatment first had their above ground biomass reduced by using a nylon string trimmer to cut the vegetation to approximately 7.5 cm above ground level. The resulting material was removed with a leaf rake. This was followed by a further reduction in above ground biomass using a walk behind rotary mower to further reduce the vegetation height to approximately 2.5 cm. and subsequent removal with a hand rake. This same treatment was carried out on the 0.5 m buffer areas between plots and the 1.0 m perimeter buffer.

3.33 Seed Bed/Seeding Preparation

Seeding took place June 18 following as close as possible to the timing of the treatment application (i.e. recommended days after herbicide application). Species selection for the seed mix was based on the following sequential selection criteria.

1. Must be a native species.
2. Must be suitable for growing conditions within the environmental gradient in which it will be planted- i.e. riparian area.
3. Must be competitive with reed canarygrass (i.e. tall/considerable biomass)
4. Must be a grass or a forb.
5. Efforts were made to ensure a diverse mix of species was selected.

With the exception of control plots, the remaining 35 plots were prepared for hand seeding with hand tools such as rakes and hoes to create adequate loosening of the soil, thus simulating cultivation. This provided for suitable seed to soil contact at the time of seeding. Seed bed preparation was made difficult as a result of no topsoil being present on site. This was confirmed by the soil analysis which identified the soil as clay. This is an undesirable circumstance that can influence the success or failure of any seeded species.

To address the soil phosphorus deficiency and assist with seedling establishment phosphorus was applied prior to seeding. Plant Products Company Ltd. Triple Superphosphate Microfine granular fertilizer (0-46-0) was applied at a rate of 2.20 kilograms of P_2O_5 per 92.9 m^2 with a walk behind drop spreader.

This was followed by seeding the Ducks Unlimited wet meadow mix (Appendix 6- see Wark et al., n.d. for species descriptions) on June 18, 2010 at a rate of 125 pure live seeds per 0.093 m^2 or approximately 8.44 kg/0.40 ha with a walk behind drop spreader. This is double the rate recommended for drill seeding and is required because of reduced seed to soil contact obtained with broadcast seeding (Gord MacKay, Personal Communication, 2010; Wark et. al., n.d.). Seed bed preparation allowed for a seeding depth of approximately 1.25 cm. Broadcast seeding on treatment plots took place by performing two diagonal passes on each treatment plot to ensure adequate coverage. Half the recommended seed was applied with each pass in order to apply the full rate.

Subsequently the fertilizer and seed was hand raked, thus simulating harrowing into the soil surface in order for the fertilizer to be available to plant roots and to establish seed to soil contact. Finally, a 20 x 20 cm. iron tamping plate was used on all plots to simulate rolling of the seed bed. The treatment areas were then left to the vagaries of mother nature.

3.4 Vegetation sampling and measurement

The projected time of year for seeding (May 30, 2010) which is the early growth stage for both graminoids and forbs, combined with the removal of vegetation from the block areas made vegetation identification challenging as a result of the vegetation not having an inflorescence, which is commonly used for vegetation identification in these circumstances. As a result, prior to conducting a vegetation analysis of the 126 sub-samples, herbarium samples of the most common species were collected and pressed.

Samples were collected by placing a 0.5 x 0.5 meter quadrat frame off of either end of each block across the area with the most obvious species diversity. Herbarium samples were then removed, washed, and pressed. This exercise was conducted at the end of each block until a sample of each of the most common species was collected. Samples were then analyzed by the University of Manitoba Botany lab for confirmation of identification. Further identification was provided by taking five specimen samples (June 14, 2010) that were headed out, from the field to Manitoba Agriculture Crop Diagnostic Centre (2010) for species confirmation.

Vegetation analysis of the 126, 2m x 1m quadrat sub-samples commenced on May 24 and concluded on May 27. This consisted of placing a 2m x 1m quadrat frame over the each of the three, 2m x 1m sub-samples contained within each of the 42 treatment plots for a total of 126 sub-samples. The quadrat frame was divided in half by a string line to facilitate a division of the quadrat to simplify the vegetation analysis procedure by breaking the area into smaller segments. Each 1 m^2 quadrat was then visually divided into 10 sub-quadrats that encompassed 5% of the total 2 x 1m meter quadrat area again to simplify the analysis of percent coverage. Ocular estimates of the percent of visual obscurity of the ground generated by each species within each sub-sample from immediately beside and above at standing height were made and recorded immediately to provide a benchmark by which to compare future measurements and thus percent change in species as a result of treatments. Subsequent analysis in August 2010, June 2011, and August 2011 were carried out in the same manner.

This method, although having the benefit of efficiency can be both subjective and influenced by species that may be in flower or those that forms clumps resulting in high cover estimates for such species. This was overcome by being cognizant of these weaknesses and exercising caution to thoroughly analyze only each rooted species (Bullock, 1996).

All percent cover measurements of all species within each sub-sample were quantified using an exact percent notation. These values provided the benchmark for subsequent comparison of the effectiveness of various treatments employed.

Quantifying the existing vegetation was paramount to creating a set of base line data from which to work from. Base line data for each treatment plot was collected to determine percent cover of reed canarygrass and all other species within each sub-sample prior to treatment application. The percent cover measurement and thus change in percent cover as a result of treatments, was being utilized to account for the fact that the entire experiment area and does not necessarily consist of monotypic levels of reed canarygrass. Three randomly located $2m^2$ sub-samples were randomly identified within each treatment plot. Sub-sample values were used to determine a mean species abundance value for each treatment when quantifying percent cover change values. Percent cover of each species was measured by ocular estimate and was recorded once in advance of and once following treatments in year one and twice in year two. The following criteria for measurement of percent cover were employed.

Percent cover was measured as follows;

- Percent cover of each species (rooted within the sub-sample) was determined by sampling three randomly selected 2m ^{2} (2 x 1 m = 2m^2) sub-samples within all 18 m^2 treatment plots.
- All measurements were carried out in a non-obtrusive manner with the intent of maintaining vegetation integrity throughout the duration of the study.

It should be noted that four genera, *Scripus* spp., *Typha* spp. *Biden* spp. and *Trifolium* spp. were all composite measures of species abundance. In the case of *Scripus* spp. *Typha* spp. this was done at the recommendation of a taxonomy expert at the University of Manitoba who suggested that accurate identification for a lab would be challenging.

3.5 Data Analysis Procedures

3.5.1 Principal Components Analysis (PCA)

Due to the large number of variables, Principal Components Analysis (PCA) in BiodiversityR (2008) was employed to simplify and identify a smaller number of variables responsible for most of the variance contained in the original data (Dunteman, 1989). PCA was based on a covariance matrix using log transformed data (log(x + 1)). PCA was used to; a.) examine the structure of the data with respect to the east and west blocks, and b.) assess temporal trajectories of reed canarygrass change per treatment. This was essential to identifying trends in species abundance and location as per objective 1 and the effectiveness of treatment as per objective 2. It should be noted that only those species with a frequency greater than 10% were included in the analysis.

3.5.2 Biodiversity measures

Rank abundance curves (BiodiversityR (2008)) were generated for each sampling period which was critical to identifying which species were most common on site as specified in objective 1. Total species abundance (percent cover) of each species by block (east or west) was used, and thus values in some instances exceeded 100%. The x-axis was ordered from most dominant to least dominant species. Graphical output was interpreted based on the shape of the Rank abundance curves. Inverse J-shaped curves indicate systems with an unequal abundance of species. In most cases such curves represented dominance by a single species.

Shannon Weiner index was used to quantify plant species diversity, as identified in objective 1 between the east and west blocks (Pielou 1975). Shannon diversity values were arrived at from the equation

$H' = -\Sigma p_i \ln p_i$ (Magurran, 2004)

Where: p_i is the proportion of individuals found in the ith species the value of which is estimated as n_i/N.

To further quantify plant species diversity as part of objective 1, differences in species relative abundance between blocks was assessed using evenness. This measure ranges from 0 to 1 and is based on a theoretical ratio using the Shannon-Wiener index (H);

$$J' = \frac{H'}{H'_{max}}$$

Where J' = Evenness measure (0-1)

H' = Shannon Wiener function equation

H'_{max}= Maximum value of H' = log S (Krebs, 1989).

This measure is simply based on the number of species present.

3.5.3 Reed canarygrass and smooth bromegrass ratios

Throughout the course of the study it was observed that as reed canarygrass (*Phalaris arundinacea)* gained dominance, it appeared to be at the expense of smooth bromegrass (*Bromus inermis)*. As smooth bromegrass (*Bromus inermis)* was the other abundant species in both the east and west blocks, mean abundance data for the two species was analyzed at the block level by treatment summarizing all measurement dates to determine a ratio value.

3.5.4 ANOVA

Differences in reed canarygrass mean abundance data (percent cover) among treatments were statistically determined using a two-way analysis of variance (ANOVA) linear design. Factors assessed included blocks and treatments as well as an interaction effect (blocks x treatments). F-ratios were used to estimate *p*-values. Data was log transformed (Log (x+1)). Analysis was carried out using CRAN R statistical package (R version 2.15.1, 2012). Between block and treatment differences were compared using only August 2011 data that was aggregated by block (east or west) for block differences and by treatment (i.e. all control data for east and west blocks was aggregated) for treatment differences.

CHAPTER 4

RESULTS

4.1 Species Abundance

In alignment with objective 1, principal component analysis (PCA) was employed to identify the most abundant species between east and west blocks. As per the initial vegetation inventory of September 2009, PCA (Figure 4.1) illustrates, as was observed in the field, an abundance of *Trifolium* spp. (Clover) in the east blocks as compared to the west blocks where *Phalaris arundinacea* (Reed canarygrass) was more abundant. Consequently, there is no positive relationship between the existence of these two species on site as indicated by the large angle between the two vectors (Figure 4.1). As a result, species composition was substantially different in the west blocks than in the east as data points on the first axis (likely time) are overlapping, while data points are separated on the second axis (likely competition) as most data points in the east blocks are positive and the west blocks are negative.

Further to objective 1, principal components analysis (PCA) was employed to identify general species abundance differences between east and west blocks. PCA illustrates (Figure 4.2), and confirms what as was observed during the September 2009 vegetation inventory and in the field during the study, a difference in species composition between east and west blocks. The east blocks exhibit a greater abundance of *Trifolium* spp. (Clover), *Schedonorus arundinacea* (Tall fescue) and *Festuca* spp. (Fescue spp.) the first two of which have a negative projection on to the first component to which most of the variation is attributed (Axis 1- 30.6%) and the latter being positively projected to the second component (Axis 2- 24.3%). Conversely, the west blocks had greater abundance of *Bromus inermis* (Smooth bromegrass) and *Phalaris arundinacea* (Reed canarygrass) with the variation of both being more closely associated with PCA Axis 2 (24.3%) and association of the two species being very close, while the variation of *Elymus trachycaulus* (Slender wheatgrass) is largely associated with PCA Axis 1 (30.6%). Consequently, there is a negative correlation between the locations of *Trifolium* spp. (Clover), *Schedonorus arundinacea* (Tall fescue) which are clustered in the east blocks and *Bromus inermis* (Smooth bromegrass) and *Phalaris arundinacea* (Reed canarygrass) which are clustered in the west blocks possibly as a result of hydrologic or nutrient gradients existing on site. With the exception of *Elymus trachycaulus* (Slender wheatgrass), the only indigenous species with a notable vector, indigenous species are of little relevance in the community. Of the total variation, 54.9% is explained by PCA Axis 1and 2.

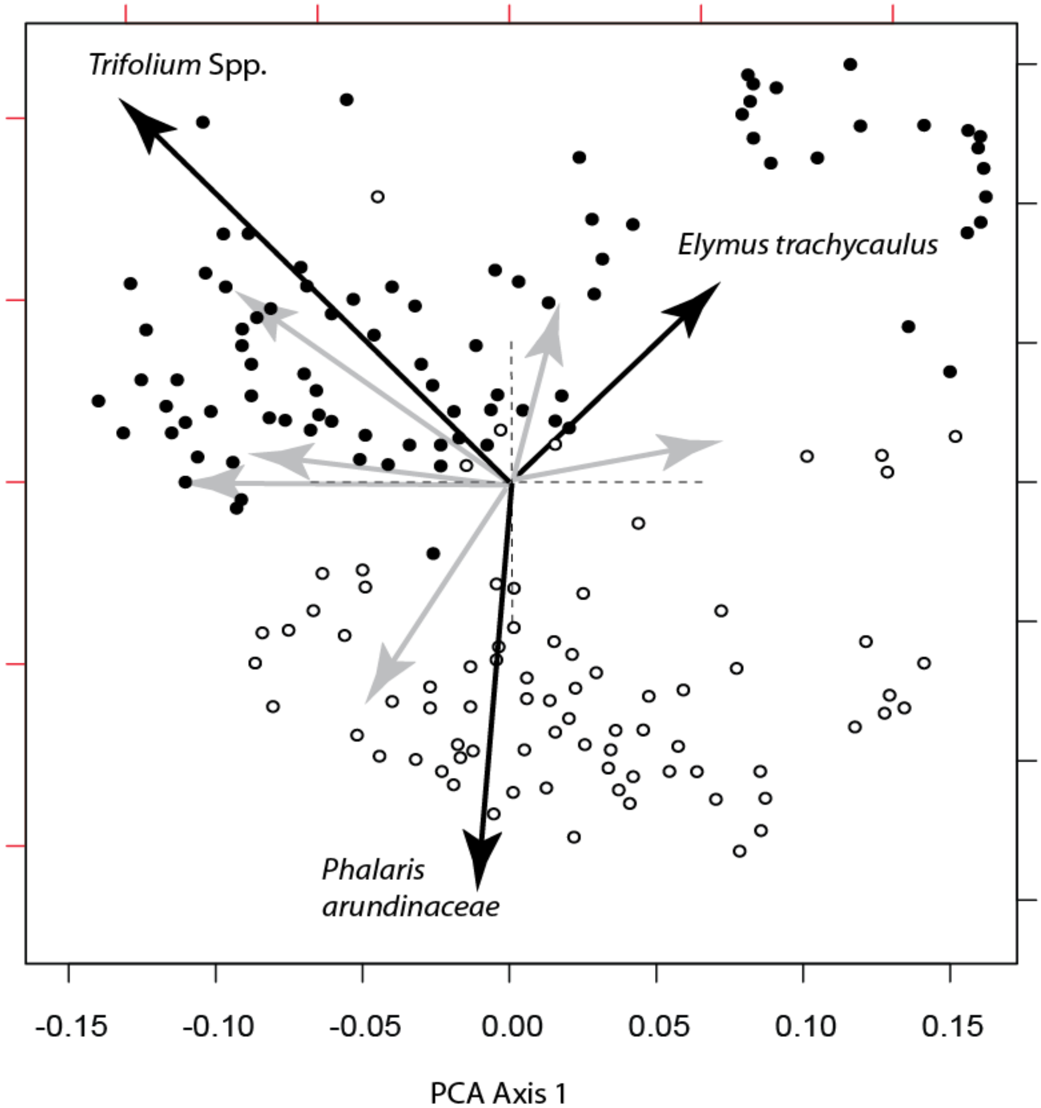

Figure 4.1. Principal Components Analysis (PCA) with data aggregated for all sites and all measurement dates, for all species with greater than 10% frequency.
PCA affirms a greater abundance of *Trifolium* spp. (Clover) and *Elymus trachycaulus* (Slender wheatgrass) in the east blocks with *Phalaris arundinacea* (Reed canarygrass) being more abundant in the west blocks.

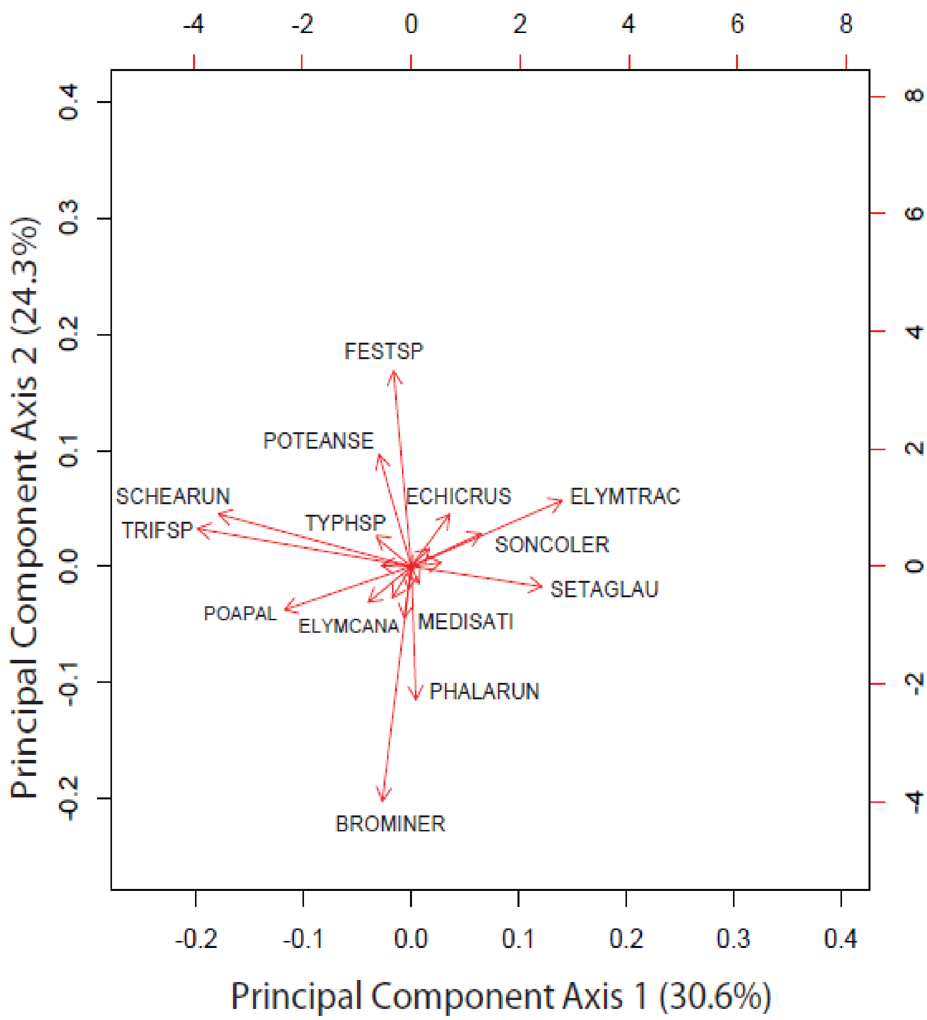

Figure 4.2. Principal Components Analysis (PCA) species trend vectors for all species with greater than 10% frequency aggregated for all sites and all measurement dates. PCA illustrates a greater abundance of *Trifolium* spp. (Clover), *Schedonorus arundinacea* (Tall fescue) and *Festuca* spp. (Fescue spp.) in the east blocks and *Bromus inermis* (Smooth bromegrass), *Phalaris arundinacea* (Reed canarygrass) and *Elymus trachycaulus* (Slender wheatgrass) more abundant in the west blocks.

4.2 Treatment Trends

As part of analyzing if treatments had an effect on reed canarygrass abundance, principal component analysis (PCA Figure 4.3) for all species at each measurement date (June 2010, August, 2010, June 2011 and August 2011) separated by block and treatment suggests that, with the exception of control plots in the west block, regardless of treatment applied or location, the responses to treatment disturbances follow distinctly similar patterns in both the east and west blocks. All treatments illustrate a return to a community status more similar to the pre-treatment state. It is likely that the first axis represents time, while the second axis represents seasonal change with treatment effect not being captured on the first or second axis. This re-iterates that treatments only had an initial effect (shift to the right from initial T1, T2, T3, and T4 values) on controlling reed canarygrass abundance and that reed canarygrass re-juvinated itself (shift to the left) and thus was resistant to treatments employed in this study. Moreover, based on the short duration of the study it is difficult to specifically identify the cause of the shifts.

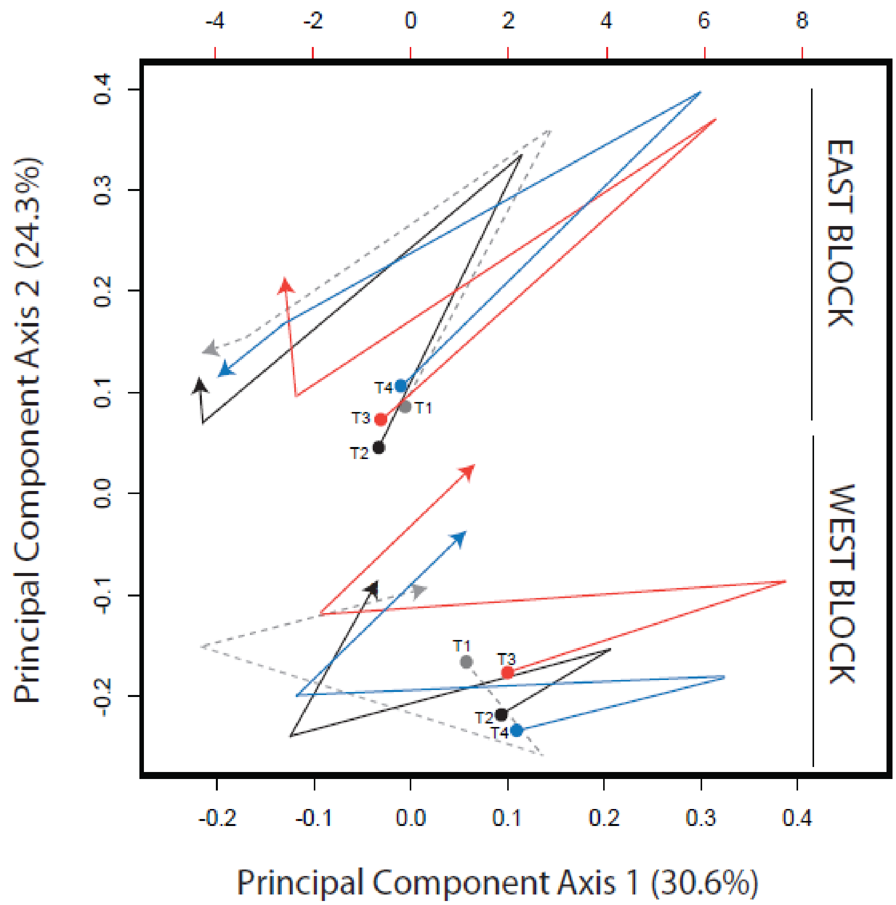

Figure 4.3. Principal components analysis trends by treatment
T1 = Control; T2 = Mowing followed by broadcast seeding; T3 = Herbicide followed by broadcast seeding; T4 = Mowing x Herbicide followed by broadcast seeding
All data for both blocks are aggregated by treatment with each line point representing respective measurement dates beginning in June 2010 and ending in August 2011. PCA showing that in all instances the community returns to status more similar to the pre-treatment state.

4.3 Dominance and diversity trends

As illustrated in Figure 4.2 (PCA species vectors) as part of determining if reed canarygrass has an effect on biodiversity, species dominance curves (Figures 4.4 a. and 4.4 b.) also indicate that a small number of species enumerated on the site were most common (Tables 4.1 a and 4.1b) suggesting that control of such species may be difficult. Table 4.2 identifies the traits associated with the most common species on the research site.

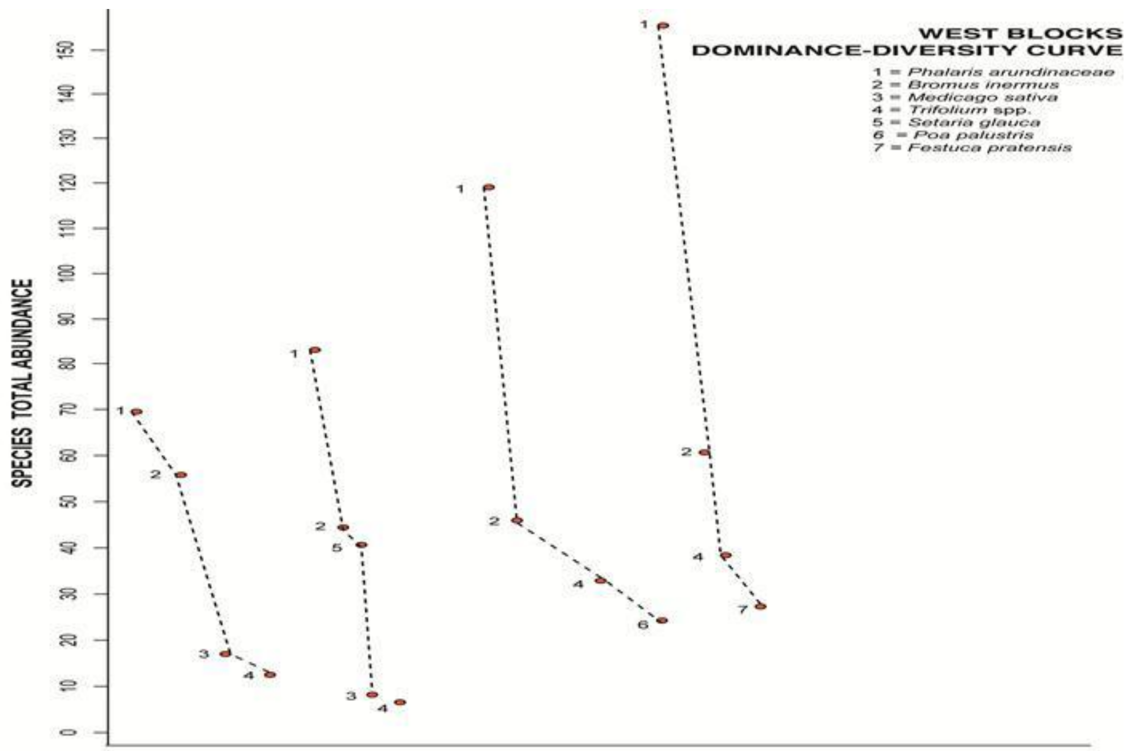

Figure 4.4 a. Seven most common species in the west blocks based on abundance. When total abundance of each species in the west blocks is measured, *Phalaris arundinacea* both increases its abundance and remains the most common species at all measurement dates. Plant species biodiversity values from June 2010 (pre-treatment) to August 2011 (final analysis) in the west blocks showed a slight decrease in from 1.47 to 1.43. Evenness values from June 2010 (pre-treatment) to August 2011 (final analysis) in the west blocks declined from 0.57 to 0.49. All species are introduced species to Canada and species 1, 4, 6, and 7 were part of the Manitoba Floodway Authority seed mix used on site.

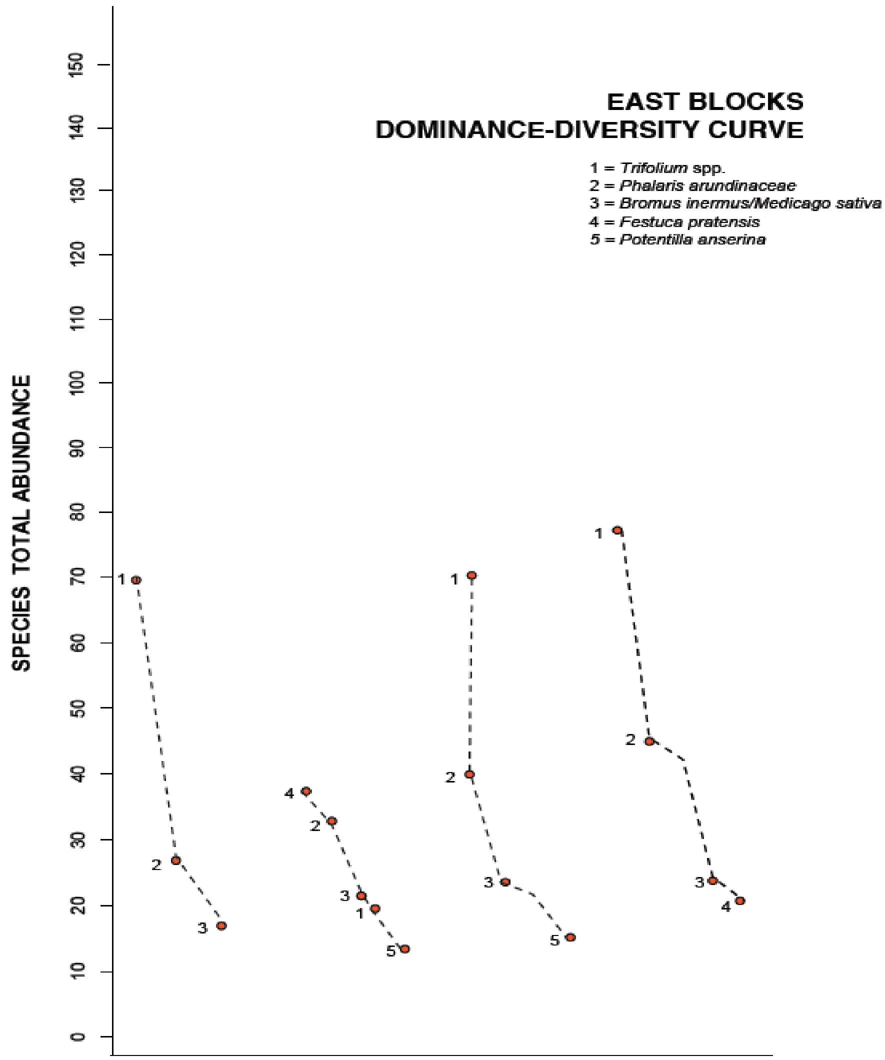

Figure 4.4 b. Six most common species in east blocks based on abundance.
When total abundance of each species in the east blocks is measured, *Phalaris arundinacea* is the second most common species at all measurement dates and increases its abundance at all measurement dates. Plant species biodiversity values from June 2010 (pre-treatment) to August 2011 (final analysis) in the east blocks increased slightly from 1.61 to 2.04. Evenness values from June 2010 (pre-treatment) to August 2011 (final analysis) in the east blocks increased slightly from 0.57 to 0.63. All species are introduced species to Canada and species 1, 2, 4 were part of the Manitoba Floodway Authority seed mix used on site. Note: *Bromus inermis* and *Medicago sativa* values although not identical were merged for visual clarity.

4.4 Plant species diversity

Plant species diversity quantification is important to identifying the effect of reed canarygrass on plant species diversity as outlined in objective 2. As revealed by the species rank abundance curves (Figures 4.5 a. and 4.5 b.) 6-7 species from a total of 31 identified during the course of the project tend to be most common. These species were found to be common to both east and west blocks throughout the growing season and from season to season (Tables 4.1 a. and 4.1 b.). Consequently, Shannon plant species diversity values range from a low of 1.47 in the west blocks in 2010 (Table 4.1 b.) to a maximum of 2.16 in the east blocks in 2011(Tables 4.1 a. and 4.1 b.). Plant species diversity values from June 2010 (pre-treatment) to August 2011 (final analysis) in the west blocks showed a slight decrease in from 1.47 to 1.43 and in the east blocks an increase from 1.61 to 2.04 (Tables 4.1 a and 4.1 b).

Evenness values ranged from a low of 0.49 in the west blocks in 2011 to a high of 0.67 in the east blocks in 2010. Evenness values from June 2010 (pre-treatment) to August 2011 (final analysis) in the west blocks declined from 0.57 to 0.49 while the east blocks increased slightly from 0.57 to 0.63 (Tables 4.1 a and 4.1 b.).

Richness in all blocks increased at each measurement date with the exception of the west blocks in August of 2011 which experienced a decrease in richness from 23 to 19 species between August 2010 and August 2011. Nonetheless, richness in all blocks increased from May 2010 through August 2011.

Table 4.1 a. Common species and plant species diversity measures JUNE 2010 and 2011

WEST BLOCKS		**EAST BLOCKS**	
JUNE 2010	**JUNE 2011**	**JUNE 2010**	**JUNE 2011**
Phalaris arundinacea L.	*Phalaris arundinacea* L.	*Trifolium spp.*	*Trifolium spp.*
Bromus inermis Leyss.)	*Bromus inermis* Leyss.)	*Phalaris arundinacea* L.	*Phalaris arundinacea* L.
Medicago sativa L.	*Trifolium spp.*	*Bromus inermis* Leyss.)	*Bromus inermis* Leyss.)
Trifolium spp.	*Poa palustris*	*Schedonorus spp.*	*Schedonorus spp.*
Vicia americana	*Elymus canadensis* L.	*Scirpus spp.*	*Potentilla spp.*
Elymus trachycaulus (Link) Malte	*Potentilla anserina* L.	*Medicago sativa* L.	*Schedonorus pratensis* (Huds.) P. Beauv.
SHANNON (H) = 1.47	SHANNON (H) = 1.74	SHANNON (H) = 1.61	SHANNON (H) = 1.98
EVENNESS (J) = 0.57	EVENNESS (J) = 0.58	EVENNESS (J) = 0.57	EVENNESS (J) = 0.65
RICHNESS = 12	RICHNESS = 20	RICHNESS = 17	RICHNESS = 21

Table 4.1 b. Common species and plant species diversity measures AUGUST 2010/2011

WEST BLOCKS		**EAST BLOCKS**	
AUGUST 2010	**AUGUST 2011**	**AUGUST 2010**	**AUGUST 2011**
Phalaris arundinacea L.	*Phalaris arundinacea* L.	*Schedonorus spp.*	*Trifolium spp.*
Bromus inermis Leyss.)	*Bromus inermis* Leyss.)	*Phalaris arundinacea* L.	*Phalaris arundinacea* L.
Setaria glauca (L.) Beauv.	*Trifolium spp.*	*Elymus trachycaulus* (Link) Malte	*Schedonorus spp.*
Elymus trachycaulus (Link) Malte	*Schedonorus pratensis* (Huds.) P. Beauv.	*Trifolium spp.*	*Bromus inermis* (Leyss.)
Medicago sativa L.	*Elymus trachycaulus* (Link) Malte	*Potentilla anserina* L.	*Schedonorus pratensis* (Huds.) P. Beauv.
Echinichloa crusgalli (L.) Beauv.	*Potentilla anserina* L.	*Sedge spp.*	*Medicago sativa* L.
SHANNON (H) = 1.73	SHANNON (H) = 1.43	SHANNON (H) = 2.16	SHANNON (H) = 2.04
EVENNESS (J) = 0.55	EVENNESS (J) = 0.49	EVENNESS (J) = 0.67	EVENNESS (J) = 0.63
RICHNESS = 23	RICHNESS = 19	RICHNESS = 21	RICHNESS = 24

Table 4.2. Common Species Traits Origin- I: Introduced; Life Cycle: A: Annual/P: Perennial; Nitrogen fixation: Y: Yes/N: No

Species	**Origin/Life Cycle**	**Reproduction/ Root system**	**Invasiveness**	**N fixing**	**Uses**
Bromus inermis	I/P	Seed/rhizomes	May become invasive when un-managed	N	Erosion control/along waterways
Medicago sativa	I/P	Seed/tap root	May become invasive when un-managed	Y	Forage
Phalaris arundinacea (seeded)	I/P	Seed/rhizomes	Invasive	N	Erosion control/along waterways
Schedonorus phoenix (seeded)	I/P	Seed/deep rooted bunch grass	May become invasive when un-managed	N	Erosion control/forage
Schedonorus pratensis (seeded)	I/P	Seed/deep rooted bunch grass	May become invasive when un-managed	N	Erosion control/forage
Setaria glauca	I/A	Seed/fibrous root system	Low	N	Not planted
Trifolium spp. (seeded)	I/P	Seed/branched taproot	Minimal	Y	Hay, pasture, soil improvement

Source: United States Department of Agriculture, 2011

As a result of four genera, *Scripus* spp., *Typha* spp. *Biden* spp. and *Trifolium* spp. being composite measures of species abundance there is the potential that plant species biodiversity values were affected by such an approach. Furthermore, in contrasting species found on site to those identified in the Classification of natural ponds and lakes in the glaciated prairie region (Stewart and Kantrud, 1971) as key indicator species for prairie wetland vegetation zones, there were eight species identified within the Parkland Mews study area that matched those suggested by Stewart and Kantrud (1971). Those species were; *Cirsium arvense*, *Poa pratensis*, *Potentilla anserina, Elymus canadensis, Festuca* spp., *Phalaris arundinacea, Scirpus* spp., and *Typha* spp. *Potentilla anserina*, *Elymus canadensis* are indigenous while *Festuca* spp., *Scirpus* spp., and *Typha* spp. could be indigenous depending on species.

4.5 Reed canarygrass/brome ratios

Analyzing of the mean abundance of each species, at each measurement date, for each treatment, indicates that with the exception of the control plots in both the east and west blocks at the initial measurement date (June 2010) there is consistently a greater abundance of reed canarygrass than smooth bromegrass regardless of location, time, or treatment. Table 4.3 illustrates the dominance of reed canarygrass over smooth bromegrass as summarized by treatment for all measurement dates.

Table 4.3. Ratio values of reed canarygrass to smooth bromegrass summarized to the block level, by treatment, for all measurement dates. Ratio values greater than 1 are bolded. Treatment1 = Control; Treatment 2 = Mowing followed by broadcast seeding; Treatment 3 = Herbicide followed by broadcast seeding; Treatment 4 = Mowing x herbicide followed by broadcast seeding

Location	**Treatment 1**	**Treatment 2**	**Treatment 3**	**Treatment 4**
West Blocks	**1.72**	**3.37**	**2.72**	**1.37**
East Blocks	**40.16**	**35.0**	**25.22**	**4.29**

Table 4.3 also suggests that treatment 2 (mowing) has the most substantial impact on reed canarygrass dominance over smooth bromegrass followed by treatment 3 (herbicide) and treatments 1 (control) and 4 (mowing x herbicide) respectively.

4.6 Changes in reed canarygrass abundance

Summaries of the mean abundance of reed canarygrass illustrate for objective 2, that all treatments, at all measurements dates, exhibit increasing abundance of reed canarygrass throughout the duration of the study in the west blocks regardless of the time of year (Figure 4.5 a.). The mowing treatment (T2) had the most substantial increase in reed canarygrass abundance in both the June 2010 to June 2011 (~ 28% to 51%) and August 2010 to August

2011 (~ 40% to 70%) comparisons and in fact reed canarygrass abundance increased more than the control. Herbicide treatments (T3) also show an increase both from June 2010 to June 2011 (~ 20-40%) and August 2010 to August 2011 (~ 20% to 50%) in mean reed canarygrass abundance. Herbicide treatments combined with mowing (T4) revealed the most promise initially with a decline from June 2010 to August 2010 (~ 28% to 20%) but similarly to other treatments reed canarygrass rejuvenates itself (~ 45% in August 2011), albeit more gradually than the other treatments. Each treatment demonstrates a trend similar to that of the control (T1).

Most treatments, at all measurements dates, demonstrated increasing trends in reed canarygrass abundance throughout the duration of the study in the east blocks regardless of the time of year (Figure 4.5 b.). The exception was the control (T1) which between August 2010 and August 2011 showed a minor decline (~ 17% to 16%). The mowing treatment (T2) had the most substantial increase in reed canarygrass abundance in both the June 2010 to June 2011(~ 10 to 17%) and August 2010 to August 2011(~ 14% to 26%) comparisons. Herbicide treatments (T3) initially show a decline from June 2010 to August 2010 in mean reed canarygrass abundance (~13% to 10%) but ultimately exhibit an increase. Herbicide treatments combined with mowing (T4) revealed the most promise between June 2010 to August 2010 (~5% to 2%), but similarly to other treatments reed canarygrass rejuvenates itself, albeit more gradually than other treatments. Nonetheless, when statistically analyzed treatment differenecs were not significant.

In analyzing objective 2 when all data from all measurement dates were aggregated, all treatments in this study showed an increasing trend in reed canarygrass mean abundance (Figures 4.6 a. and 4.6 b.). Evaluating only the August 2011 data, ANOVA at

P value <0.05 demonstrates that regardless of the trends revealed through PCA analysis, the mean abundance of reed canarygrass did not significantly change during the course of the study as a result of treatments ($P = 0.4067$). Moreover, there was no significant difference in reed canarygrass abundance at the treatment level within blocks ($P = 0.9625$). However, there was a significant difference in reed canarygrass abundance between blocks ($P = \leq 0.0001$; Table 4.4). However, it is important to note that reed canarygrass trended toward increasing abundance on site throughout the duration of the study and consequently one more year of data would have revealed if this trend continued.

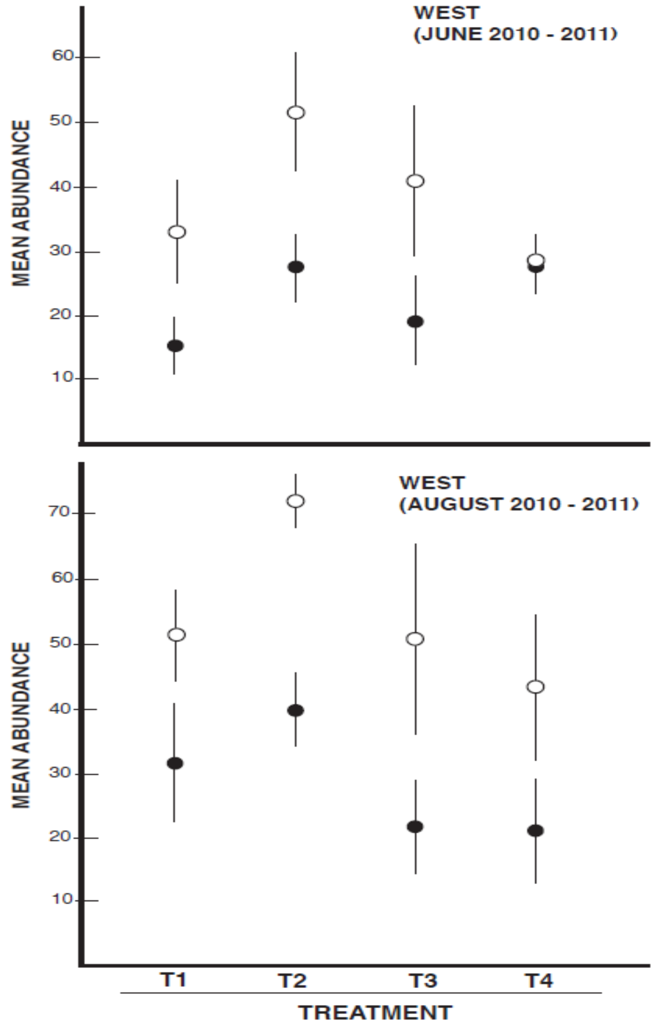

Figure 4.5 a. Trends in west block reed canarygrass mean abundance based on treatment comparing June 2010 to June 2011 and August 2010 to August 2011.
T1 = Control; T2 = Mowing followed by broadcast seeding; T3 = Herbicide followed by broadcast seeding; T4 = Mowing x Herbicide followed by broadcast seeding
All data for the west blocks are aggregated by treatment and vertical lines are standard error bars.

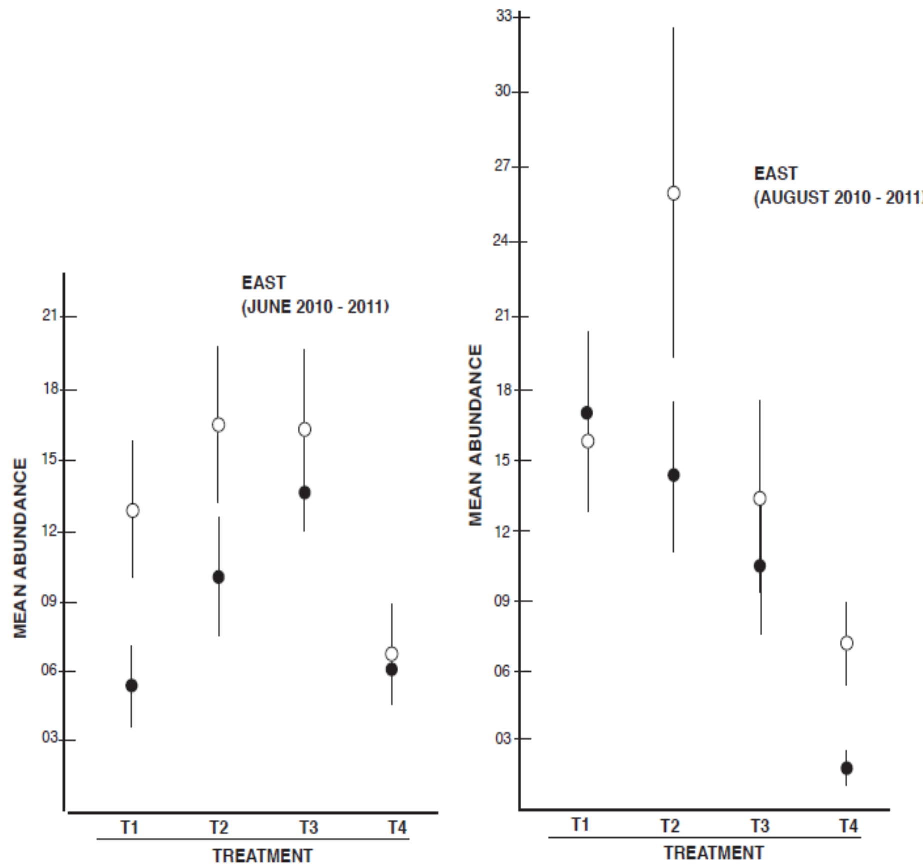

Figure 4.5 b. Trends in east block reed canarygrass mean abundance based on treatment comparing June 2010 to June 2011 and August 2010 to August 2011.
T1 = Control; T2 = Mowing followed by broadcast seeding; T3 = Herbicide followed by broadcast seeding; T4 = Mowing x Herbicide followed by broadcast seeding
All data for the east blocks are aggregated by treatment and vertical lines are standard error bars.

Table 4.4- ANOVA results for differences in treatment effects (Treatment), differences in treatments at the block level (Treatment/Block) and differences between blocks (Block)

Source	df	Sum of Squares	Mean Square	F-ratio	Probability
Treatment	3	1866.32	622.108	0.99549	0.4067
Treatment/Block	3	177.445	59.1482	0.09465	0.9625
Block	1	13578.1	13578/1	21.728	**≤ 0.0001**

CHAPTER 5

DISCUSSION

5.1 Species trends

Principal components analysis affirmed what was witnessed in the field, and that is, that there are distinct differences between the west and east blocks in community composition. The most common species in the west blocks are reed canarygrass and smooth brome grass while those most common in the east blocks are clover and fescues. This is likely due to inheritably different growing environments. Furthermore, the presence of indigenous species was negligible.

PCA analysis also revealed patterns whereby reed canarygrass, and the plant community in general, diverge from their initial state following treatment, but then converge back toward the pre-treatment state, as does the control. This may be due to treatment response but is more likely due to seasonality or possibly stabilization of the community following construction disturbance experienced when the wetland cells were constructed. Underlying this pattern is the fact that regardless of treatment, there is an aggregate increase in reed canarygrass which is similar to the findings of Kercher and Zedler (2004). Many factors, both independently and synergistically, can influence successional patterns following disturbance including; age of species in the community, disturbance frequency and intensity, the composition of the community, life history stage, resource availability, propagule availability, and the time of measurement (Armesto and

Pickett, 1985; Li et al., 2004; Renne et al., 2006). Nonetheless, reed canarygrass proliferates in this study environment.

The Parkland Mews site may be influenced by seed bank activity. Soil excavated to construct wetlands was used to build up areas adjacent to the wetland cells. Being that undisturbed sections of the west drain located both to the west and the east of the study site are inundated with reed canarygrass it is likely that the portion of the drain that traversed through the study site prior to wetland construction was also inundated with reed canarygrass. Consequently, it is assumed that in addition to reed canarygrass being a component of the seed mix used to vegetate the site, reed canarygrass seed was also prevalent in the soil seed bank. The construction disturbance, bare ground existing on site and initial competition reduced by treatments, would also create environment conducive to other species establishing, albeit temporarily. This notion is in alignment with the notion of the 'intermediate disturbance hypothesis' proposed by Grime (1973).

Grime (1973) suggests that plant species diversity is greatest when there is a moderate amount of disturbance. Consequently, when there are 'intermediate' levels of disturbance a balance in species diversity is struck in the presence of r-selected species, which are short-lived and opportunistic, and k-selected species, which are longer-lived and more competitive (Grime, 1973). As a result, if disturbances are not too severe or widespread they can contribute to biodiversity by introducing new species (Leveque and Mounolou, 2003; Zeigler, 2007). In fact, Thompson and Grime (1979) propose that in years following a disturbance that it seems reasonable to expect that the plants represented by the buried seed bank will be more prevalent. Thus those sites that contain plant types with persistent seed banks are likely dominate those sites where a disturbance occurs. Furthermore, the ultimate success of these plants will also be influenced by the environmental conditions in the year or season in which they are attempting to establish themselves. In turn, Margalef (1963) proposes that it is those ecosystems that do not experience strong outside disturbances that progress in definitive and predictable ways that lead to a mature state. Consequently, succession possesses a chronological history when the environment is stable, while outside disturbance interrupts the successional process leading to less mature ecosystems. In turn, less mature ecosystems are less predictable, less stable and possess greater uncertainty. Thus, less mature ecosystems are more susceptible to change and easily influenced by external inputs whereas mature ecosystems are more resilient to such change, which buffers the system from disturbance and thus succession (Connell and Slayter, 1977; Margelef, 1963). Concurring with this are the findings of Michaela and Schmidt (2009) who established that as disturbance intensity and frequency decreased, so did the ability of the plant community to regenerate from the seed bank and that species richness was higher in plots receiving an annual disturbance regime similar to fluctuations witnessed at Parkland Mews. Being that the Parkland Mews study site is still in its ecological infancy, greater instability and plant species diversity should be expected at this stage. This however, is not the only process influencing successional patterns.

Hubbard and Nicholson (1968) found that when establishing reed canarygrass with Ladino clover that the clover provided nitrogen for reed canarygrass and resulted in increased forage production. Consequently, in an instance like Parkland Mews where clover was part of the seed mix, nitrogen fixation has the potential to enhance reed canarygrass production on what is currently a nitrogen limited soil. Additionally, Stern and MacDonald (1962) confirmed earlier findings by Blackman and Templeman that as a result of increased grass growth due to greater nitrogen availability, grasses outcompeted clover for light and over time increased grass yields. Thus, based on the results of the Parkland Mews study it may be that *Trifolium* spp. may be more competitive than reed canarygrass at lower nitrate levels while the opposite may be true at higher nitrate levels.

Numerous other studies have also demonstrated a return of reed canarygrass following treatment disturbance (Mahanney et al., 2004; Annen et al., 2005; Foster and Wetzel, 2005; Adams and Galatowitsch, 2006; Wilcox, 2007; Miller et al., 2008).

Analysis of reed canarygrass specifically indicates that on the Parkland Mews site all treatments, at all measurements dates, demonstrated increasing trends in reed canarygrass abundance throughout the duration of the study in the west blocks regardless of the time of year. Most treatments, at all measurements dates, demonstrated increasing trends in reed canarygrass abundance throughout the duration of the study in the east blocks regardless of the time of year. The exception was the control (T1) which between August 2010 and August 2011 which showed a slight decline but was not statistically significant.

The mowing treatment (T2) had the most substantial increase in reed canarygrass abundance in both the June and August comparisons and in fact increased reed canarygrass abundance more than the control. This trend is consistent with findings of Wells (1971) and Horrocks and Washko (1971) who found that reed canarygrass tillering increased with spring mowing and those of Miller et al. (2008) who found that reed canarygrass abundance increased a short time (5 months) after mowing.

Herbicide treatments (T3) initially indicated a decline in mean reed canarygrass abundance but ultimately exhibited an increase. Other studies have revealed similar trends whereby herbicide applications initially demonstrate control of reed canarygrass above ground biomass, only to see reed canarygrass return to previous or greater than previous abundance levels (Annen et al., 2005; Wilcox et al., 2007; Miller et al. 2008).

Herbicide treatments combined with mowing (T3) revealed the most promise initially, but similarly to other treatments reed canarygrass rejuvenates itself, albeit more gradually than the other treatments. Although there is an insubstantial amount of research in this area, the findings of work by Mahaney et al. (2004), and recommendations of Stannard and Crowder (2001) and Wilson and Gerry (1995) suggest that combination treatments are the most effective but that re-application of such combinations will likely be necessary.

Each of the aforementioned treatments demonstrates a trend similar to that of the control (T1).

5.2 Reed canarygrass and plant species diversity

The results of this study suggest that reed canarygrass keeps plant species diversity at low levels. Diversity curves establish that both west and east blocks are host to a small number of species with reed canarygrass being the most common species in the west blocks and clover in the east blocks. Furthermore, the abundance of indigenous species is negligible while reed canarygrass illustrated increasing levels of abundance.

The maximum biodiversity value witnessed during the course of the study was 2.16 with a mean value of 1.77 across all blocks for all measurement events. Krebs (1989) suggests that Shannon-Wiener values seldom exceed 5 in most biological communities. Magurran (2004) suggests that values seldom exceed 4 and frequently lie between 1.5 and 3.5 with 1.5 representing locations with low species richness and evenness and 3.5 for those with greater plant species diversity. By comparing these suggested values and ranges, one can infer that plant species diversity on the Parkland Mews site is low. This is further supported by the mean evenness value of 0.59 across all blocks for all measurement events indicating dominance by a small number of species. The decrease in evenness values in the west blocks from 0.57 to 0.49 over the duration of the study infers that species biodiversity declines when an increase in Reed canarygrass is experienced as noted in the west blocks.

Although all blocks experienced an increase in richness over the duration of the study from May 2011 to August 2011, the west blocks experienced a decrease, from 23 to 19 species, between August 2010 and August 2011.

Galatowitsch et al. (1999) found that habitats containing reed canarygrass generally had reduced indigenous species diversity and this is further supported by Werner and Zedler (2002). Wilcox et al. (2007) demonstrated an initial increase in species richness following herbicide or burning treatments in combination with seeding, yet reed canarygrass rapidly regained dominance. Kercher and Zedler (2004) witnessed a reduction in richness as a result of sediment and nutrient additions associated with intermittent flooding both of which appear to favor an influx in reed canarygrass, while Green and Galatowitsch (2002) discovered declines in diversity and evenness both where reed canarygrass was present and where its proliferation was further manifested by nitrate enrichment.

Critically, a longer term (19 year) study strongly suggests that reductions in species biodiversity are largely as a result of invasive species such as reed canarygrass (Aronson and Galatowitsch, 2008) and in a study measuring $1m^2$ plots across 24 wetland communities Schooler et al. (2006) identified that reed canarygrass has a negative influence on plant community diversity while Molofsky et al. (1999) propose similar findings.

It is also believed that invasive species have played a part in influencing about half of those species that are now considered to be rare, endangered, threatened, or extinct (Bhowmik, 2005; Wetzel, 2005). Species diverse ecosystems possess greater resistance to invasion, yet once invasion does occur, it occurs rapidly and at the expense of species diversity (Wetzel, 2005).

Common species identified in the pre-treatment analysis of the research site tended to stay common throughout the course of the study, and as a result substantial fluctuations in biodiversity and evenness values were not witnessed. These common species are generally pioneer species (those capable of nitrogen fixation and invasive species). In fact, all of the common species identified are introduced species, five of the seven are grasses, while four of five grasses have the potential to become invasive when un-managed (United States Department of Agriculture, 2011) revealing a consistent theme among the common species found on site. Furthermore, many of these species were species that were seeded by the Manitoba Floodway Authority with the intent of rapid establishment to minimize erosion upon completion of dike construction.

As many of the species identified in the plots across all sample dates were also part of the floodway authority base mix, which was actually seeded upslope from the location of test plots, it appears as though erosion may have displaced the base mix seed down the slope. This may have occurred during spring runoff as dormant seeding took place in late October of 2009. As a result, base mix species tended to dominate the zone that wet meadow species were seeded. Likely adding to the successful establishment of base mix species in the wet meadow zone is the fact that those species contained in the wet meadow mix are exceptionally slow to establish and germinate. Moreover, the abundance of these species indicates that the success rate of controlling any of the species with the treatment techniques employed- spring mowing, spring herbicide, spring mowing-herbicide each followed by broadcast seeding with indigenous species had no effect on reducing *Phalaris arundinacea* or positively influencing indigenous species composition within plots.

Phalaris arundinacea and *Bromus inermis* were two abundant species across all blocks. *Phalaris arundinacea* abundance is likely the result of being seeded as part of the base mix, the high probability of a seed source from locations in the west drain upstream of the research site and nitrate loading. As for *Bromus inermis* its abundance was most probably the result of its prevalence along roadways, rights of ways, field edges and ditches in areas adjacent to the research site (Vance et al., 1999) providing a seed source. *Trifolium, Medicago sativa* L., *Schedonorus* species were also species that reoccurred both spatially and temporally. All are non-native species, were part of the Manitoba Floodway Authority base mix and commonly inhabit waste areas, ditches, roadsides and wet meadow areas (Vance et al., 1999). *Elymus canadensis* and *Elymus trachycaulus* were the only two desirable seeded species that appear to demonstrate in this instance, any ability to establish and persist. The ability of *Elymus canadensis and Elymus trachycaulus* to establish and prevail in this system is likely due to their rapid growth rate, moderate re-growth rate, adaptations to fine-textured soils, good seedling vigor and shade tolerance (USDA, 2012).

Still, results indicate that the establishment of the wet meadow mix seeded by the Manitoba Floodway Authority was unsuccessful either as a result of not establishing due to spring erosion or as a result of high bulk density or low levels of organic carbon contained in the soil which effect indigenous wet meadow species ability to establish (Galatowitsch and van der Valk, 1996). It is also possible that the wet meadow species did not establish due to being out competed for resources. Additionally, initial site conditions (Galatowitsch and van der Valk, 1996; Hausman et al., 2007; Ballantine et al.; 2012) or site location (Luckeydoo, 2006; Matthews et al., 2009; Alsfeld et al., 2010) have shown to have a significant impact on species composition on a site.

Interestingly, during the course of the study, regardless of location (i.e. west versus east block locations) reed canarygrass displayed a competitive dominance over smooth bromegrass. So much so, that the increase in reed canarygrass appears to be at the expense of smooth bromegrass, another invasive species. This is a critical finding, as it supports the notion that reed canarygrass is an aggressive specimen whose clonal traits (Lahring, 2003), horizontal arrangement leaf arrangement and prolific seed production (Kaufman and Kaufman, 2007) provides it with a competitive advantage over other species. Furthermore, it has the ability to persist on any upland area typically receiving more than 500mm of annual precipitation while able to withstand ponding for 6-8 weeks where leaves can get above the water surface (Looman, 1983). These traits coupled with its ability to outcompete indigenous species through its dense monospecific colonization, particularly in situations of high soil or water nitrogen levels illustrate that its control is very difficult (Kaufman and Kaufman, 2007).

Reed canarygrass also has a more horizontally orientated leaf orientation (which favors light absorption) than smooth bromegrass and has demonstrated substantial phenotypic variation (Maurer and Zedler, 2002; Zedler and Miller, 2003; Lavergne and Molofsky, 2007). Although no studies have indicated so, it is also possible that reed canarygrass possesses greater below ground biomass than smooth bromegrass. This would not only make it more resistant to disturbance, but also more responsive to recovery following disturbance and thus dominance under the reduced competitive environment experienced following a disturbance.

Consequently, it should not be unreasonable to suggest that one invasive species, reed canarygrass, can gain dominance over another invasive species, smooth bromegrass.. Growing adjacent to each other, reed canarygrass is obviously larger than smooth bromegrass which is likely the result of reed canarygrass being more competitive than smooth bromegrass particularly in disturbed (herbicide and/or mowing treatments) environments where competition is minimized or resources are low.

Hence, the survivability of reed canarygrass in a variety of environments and forms may preclude it from general theories, making it unique in this capacity.

Nonetheless, in a thorough analysis of research literature on the control of smooth brome in tallgrass prairies, Salesman and Thomsen (2011) found that there is considerable uncertainty surrounding the control of smooth brome with late-spring control measures such as burning, herbicide, mowing, grazing or various treatment combinations aimed at defoliation. As a result, this pattern witnessed at the Parkland Mews site may be an anomaly and thus further investigation of this finding is warranted.

5.3 Reed canarygrass treatment effects for all blocks

None of the four treatments employed exhibited any significant change in reed canarygrass populations. The results of treatments employed in this study are not surprising considering there is little evidence (Horn, 1971; van der Valk, 1992[1]; van der Valk, 1992[2] ; Molofsky et al. 1999; Shaw, 2000; Miller and Zedler 2003; Sahramaa and Jauhiainen, 2003; Kercher and Zedler 2004; Mahaney et. al. 2004; Perry and Galatowitsch, 2004; Adams and Galatowitsch, 2006; Lavergne and Molofsky 2007; Adams and Galatowitsch, 2008; Annen, 2008; Aronson and Galatowitsch, 2008) that native species when broadcast seeded in mesocosm studies have the ability to outcompete an established stand of reed canarygrass even when treatments are made in advance of seeding in an attempt to reduce reed canarygrass populations and competiveness. Note that the purpose of using only August 2011 data was i) to eliminate differences due to seasonal effect (i.e., June-August comparisons), ii) to eliminate comparisons with pre-treatment vegetation analysis (June 2010) and post treatment vegetation analysis (August 2011) (i.e., June 2010 – August 2011 comparison), and iii) to allow the community to stabilize following treatment disturbance; the August 2010 treatments plots were in a state of post-disturbance recovery and thus not phenologically comparable to the same plots the following year (i.e., August 2010 – 2011 comparison).

Although there was an increase in reed canarygrass mean abundance in the control plots over the course of the study there was no significant change in reed canarygrass mean abundance based on the August 2011 data. This increasing trend could be the result of any number of factors including; establishment of reed canarygrass seed as a result of local and upstream seed sources, clonal ramet expansion, or the gradual succession of species composition as the result of the Parkland Mews site being in its infancy. It is likely that clonal expansion played a prominent role.

The primary means of reproduction of invasive species in wetland environments is via vegetative reproduction as the result of plant fragmentation (Bhowmik, 2005; Wetzel, 2005). Most invasive emergent wetland species invade successfully as a result of possessing large structural roots and rhizomes that are not only capable of spreading new plants as in the case of rhizomes, but also have a slower turnover rates as a result of large structural roots when compared with those plants possessing finer roots. The process of invasion is most often acknowledged as one first of population expansion, followed by confinement and consolidation of the invasive species, followed by population fragmentation and integration with indigenous species. Invasive species are not always superior competitors, but often have different physiological requirements for water, nutrients and light. However, under conditions of high fertility and plant densities, the competition for light is usually the dominant force on plant species interaction. For example, *Typha* and *Phragmites* are two such invasive wetland species that possess a strong ability to compete for light reducing its potential to reach other plants by as much as 70-90% (Wetzel, 2005).

Furthermore, most successful invasive species possess aggressive and overlapping clonal reproduction and rapid growth.. The result of this aggressive growth is a continual senescence of above ground biomass that is relatively resistant to prompt degradation as a result of remaining standing or due to anaerobic conditions. It is thus presumed that the complex biological interactions that occur as a result, contribute to enhancing the success of invasive species ability to compete with indigenous species (Wetzel, 2005).

Consequently, doing nothing (control) appears to be just effective as the other three treatments employed in this study in minimizing reed canarygrass expansion demonstrating the resilience common to invasive species in disturbed systems (Folke et al., 2004; Richardson et al., 2007; Herschner and Havens, 2008).

In all instances mowing followed by broadcast seeding indicated the most consistent trend increases in reed canarygrass mean abundance across both east and west blocks on all measurement dates, although ANOVA revealed none to be significant based on August 2011 data.

The increases are likely the result of the stimulation of adventitious buds in response to decreased above-ground biomass resulting in increased light availability. Furthermore, is the natural growth response of graminoids to replacing above ground biomass in order to sustain production (photosynthesis).

Wells (1971) clearly demonstrated that species respond individually to various mowing treatments. For example, frequency of mowing, time of year and soil type all influenced how various species performed following mowing. Early spring mowing may stimulate tillering in only a few species resulting in a decrease in species diversity as is true for leaving an area uncut when only a few tall, aggressive species tend to dominate. Multiple mowing’s throughout the growing season generally appears to promote the greatest species diversity.

Lawrence and Ashford (1969) arrived at similar conclusions when they determined that reed canarygrass cut at a younger stage of development produced less above-ground biomass than more mature stands. Yet both frequency and timing of mowing events is important for a number of reasons. These include minimizing the promotion of tillering in grasses which can, if promoted, particularly in the spring, as witnessed at Parkland Mews, increase vegetative growth. This allows reed canarygrass to out-compete adjacent species thereby reducing species diversity. Conversely, infrequent mowing activity can also result in a decline in diversity (Luken, 1992).

Miller et al. (2008) found that multiple mowings (twice in summer of year one; three times in spring- summer of year two) of reed canarygrass to within 2.5 cm of the soil surface provided 88% and 90% control respectively five-months after planting. However, 14 and 17 months after planting control levels declined to 45% and 72% respectively.

Sheaffer et al. (1992) found that reed canarygrass under both drought and well watered situations is one of the most productive cool season grasses. Marten and Hoven (1980) in analyzing the persistence of four grasses under a variety of mowing regimes also found that reed canarygrass performed best in terms of persistence and the production of above-ground biomass at all cutting regimes and in fact mowing two, three or four times per year had no effect on the stand. Additionally, mowing acted as a hindrance to other species in the study such as tall fescue and orchardgrass.

Moreover, Horrocks and Washko (1971) found that clipping reed canarygrass early in its development stage (i.e. 5 or 10 cm above the soil) produced significantly more tillers than those plants clipped when the growing point was 20 cm above the soil surface and clipping three weeks following this. Subsequently, they also found that tiller production following mowing increased as a result of nitrogen fertilization as did the United States Department of Agriculture (Stannard and Crowder, 2001). Importantly, winter injury following low mowing in the fall (3.8cm) also caused the greatest winter injury to reed canarygrass when compared against intermediate wheatgrass or bromegrass (Horrocks and Washko, 1971). This provides optimism for the use of properly timed mowing at the correct height to aid in the reduction of reed canarygrass.

Therefore, mowing more frequently at the correct growing phase may reduce tiller productivity of reed canarygrass, but this is unclear and may be affected by any other number of factors and thus at this point would be considered a trial and error management tool in an attempt to reduce reed canarygrass.

Mowing treatments although likely more effective later in the season can be challenging at that time when above-ground biomass of reed canarygrass can reach heights upward of 2.1 meters (Stannard and Crowder, 2001). It appears as though mowing on its own is not a realistic approach to reed canarygrass control. Consequently, in this instance spring mowing is not advisable for the control of reed canarygrass.

Spring herbicide treatments followed by broadcast seeding initially illustrated small decreases in reed canarygrass mean abundance but ultimately exhibited an increase, although ANOVA validated that none were significant.

These small decreases may be the result of a number of factors including significant rain events and below average temperatures in the days and weeks following treatment thus minimizing herbicide translocation throughout the plant. Importantly, the timing of herbicide application (i.e. spring versus late summer-fall) may have influenced control success as well. In the spring the plant is focused on above-ground biomass production, while in the late summer the focus shifts to the consumption of energy resources toward seed and flower production. Furthermore, the late fall storage of carbohydrates in the root system and thus increased herbicide translocation to below-ground reproductive structures may all contribute to a more susceptible plant. Although herbicides can be an effective vegetation management strategy, use of such products often does little for successful succession management due to the non-selective nature of their control (Luken, 1990) as is the case with Roundup Transorb.

Wilcox et al. (2007) experienced a reduction in reed canarygrass after two glyphosate treatments but similarly to the Parkland Mews research, levels of reed canarygrass were the same as that of control plots two years following treatment. Conversely, in a two-year study Miller et al. (2008) realized excellent control of reed canarygrass when individual reed canarygrass plants were spot treated with glyphosate in the fall of each year. In fact spot treatment control levels of reed canarygrass attained at two different sites were 89% and 98% respectively, five-months after planting. However on one site in year two, control levels declined to 68% with reduced success likely attributed to the vagaries of weather, while control on the other site remained at 94%.

On experimental plots consisting of 75% to 100% reed canarygrass cover, Adams and Galatowitsch (2006) found that one herbicide (Glyphosate) treatment in either late August or late September provided better control than two herbicide applications in May. Any new growth following fall treatments was a result of new plants from the reed canarygrass seed bank as opposed to both rhizome and seedling growth following the spring application. Plots receiving herbicide applications on successive years also demonstrated a significant reduction in the reed canarygrass seed bank. Moreover, they found one year after herbicide treatments that biomass associated with reed canarygrass was still lower in plots receiving late fall herbicide treatments as opposed to spring treatments which were comparable to control plots. Miller et al. (1995) had similar findings where successive fall applications of glyphosate generated excellent control of reed canarygrass whereas any other glyphosate treatments were deemed ineffective.

Foster and Wetzel (2005) indicate that herbicide treatments were effective in reducing reed canarygrass shoot biomass for the majority of two growing seasons (37% and 19% in years one and two respectively), but reed canarygrass was as dominant in treatment plots as control plots by the end of the second season. Knezevic et al. (2004), had similar findings when evaluating the control of purple loosestrife with glyphosate.

Paveglio and Kilbride (2000) were successful with reducing reed canarygrass populations by spraying and disking in one season followed by a subsequent application of glyphosate the following season. Importantly, Wilson and Gerry (1995) found significantly greater densities of native grass seedlings in plots that were sprayed with glyphosate than those that were not. In fact, those that were not sprayed exhibited "no native seedlings".

Although Annen et al. (2005) found that a single application of a post-emergent selective herbicide can reduce the number of viable propagules and reduced above-ground biomass these effects were only prevalent in the year of application. Following this reed canarygrass, as it did at Parkland Mews, continued its expansion. Although the results of herbicide use show some promise, the timing and frequency of application require further evaluation.

One point of interest on the research site was the observation and recorded values of the prevalence of *Setaria glauca* in those plots receiving herbicide treatment particularly in blocks one, two and three. *Setaria glauca*, an introduced species, is a graminoid, member of the Poaceae family and is considered a noxious weed in the United States, Canada, and Manitoba (Manitoba Agriculture, 2010; United States Department of Agriculture, 2010). It is classified as a summer annual that does not spread horizontally thus grows in clumps particularly on fertile soils (Virginia Tech, 2010).

It is presumed that its prevalence in those plots receiving herbicide treatment was the result of the vegetative canopy being reduced to the point where conditions were conducive to the germination of *Setaria glauca* seed, which lay dormant in the soil seed bank. Dawson and Burns (1962) found that although few seeds of *Echinochloa crusgalli* (L.) Beauv. (Barnyardgrass) and *Setaria glauca* (L.) Beauv. (Yellow foxtail) germinated on the surface, most germinated from a depth of 3.75 cm. with some individuals of *Setaria glauca* germinating from a 15 cm depth. Furthermore, seventy-five percent of *Setaria glauca* seedlings grew as much as 1.8 cm in the soil all of which may contribute to *Setaria glauca's* observed ability to germinate and establish following glyphosate treatments on the Parkland Mews site (personal observation). Similarly, Miller et al. (2008) and Gerry and Wilson (1995) found a general increase in the establishment of broadleaf and annual weeds which coincided with an increase in reed canarygrass control. Moreover, Sheley at al. (2006) determined that in nearly all instances where herbicides were applied, exotic grass abundance increased, likely due to their responsiveness to an increase in nutrients as competition waned.

Spring mowing and spring herbicide followed by broadcast seeding initially trended toward reductions in reed canarygrass mean abundance but ultimately reed canarygrass prevailed and ANOVA revealed that treatment three produced no significant change in reed canarygrass mean abundance.

This immediate decline is likely the result of the plant initially being debilitated through mowing and herbicide applications. This would have a greater impact both as a result of a weaker plant in response to mowing and the herbicide application not having to contend with the dense above-ground biomass, when compared to being applied in the absence of mowing, thus creating greater herbicide penetration to the lower portions of the plant canopy.

Wilcox (2007) found that in experimental quadrats receiving a treatment of glyphosate, followed by burning and spring seeding, that reed canarygrass frequency and biomass initially decreased while seeded native species increased. However, although native species richness increased, over the course of the next two years reed canarygrass frequency and biomass increased substantially. As a result, treatments did not produce a desirable dense stand of native species as reed canarygrass regained dominance after three years of study. In fact, after only two years, reed canarygrass height and cover in treatment macroplots were similar to control plots as was illustrated at Parkland Mews. Furthermore, Wilcox (2007) found that reed canarygrass populations did not decrease as a result of clipping reed canarygrass, as a result of seeding, or through the use of a selective herbicide. Ultimately the only treatment that demonstrated reduced reed canarygrass was herbicide use combined with burning. However, this was short-lived as reed canarygrass eventually regained dominance. Wilcox (2007) found that spring herbicide applications combined with mowing were deemed to be ineffective in this study. Additionally, while reed canarygrass produces copious amounts of seed it also appears to retain its ability to maintain a level of stored food reserves that allow it to escape treatments aimed at its eradication.

Nonetheless, the findings of work by Mahaney et al. (2004) and recommendations of Stannard and Crowder (2001) suggest that combination treatments are the most effective but that re-application of such combinations will likely be necessary.

5.4 Treatment results summary

As a result, objective two of this study; “To assess the effectiveness of three treatments on reed canarygrass control”, was largely consistent with that of other studies whereby the control of reed canarygrass may be attainable with repeated combination treatments (mowing and herbicide) if other conditions (i.e. low soil nitrogen levels) and minimal disturbance are also present (Stannard and Crowder; 2001; Kercher and Zedler, 2004; Mahaney et al., 2004;) It also re-enforces the fact that rhizome geophytes are difficult to control with the application of herbicides and/or mowing treatments when such species possess the ability to regenerate above-ground biomass from adventitious buds located below ground (Wilmanns, 1993). Furthermore, is the importance of the notion that early eradication of small patches at a pre-determined threshold levels are critical the control and management of reed canarygrass (Maurer et al., 2003; Myers and Bazely, 2003).

Importantly, this study only occurred over two growing seasons. Subsequent analysis in the 2012 growing season would have most likely revealed that reed canarygrass abundance continued to increase.

CHAPTER 6

SUMMARY, CONCLUSIONS, and RECOMMENDATIONS

6.1 Summary

The findings of this study illustrate the ineffectiveness of the treatments aimed at reed canarygrass control which is supported by a significant body of literature as described in Chapter 5. In fact, reed canarygrass abundance generally increased throughout the duration of the study, only a small number of species were identified as being common throughout the site, while treatments demonstrated no significant change in reed canarygrass abundance. As a result, the findings of this study support the belief that reed canarygrass is an aggressive and dominant invasive wetland species that tolerates a wide range of treatments and environmental conditions as identified in chapter 1.

6.2 Conclusions

It is not surprising that in this study that plant species biodiversity values were low as Perry and Vanderklein (1996) have demonstrated that vegetation in created wetlands is almost always lower in density and species composition than that found in natural wetlands. Furthermore, they indicate that it may be a period of decades if at all, before a created wetland provides the functions of storing carbon and nutrients. The presence of a dominant invasive species such as reed canarygrass makes this task more daunting.

The findings of Lavoie et al., (2005) support that reed canarygrass invasions are not the result of a single event, but rather a series of simultaneous events. Their findings in

Quebec point to the likelihood of invasion being the result of anthropogenic disturbances that have produced nitrate pollution, water level fluctuations, and general construction disturbance. The latter two findings are in alignment with activities undertaken by the Manitoba Floodway Authority who both constructed the site and manage water levels at Parkland Mews. This presents a complexity of hindrances to controlling invasions of reed canarygrass in wetland environments that attempt to enhance the diversity of the wetland vegetation community particularly within the emergent species gradient.

Consequently, there is a dichotomy between the traits and adaptability of reed canarygrass and desirable indigenous species which allow reed canarygrass to prevail in spite of efforts aimed at its eradication. Its adaptability and inherent traits will need to be recognized, such that there is an awareness that its eradication will not be without considerable inputs and that current approaches aimed at its control are inadequate in their current form. It must also be acknowledged that invasive species control is often difficult, expensive, and in instances of significant invasion, impossible (Luken, 1990). Critically, Connell (1978) emphasizes that increasing the frequency and intensity of disturbances, many of which may be anthropogenically influenced, could lead to communities that consist largely of species capable of quickly reaching maturity. It again seems reasonable to predict that such occurrences would favor the further proliferation of reed canarygrass.

Numerous and multiple ecosystem disturbances including nitrogen loading, flooding, sedimentation, mowing and herbicide applications facilitate the invasion of reed canarygrass. In fact, the literature strongly suggests that flooding (Barclay and Crawford, 1982; Conchou and Patou, 1987; Klimesova, 1994) and nutrient loading (Wilson and

Gerry, 1995; Green and Galatowitsch, 2002; Kercher and Zedler, 2004; Martina and von Ende, 2008) are primary drivers of reed canarygrass proliferation. Consequently, managing disturbance regimes in a way that deters reed canarygrass is critical to the establishment of desirable species. Furthermore, it has also been recognized that although species invasion is often correlated to disturbance, differences in soil pathogens or biota, such that there is no pathogenic impact on the invader, may in fact be facilitating the invasiveness of some species outside of their native range (Hierro et al., 2006; Reinhart and Callaway, 2006).

Exacerbating this phenomena, is the fact ecosystem resilience is also influenced by disturbance history and in some instances may be required to maintain ecosystem resilience to more severe disturbances (Davies et al., 2009). As such, those systems that are stressed due to disturbances that are too severe, inadequate, or non-historical place stresses on ecosystems that provide opportunities for the invasion of introduced species (Herschner and Havens, 2008). This is particularly true in today's anthropogenic landscapes, where frequency, magnitude and duration of disturbance regimes has been altered thus influencing the resilience and success of the retention of historical ecosystem patterns and communities (Folke et al., 2004; Richardson et al., 2007).

Invasive species are a threat to wetland ecosystems that require management strategies aimed at eradicating or controlling their invasion and influence on ecosystem diversity. However, understanding the species or site characteristics which make an ecosystem susceptible to invasion is complex. This is likely especially true for the Parkland Mews study site where considerable disturbance occurred during construction activities. Predicting which species will invade and thrive in specific ecosystems is confounded by innumerable factors and interactions. However, there exists a paradox whereby natural disturbance regimes are required for the management of indigenous species, but at the same time often provides the opportunity for the introduction of invasive species.

There are a multitude of challenges facing the manager tasked with controlling invasive species. Identifying threshold levels at which it is still both economically and logistically feasible to eradicate an invasive species from an ecosystem is essential to successful control. A lack of research findings results in the inability to plan accordingly, particularly if one is considering a logical decision making process that is analytical and process oriented. Consequently stakeholder involvement, goal identification, consensus, cost-benefit analysis, an extensive and appropriate plan, adequate resources, education, research, and adaptive management are all likely to be components of a successful invasive species control program. The challenge for humans in retaining the diversity of the planet is not so much a natural one but a functional and thus philosophical one. Where one individual views earth’s nature and resources as expendable, others see them as requiring largely complete preservation, and there are those who view the earth’s nature and resources to be consumed at a sustainable level. Although the former two schools of thought can provide balance in this debate, it is likely that the latter approach that should be considered when evaluating means by which to reduce the impact of invasive species and the retention of biodiversity (Elton, 1958).

Improving identification and interception of those species that are inconspicuously introduced through activities like trade, initial testing of impacts of those species that are to be consciously introduced, and records of the influence of both natural and introduced disturbances on the proliferation of the introduced species are elementary approaches to limiting the impact of invasive species (Myers and Bazely, 2003).

Last is the requirement for government and government agencies such as the Manitoba Floodway Authority (who constructed and seeded the site and are responsible for water managing water levels) to take responsibility for better controlling the introduction of invasive species most of which have been intentionally introduced for the benefit and/or pleasure of humans. Ultimately, invasive species management is a challenge created by humans that must now be managed by humans. This initiative is the responsibility of all levels of government for it is they who currently regulate global and local trade and transportation of goods and possess the capacity to impose sanctions on the identification and introduction of undesirable species. Yet this presents another set of challenges including; changing political parties, economic wherewithal, public pressure, shifting priorities, and political will, that dot the political landscape and thus influence governments' ability to be an active and consistent participant in the task of controlling invasive species. Thus, public education, communication and awareness, and ecological and economic quantification of the impact of invasive species are required to bring the issue of invasive species to the environmental, public and political forefront (Bhowmik 2005). This can be accomplished through post-secondary research and the involvement of stakeholders such as the agricultural community, invasive species groups, and non-governmental organizations. Further documentation, research and experimentation are required to exploit the characteristics of specific invasive species to further the development of eradication and management strategies aimed at such species (Myers and Bazely, 2003).

More specifically, Parkland Mews must decide whether it is committed to reed canarygrass reduction and management and furthermore whether to focus on vegetation management (short term control) or succession management. Had another year of data been collected, it is likely that reed canarygrass abundance would continue to illustrate an increase. As a result, it is unlikely that there is any probable potential for Parkland Mews to manage reed canarygrass with the current upstream seed source, the high potential for nutrient and sediment laden runoff to be introduced to the Parkland Mews ecosystem, fluctuating water levels and its already substantial establishment on site.

Parkland Mews, although existing as an entity for some time, represents a new site ecologically. The intent of the entity, infrastructure, and resident ecosystem is for the captive breeding and research of Peregrine falcons. Consequently, the resident wetlands and surrounding vegetation are secondary to main objective of the operation. Yet, the function of this ecosystem is inextricably linked to the Peregrines, as it is intended to provide an opportunity for the Peregrines to obtain the avian hunting skills required to survive on their own. However, Peregrine sustenance is supported by a wide range of avian species and thus they are not particularly selective when it comes to their prey. As a result, maintaining a diverse array of indigenous vegetative species with the intent of attracting a broad range of avian species may be in vain based on the invasive nature of reed canarygrass and the apparent low requirement of the Peregrines to have access to such avian diversity. Consequently, the results of this study may allow Parkland Mews to re-define its vegetation management goals.

6.3 Recommendations

6.31 Management of reed canarygrass in existing vegetation communities

Active management of indigenous species communities that are threatened by reed canarygrass requires a multifaceted approach applying treatments to small patches early prior to their spread.

Multiple mowings in the late summer just before flowering and again before winter at a low mowing height with the intent of weakening the plant at the seed set stage and reducing the probability of complete winter survival (Lawrence and Ashford, 1969; Stannard and Crowder, 2001) seems to demonstrate the most promise.

Moreover, approaches suggested by Knezevic et al. (2004) and Sheley et al., (2006) who recommend diverse and repetitive management techniques aimed at reed canarygrass control and those of Ramula et al. (2008) who suggest that with long-lived species, like reed canarygrass, both survival and growth periods must be targeted with the intent of simultaneous reductions with repeated control measures, are likely to be required to demonstrate management success. As a result, control strategies that utilize herbicides which show an initial trend toward reduction may need to be combined with other treatments (i.e. mowing and re-seeding) and repeated which could conceptually result in a reed canarygrass reduction pattern as conceptually illustrated in Figure 6.1.

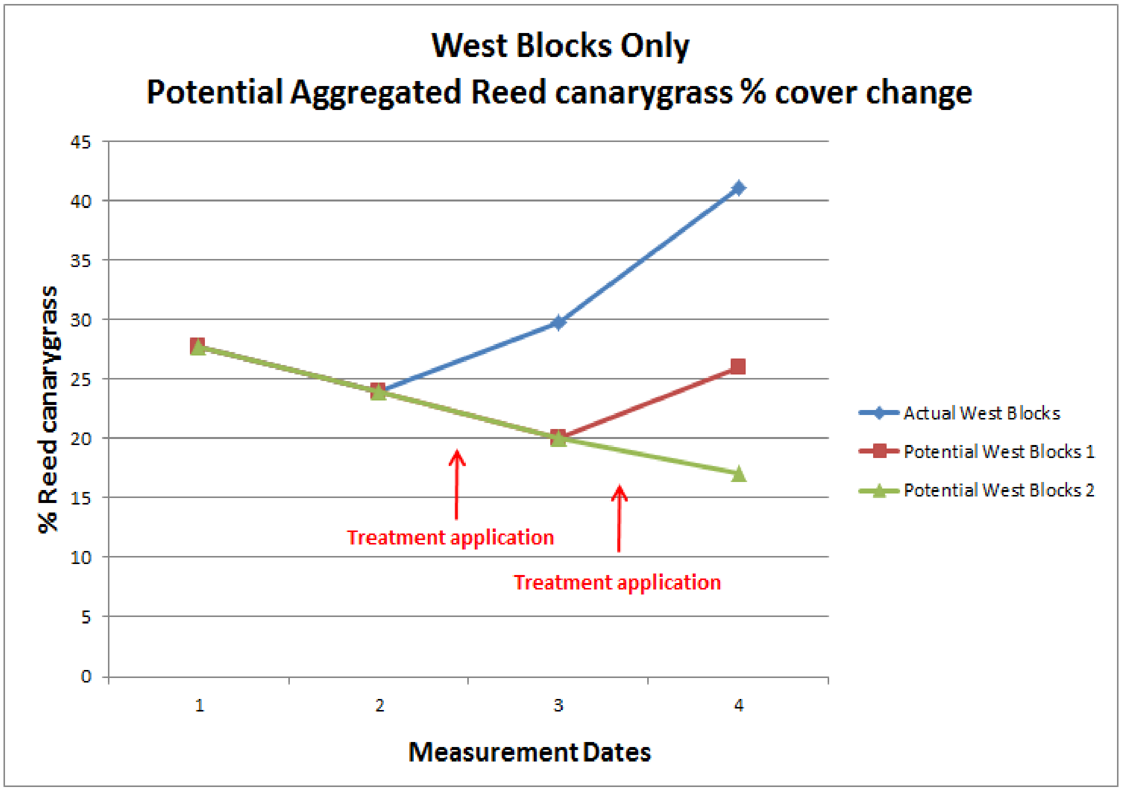

Figure 6.1. Conceptual image of reed canarygrass mean abundance levels with repeated treatment applications made at the point where reed canarygrass abundance levels are reduced following previous treatment applications. It is suggested that for effective management of reed canarygrass that combined and repeated treatment applications be timed to coincide with reed canarygrass resurgence following initial treatments.

6.32 Management philosophy

What should be identified is a management philosophy that organizes an approach to managing Parkland Mews and other ecosystems challenged by invasive species. The philosophy and associated strategies should be goal oriented, realistic, attainable, measureable (including threshold levels) and useful. The development and implementation of an action plan followed by monitoring of progress and adaptation as a result of observation is the only effective means of identifying both successes and failures.

Planning is also critical to address a seemingly unacknowledged challenge and that is the reality of a narrow time frame for the seeding of indigenous species; a three week window (June 1-June 21) that is both dry enough to prepare a seed bed and moist enough for germination in areas adjacent to wetlands can be challenging in the absence of water level control.

Consequently, vegetation manager’s need to consider a multitude of techniques including proper wetland management coupled with limiting resource availability, herbicide use, mowing, grazing, burning, soil and water testing, and inundation and drawdowns in wetland environments. Moreover, it is important that ecosystem, and in this instance wetland managers, establish threshold levels whereby control measures are implemented prior to the invasion progressing too far.

6.33 Legislation

Using reed canarygrass as a forage species (economic goal), and as a means of bank stabilization or controlling erosion (engineering goals) conflicts with ecosystem management goals. Anthropogenic priorities seem to preclude the urgency that should be attributed to mitigating invasive spread of reed canarygrass not only adjacent to wetlands but in upland and rangelands as well. Although there is more to learn with regard to eradicating reed canarygrass where it is already established, from a broader ecosystem perspective, the use of reed canarygrass as either a forage crop or as a means of erosion control, by agencies such as the Manitoba Floodway Authority should be eliminated. This will require awareness and education, political will and leadership and as such alternative approaches to bank stabilization and erosion control should be acknowledged. In fact, approaches such as the use of straw mulch or temporary cover crops such as oats (or possibly annual ryegrass) have been employed by the Manitoba Floodway Authority on portions of the floodway expansion project. Based on the results of this study and the literature, controlling reed canarygrass after it has established appears futile.

6.34 Water level control

The ability of Parkland Mews and other wetland systems to secure some form of water level control particularly in light of the fact that the site contains multiple wetland/floodplain complexes would allow for greater water level management flexibility. This would further provide the Parkland Mews site with the ability to take a more active role in the management of vegetation species existing adjacent to the wetland cells, with the intent of encouraging a more diverse vegetation inventory (Fredrickson and Taylor, 1982). However, this management approach will only be successful if both water level control and other means are used to control the establishment and persistence of reed canarygrass which not only exists on site in the vegetative form but also within the seed bank. This appears to be an issue for Parkland Mews with its hydrologic link to an upstream seed source and its inability to control water levels which are currently managed by the Manitoba Floodway Authority for flood protection purposes.

6.35 Water Quality

Although defining ‘ideal’ water quality parameters is not likely to happen, utilizing the research available for reed canarygrass combined with baseline data requirements for other vegetation species will allow for the comparison of said parameters to the data collected from Parkland Mews. This would allow for the identification of apparent problems and potential recommendations to mitigate these challenges.

6.36 Seeding

Preliminary site preparation is paramount to the successful establishment of native prairie species but is particularly so when pre-existing invasive species such as reed canarygrass, quackgrass, and birds' foot trefoil already exist on site as was the case with the two former species on the Parkland Mews site (Svedarsky et al., 2002*).* Thus, a multi-year approach to control and continual after care of sites consisting of substantial amounts of reed canarygrass are required to be successful.

Proper seed bed preparation is critical to proper seed placement and thus the success of seeded species that are considered fastidious. This includes minimizing the amount of above ground biomass that presents difficulties both to seed bed preparation and generating competition between existing vegetation and that which is seeded. A 'scorched earth policy' may be the best option for the control and management of an invasive species. This could consist of either herbicide applications in combination with cultivation or the use of fire to establish a vegetation-free seed bed that can be adequately prepared in terms of grading and seed bed preparation by eliminating above ground biomass first. Tillage combined with herbicide treatments may be a reasonable approach if vegetation selectiveness is not a concern.

Drill seeding should be the chosen method of seeding to increase seed to soil contact with the intent of increasing germination success. Furthermore, those species that germinate and establish rapidly and thus are competitive that are best suited to developing dense canopies that reduce light availability to reed canarygrass thus minimizing its competitiveness.

The aforementioned approaches to seed bed preparation would require a new start at Parkland Mews.

6.37 Expanded research

There is a lack of data that demonstrate successful suppression of reed canarygrass. Furthermore, no threshold level of reed canarygrass has been identified after which control of said species becomes cost and time prohibitive. In fact, it appears that the control and management of reed canarygrass has taken a somewhat onerous and ineffective approach to its eradication. While a few studies have included or have analyzed the effects of sedimentation, phosphorus, and nitrogen (Horrocks and Washko, 1971; Wilson and Gerry, 1995; Green and Galatowitsch, 2002; Miller and Zedler, 2003; Kercher and Zedler, 2004; Mahaney et al. 2004; Lavoie and Dufresne, 2005; Martina and von Ende, 2008) it is widely acknowledged that these constituents have a substantial influence on the invasibility and success of reed canarygrass communities. Furthermore, it is widely accepted that pH and salinity have an effect on plant growth. Consequently, analyzing the aforementioned constituents in both soil and water can provide insight into the potential for indigenous vegetation success or alternatively the success of reed canarygrass. Further research on the use of herbicides or conversely more holistic approaches should also be considered.

The use of herbicides and associated application methods could be further explored. For example, the use of non-selective herbicides and or the use of a wick applicator are worthy of further study. Still, although management of invasive species has focused primarily on their control, a shift toward a more holistic approach should be considered including managing influences such as disturbance and resource availability. Moreover, is the requirement for the further investigation on the effects of plant competition for light.

The ability of reed canarygrass to produce predominantly vertically oriented leaves from its stems increases the leaf surface area exposed to sunlight, creates a favorable light absorption angle, and casts more shade on adjacent and underlying species (personal observation). Antieau (1998), Maurer and Zedler (2002) and Kim et al. (2006), all found that reed canarygrass survival and biomass was lower in situations of reduced light, suggesting that this may be a means of control. However, when light is reduced for reed canarygrass it is likely reduced for many other desirable species as well. Observation of the impact of climate change is also worth noting.

Climate change may also have future effects on the persistence of invasive species. If the climate is no longer suitable for the invasion and persistence of certain invasive species this may provide an opportunity for restoration (Bradley and Wilcox, 2009) and thus adaptive management will be essential. However, this is a long term approach and one that is unknown. Unfortunately, vegetation managers require an immediate response to what is clearly a dominant invasive plant in the case of reed canarygrass.

6.38 Aerial Imagery

In this study neither analysis of the raw aerial imagery on its own, nor with the use of eCognition object based analysis on its own, or through cross-referencing eCognition imagery with data collected on the ground for June 14, 2011 imagery and data, provided any ability to use the aerial imagery as a means of identifying Reed canarygrass in this study. However, the use or aerial imagery from two subsequent years may improve the potential of this approach.

It is recommended that if a similar aerial imagery approach were to be investigated again, that an image be collected in August of one growing season for comparison against a second image collected in August of second growing season. This was not possible in this instance due to not having project funding secured until September of 2011.

6.39.3 Biological control

Biological control agents may be the preferred approach to control for some managers but there does not appear to be any immediate promise for the biological control of reed canarygrass. A lack of data related to insect control (Galatowitsch et al., 1999), a lack of natural enemies and cultivation of varieties for use as forage species has resulted in breeding to reduce or eliminate the effect of biological agents. This coupled with the potential that biological introduction intended to assist with eradication in invasion circumstances increases the potential for invasion into environments where reed canarygrass is being used for erosion control and or forage production. Moreover, the wide diversity of genotypes that exist within reed canarygrass populations combined with their phenotypic plasticity present challenging circumstances with regard to biological control.

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APPENDICES

Appendix 1 – Manitoba Floodway Authority Base mix species list

Base Mix						
Family	**Botanical Name**	**Common Name**	**Life Cycle**	**Native/Non-Native**	**Wetland Desirability (D-Desirable) ND-Non-desirable)**	**Cool Season (C3) or Warm Season (C4)**
Fabaceae	*Trifolium repens* L.	Alsike Clover (5%)	Perennial	Non-Native	ND	C3
Fabaceae	*Trifolium pratense* L.	Red Clover (5%)	Perennial	Non-Native	ND	C3
Fabaceae	*Astragalus cicer* L.	Cicer Milk Vetch (5%)	Perennial	Non-native	ND	C3
Poaceae	*Agropyron dasytachyum* (Hook.) Scribn	Northern Wheatgrass (15%)	Perennial	Native	D	C3
Poaceae	*Festuca rubra*	Creeping Red Fescue (15%)	Perennial	Non-Native	ND	C3
Poaceae	*Thinopyrum ponticum*	Tall wheatgrass (15%)	Perennial	Non-Native	ND	C3
Poaceae	*Phalaris arundinacea* L.	Reed Canary Grass (10%)	Perennial	Native/Non-Native	ND	C3
Poaceae	*Agropyron trachycaulum* (Link) Malte	Slender Wheatgrass (10%)	Perennial	Native	D	C3
Poaceae	*Phleum pretense* L.	Timothy (10%)	Perennial	Non-Native	ND	C3
Poaceae	*Festuca arundinaceae*	Tall fescue (10%)	Perennail	Non-Native	ND	C3

Source: Manitoba Floodway Authority. (2009). *Seed Specifications.* Provided by Rick Hay and Brian Lacey, October 2, 2009.

Appendix 2 - Floodway Authority Wet meadow mix species list

Wet meadow Mix						
Family	**Botanical Name**	**Common Name**	**Life Cycle**	**Native/Non-Native**	**Wetland Desirability (D-Desirable) ND-Non-desirable)**	**Cool Season (C3) or Warm Season (C4)**
Poaceae	*Pascopyrum smithii* (Rydb.) A. Love	Western Wheatgrass (12.7%)	Perennial	Native	D	C3
Poaceae	*Spartina pectinata* Link	Cord Grass (25%)	Perennial	Native	D	C4
Poaceae	*Beckmannia syzigachne* (Steud.) Fern.	Slough Grass (12.7%)	Annual	Native	D	C3
Poaceae	*Scholochloa festucacea*	Whitetop (12.7%)	Perennial	Native	D	C3
Poaceae	Agrostis scabra	Tickle grass (1.3%)	Perennial	Native	D	C3
Poaceae	*Glyceria grandis*	Tall Manna Grass (2.6%)	Perennial	Native	D	C3
Poaceae	*Poa palustris* L.	Fowl Blue Grass	Perennial	Native	D	C3
Poaceae	*Panicum virgatum* L.	Switchgrass (6.3%)	Perennial	Native	ND	C4
Poaceae	*Thinopyrum ponticum*	Tall Wheat Grass (12.7%)	Perennial	Non-native	ND	C3
Poaceae	*Agropyron trachycaulum (Link) Malte*	Slender Wheatgrass (12.7%)	Perennial	Native	D	C3

Source: Manitoba Floodway Authority. (2009). *Seed Specifications.* Provided by Rick Hay and Brian Lacey, October 2, 2009.

Appendix 3- September 2009 Vegetation Inventory and Descriptions- Cell Two

Cell Two						
Family	**Botanical Name**	**Common Name**	**Life Cycle**	**Native/Non-Native**	**Wetland Desirability (D-Desirable) ND-Non-desirable)**	**Cool Season (C3) or Warm Season (C4)**
Altismataceae	*Alisma trivial Pursh*	Broad-leaved Water-Plantain	Perennial	Native	D	C3
Asteraceae	*Ambrosia artemesiifolia* L.	Common Ragweed	Annual	Native	ND	C3
Brassicaceae	*Synapsis arvensis* L. spp. *arvensis*	Wild Mustard	Perennial	Native/Non-native	ND	C3
Brassicaceae	*Thlapsi arvense L.*	Stinkweed	Annual or Winter Annual	Non-Native	ND	C3
Brome Family	*Bromus inermis Leyss*	Smooth Brome	Perennial	Non-Native	ND	C3
Composite	*Mentua arvensis* L.	Many Flowered Aster	Perennial	Native	D	C3
Composite	*Sonchus arvensis* L.	Perennial Sow-Thistle	Perennial	Native	ND	C3
Composite	*Cirsium arvense* (L.) Scop.	Canada Thistle	Perennial	Non-Native	ND	C3
Composite	*Solidago spp.*	Goldenrod spp.	Perennial	Native	D	C3
Cyperaceae	Unknown	Bulrush spp.	Perennial	Unknown	Unknown	Unknown
Cyperaceae	Unknown	Sedge	Perennial	Unknown	Unknown	Unknown
Fabaceae	*Vicia americana*	American Vetch	Perennial	Native	D	C3
Lamiaceae	*Mentura arvensis* L.	Wild Mint	Perennial	Native	D	C3
Legume	*Medicago sativa* L.	Alfalfa	Perennial	Non-Native	ND	C3
Legume	*Melilotus alba* Desr.	White Sweet-Clover	Perennial	Non-Native	ND	C3
Legume	*Trifolium pretense* L.	Red Clover	Perennial	Non-Native	ND	C3

Legume	*Petalostemon purpureum* (Vent.) Rydb.	Purple Prairie Clover (1 Plant)	Perennial	Native	D	C4
Plantain	*Plantago major* L.	Common Plantain	Perennial	Native	ND	C3
Poaceae	*Becjmannia syzigachne (Steud) Fernald*	Slough Grass	Annual	Native	D	C3
Poaceae	*Phalaris arundinacea* L.	Reed Canary Grass	Perennial	Native/Non-Native	ND	C3
Poaceae	*Agropyron trachycaulum*	Slender Wheat Grass	Perennial	Native	D	C3
Poaceae	*Agrostis scabra* Willd	Hair Grass	Perennial	Native	D	C3
Poaceae	*Avena fatua* L.	Wild Oat	Annual	Non-Native	ND	C3
Poaceae	*Beckmannia syzigachne* (Steud.) Fern.	Slough Grass	Annual	Native	D	C3
Poaceae	*Elymus Canadensis* L.	Canadian Wild Rye	Perennial	Native	ND	C3
Poaceae	*Hordeum jubatum* L.	Wild Barley	Perennial	Native	ND	C3
Poaceae	*Setaria viridis* (L.) Beauv.	Green Foxtail	Annual	Non-Native	ND	C4
Poaceae	*Festica arundinacea*	Tall Fescue	Perennial	Non-Native	ND	C3
Polygonaceae	*Rumex crispus* L	Curled Dock	Perennial	Non-Native	ND	C3
Roseaceae	*Potentilla arguta* Pursh	White Cinquefoil	Perennial	Native	D	C3
Salix spp.	*Saliz spp.*	Willow spp.	Perennial	Native	D	C3
Salix spp.	*Populus tremuloides* Michx.	Aspen Poplar	Perennial	Native	D	C3
Typhaceae	*Typha latifolia* L.	Common Cattail	Perennial	Native	D	C3

September 2009 Vegetation Inventory- Cell One

Cell One						
Family	**Botanical Name**	**Common Name**	**Life Cycle**	**Native/Non-Native**	**Wetland Desirability (D-Desirable) ND-Non-desirable)**	**Cool Season (C3) or Warm Season (C4)**
Brassicaceae	*Synapsis arvensis* L. spp. *arvensis*	Wild Mustard	Perennial	Native/Non-native	ND	C3
Composite	*Aster pansus* (Blake) Cronq.	Many Flowered Aster	Perennial	Native	D	C3
Composite	*Sonchus arvensis* L.	Perennial Sow-Thistle	Perennial	Non-Native	ND	C4
Cyperaceae	*?*	Bulrush spp.	Perennial	Native	D	C3
Fabaceae	*Vicia americana*	American Vetch	Perennial	Native	D	C3
Legume	*Medicago sativa* L.	Alfalfa	Perennial	Non-Native	ND	C3
Legume	*Melilotus alba* Desr.	White Sweet-Clover	Perennial	Non-Native	ND	C3
Legume	*Trifolium pretense* L.	Red Clover	Perennial	Non-Native	ND	C3
Poaceae	*Phalaris arundinacea* L.	Reed Canary Grass	Perennial	Non-Native	ND	C3
Poaceae	*Agrostis scabra* Willd	Hair Grass	Perennial	Native	D	C3
Poaceae	*Avena fatua* L.	Wild Oat	Annual	Non-Native	ND	C3
Poaceae	*Elymus Canadensis* L.	Canadian Wild Rye	Perennial	Native	D	C3
Poaceae	*Setaria viridis* (L.) Beauv.	Green Foxtail	Annual	Non-Native	ND	C4
Poaceae	*Festica arundinacea*	Tall Fescue	Perennial	Non-Native	ND	C3
Polygonaceae	*Rumex crispus* L	Curled Dock	Perennial	Non-Native	ND	C3
Roseaceae	*Potentilla alba* L.	White Cinquefoil	Perennial	Native	D	C3

September 2009 Vegetation Inventory- Floodplain two

Floodplain Two						
Family	**Botanical Name**	**Common Name**	**Life Cycle**	**Native/Non-Native**	**Wetland Desirability (D-Desirable) ND-Non-desirable)**	**Cool Season (C3) or Warm Season (C4)**
Composite	*Sonchus arvensis* L.	Perennial Sow-Thistle	Perennial	Non-Native	ND	C4
Composite	*Cirsium arvense* (L.) Scop.	Canada Thistle	Perennial	Non-Native	ND	C4
Legume	*Medicago sativa* L.	Alfalfa	Perennial	Non-Native	ND	C3
Legume	*Melilotus alba* Desr.	White Sweet-Clover	Perennial	Non-Native	ND	C3
Poaceae	*Phalaris arundinacea* L.	Reed Canary Grass	Perennial	Non-Native	ND	C3
Poaceae	*Elymus Canadensis* L.	Canadian Wild Rye	Perennial	Native	D	C3
Poaceae	*Setaria viridis* (L.) Beauv.	Green Foxtail	Annual	Non-Native	ND	C4
Poaceae	*Festica arundinacea*	Tall Fescue	Perennial	Non-Native	ND	C3
Polygonaceae	*Rumex crispus* L	Curled Dock	Perennial	Non-Native	ND	C3
Salix spp.	*Salix spp.*	Salix spp.	Perennial	?	D	C3
Typhaceae (Cattail Family)	*Typha latifolia* L.	Common Cattail	Perennial	Native	D	?

September 2009 Vegetation Inventory- Floodplain one

Floodplain One						
Family	**Botanical Name**	**Common Name**	**Life Cycle**	**Native/Non-Native**	**Wetland Desirability (D-Desirable) ND-Non-desirable)**	**Cool Season (C3) or Warm Season (C4)**
Composite	*Sonchus arvensis* L.	Perennial Sow-Thistle	Perennial	Non-Native	ND	C4
Fabaceae	*Vicia americana*	American Vetch	Perennial	Native	?	?
Legume	*Medicago sativa* L.	Alfalfa	Perennial	Non-Native	ND	C3
Legume	*Melilotus alba* Desr.	White Sweet-Clover	Perennial	Non-Native	ND	C3
Poaceae	*Phalaris arundinacea* L.	Reed Canary Grass	Perennial	Non-Native	ND	C3
Poaceae	*Elymus Canadensis* L.	Canadian Wild Rye	Perennial	Native	D	C3
Poaceae	*Festica arundinacea*	Tall Fescue	Perennial	Non-Native	ND	C3
Polygonaceae	*Rumex crispus* L	Curled Dock	Perennial	Non-Native	ND	C3

Source: Inventory identified by Rob Officer and Gord MacKay September 29 and October 6, 2009.

Appendix 4- Soil Test Results

Soil Analysis results from Blocks I and II taken on April 27, 2010.

Sample Details/Parameters	Result	Units	Detectable Limit
Miscellaneous Parameters			
Alkalinity, Total	77	mg/L	10
Bicarbonate (HCO3)	93.4	mg/kg	2.0
Organic Matter	5.8	%	1.0
Available Micronutrients			
Copper (Cu)	2.02	ug/cc	0.020
Iron (Fe)	17.1	ug/cc	2.0
Manganese (Mn)	1.92	ug/cc	0.040
Zinc (Zn)	0.72	ug/cc	0.20
Available Boron (B)	0.58	ug/cc	0.10
Available Chloride	74.5	ug/cc	0.20
Available Macronutrients			
Available Nitrate-N	1.62	ug/cc	0.40
Available Phosphate-P	5.1	ug/cc	1.0
Available Potassium-K	345	ug/cc	2.0
Available Sulfate-S	278	ug/cc	2.0
Qualitative Soil Texture (Hand Texture)	Clay	n/a	n/a
pH and EC 1:2 soil to water (Ag method)			
pH (1:2 soil:water)	8.04	pH	0.10
Conductivity (1:2)	0.830	ds m-1	0.010
Sodium Adsorption Ratio(SAR and Cations in saturated soil)			
Calcium (Ca)	297	mg/L	1.2

Potassium (K)	17.2	mg/L	2.0
Magnesium (Mg)	169	mg/L	1.0
Sodium (Na)	120	mg/L	4.0
SAR	1.38	SAR	0.10
Saturated Paste			
% Saturation	86.4	%	1.0

Source: ALS Laboratory Analytical Group, 2010

Soil Analysis results from Block III taken on April 27, 2010.

Sample Details/Parameters	Result	Units	Detectable Limit
Miscellaneous Parameters			
Alkalinity, Total	82	mg/L	10
Bicarbonate (HCO3)	99.5	mg/kg	2.0
Organic Matter	6.8	%	1.0
Available Micronutrients			
Copper (Cu)	1.59	ug/cc	0.020
Iron (Fe)	22.6	ug/cc	2.0
Manganese (Mn)	2.75	ug/cc	0.040
Zinc (Zn)	0.66	ug/cc	0.20
Available Boron (B)	0.70	ug/cc	0.10
Available Chloride	58.1	ug/cc	0.20
Available Macronutrients			
Available Nitrate-N	1.31	ug/cc	0.40
Available Phosphate-P	14.3	ug/cc	1.0
Available Potassium-K	397	ug/cc	2.0
Available Sulfate-S	504	ug/cc	2.0
Qualitative Soil Texture (Hand Texture)	Clay	n/a	n/a

pH and EC 1:2 soil to water (Ag method)			
pH (1:2 soil:water)	7.80	pH	0.10
Conductivity (1:2)	1.09	ds m-1	0.010
Sodium Adsorption Ratio(SAR and Cations in saturated soil)			
Calcium (Ca)	221	mg/L	1.2
Potassium (K)	15.2	mg/L	2.0
Magnesium (Mg)	108	mg/L	1.0
Sodium (Na)	68.5	mg/L	4.0
SAR	0.94	SAR	0.10
Saturated Paste			
% Saturation	97.5	%	1.0

Source: ALS Laboratory Analytical Group, 2010

Soil Analysis results from Blocks IV, V, VI taken on April 27, 2010.

Sample Details/Parameters	Result	Units	Detectable Limit
Miscellaneous Parameters			
Alkalinity, Total	68	mg/L	10
Bicarbonate (HCO3)	82.8	mg/kg	2.0
Organic Matter	3.6	%	1.0
Available Micronutrients			
Copper (Cu)	1.91	ug/cc	0.020
Iron (Fe)	9.3	ug/cc	2.0
Manganese (Mn)	1.13	ug/cc	0.040
Zinc (Zn)	0.31	ug/cc	0.20
Available Boron (B)	0.58	ug/cc	0.10
Available Chloride	82.5	ug/cc	0.20

Available Macronutrients			
Available Nitrate-N	0.63	ug/cc	0.40
Available Phosphate-P	9.5	ug/cc	1.0
Available Potassium-K	300	ug/cc	2.0
Available Sulfate-S	180	ug/cc	2.0
Qualitative Soil Texture (Hand Texture)	Clay	n/a	n/a
pH and EC 1:2 soil to water (Ag method)			
pH (1:2 soil:water)	8.41	pH	0.10
Conductivity (1:2)	0.624	ds m-1	0.010
Sodium Adsorption Ratio(SAR and Cations in saturated soil)			
Calcium (Ca)	158	mg/L	1.2
Potassium (K)	10.2	mg/L	2.0
Magnesium (Mg)	137	mg/L	1.0
Sodium (Na)	96.6	mg/L	4.0
SAR	1.36	SAR	0.10
Saturated Paste			
% Saturation	75.0	%	1.0

Source: ALS Laboratory Analytical Group, 2010

Appendix 5- 2010-2011 Water Test Results

Water Sample Analytical Results- October 7, 2010 Samples

LOCATION 1

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	357	0.40	umhos/cm
Phosphorus, total	0.229	0.0020	mg/L
Turbidity	55.0	0.10	NTU
pH	8.36	0.10	pH units
Alkalinity, Total (as CaC03)	129	1.0	mg/L
Bicarbonate (HCO3)	153	2.0	mg/L
Carbonate (CO3)	2.03	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	1.26	0.20	mg/L
Total Nitrogen (Calculated)	1.26	0.20	mg/L
Sodium Adsorption Ratio	0.27	0.030	mg/L
Calcium (Ca)-Total	35.3	0.20	mg/L
Magnesium (Mg)- Total	16.9	0.050	mg/L
Sodium (Na) Total	7.88	0.050	mg/L

LOCATION 2

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	317	0.40	umhos/cm
Phosphorus, total	0.393	0.0020	mg/L
Turbidity	82.7	0.10	NTU
pH	8.31	0.10	pH units
Alkalinity, Total (as CaC03)	114	1.0	mg/L
Bicarbonate (HCO3)	138	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	1.62	0.20	mg/L
Total Nitrogen (Calculated)	1.62	0.20	mg/L
Sodium Adsorption Ratio	0.23	0.030	mg/L
Calcium (Ca)-Total	31.3	0.20	mg/L
Magnesium (Mg)- Total	14.9	0.050	mg/L
Sodium (Na) Total	6.32	0.050	mg/L

LOCATION 3

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	322	0.40	umhos/cm
Phosphorus, total	0.358	0.0020	mg/L
Turbidity	47.4	0.10	NTU
pH	8.31	0.10	pH units
Alkalinity, Total (as CaC03)	114	1.0	mg/L
Bicarbonate (HCO3)	138	2.0	mg/L
Carbonate (CO3)	0.79	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	0.086	0.050	mg/L
Nitrate and Nitrite as N	0.086	0.071	mg/L
Total Kjeldalh Nitrogen	1.36	0.20	mg/L
Total Nitrogen (Calculated)	1.45	0.20	mg/L
Sodium Adsorption Ratio	0.24	0.030	mg/L
Calcium (Ca)-Total	31.8	0.20	mg/L
Magnesium (Mg)- Total	15.1	0.050	mg/L
Sodium (Na) Total	6.53	0.050	mg/L

LOCATION 4

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	297	0.40	umhos/cm
Phosphorus, total	0.386	0.0020	mg/L
Turbidity	65.6	0.10	NTU
pH	8.04	0.10	pH units
Alkalinity, Total (as CaC03)	114	1.0	mg/L
Bicarbonate (HCO3)	139	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	0.072	0.050	mg/L
Nitrate and Nitrite as N	0.072	0.071	mg/L
Total Kjeldalh Nitrogen	1.59	0.20	mg/L
Total Nitrogen (Calculated)	1.66	0.20	mg/L
Sodium Adsorption Ratio	0.25	0.030	mg/L
Calcium (Ca)-Total	39.9	0.20	mg/L
Magnesium (Mg)- Total	18.4	0.050	mg/L
Sodium (Na) Total	7.68	0.050	mg/L

Source: ALS Labs, Winnipeg, 2010

Mean Values Locations 1-4, October 10, 2010

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	323.25	0.40	umhos/cm
Phosphorus, total	0.3415	0.0020	mg/L
Turbidity	62.68	0.10	NTU
pH	8.26	0.10	pH units
Alkalinity, Total (as CaC03)	118	1.0	mg/L
Bicarbonate (HCO3)	142	2.0	mg/L
Carbonate (CO3)	0.95	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	1.46	0.20	mg/L
Total Nitrogen (Calculated)	1.50	0.20	mg/L
Sodium Adsorption Ratio	.25	0.030	mg/L
Calcium (Ca)-Total	34.75	0.20	mg/L
Magnesium (Mg)- Total	16.3	0.050	mg/L
Sodium (Na) Total	7.10	0.050	mg/L

Water Sample Analytical Results- April 7, 2011 Samples

LOCATION 1

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	202	0.40	umhos/cm
Phosphorus, total	.277	0.0020	mg/L
Turbidity	24.6	0.10	NTU
pH	7.99	0.10	pH units
Alkalinity, Total (as CaC03)	70.6	1.0	mg/L
Bicarbonate (HCO3)	86.1	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	1.34	0.050	mg/L
Nitrate and Nitrite as N	1.34	0.071	mg/L
Total Kjeldalh Nitrogen	.97	0.20	mg/L
Total Nitrogen (Calculated)	2.30	0.20	mg/L
Hardness	80.4	0.20	mg/L
Sodium Adsorption Ratio	0.2	0.030	mg/L
Calcium (Ca)-Total	17.7	0.20	mg/L
Magnesium (Mg)- Total	8.77	0.050	mg/L
Sodium (Na) Total	4.09	0.050	mg/L

LOCATION 2

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	215	0.40	umhos/cm
Phosphorus, total	0.291	0.0020	mg/L
Turbidity	27.1	0.10	NTU
pH	7.95	0.10	pH units
Alkalinity, Total (as CaC03)	70.4	1.0	mg/L
Bicarbonate (HCO3)	85.9	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	1.39	0.050	mg/L
Nitrate and Nitrite as N	1.39	0.071	mg/L
Total Kjeldalh Nitrogen	0.97	0.20	mg/L
Total Nitrogen (Calculated)	2.36	0.20	mg/L
Hardness	85.0	0.20	mg/L
Sodium Adsorption Ratio	0.22	0.030	mg/L
Calcium (Ca)-Total	18.8	0.20	mg/L
Magnesium (Mg)- Total	9.26	0.050	mg/L
Sodium (Na) Total	4.61	0.050	mg/L

LOCATION 3

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	205	0.40	umhos/cm
Phosphorus, total	0.276	0.0020	mg/L
Turbidity	28.3	0.10	NTU
pH	7.92	0.10	pH units
Alkalinity, Total (as CaC03)	71.2	1.0	mg/L
Bicarbonate (HCO3)	86.9	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	1.38	0.050	mg/L
Nitrate and Nitrite as N	1.38	0.071	mg/L
Total Kjeldalh Nitrogen	1.07	0.20	mg/L
Total Nitrogen (Calculated)	2.45	0.20	mg/L
Hardness	81.7	0.20	mg/L
Sodium Adsorption Ratio	0.2	0.030	mg/L
Calcium (Ca)-Total	18.0	0.20	mg/L
Magnesium (Mg)- Total	8.92	0.050	mg/L
Sodium (Na) Total	4.18	0.050	mg/L

LOCATION 4

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	202	0.40	umhos/cm
Phosphorus, total	0.296	0.0020	mg/L
Turbidity	31.7	0.10	NTU
pH	7.89	0.10	pH units
Alkalinity, Total (as CaC03)	69.7	1.0	mg/L
Bicarbonate (HCO3)	85.0	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	1.40	0.050	mg/L
Nitrate and Nitrite as N	1.40	0.071	mg/L
Total Kjeldalh Nitrogen	0.91	0.20	mg/L
Total Nitrogen (Calculated)	2.30	0.20	mg/L
Hardness	80.9	0.20	mg/L
Sodium Adsorption Ratio		0.030	mg/L
Calcium (Ca)-Total	17.8	0.20	mg/L
Magnesium (Mg)- Total	8.86	0.050	mg/L
Sodium (Na) Total	4.19	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Mean Values- Locations 1-4, April 7, 2011

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	206	0.40	umhos/cm
Phosphorus, total	.285	0.0020	mg/L
Turbidity	27.93	0.10	NTU
pH	7.94	0.10	pH units
Alkalinity, Total (as CaC03)	70.47	1.0	mg/L
Bicarbonate (HCO3)	85.98	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	1.38	0.050	mg/L
Nitrate and Nitrite as N	1.38	0.071	mg/L
Total Kjeldalh Nitrogen	.98	0.20	mg/L
Total Nitrogen (Calculated)	2.35	0.20	mg/L
Sodium Adsorption Ratio	0.2	0.030	mg/L
Hardness	82	0.20	mg/L
Calcium (Ca)-Total	18.07	0.20	mg/L
Magnesium (Mg)- Total	8.95	0.050	mg/L
Sodium (Na) Total	4.27	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Water Sample Analytical Results- May 16, 2011 Samples

LOCATION 1

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	769	0.40	umhos/cm
Phosphorus, total	.055	0.0020	mg/L
Turbidity	12.2	0.10	NTU
pH	8.63	0.10	pH units
Alkalinity, Total (as CaC03)	167	1.0	mg/L
Bicarbonate (HCO3)	185	2.0	mg/L
Carbonate (CO3)	9.69	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.050	0.071	mg/L
Total Kjeldalh Nitrogen	.95	0.20	mg/L
Total Nitrogen (Calculated)	0.95	0.20	mg/L
Hardness	352	0.20	mg/L
Sodium Adsorption Ratio	0.65	0.030	mg/L
Calcium (Ca)-Total	68.3	0.20	mg/L
Magnesium (Mg)- Total	44.0	0.050	mg/L
Sodium (Na) Total	27.9	0.050	mg/L

LOCATION 2

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	595	0.40	umhos/cm
Phosphorus, total	0.080	0.0020	mg/L
Turbidity	11.6	0.10	NTU
pH	9.02	0.10	pH units
Alkalinity, Total (as CaC03)	141	1.0	mg/L
Bicarbonate (HCO3)	131	2.0	mg/L
Carbonate (CO3)	20.4	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.050	0.071	mg/L
Total Kjeldalh Nitrogen	1.52	0.20	mg/L
Total Nitrogen (Calculated)	1.52	0.20	mg/L
Hardness	268	0.20	mg/L
Sodium Adsorption Ratio	0.55	0.030	mg/L
Calcium (Ca)-Total	51.6	0.20	mg/L
Magnesium (Mg)- Total	33.9	0.050	mg/L
Sodium (Na) Total	20.6	0.050	mg/L

LOCATION 3

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	618	0.40	umhos/cm
Phosphorus, total	0.149	0.0020	mg/L
Turbidity	11.6	0.10	NTU
pH	8.86	0.10	pH units
Alkalinity, Total (as CaC03)	161	1.0	mg/L
Bicarbonate (HCO3)	163	2.0	mg/L
Carbonate (CO3)	16.5	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.050	0.071	mg/L
Total Kjeldalh Nitrogen	1.64	0.20	mg/L
Total Nitrogen (Calculated)	1.64	0.20	mg/L
Hardness	288	0.20	mg/L
Sodium Adsorption Ratio	0.53	0.030	mg/L
Calcium (Ca)-Total	59.9	0.20	mg/L
Magnesium (Mg)- Total	33.6	0.050	mg/L
Sodium (Na) Total	20.7	0.050	mg/L

LOCATION 4

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	610	0.40	umhos/cm
Phosphorus, total	0.117	0.0020	mg/L
Turbidity	19.1	0.10	NTU
pH	8.80	0.10	pH units
Alkalinity, Total (as CaC03)	170	1.0	mg/L
Bicarbonate (HCO3)	176	2.0	mg/L
Carbonate (CO3)	15.2	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.050	0.071	mg/L
Total Kjeldalh Nitrogen	1.35	0.20	mg/L
Total Nitrogen (Calculated)	1.35	0.20	mg/L
Hardness	281	0.20	mg/L
Sodium Adsorption Ratio	0.5	0.030	mg/L
Calcium (Ca)-Total	60.1	0.20	mg/L
Magnesium (Mg)- Total	31.7	0.050	mg/L
Sodium (Na) Total	19.1	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Mean Values- Locations 1-4, May 16, 2011

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	648	0.40	umhos/cm
Phosphorus, total	.100	0.0020	mg/L
Turbidity	13.63	0.10	NTU
pH	8.83	0.10	pH units
Alkalinity, Total (as CaC03)	160	1.0	mg/L
Bicarbonate (HCO3)	163.75	2.0	mg/L
Carbonate (CO3)	15.45	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.050	0.071	mg/L
Total Kjeldalh Nitrogen	1.37	0.20	mg/L
Total Nitrogen (Calculated)	1.37	0.20	mg/L
Sodium Adsorption Ratio	0.56	0.030	mg/L
Hardness	297.25	0.20	mg/L
Calcium (Ca)-Total	59.98	0.20	mg/L
Magnesium (Mg)- Total	35.8	0.050	mg/L
Sodium (Na) Total	22.08	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Water Sample Analytical Results- June 16, 2011 Samples

LOCATION 1

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	847	0.40	umhos/cm
Phosphorus, total	.103	0.0020	mg/L
Turbidity	3.98	0.10	NTU
pH	9.87	0.10	pH units
Alkalinity, Total (as CaC03)	105	1.0	mg/L
Bicarbonate (HCO3)	19.2	2.0	mg/L
Carbonate (CO3)	53.7	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	n/a	0.20	mg/L
Total Nitrogen (Calculated)	n/a	0.20	mg/L
Hardness	360	0.20	mg/L
Sodium Adsorption Ratio	0.94	0.030	mg/L
Calcium (Ca)-Total	56.2	0.20	mg/L
Magnesium (Mg)- Total	53.3	0.050	mg/L
Sodium (Na) Total	41.1	0.050	mg/L

LOCATION 2

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	667	0.40	umhos/cm
Phosphorus, total	0.256	0.0020	mg/L
Turbidity	12.4	0.10	NTU
pH	8.85	0.10	pH units
Alkalinity, Total (as CaC03)	109	1.0	mg/L
Bicarbonate (HCO3)	107	2.0	mg/L
Carbonate (CO3)	13	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	n/a	0.20	mg/L
Total Nitrogen (Calculated)	n/a	0.20	mg/L
Hardness	350	0.20	mg/L
Sodium Adsorption Ratio	0.69	0.030	mg/L
Calcium (Ca)-Total	46.4	0.20	mg/L
Magnesium (Mg)- Total	34	0.050	mg/L
Sodium (Na) Total	27.5	0.050	mg/L

LOCATION 3

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	747	0.40	umhos/cm
Phosphorus, total	0.226	0.0020	mg/L
Turbidity	19.8	0.10	NTU
pH	8.05	0.10	pH units
Alkalinity, Total (as CaC03)	156	1.0	mg/L
Bicarbonate (HCO3)	191	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	n/a	0.20	mg/L
Total Nitrogen (Calculated)	n/a	0.20	mg/L
Hardness	350	0.20	mg/L
Sodium Adsorption Ratio	0.69	0.030	mg/L
Calcium (Ca)-Total	74.4	0.20	mg/L
Magnesium (Mg)- Total	39.9	0.050	mg/L
Sodium (Na) Total	29.5	0.050	mg/L

LOCATION 4

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	742	0.40	umhos/cm
Phosphorus, total	0.184	0.0020	mg/L
Turbidity	15.9	0.10	NTU
pH	8.05	0.10	pH units
Alkalinity, Total (as CaC03)	153	1.0	mg/L
Bicarbonate (HCO3)	187	2.0	mg/L
Carbonate (CO3)	<0.60	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	n/a	0.20	mg/L
Total Nitrogen (Calculated)	n/a	0.20	mg/L
Hardness	362	0.20	mg/L
Sodium Adsorption Ratio	0.71	0.030	mg/L
Calcium (Ca)-Total	70.3	0.20	mg/L
Magnesium (Mg)- Total	37.7	0.050	mg/L
Sodium (Na) Total	29.6	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Mean Values- Locations 1-4, June 16, 2011

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	751	0.40	umhos/cm
Phosphorus, total	.192	0.0020	mg/L
Turbidity	13.02	0.10	NTU
pH	8.71	0.10	pH units
Alkalinity, Total (as CaC03)	131	1.0	mg/L
Bicarbonate (HCO3)	126.05	2.0	mg/L
Carbonate (CO3)	16.67	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	n/a	0.20	mg/L
Total Nitrogen (Calculated)	n/a	0.20	mg/L
Sodium Adsorption Ratio	0.76	0.030	mg/L
Hardness	355.5	0.20	mg/L
Calcium (Ca)-Total	61.82	0.20	mg/L
Magnesium (Mg)- Total	41.2	0.050	mg/L
Sodium (Na) Total	31.93	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

June 27 and July 29 2011- Organochlorine pesticides, phenoxyacid herbicides, Glyphosate and AMPA testing

Samples collected at each of the four sample sites were tested for organochlorine pesticides, phenoxyacid herbicides, and glyphosate and AMPA. Results for all samples indicated no detectable levels of either the organochlorine pesticides or phenoxyacid herbicides. Low levels of glyphosate and AMPA were detected at each sample site generating mean values of 7.84 ug/l and 3.99 ug/l respectively across all sites.

Water Sample Analytical Results- July 12, 2011 Samples

LOCATION 1

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	415	0.40	umhos/cm
Phosphorus, total	0.194	0.0020	mg/L
Turbidity	1.85	0.10	NTU
pH	9.84	0.10	pH units
Alkalinity, Total (as CaC03)	146	1.0	mg/L
Bicarbonate (HCO3)	64.1	2.0	mg/L
Carbonate (CO3)	55.7	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	1.67	0.20	mg/L
Total Nitrogen (Calculated)	1.67	0.20	mg/L
Hardness	214	0.20	mg/L
Sodium Adsorption Ratio	0.61	0.030	mg/L
Calcium (Ca)-Total	38.5	0.20	mg/L
Magnesium (Mg)- Total	28.6	0.050	mg/L
Sodium (Na) Total	20.4	0.050	mg/L

LOCATION 2

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	412	0.40	umhos/cm
Phosphorus, total	0.185	0.0020	mg/L
Turbidity	6.88	0.10	NTU
pH	9.96	0.10	pH units
Alkalinity, Total (as CaC03)	100	1.0	mg/L
Bicarbonate (HCO3)	27.1	2.0	mg/L
Carbonate (CO3)	46.7	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	0.438	0.050	mg/L
Nitrate-N	4.32	0.050	mg/L
Nitrate and Nitrite as N	4.76	0.071	mg/L
Total Kjeldalh Nitrogen	2.19	0.20	mg/L
Total Nitrogen (Calculated)	6.95	0.20	mg/L
Hardness	190	0.20	mg/L
Sodium Adsorption Ratio	0.48	0.030	mg/L
Calcium (Ca)-Total	41.3	0.20	mg/L
Magnesium (Mg)- Total	21.2	0.050	mg/L
Sodium (Na) Total	15.1	0.050	mg/L

LOCATION 3

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	416	0.40	umhos/cm
Phosphorus, total	0.387	0.0020	mg/L
Turbidity	14.3	0.10	NTU
pH	9.17	0.10	pH units
Alkalinity, Total (as CaC03)	119	1.0	mg/L
Bicarbonate (HCO3)	68.9	2.0	mg/L
Carbonate (CO3)	37.4	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	1.97	0.20	mg/L
Total Nitrogen (Calculated)	1.97	0.20	mg/L
Hardness	197	0.20	mg/L
Sodium Adsorption Ratio	0.6	0.030	mg/L
Calcium (Ca)-Total	39.8	0.20	mg/L
Magnesium (Mg)- Total	23.7	0.050	mg/L
Sodium (Na) Total	19.3	0.050	mg/L

LOCATION 4

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	432	0.40	umhos/cm
Phosphorus, total	0.166	0.0020	mg/L
Turbidity	12.8	0.10	NTU
pH	9.49	0.10	pH units
Alkalinity, Total (as CaC03)	107	1.0	mg/L
Bicarbonate (HCO3)	64.1	2.0	mg/L
Carbonate (CO3)	32.5	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	0.534	0.050	mg/L
Nitrate-N	0.968	0.050	mg/L
Nitrate and Nitrite as N	1.50	0.071	mg/L
Total Kjeldalh Nitrogen	2.89	0.20	mg/L
Total Nitrogen (Calculated)	4.39	0.20	mg/L
Hardness	191	0.20	mg/L
Sodium Adsorption Ratio	0.52	0.030	mg/L
Calcium (Ca)-Total	39.4	0.20	mg/L
Magnesium (Mg)- Total	22.4	0.050	mg/L
Sodium (Na) Total	16.4	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Mean Values- Locations 1-4, July 12, 2011

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	418.75	0.40	umhos/cm
Phosphorus, total	0.233	0.0020	mg/L
Turbidity	8.96	0.10	NTU
pH	8.95	0.10	pH units
Alkalinity, Total (as CaC03)	118	1.0	mg/L
Bicarbonate (HCO3)	56	2.0	mg/L
Carbonate (CO3)	43.07	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	0.134	0.050	mg/L
Nitrate-N	.242	0.050	mg/L
Nitrate and Nitrite as N	.38	0.071	mg/L
Total Kjeldalh Nitrogen	3.51	0.20	mg/L
Total Nitrogen (Calculated)	3.75	0.20	mg/L
Sodium Adsorption Ratio	.55	0.030	mg/L
Hardness	198	0.20	mg/L
Calcium (Ca)-Total	39.75	0.20	mg/L
Magnesium (Mg)- Total	23.98	0.050	mg/L
Sodium (Na) Total	17.8	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Water Sample Analytical Results- July 29, 2011 Samples

LOCATION 1

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	491	0.40	umhos/cm
Phosphorus, total	0.285	0.0020	mg/L
Turbidity	7.89	0.10	NTU
pH	9.62	0.10	pH units
Alkalinity, Total (as CaC03)	171	1.0	mg/L
Bicarbonate (HCO3)	101	2.0	mg/L
Carbonate (CO3)	52	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	1.87	0.20	mg/L
Total Nitrogen (Calculated)	1.87	0.20	mg/L
Hardness	239	0.20	mg/L
Sodium Adsorption Ratio	0.62	0.030	mg/L
Calcium (Ca)-Total	41.8	0.20	mg/L
Magnesium (Mg)- Total	32.6	0.050	mg/L
Sodium (Na) Total	21.8	0.050	mg/L

LOCATION 2

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	408	0.40	umhos/cm
Phosphorus, total	0.212	0.0020	mg/L
Turbidity	3.52	0.10	NTU
pH	9.48	0.10	pH units
Alkalinity, Total (as CaC03)	98.2	1.0	mg/L
Bicarbonate (HCO3)	62.1	2.0	mg/L
Carbonate (CO3)	27.8	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	0.257	0.050	mg/L
Nitrate-N	0.529	0.050	mg/L
Nitrate and Nitrite as N	0.786	0.071	mg/L
Total Kjeldalh Nitrogen	2.74	0.20	mg/L
Total Nitrogen (Calculated)	3.53	0.20	mg/L
Hardness	179	0.20	mg/L
Sodium Adsorption Ratio	0.45	0.030	mg/L
Calcium (Ca)-Total	35.9	0.20	mg/L
Magnesium (Mg)- Total	21.8	0.050	mg/L
Sodium (Na) Total	0.45	0.050	mg/L

LOCATION 3

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	442	0.40	umhos/cm
Phosphorus, total	0.505	0.0020	mg/L
Turbidity	7.39	0.10	NTU
pH	9.32	0.10	pH units
Alkalinity, Total (as CaC03)	121	1.0	mg/L
Bicarbonate (HCO3)	92	2.0	mg/L
Carbonate (CO3)	26.8	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	2.16	0.20	mg/L
Total Nitrogen (Calculated)	2.16	0.20	mg/L
Hardness	192	0.20	mg/L
Sodium Adsorption Ratio	0.54	0.030	mg/L
Calcium (Ca)-Total	40.4	0.20	mg/L
Magnesium (Mg)- Total	22.1	0.050	mg/L
Sodium (Na) Total	17.2	0.050	mg/L

LOCATION 4

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	463	0.40	umhos/cm
Phosphorus, total	0.583	0.0020	mg/L
Turbidity	16.4	0.10	NTU
pH	8.67	0.10	pH units
Alkalinity, Total (as CaC03)	123	1.0	mg/L
Bicarbonate (HCO3)	133	2.0	mg/L
Carbonate (CO3)	7.79	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	<0.050	0.050	mg/L
Nitrate-N	<0.050	0.050	mg/L
Nitrate and Nitrite as N	<0.071	0.071	mg/L
Total Kjeldalh Nitrogen	3.42	0.20	mg/L
Total Nitrogen (Calculated)	3.42	0.20	mg/L
Hardness	195	0.20	mg/L
Sodium Adsorption Ratio	0.52	0.030	mg/L
Calcium (Ca)-Total	39.3	0.20	mg/L
Magnesium (Mg)- Total	23.5	0.050	mg/L
Sodium (Na) Total	16.6	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Mean Values- Locations 1-4, July 29, 2011

TEST DESCRIPTION	RESULT	DETECTABLE LIMIT	UNITS
Conductivity	451	0.40	umhos/cm
Phosphorus, total	0.396	0.0020	mg/L
Turbidity	8.8	0.10	NTU
pH	9.27	0.10	pH units
Alkalinity, Total (as CaC03)	128.3	1.0	mg/L
Bicarbonate (HCO3)	97.02	2.0	mg/L
Carbonate (CO3)	28.6	0.60	mg/L
Hydroxide (OH)	<0.40	0.40	mg/L
Nitrite-N	0.064	0.050	mg/L
Nitrate-N	0.132	0.050	mg/L
Nitrate and Nitrite as N	0.197	0.071	mg/L
Total Kjeldalh Nitrogen	2.55	0.20	mg/L
Total Nitrogen (Calculated)	2.55	0.20	mg/L
Sodium Adsorption Ratio	0.53	0.030	mg/L
Hardness	201.25	0.20	mg/L
Calcium (Ca)-Total	157.4	0.20	mg/L
Magnesium (Mg)- Total	25	0.050	mg/L
Sodium (Na) Total	17.35	0.050	mg/L

Source: ALS Labs Analytical Group, Winnipeg, 2011

Appendix 6- Ducks Unlimited Wet Meadow Mix species list and descriptions- %C3 in the mix = 64%; C4 in the mix = 36%; Wet meadow species- 83%.

Northern Wheatgrass	*Elymus lanceolatus*	C3
Western Wheatgrass	*Pascopyrum smithii*	C3
Canada Wildrye	*Elymus candensis*	C3
Slender Wheatgrass	*Elymus Trachycaulus sub. Trachycaulus*	C3
Awned Wheatgrass	*Elymus trachycaulus sub. subsecundus*	C3
Tickle Grass	*Agrostis scabra*	C3
Tufted Hairgrass	*Deschampsia caespitosa*	C3
Switchgrass	*Panicum virgatum*	C4
Cord Grass	*Spartina pectinanta*	C4
Slough Grass	*Beckmannia syzigachne*	C3
Tall Mana Grass	*Glyceria grandis*	C3

Source: Native Plant Solutions, 2010