University of Alberta

Created stormwater wetlands as wetland compensation and a floristic quality approach to wetland condition assessment in central Alberta

by

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ABSTRACT

In Alberta, almost all created wetlands accepted as compensation have been naturalized stormwater management facilities. Our investigation of 32 created and natural wetlands in central Alberta determined that created wetlands have steeper shoreline slopes, largely as a result of their primary function as stormwater retention ponds. This resulted in distinctly different vegetation zonation, with the steeper slopes of created wetlands resulting in fewer, narrower wetland vegetation zones. This was reflected in reduced species richness and abundance of wetland songbirds at created wetlands. This study also discusses the development of a Floristic Quality Assessment (FQA) approach, a standardized, quantitative approach to measuring wetland condition, for Alberta's Parkland and Boreal natural regions. I present plant survey data from the 32 wetlands as validation of the effectiveness of this approach. This study provides information on current wetland compensation practices and a potential wetland assessment tool; both topics that are directly relevant to the implementation of wetland compensation policies in Alberta.

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CHAPTER 1

General Introduction: Created wetlands as wetland compensation and methods used to measure compensatory success

Background

Although the recognition of wetlands as significant components of the landscape has increased in recent years (Kennedy and Mayer 2002, Matthews and Endress 2008), the loss of wetlands has been widespread. In the continental United States, it is estimated that approximately 52% of wetlands have been lost (Dahl 2000). Within the settled, southern half of Alberta, the loss of wetlands is estimated at between 60 and 70% (SOCE 1991, Strong and Leggat 1992). In response to this trend, governments have developed policies intended to prevent further wetland loss (Matthews and Endress 2008). In the United States, a wide range of local, state, federal and private programs exist in support of the primary policy of ‘no net loss’ of wetlands (Whigham 1999). Since amendments were made to the United States Clean Water Act, the nation’s primary piece of wetland legislation, in the late 1970’s, compensatory wetland mitigation has become an important part of realizing the policy of ‘no net loss’ and wetland compensation is now common across much of the United States (Spieles 2005). In Alberta, Canada, the Provincial Government similarly developed a wetland policy in 1993 with a goal to “restore or create” wetlands with the overall intent of sustaining “the social, economic and environmental benefits that functioning wetlands provide” (Alberta Water Resources Commission 1993). Although wetland policy development has been delayed in Alberta, the practice of creating wetlands as a form wetland compensation for the loss of natural wetlands is also now common across southern and central Alberta.

Wetland Compensation

The heavy reliance of existing wetland policies on wetland compensation has made it necessary to quantify the success of wetland creation and restoration projects at mitigating the lost area and function of natural wetlands. Accordingly, the question of whether or not restored and created wetlands are structurally and functionally equivalent to natural wetlands has been the focus of extensive research in the United States (Stolt et al. 2000, Spieles 2005, Hartzell et al. 2007, Fennessy et al. 2008, Hoeltje and Cole 2009). In contrast, very limited compensation research has been completed in Canada. There is still much debate over how to best assess wetland compensation, however, wetland bioassessments in the US have been used extensively to determine wetland condition and, in turn, the success of wetland mitigation. In bioassessments, biological indicators are used as surrogates for the complex ecological processes and wetland functions that are difficult to measure directly. Although this research has produced equivocal results, perhaps as a function of the great variety in the types of created wetlands or the variety of methods used to measure compensation success, there is an overall sense that created wetlands often fail in achieving their desired ecological objectives (Spieles 2005, Alsfeld et al. 2009).

In Alberta, almost all created wetlands accepted as compensation have been naturalized stormwater management facilities (NSWMFs). These facilities are designed and built with the primary purpose of providing stormwater management. Their primary function is the provision of stormwater storage (and attenuation of downstream flows) with a secondary function of water quality improvement (e.g., sediment settling, contaminant uptake). In situations when the design and construction of stormwater management facilities also include an effort to create wetland habitat (e.g., creation of shallow water areas for emergent vegetation, riparian shrub plantings) the facilities are often accepted by the Province as wetland compensation. The acceptance of naturalized stormwater wetlands (i.e., created wetlands) as a form of wetland compensation continues despite the absence of scientific evidence that they successfully replace natural wetland functions.

Wetland Assessment

Not only are there no published studies investigating the effectiveness (i.e., success) of wetland creation in central Alberta or the broader Canadian Parkland ecoregion, there is no standardized approach for the assessment of wetland condition nor are there any standardized quantitative ecological performance standards that could be used in such assessments. In the literature, wetland birds and plants have both been previously studied and found to be effective indicators of wetland condition in selected areas (e.g., Adamus 1996, Puchniak 2002). Further, many wetland managers agree that the combination of a healthy plant and animal community indicates an ecologically functional wetland (Gray et al. 1999).

Wetland plants are among the best developed and commonly used biological indicators (US EPA 2002, Spieles 2005, Miller et al. 2006). Wetland plants have well-documented response thresholds to wetland degradation (e.g., DeLuca et al. 2004), integrate disturbances at numerous biological scales (Ervin et al. 2006, Mack 2007) and are relatively easy to survey and monitor over time (Bowers and Boutin 2008). Wetland birds are also potentially attractive as indicators, primarily because they are relatively easy to monitor and bird survey protocols are well established (Adamus 1996), but also because birds hold great value with the general public, are of interest to conservation managers and because the provision of bird habitat is considered one of the most important functions of wetlands in Alberta (Wray and Bayley 2006).

This present study focuses on created wetlands as a form of wetland compensation and methods used to measure compensatory success in central Alberta. Specifically, this thesis includes two primary chapters that investigate particular aspects of this topic. Chapter 2 focuses on the investigation of the characteristic steep-sided basin design of created stormwater wetlands and whether or not there are differences in the vegetation zonation, wetland bird community and plant community between created and natural wetlands. This chapter provides information that could be used to inform decisions about how wetland creation should be implemented as part of wetland compensation program in Alberta. Chapter 3 discusses the development and evaluation of a Floristic Quality Assessment (FQA) approach to wetland condition assessment in Alberta. In many jurisdictions across the United States, FQA has been repeatedly shown to be highly correlated to various independently developed disturbance ratings and other measures of wetland site condition (e.g., Lopez and Fennessy 2002). Development of such a system in Alberta provides a much needed assessment tool that could be used as part of a standardized approach to assessing wetland conditionin the Province.

As Alberta continues to develop its wetland policy and related implementation plans, the issues of wetland compensation and wetland condition assessment will only grow in importance. The question of whether or not created wetlands are successfully replacing natural wetlands is integral to ensuring that policy objectives are met. Standardizing an approach to quantitative wetland condition assessment, whether it is to help determine suitable wetland compensation or to measure impacts of a proposed development project, is likely to become increasingly important in the future. The need for additional research in support of an effective wetland policy remains, however, this study provides relevant results that begin to fill in the necessary data gaps and furthers the information base available for wetland management in Alberta.

Literature Cited

Adamus, P.R. 1996. Bioindicators for assessing ecological integrity of prairie wetlands. *EPA/600/R-96/082*. Corvallis, OR: U.S. Environmental Protection Agency, Environmental Research Laboratory.

Alberta Water Resources Commission. 1993. Wetland management in the settled area of Alberta: An interim policy. Alberta Water Resources Commission. Edmonton, AB.

Alsfled, A.J., J.L. Bowman and A. Deller-Jacobs. 2009. Effects of woody debris, microtopography, and organic matter amendmnets on the biotic community of constructed depressional wetlands. Biological Conservation 142: 247-255.

Bowers, K. and C. Boutin. 2008. Evaluating the relationship between floristic quality and measures of plant biodiversity along stream bank habitats. Ecological Indicators 8: 466-475.

Dahl, T.E. 2000. Status and Trends of Wetlands in the Conterminous United States 1986 to 1997. Washington, DC: U.S. Department of the Interior, Fish and Wildlife and Service.

DeLuca, W.V, C.E. Studds, L.L. Rockwood and P.P. Marra. 2004. Influence of land use on the integrity of marsh bird communities of Chesapeake Bay, USA. Wetlands 24(4): 837-847.

Ervin, G.N., B.D. Herman, J.T. Bried and D.C. Holly. 2006. Evaluating non-native species and wetland indicator status as components of wetlands floristic assessment. Wetlands 26(4): 114-1129.

Fennessy, M.S., A. Rokosh and J.J. Mack. 2008. Patterns of plant decomposition and nutrient cycling in natural and created wetlands. Wetlands 28(2): 300-310.

Gray, B.T., R.W. Coley, R.J. McFarlane, A.J. Puchniak, D.A. Sexton, and G.R. Stewart. 1999. Restoration of prairie wetlands to enhance bird habitat: A Ducks Unlimited Canada perspective. IN Aquatic restoration in Canada. T. Murphy and M. Munawar editors. Backhuys Publishers, Leiden, the Netherlands. Pp 171-194.

Hartzell, D., J.R. Bidwell and C.A. Davis. 2007. A comparison of natural and created depressional wetlands in central Oklahoma using metrics from indices of biological integrity. Wetlands 27(4): 794-805.

Hoeltje, S.M. and C.A. Cole. 2009. Comparison of function of created wetlands of two age classes in central Pennsylvania. Environmental Management 43: 597-608.

Kennedy, G. and T. Mayer. 2002. Natural and constructed wetlands in Canada: An overview. Water Quality Research Journal of Canada 37(2): 295-325.

Lopez, R. D. and M. S. Fennessy. 2002. Testing the floristic quality assessment index as an indicator of wetland condition. Ecological Applications 12:487–497.

Mack, J. J. 2007. Developing a wetland IBI with statewide application after multiple testing iterations. Ecological Indicators 7:864–81.

Matthews, J.W. and A.G. Endress. 2008. Performance criteria, compliance success, and vegetation development in compensatory mitigation wetlands. Environmental Management 41: 130-141.

Miller, S.J., D.H. Wardrop, W.M. Mahaney and R.P. Brooks. 2006. A plant-based index of biotic integrity (IBI) fro headwater wetlands in central Pennsylvania. Ecological Indicators 6: 290-312.

Puchniak, A.J. 2002. Recovery of bird and amphibian assemblages in restored wetlands in Prairie Canada. M.Sc. Thesis, University of Alberta. 220 pages

SOCE. 1991. The state of Canada's environment. The Government of Canada. Ottawa, Ontario.

Spieles, D.J. 2005. Vegetation development in created, restored and enhanced mitigation wetland banks of the United States. Wetlands 25(1): 51-63.

Stolt, M.H., M.H. Genthner, L. Daniels, V.A. Groover, S. Nagle and K.C. Haering. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands 20(4): 671-683.

Strong, W.L. and K.R. Leggat. 1992. Ecoregions of Alberta. Alberta Forestry, Lands and Wildlife, Edmonton, Alberta, Canada. 55 pp + App

United States Environmental Protection Agency. 2002. Methods for evaluating wetland condition: Using vegetation to assess environment condition in wetlands. Office of Water, U.S. Environmental Protection Agency, Washington, DC. EPA-822-R-02-02-020.

Whigham, D.F. 1999. Ecological issues related to wetland preservation, restoration, creation and assessment. The Science of the Total Enviroment 240: 31-40.

Wray, H.E. and S.E. Bayley. 2006. A review of indicators of wetland health and function in Alberta's prairie, aspen parkland and boreal dry mixedwood regions. Report prepared for The Water Research Users Group, Alberta Environment.

CHAPTER 2

The effect of shoreline slope on wetland zonation, birds and vegetation in created and natural open water marsh wetlands

Introduction

Although the recognition of wetlands as significant components of the landscape has increased in recent years (Kennedy and Mayer 2002, Matthews and Endress 2008), the loss of wetlands has been widespread. In the continental United States, it is estimated that approximately 52% of wetlands have been lost (Dahl 2000). Within the settled, southern half of Alberta, the loss of wetlands is estimated at between 60 and 70% (SOCE 1991, Strong and Leggat 1992). In response to wetland loss, and to provide direction for wetland-related decision making, the Government of Alberta developed a wetland policy "to restore and create" wetlands in 1993 (Alberta Water Resources Commission 1993). Many wetlands have since been restored (i.e., restoring previously drained wetlands) to compensate for wetland losses. More recently, creating stormwater wetlands (i.e., constructing wetlands where none previously existed) has also been accepted as compensation.

The question of whether or not restored and created wetlands are structurally and functionally equivalent to natural wetlands has been the focus of extensive research in the United States (Stolt et al. 2000, Spieles 2005, Hartzell et al. 2007, Fennessy et al. 2008, Hoeltje and Cole 2009). Studies of restored wetlands have generated equivocal results in terms of their success (Brown and Smith 1998, Ratti et al. 2001). Some studies have found restored wetlands to have reduced numbers of native plant and bird species relative to natural reference wetlands (Delphey and Dinsmore 1993), some have found comparable bird and plant communities (Puchniak 2002) and some have, in fact, found restored wetlands to have increased species richness (Ratti et al. 2001). For created wetlands, reported results have been equally varied. Balcombe (2003) and Hartzell et al. (2007), among others, have found that plant species richness was similar between natural and created wetlands. In contrast, Campbell et al. (2002) and Kellog and Bridgham (2002) observed a difference in plant species richness between created and natural wetlands. Analysis of the macroinvertebrate community between natural and created wetlands have included studies reporting similar species composition (Streever et al. 1996), but also studies that have observed differences (Stanczak and Keiper 2004). Equivocal results have also been reported in studies that have focused on the bird community. Similar avian species richness (Brown and Smith 1998, Juni and Berry 2001, Ratti et al. 2001, Balcombe et al. 2005) and diversity (Juni and Berry 2001, Ratti et al. 2001) have been reported for natural and created depressional wetlands. Other studies (e.g., Brown and Smith 1998) report higher bird densities at natural wetlands. Despite the generally equivocal nature of the results reported in the literature, there is an overall sense that created wetlands often fail in achieving their desired ecological objectives (Spieles 2005).

It has been said that the greatest challenge for the successful creation of stormwater wetlands is the establishment of natural vegetation zonation, and that hydroperiod is the key factor (Jenkins and Greenway 2007). An inappropriate or unnatural hydrologic regime is commonly reported as a key factor in failed wetland mitigation (Spieles et al. 2006, Hoeltje and Cole 2009). In Washington State, both an unsuitable hydrology and wetland topography were blamed for the lack of hydric soil development and consequently overall mitigation failure (Kunz et al. 1988). Early work on surface-mine impoundments and agricultural stock ponds in the American Great Plains outlined the importance of wetland topography, and particularly the steepness of side slopes as a factor in the development of wetland vegetation (Rumble et al. 1984) and waterfowl use (Uresk and Severson 1988; Weber et al. 1982). In a guideline document for land resource managers, Olson (1999) also noted the importance of slope as a key factor influencing wetland vegetation in created wetlands.

In Alberta, most created wetlands accepted as compensation have been naturalized stormwater management facilities. These facilities are designed and built with the primary purpose of providing stormwater management. Their primary function is the provision of stormwater storage (and attenuation of downstream flows) with a secondary function of water quality improvement (e.g., sediment settling, contaminant uptake). In situations when the design of stormwater facilities also includes an effort to create wetland habitat (e.g., creation of shallow water areas for emergent vegetation, aquatic plantings) the facilities are often accepted by the Province as wetland compensation. The acceptance of naturalized stormwater wetlands (i.e., created wetlands) as a form of wetland compensation continues despite the absence of regionally specific scientific justification regarding the similarity of these systems to natural wetland systems.

Our primary objective was to investigate if the characteristic steep-sided basin design of created stormwater wetlands has resulted in important morphological and functional differences between created and natural wetlands. We assessed slope steepness, wetland zonation and surveyed the vegetation and birds at two types of created wetlands and at two categories of natural wetlands. We hypothesized that the steep-sided basin design of created wetlands would result in reduced wetland zonation which could, in turn, have an effect on the bird community. We also hypothesized that slope, through its interaction with water level, could influence wetland abiotic parameters and impact vegetation establishment. If true, differences in slope between wetland types could result in differences in plant community composition and vegetation structure.

No published studies have investigated the effectiveness (i.e., success) of wetland creation in Alberta or the broader Canadian Parkland ecoregion. As part of this investigation, we wanted to report on any differences between created and natural wetlands; information that could then be used to inform decisions about how wetland creation should be implemented as part of wetland compensation in Alberta.

Methods

Study Area

Fieldwork for this study was conducted in the summer (May-August) of 2008 at 32 created and natural wetlands in and around the City of Edmonton, Alberta, Canada. Edmonton (53.54°N latitude and 113.50°W longitude) is located in the Parkland Natural Region (Natural Regions Committee 2006) at the northern extreme of the North American Great Plains Ecoregion (Commission for Environmental Cooperation 1997). Marsh wetlands, located in undrained post-glacial depressions are the most common natural wetland type in this area of Alberta (Natural Regions Committee 2006).

Wetland Selection

Created wetlands were one of two types of stormwater management facilities: naturalized stormwater wetlands or wet ponds. Naturalized stormwater wetlands (NSWs) include naturalization features such as areas of shallow water, placement of salvaged wetland soils and planting of aquatic and riparian plant species, all intended to create more typical marsh habitat. NSWs are currently accepted as a form of compensation for the destruction of natural wetlands. Wet ponds (WPs) represent the old standard stormwater facility type that was typically constructed during the 1980's and 1990's that have no wetland naturalization and have stone riprap along their shorelines to protect against erosion. The riprap inadvertently, but effectively impedes the establishment of significant amounts of shoreline vegetation. Wet ponds are usually not accepted as wetland compensation. Both NSWs and WPs are characterized by permanent open water occupying the majority of the wetland basin. In this regard, they would be classified as permanent, open water marsh wetlands (Class V) following the Stewart and Kantrud (1971) wetland classification system. Eight NSWs and 8 WPs were selected from a list of potential sites in the Edmonton area. Because of their stormwater functions, all created wetlands were situated in urban areas or in close proximity to provincial roadways. All created wetlands were at least 3 years old; increasing the minimum age criteria beyond 3 years would have greatly reduced the number of potential sites. As required by government standards (Alberta Environment Protection 1999, City of Edmonton 2008), shoreline slopes of stormwater management facilities typically measure between 5:1 (horizontal run [H]:vertical rise [V]) to 7:1. Hereafter, the term 'created wetland' will be used to represent the broader stormwater wetland type, while 'naturalized stormwater wetland' (NSW) and 'wet pond' (WP) will be used as the terms identifying the two specific sub-types of created wetland.

Sixteen natural wetlands were selected as a comparative wetland type. Natural wetlands were sub-divided into two categories: agricultural wetlands (n=8) and undisturbed reference wetlands (n=8). Agricultural wetlands were defined as wetlands having at least 50% of the land within a 500m buffer under cultivation or pasture. Because cultivated lands occupy a significant portion of the parkland landscape (Natural Regions Committee 2006), agricultural wetlands represent the most common wetland 'type' on the landscape in east-central Alberta and those most vulnerable to current threats of agricultural land conversion and urban expansion. Undisturbed reference wetlands were defined as wetlands located within a protected area and having a maximum of 10% of the land within 500m under cultivation or pasture, with no agricultural land directly bordering the wetland. Hereafter, the term 'natural wetland' will be used to represent the broader wetland type, while 'reference wetland' and 'agricultural wetland' will be used as the terms identifying the two specific sub-types of natural wetland.

All natural wetlands were permanent, open water marsh wetlands (Class V; Stewart and Kantrud 1971). Natural wetlands were identified using high resolution (0.25m pixels) 2007 digital aerial photography and then randomly selected from a list of potential sites located within 60km of Edmonton. Permanency of selected sites was confirmed through a review of aerial photographs from multiple years. All sites were ground truthed to confirm suitability for inclusion in the study. To minimize variation between treatment types, all selected wetlands were between 1 ha and 10 ha in size.

Bird Surveys

Birds were surveyed at each wetland three times during the breeding season (20 May 2008 to 01 July 2008). Surveys were conducted between 30 minutes before sunrise and 10:00 AM and were only completed in suitable weather (i.e., no heavy rain or strong winds). Each bird survey consisted of a visual survey and an auditory survey.

Visual surveys focused on the detection of waterfowl and other conspicuous wetland birds using the open water area of the wetland. Visual surveys were conducted from locations well away from the wetland edge using binoculars and a spotting scope (15-60x zoom) to minimize the risk of flushing birds. The number of viewing locations at each wetland varied based on wetland size and configuration, but in all cases resulted in full coverage (>95%) of the open water area. All visible non-passerines were identified to species and counted. To assess the level of breeding activity, waterfowl were identified as belonging to a pair, single males, single females, belonging to a group of males or unknown. Behavioral cues indicative of possible breeding (e.g., nest building, courting and territorial displays) were also noted when observed.

Auditory surveys focused on the detection of wetland-dependent songbirds and inconspicuous non-passerine wetland birds (e.g., Sora; *Porzana carolina*). The auditory survey consisted of two 8-minute, 50m fixed-radius point counts conducted at the transition between emergent and wet meadow vegetation zones to maximize coverage of wetland habitat. The separation between point count stations was maximized to reduce the risk of double counting individual birds and a minimum separation of 150 m was used in all cases. All target bird species detected by sight or sound were recorded. Bird species nomenclature follows the American Ornithologist's Union standard.

Vegetation Surveys

Vegetation surveys were conducted at each wetland between 31 July 2008 and 27 August 2008. Prior to fieldwork, three survey transects were identified at each wetland using recent (2007) aerial photographs. Transect locations were chosen by drawing three lines separated by 120° extending outward from the center of the open water area. The angle of the first line was randomly chosen. Where the three lines bisected the edge of the open water, transects were drawn perpendicular to the shoreline from the edge of the open water to the upland-wetland interface.

At each transect, the number of distinct vegetation zones between the open water and upland-wetland interface was determined. Zones were differentiated based on the composition of dominant species and were categorized as either emergent (EM) or wet meadow (WM). Our definition of the emergent zone is analogous to the Stewart and Kantrud (1971) deep marsh zone classification, while our wet meadow category encompasses their shallow marsh, wet meadow and wetland-low prairie zones. Zone width was measured using a 50m tape measure. Within each zone, two 1 m^2 quadrats were placed at the mid-point of each zone; individual quadrats were separated by 5 m in a direction perpendicular to the transect. Depending on the number of zones at the wetland, the total number of quadrats sampled ranged from 6 to 20 (i.e., average of one zone to greater than three zones per transect). All plants within each quadrat were identified to species and percent cover was estimated using a modified Daubenmire (1959) cover-abundance scale. To ensure that a comprehensive inventory of all plant species at each wetland was obtained, a time-restricted walkabout was completed following the quadrat surveys (Locky and Bayley 2006). At each WM quadrat a visual obstruction measurement was taken as a relative measure of above ground biomass following Robel et al. (1970). The use of a visual obstruction measure (VOM) effectively combines plant density and height into a single measurement, making it a good relative measure of above ground biomass and an effective single parameter for the analysis of vegetation structure. Visual obstruction readings were not possible in the emergent zone because the vegetation heights were too great. Maximum vegetation height was also measured at all quadrats.

Multiple plant identification guides were used for species identification in the field (Johnson et al. 1995, Lahrig 2003). For species that we were not able to identify in the field, a specimen was collected for later identification using the Flora of Alberta (Moss 1983) and other available resources. Nomenclature of plant species closely follows the Integrated Taxonomic Information System (ITIS; www.itis.gov), an American national nomenclature source used by the US Environmental Protection Agency. The native status of species follows the Alberta Natural History Information Center's determination.

Shoreline Slope

Shoreline slope was measured along each of the three vegetation survey transects at each wetland. Two components of slope were measured: above-water (AW) and below-water (BW), with the transition between the two defined as the edge of the open water. For each slope component, three measurements were made at known distances (3m, 5m and 10m) from the edge of the open water. AW slope was calculated using measurements taken with a leveled laser beam and a 3m ruler held vertical at the edge of the open water. BW slope was calculated using measurements of water depth extending out towards the center of the wetland. Trigonometry was used to calculate the percent slope. The percent slope from the three transects was averaged to obtain a site value for above water, below water and overall shoreline slope.

Hydrology

HOBO® water level data loggers were installed at half of the selected wetlands, with equal distribution among wetland types. Data loggers recorded water depth measurements at 6hr intervals from mid-May through to the end of the September. Data from the loggers were used to develop hydrographs for each wetland and to measure seasonal amplitude (i.e., drawdown) and the rate of water level fluctuation (i.e., water level variation). Water level gauges were installed in sites without the data loggers. Water depth readings were taken during each field visit to the site. These data enabled measurements of seasonal amplitude, but it was not possible to calculate the rate of water level fluctuation.

Data Analysis

All analyses were conducted using either SPSS 16.0® or SYSTAT® 12. Categorical analysis of variables between wetland types was conducted using one-way analysis of variance (ANOVA) ($\alpha = 0.05$). ANOVAs were chosen as they are known to be extremely robust, particularly with respect to violations of the assumption of normality, under circumstances of equal sample size, as is the case with this study (Ito 1980). Following significant ANOVA results, Tukey's post hoc tests ($\alpha = 0.05$) were completed to make pairwise comparisons among wetland types. Linear regression was used to investigate correlation between abiotic and biotic wetland components.

Results

Shoreline Slope

As expected, shoreline slopes were steeper in created wetlands compared to the natural wetland types (Table 2-1). Shoreline slopes (averaging above and below water slope) were shallowest in reference wetlands at approximately 3% slope (i.e., 31:1 horizontal run [H]:vertical rise [V]), followed closely by agricultural wetlands with slopes of approximately 4% (i.e., 23H:1V). Slopes of NSWs (10%; 10H:1V) and WPs (14%; 7H:1V) were much steeper. Based on the average wetland slope values calculated using the 5m measurements, the differences in shoreline slope were significant between wetland types for both the below water ($p<0.001$, $F_{3,28}=20.81$) and above water component ($p<0.001$, $F_{3,28}=32.02$). The results were unchanged when analyzed using average slope values calculated from all three of the slope measurements taken (BW: $p<0.001$, $F_{3,28}=18.12$; AW: $p<0.001$, $F_{3,28}=18.73$). In all cases, pairwise comparisons between natural and created wetland types were significant ($p\leq0.024$), with steeper slopes characterizing both the above and below water components of created wetlands. Among natural wetland types, the slope of agricultural and reference wetlands were not significantly different from each other for both below ($p=1.00$) and above-water slope components ($p=0.43$). In contrast, among created wetlands, naturalized stormwater wetlands and wet ponds differed in terms of below water slope ($p<0.01$), but not above water slope ($p=0.14$).

Wetland Vegetation Zones

The number of distinct vegetation zones at a site varied depending on wetland type ($p<0.001$, $F_{3,28}=19.04$). Wetland zonation was highest in natural reference wetlands with an average of 2.6 zones (range 2.00 to 3.00), agricultural wetlands averaged 2.4 zones (range 1.67 to 3.00), NSWs averaged 1.9 zones (range 1.33 to 2.33) and WPs averaged 1.3 zones (range 1.00 to 2.00).

The four wetland types also differed significantly in terms of the width of their wetland vegetation zones (Figure 2-1) for both the wet meadow ($p<0.001$, $F_{3,28}=18.72$) and emergent zone ($p=0.003$, $F_{3,28}=6.85$). In general, vegetation zones were much wider in the natural wetlands compared to the created wetland types. Comparing wet meadow zone widths among wetland types, there were significant ($p\leq0.03$) pairwise differences between all wetland types except between NSWs and WPs ($p=0.95$), suggesting similarity in width of WMs among created wetland types. In contrast, emergent zone widths were comparable across most pairwise comparisons of wetland type, including NSWs and agricultural ($p=0.093$) and reference wetlands ($p=0.091$). The only significant pairwise comparisons were between WPs and the two natural wetland types ($p=0.005$). Although not statistically significant ($p\geq0.09$), the differences in zone width between the different wetland types (Table 2-1) were still considerable and likely represent a biologically meaningful result. The emergent zone of reference (mean = 15.1m) and agricultural wetlands (mean = 15.1m) was almost three times as wide as the mean emergent zone width of NSWs (mean = 5.7m). The difference in mean zone width was also considerable between NSW's (mean = 5.7m) and WP's (mean = 0.8m).

The width of wetland vegetation zones had a strong negative correlation to shoreline slope, with narrower vegetation zones correlated with steep slopes and wider zones correlated with shallow slopes. The correlation was strongest in the wet meadow zone (r^2=0.769, $p<.001$; Figure 2-2a) and only slightly weaker in the emergent zone (r^2=0.710, $p<.001$; Figure 2-2b). The complete overlap in average zone width and shoreline slope between the reference and agricultural wetlands in the regression plots (Figures 2-2a and 2-2b) mimics the pattern observed in the ANOVA analysis. In a similar way, the relatively tight cluster of NSW and WP sites in Figure 2-2a illustrates the structural similarities between these two created wetland types in terms of above water slope and the correspondingly similar width of their wet meadow zones. In contrast, NSWs and WPs show considerable separation in terms of their emergent zone (Figure 2-2b). Six of the eight WPs did not have an emergent zone, while all NSWs had an emergent zone averaging at least 2.0 m wide (Table 2-1). In Figure 2-2b, NSW sites are more closely grouped with the agricultural and reference wetlands, including three NSWs where the below water slope and emergent zone widths were similar to those of natural wetlands.

Birds

Including both visual and auditory surveys, we detected a total of 44 wetland dependent bird species. Wetland bird species were defined as species that can be considered either obligate or facultative wetland species following similar criteria as the United States Department of Agriculture's wetland plant indicator status using descriptions of bird habitat associations from The Atlas of Breeding Birds of Alberta (Federation of Alberta Naturalists 2007) and Birds of Alberta (Fisher and Acorn 1998). For 5 of the 44 species, the only records were single individuals observed on one occasion; these species were excluded in subsequent analyses. Of the 39 species observed on more than one occasion, 28 were waterfowl or waterbirds (hereafter referred to as waterbirds) and 11 were songbirds. Table 2-2 summarizes the number of species observed in each wetland type.

Overall bird species richness did not differ between 3 of the 4 wetland types (Figure 2-3). The only exception was for wet ponds which had significantly lower species richness compared to all other wetland types ($p \leq 0.01$). Looking only at waterbird species, there was a significant difference among wetland types ($p=0.036$, $F_{3,28}=3.26$), but the only significant pairwise comparison among wetland types was between agricultural wetlands and wet ponds ($p=0.048$). The difference in number of waterbird species between NSWs and WPs was only marginally non-significant ($p=0.06$), with NSWs averaging more than 3 species more than WPs (mean=8.8 vs. 5.2).

For wetland dependent songbirds, our results show that natural wetlands have an increased richness of wetland dependent songbirds relative to created wetlands ($p<0.001$, $F_{3,28}=15.34$). In terms of species richness, both natural wetland types (i.e., reference and agricultural) were statistically similar ($p=0.25$) as were both created wetland types (i.e., NSWs and WPs; $p=0.33$). The difference between NSWs (mean=1.8 spp.) and agricultural wetlands (mean=3.2 spp.), although slightly non-significant ($p=0.06$), likely represents a biologically significant division between natural and created wetlands in terms of their ability to support wetland dependent songbirds.

The analysis of species totals between wetland types reveals differences in the ability of the different wetland types to support wetland birds. NSWs, as a wetland type, supported 8 (73%) of the 11 wetland songbirds observed throughout the study (Table 2-2). The 3 species not found in any of the NSWs were Alder Flycatcher (*Empidonax alnorum*), Common Yellowthroat (*Geothlypis trichas*) and Swamp Sparrow (*Melospiza georgiana*). In contrast, WPs supported only 2 (18%) wetland dependent songbird species: Song Sparrow (*Melospiza melodia*) and Red-winged Blackbird (*Agelaius phoeniceus*); both generalist wetland species common throughout central Alberta (Federation of Alberta Naturalists 2007). For waterbirds, 25 (89%) of the total 28 waterbird species were found at NSWs; 17 (61%) of all waterbird species were found at WPs. The 3 species found at natural wetlands (i.e., either reference or agricultural wetlands) that were not found at NSWs were Black Tern (*Chlidonias niger*), Eared Grebe (*Podiceps nigricollis*) and Ring-necked Duck (*Aythya collaris*). A single species, Franklin's Gull (*Leucophaeus pipixcan*), was observed at NSWs but neither of the natural wetland types.

The differences in wetland dependent songbird species richness among wetland types mirrored the differences in wet meadow zone width among wetland types. In fact, analysis of seven bird species considered to be specifically dependent on wet meadow habitat [i.e., Wilson's Snipe (*Gallinago delicate*), Common Yellowthroat, Le Conte's Sparrow (*Ammodramus leconteii*), Nelson's Sparrow (*Ammodramus nelsoni*), Song Sparrow, Lincoln's Sparrow (*Melospiza lincolnii*) and Swamp Sparrow; Federation of Alberta Naturalists 2007] found that increases in zone width had a significant correlation to the number of wet meadow species (total # species observed over the three surveys) at a wetland ($p<0.001$; $r^2=0.539$; Figure 2-4a). Thus, wider wet meadow zones resulted in a greater number of wet meadow dependent species. The strength and direction of this correlation also held true in terms of abundance of those same species ($p<0.001$; $r^2=0.538$; Figure 2-4b).

Plant Community Attributes

Total plant species richness (combining plot data and walkabout data) was significantly different among wetland types ($p=0.025$, $F_{3,28}=3.64$), but this was driven primarily by the low species richness of WPs. Even so, WPs were only significantly different from reference wetlands ($p=0.014$); WP plant species richness was comparable to that of NSWs ($p=0.371$) and agricultural wetlands ($p=0.259$). Native plant species richness was also significantly different among wetland types ($p<0.001$, $F_{3,28}=14.09$). Within just the species rich wet meadow zone, plant species richness (quadrat data only) was not different between wetland types ($p=0.533$, $F_{3,28}=0.75$). In contrast, in the relatively species poor emergent zone, the differences in species richness (quadrat data only) between wetland types was strongly significant ($p<0.001$, $F_{3,28}=15.84$). Natural reference (mean=13.3 spp.) and agricultural wetlands (mean=11.6 spp.) had similar species richness ($p=0.865$). NSWs had on average approximately half as many species (mean=6.0). In the few WP sites where an emergent zone was present, only a single species (common cattail, *Typha latifolia*) was recorded within plots.

Many previous studies caution against the use of species richness as a measure of mitigation success (Spieles 2005). Indeed, further analysis of the data revealed that interpretation of species richness alone can be misleading because it ignores the contribution of exotic, non-native species to the plant community. The proportion of the plant community consisting of exotic plant species differed dramatically between wetland types ($p<0.001$, $F_{3,28}=54.17$). The plant community of WPs consisted of over 30% non-native species (31.0%; Figure 2-5). The non-native proportion of the plant community was significantly lower ($p\leq0.001$) at NSWs (22.3%) and agricultural wetlands (14.9%). Reference wetlands showed a marked reduction in non-native species presence, with approximately half the non-native species contribution (7.7%) of agricultural wetlands ($p=0.004$). All pairwise comparisons among wetland types were strongly significant ($p<0.004$).

Vegetation structure metrics were based on vegetation height and density. Within the emergent zone of marshes, the height and density of the dominant species (typically common cattail) are two parameters that can be used to quantitatively describe the habitat. The average height (above water) of cattails ($p=0.243$, $F_{3,72}=1.42$) and the average density of cattail stems ($p=0.533$, $F_{3,63}=0.74$) did not differ between wetland types. Even at WPs, at the few sites that supported an emergent cattail zone, the structural parameters of that zone were not significantly different than those from cattail zones in agricultural and reference wetlands. In contrast, the structure of the wet meadow zone showed striking differences between wetland types.

The wet meadow visual obstruction measures (surrogate measure for above ground biomass) made following Robel et al. (1970) differed significantly among wetland types ($p<0.001$, $F_{3,28}=38.26$; Figure 2-5), however, it was only the sparse vegetation and corresponding low visual obstruction measurements of WPs that drove this difference. The visual obstruction measurements at NSWs were lower than both agricultural and reference wetlands, but not statistically different ($p\geq0.102$). Not surprisingly, analyses based on vegetation height showed a similar pattern ($p<0.001$, $F_{3,28}=17.75$). Wet meadow vegetation in WPs was not only more sparse, but shorter (all pairwise comparisons $p<0.001$) than the vegetation in reference, agricultural and NSW sites, which were not statistically different ($p\geq0.527$).

Hydrology

The hydrology of created stormwater wetlands differed dramatically compared to natural wetlands. The hydrology of created wetlands was characterized by multiple pulsed peaks, each with a steady, controlled drawdown in water depth back to the engineered normal water level. There was no seasonal drawdown in created wetlands. In contrast, natural wetland hydrology was characterized by a steady drawdown throughout the summer with very small, short term water level fluctuations. As is typical of the region, the increased evaporation relative to precipitation during the summer results in the lowering of wetland water levels (Winter and Rosenberry 1995). Among our study sites, seasonal drawdown in natural wetlands was considerable, averaging 0.49m and 0.39m for reference and agricultural wetlands respectively. Not surprisingly, the hydrological differences between natural and created wetlands produced strongly significant results when comparing seasonal amplitude across wetland types ($p \leq 0.001$, $F_{3,12}=13.79$). Pairwise comparisons were significant ($p \leq 0.03$) between created wetland types and natural wetland types, but not within the two broader wetland types ($p \geq 0.71$).

Interestingly, despite the different mechanisms controlling their maximum amplitude (maximum water depth minus minimum water depth), this metric did not differ among wetland types ($p=0.153$, $F_{3,12}=2.11$). The maximum amplitude of created wetlands was a result of the short-term increases in water depth caused by the storage of stormwater. In natural wetlands, the maximum amplitude came as a result of the seasonal lowering of water levels.

Discussion

The definitive benchmark for successful wetland compensation is often described as achieving a similarity between compensation (i.e., restored or created) wetlands and natural wetlands in the surrounding region (Galatowitsch and Van der Valk 1994). Ideally, created wetlands should mimic natural wetlands in terms of both structure and function. Some approaches, such as the hydrogeomorphic assessment approach (Smith et al. 1995) attempt to measure function directly, although wetland functions and associated processes are not easily measured (Fennessy et al. 2008). Accordingly, many studies focus on measuring elements of wetland structure, making the assumption that wetlands having similar structural characteristics should function in a similar or equivalent manner (Zampella and Laidig 2003). Although this assumption may not always hold true, the notion that improper wetland structure will negatively affect the functional capabilities of the wetland is well accepted (Hoeltje and Cole 2009). Thus, our investigation of created stormwater wetlands focused on shoreline slope as a key element of wetland structure that could possibly lead to differences in wetland zonation, bird community and vegetation structure between created and natural wetlands.

Shoreline slope and Wetland Zonation

As we had predicted, shoreline slopes of created wetlands were much steeper than natural wetlands. With natural wetland slopes averaging between 20:1 to 50:1 (H:V), the 5:1 to 10:1 slopes of created wetlands fell well short of replicating the shallow slopes characteristic of natural marsh wetlands. This failure in the design of created wetlands resulted in distinctly different wetland zonation, with the steeper slopes of created wetlands resulting in fewer, narrower wetland vegetation zones.

The link between steeper slopes and different zonation is not unique to our study. Zampella and Laidig (2003) found that the steeper bank slopes of excavated coastal plain ponds in New Jersey affected water depths along the shoreline, resulting in the lack of plant zonation typical of natural ponds. Jenkins and Greenway (2007) reported that wetland bathymetry, through its interaction with wetland hydrology, influences the length of inundation and soil saturation levels along shoreline areas. These processes act at relatively small scales such that wetland vegetation zones are known to change from one to the next over relatively small elevation changes (Sanderson et al. 2008). It is through this interaction of slope and water levels that wetland bathymetry exerts a strong influence over the patterns of vegetation zonation observed at created wetlands. Acknowledging the strong influence of this structural attribute, Jenkins and Greenway (2007) also showed that modifications to the bathymetry of the wetland can result in improvements to wetland inundation characteristics and, in turn, to broader, more natural vegetation zones. In addition to differences of shoreline slope, differences in the permeability of shoreline soils has been suggested as a contributing factor to narrower vegetation zones in created wetlands (McKinstry and Anderson 2002). It has also been suggested that some of the issues of poor soil quality in created wetlands may be mitigated by time (Uresk and Severson 1988) through decomposition and accumulation of organic matter. Zampella and Laidig (2003) argue, however, that time is not sufficient to overcome differences in shoreline slope.

Despite the dramatic differences in shoreline slope between natural and created wetlands, the differences among created wetland types suggest that naturalization efforts taken during the design and construction of NSWs have resulted in some improvements relative to WPs. The shallower below water slopes of NSWs illustrate that design efforts to create areas of shallow water are effective. Above normal water level, however, the slopes of NSWs and WPs did not differ. As stormwater management facilities, both created wetland types provide their capacity for stormwater storage in the area above normal water level. Having steep slopes extending out from the edge of the open water provides for the required storage and minimizes the area required to build the facility, a desirable situation considering that most stormwater wetlands are located in urban areas where land costs are high and the desire to maximize development potential is a strong factor in land-use planning decisions.

Wetland Birds

Our results show that the influence of shoreline slopes in created wetlands is not restricted to the pattern of vegetation zonation. From the perspective of wetland wildlife, the reduction in the number of wetland zones caused by steep shoreline slopes represents a reduction in habitat diversity. Similarly, the reduction in the width of wetland zones represents a reduction in habitat area. Combined, these factors resulted in reduced species richness and abundance of wet-meadow dependent songbirds among our created wetlands. Wider riparian vegetation zones have been shown to have increased structural complexity of vegetation (Cooke and Zack 2008). When one considers that the structure and cover pattern of vegetation can be more important than the actual plant species composition for influencing bird use of wetlands (Fairbairn and Dinsmore 2001), the observed link between wider wet meadow zones and increased richness and abundance of wet meadow dependent species makes sense. Indeed, the three wet meadow dependent bird species recorded at natural wetlands but not created wetlands (i.e., Alder Flycatcher, Common Yellowthroat and Swamp Sparrow) occur most commonly at vegetatively complex wetlands that include dense herbaceous vegetation and riparian shrubs (Federation of Alberta Naturalists 2007).

Interestingly, despite their narrower width, the emergent zone of naturalized stormwater wetlands was seemingly sufficient to support a waterbird species richness comparable to natural wetlands. The differences between the wet meadow and emergent zone and their differing influence on bird species richness, suggest that there may be some threshold for zone width above which most species typical of that habitat type can be supported. Currently, it is clear that naturalized stormwater wetlands fall below this threshold in terms of a wet meadow zone, but appear to be above the threshold in terms of having a sufficiently wide emergent zone. Alternatively, it could be that waterbirds are generally more tolerant of variable habitat conditions (Batt et al. 1992) compared to songbirds and, thus, more tolerant of variation in the emergent zone. Many waterbird species use elements of upland habitat, emergent vegetation and the open water, while wetland songbirds are more typically reliant on the habitat provided by just a single zone (i.e, wet meadow).

Plant Community Attributes

The most dramatic difference in the plant community among wetland types was the presence of numerous non-native species in created wetlands. Non-native species are widely acknowledged as indicators of environmental stress and generally considered to be detrimental to wetland function (Cohen et al. 2004). There are likely a number of factors that combine to result in the widespread presence of non-native species in created wetlands. The manicured and often weedy upland areas surrounding created wetlands provide an immediately adjacent and abundant seed source from which non-natives species can spread. Steep shoreline slopes likely result in relatively rapid gradients from saturated soils to drier soils, allowing upland non-native species to grow in close proximity to the water's edge. In our study, the vast majority of the non-native species were upland species that had established in the wet meadow zone, a pattern also seen in Bowers and Boutin (2008) for their work in southern Ontario, Canada. In contrast, regardless of wetland type, the inundated emergent zone was relatively free of non-native species. Although this was the case in our study area, non-native species grow as emergents in other geographic areas, including *Typha* spp. hybrids, purple loosestrife and *Phragmites australis* (T'ulbure et al. 2007). It is also possible that the soils of created wetlands are of poorer quality (Uresk and Severson 1988, Stolt et al. 2000) and thus facilitate the establishment of opportunistic non-native species relative to native species. Once established, there appears to be few natural mechanisms that control non-native species in created wetlands. The rapid drawdown in water level following storm events in created stormwater wetlands results in only short-term inundation which appears to be insufficient to kill non-native upland species.

Vegetation structure is known to be important for structuring wetland wildlife communities (Fairbairn and Dinsmore 2001). Despite the differences in zone width, emergent zones had similar structure (i.e., vegetation height and stem density) across all wetland types in our study. Common cattail was by far the dominant species in emergent zones of all wetland types. We believe that the similarity among emergent zones is largely a reflection of the growth pattern of this dominant species, as cattails often grow as dense, near monocultures. In contrast, important differences were apparent among wet meadow zones. Not surprisingly, the rip-rap along the shorelines of WPs inhibited vegetation growth, resulting in sparse, low-growing vegetation dominated by weedy, non-native species. At NSWs, despite having a high proportion of non-native species compared to natural wetlands, the wet meadow zone supported sufficient numbers of native wetland plant species to result in comparable structural metrics as the tall, dense herbaceous growth typical of wet meadow vegetation in natural wetlands.

Hydrology

Hydrology is frequently cited as one of the most important factors in determining success for wetland mitigation (Spieles et al. 2006, Jenkins and Greenway 2007, Hoeltje and Cole 2009). Although our study did not focus on hydrology, it is clear that the hydrology of created stormwater wetlands is fundamentally different than that of natural marsh wetlands. Considering this, it is expected that hydrology plays a role in structuring the zonation of created wetlands. The question of exactly how hydrology interacts with shoreline slope and which plays a more significant role, is more difficult to ascertain. The lack of seasonal drawdown in created wetlands is one obvious source of influence, and this has been hypothesized as a factor contributing to the narrow wetland zonation of created wetlands (McKinistry and Anderson 2002). It is, however, not unreasonable to think that the pulsed hydrology of stormwater wetlands should be capable of sustaining natural wetland zonation. With shallow shoreline slopes and maximum amplitudes of the same magnitude as natural wetlands, the pulsed hydrology of stormwater wetlands should result in frequent, short-term flooding of wide areas surrounding the normal water level. These periodic flooding events could maintain sufficient soil saturation levels for long enough to sustain a healthy wet meadow zone. Bonilla-Warford and Zedler (2002), however, highlighted the problem of fluctuating water levels and noted that our limited knowledge of plant flooding tolerances and the associated poor choices made during wetland design are among the main reasons for poor species establishment and lack of native wetland plant diversity at created stormwater wetlands. Thus, in the design of created wetlands, it is necessary to carefully consider not only water depth, but other hydrological parameters including drawdown and length of inundation. Through these mechanisms, hydrology holds strong potential to influence the zonation at created wetlands. However, even with proper hydrology, artificially steep shoreline slopes will restrict the establishment of naturally patterned vegetation zones.

Management Implications

It has been argued that it is unreasonable to expect natural levels of ecological functioning to be sustained by created wetlands, particularly in highly urbanized settings (Brinson and Rheinhardt 1996). Our data show that created stormwater wetlands do indeed fall short of replicating natural wetlands across many of the parameters we measured. Current design standards for created wetlands, which outline unnaturally steep shoreline slopes and are largely guided by the need to provide stormwater storage, are acting as a major impediment to the creation of natural wetland vegetation zonation at stormwater wetlands. Fundamental to successful marsh wetland creation is the establishment the shallow slopes observed in natural wetlands. Steep shoreline slopes will impede the establishment of natural wetland zonation, even if the hydrology, soils and other factors are all suitable. Created wetlands designed and built with shallower shoreline slopes will likely develop more natural vegetation zonation and, thus, be more likely to support bird and plant communities similar to natural wetlands of the region.

Literature Cited

Alberta Environment Protection. 1999. Stormwater management guidelines for the Province of Alberta. Municipal Program Development Branch, Environmental Sciences Division, Environmental Service. Edmonton, AB.

Alberta Water Resources Commission. 1993. Wetland management in the settled area of Alberta: An interim policy. Alberta Water Resources Commission. Edmonton, AB.

Balcombe, C.K. 2003. An evaluation of vegetation and wildlife communities in mitigation and natural wetlands of west Virginia. M.Sc. Thesis. West Virginia University, 417 pages

Balcombe, C. K., J. T. Anderson, R. H. Fortney, J. S. Rentch, W. N. Grafton, and W. S. Kordek. 2005. A comparison of plant communities in mitigation and reference wetland in the mid-Appalachians. Wetlands 25:130–42.

Batt, B.D, A.D. Afton, M.G. Anderson, C.D. Ankney, D.H. Johnson, J.A. Kadlec and G.L. Krapu. 1992. Ecology and management of breeding waterfowl. University of Minnesota Press. Minneapolis, MN.

Bonilla-Warford, C.M. and J.B. Zedler. 2002. Potential for using native plant species in stormwater wetlands. Environmental Management 29: 385-394

Bowers, K. and C. Boutin. 2008. Evaluating the relationship between floristic quality and measures of plant biodiversity along stream bank habitats. Ecological Indicators 8:466-475.

Brinson, M.M. and R. Rheinhardt. 1996. The role of reference wetlands in functional assessment and mitigation. Ecological Applications 6: 69-76.

Brown, S.C. and C.R. Smith. 1998. Breeding season bird use of recenlty restored versus natural wetlands in New York. Journal of Wildlife Management 62: 1480-1491.

Campbell, D.A., C.A. Cole, and R.P. Brooks. 2002. A comparison of created and natural wetlands in Pennsylvania, USA. Wetlands Ecology and Management 10: 41–49.

City of Edmonton. 2008. Design and construction standards. Volume 3: Drainage. City of Edmonton. Edmonton, AB.

Cohen, M.J., S. Carstenn and C.R. Lane. 2004. Floristic quality indices for biotic assessment of depressional marsh condition in Florida. Ecological Applications 14: 784-794.

Commission for Environmental Cooperation. 1997. Ecological regions of North America: toward a common perspective. Commission for Environmental Cooperation, Montreal, Quebec, Canada. 71pp.

Cooke, H.A. and S. Zack. 2008. Influence of beaver dam density on riparian areas and riparian birds in shrubsteppe of Wyoming. Western North American Naturalist 68: 365-373.

Dahl, T.E. 2000. Status and Trends of Wetlands in the Conterminous United States 1986 to 1997. Washington, DC: U.S. Department of the Interior, Fish and Wildlife and Service.

Daubenmire, R. F. 1959. A canopy-cover method of vegetational analysis. Northwest Science 33:43–46.

Delphey, P.J. and J.J. Dinsmore. 1993. Breeding bird communities of recently restored and natural prairie potholes. Wetlands 13: 200-206.

Fairbairn, S.E. and J.J Dinsmore. 2001. Local and landscap-level influences on wetland bird communities of the prairie pothole region of Iowa, USA. Wetlands 21: 41-47.

Federation of Alberta Naturalists. 2007. The atlas of breeding birds of Alberta: A second look. Federation of Alberta Naturalists. Edmonton, AB.

Fennessy, M.S., A. Rokosh and J.J. Mack. 2008. Patterns of plant decomposition and nutrient cycling in natural and created wetlands. Wetlands 28: 300-310.

Fisher, C.C. and J. Acorn. 1998. Birds of Alberta. Lone Pine Publishing. Edmonton, AB. 384 pgs.

Galatowitsch, S.M. and A.G. van der Valk. 1994. Restoring Prairie Wetlands. Ames, Iowa. Iowa State University Press. 246 pp.

Hartzell, D., J.R. Bidwell and C.A. Davis. 2007. A comparison of natural and created depressional wetlands in central Oklahoma using metrics from indices of biological integrity. Wetlands 27: 794-805.

Hoeltje, S.M. and C.A. Cole. 2009. Comparison of function of created wetlands of two age classes in central Pennsylvania. Environmental Management 43: 597-608.

Ito, P.K. (1980). Robustness of ANOVA and MANOVA test procedures. In Handbook of statistics 1. Krishnaiah, P.R. (ed.). North Holland. Amsterdam, Holland. 1002 pp.

Jenkins, G.A. and M. Greenway. 2007. Restoration of a constructed stormwater wetland to improve its ecological and hydrological performance. Water Science and Technology 56: 109-116.

Johnson, D., L. Kershaw and A. MacKinnon. 1995. Plants of the western boreal forest and aspen parkland. Lone Pine Publishing. Edmonton, AB, Canada.

Juni, S. and C. R. Berry. 2001. A biodiversity assessment of compensatory mitigation wetlands in eastern South Dakota. Proceedings of the South Dakota Academy of Science 80:185–200

Kellogg, C. H., and S. D. Bridgham. 2002. Colonization during early succession of restored freshwater marshes. Canadian Journal of Botany 80: 176-185.

Kennedy, G. and T. Mayer. 2002. Natural and constructed wetlands in Canada: An overview. Water Quality Research Journal of Canada 37: 295-325.

Kunz, K., M. Rylko and E. Somers. 1988. An assessment of wetland mitigation practices in Washington State. National Wetlands Newsletter 10: 2-4.

Lahring, H. 2003. Water and wetland plants of the prairie provinces. Canadian Plains Research Center. Regina, SK, Canada.

Locky, D.A. and S.E. Bayley. 2006. Plant diversity, composition, and rarity in the southern boreal peatlands of Manitoba, Canada. Canadian Journal of Botany 84: 940-955.

Matthews, J.W. and A.G. Endress. 2008. Performance criteria, compliance success, and vegetation development in compensatory mitigation wetlands. Environmental Management 41: 130-141.

McKinstry, M. and S. Anderson. 2002. Creating wetlands for waterfowl in Wyoming. Ecological Engineering 18: 293-304.

Moss, E.H. 1983. Flora of Alberta. University of Toronto Press. Toronto. 687pp.

Natural Regions Committee 2006. Natural regions and subregions of Alberta. Compiled by D.J. Downing and W.W. Pettapiece. Government of Alberta. Pub. No. T/852.

Olson, R. 1999. Constructing wetlands in the intermountain west: Guidelines for land resource managers. University of Wyoming. Laramie, Wyoming. 52 pgs.

Puchniak, A.J. 2002. Recovery of bird and amphibian assemblages in restored wetlands in Prairie Canada. M.Sc. Thesis, University of Alberta. 220 pages

Ratti, J.T., A.M. Rocklage, J.H. Giudice, E.O. Garton and D.P. Golner. 2001. Comparison of avian communities on restored and natural wetlands in North and South Dakota. Journal of Wildlife Management 65: 676-684.

Robel, R.J., J.N. Briggs, A.D. Dayton, and L.C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23:295–297.

Rumble, M. A., M. T. Anderson and C. L. Hawkes. 1985. Morphometry of coal and bentonite surface mine and livestock impoundments in the northern high plains. Reclamation and Revegetation Research 3:293-300.

Sanderson. J.S., N.B. Kotliar and D.A. Steingraeber. 2008. Opposing environmental gradients govern vegetation zonation in an intermountain playa. Wetlands 28: 1060-1070.

Smith, R.D., A. Ammann, C. Bartoldus, and M.M. Brinson. 1995. An approach for assessing wetland functions using hydrogeomorphic classification, reference wetlands, and functional indices. Technical Report WRP–DE–9, U.S. Corps of Engineers, Army Engineer Waterways Experiment Station, Vicksburg, MS.

SOCE. 1991. The state of Canada's environment. The Government of Canada. Ottawa, Ontario.

Spieles, D.J. 2005. Vegetation development in created, restored and enhanced mitigation wetland banks of the United States. Wetlands 25: 51-63.

Spieles, D.J., M.Coneybeer and J. Horn. 2006. Community structure and quality after 10 years in two central Ohio mitigation bank wetlands. Environmental Management 38:837-852.

Stanczak, M and J.B. Keiper. 2004. Benthic invertebrates in adjacent created and natural wetlands in northeastern Ohio, USA. Wetlands 24: 212-218.

Stewart, R.E. and H.A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. Resource Publication 92, Bureau of Sport Fisheries and Wildlife, U.S. Fish and Wildlife Service, Washington, D.C. 57pp

Stolt, M.H., M.H. Genthner, L. Daniels, V.A. Groover, S. Nagle and K.C. Haering. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands 20: 671-683.

Strong, W.L. and K.R. Leggat. 1992. Ecoregions of Alberta. Alberta Forestry, Lands and Wildlife, Edmonton, Alberta, Canada. 55 pp + App

Streever, W. J., K. M. Portier, and T. L. Crisman. 1996. A comparison of dipterans from ten created and ten natural wetlands. Wetlands 16: 416–428.

T'ulbure, M. G., C. A. Johnston and D.L. Auger. 2007. Rapid invasion of a Great Lakes coastal wetland by non-native *Phragmites australis* and *Typha*. Journal of Great Lakes Research 33: 269-279.

Uresk, D.W. and K. Severson. 1988. Waterfowl and shorebird use of surface-mined and livestock water impoundments on the northern Great Plains. Great Basin Naturalist 48: 353-357.

Weber, M. J., P. A. Vohs, Jr., and L. D. Flake. 1982. Use of prairie wetlands by selected bird species in South Dakota. Wilson Bulletin 94:550-554.

Winter, T. C and D. O. Rosenberry. 1995. The interaction of ground water with prairie pothole wetlands in the Cottonwood Lake area east-central North Dakota, 1979-1990. Wetlands 15: 193-211.

Zampella, R. A. and K. J. Laidig. 2003. Functional equivalency of natural and excavated coastal plain ponds. Wetlands 23:860-876.

Table 2-1. Vegetation zone width and slope characteristics of wetlands surveyed in central Alberta.

	Emergent zone width (m)		Wet meadow zone width (m)		Above-water slope (%)		Below-water slope (%)	
Wetland Type	Mean ± SD	Range	Mean ± SD	Range	Mean ± SD	Range	Mean ± SD	Range
Reference	15.1 ± 10.7	5.0 – 31.0	20.4 ± 8.1	13.3 – 32.3	2.0 ± 1.4	0.3 – 4.7	4.4 ± 2.5	0.9 – 8.0
Agricultural	15.1 ± 10.1	2.0 – 31.0	12.1 ± 7.1	3.3 – 23.3	4.3 ± 4.3	0.7 – 14.1	4.5 ± 2.2	1.4 – 7.7
Naturalized Stormwater	5.7 ± 4.4	2.0 – 14.0	3.6 ± 2.4	1.5 – 8.3	11.3 ± 2.7	7.9 – 16.1	8.9 ± 4.5	3.8 – 15.2
Wet Pond	0.8 ± 1.4	0.0 – 4.0	2.2 ± 0.6	1.3 – 3.0	14.6 ± 2.7	10.3 – 17.6	14.3 ± 1.7	11.1 – 16.3

Table 2-2. Summary of mean bird species richness observed at different wetland types.

	Wetland Type					
Species Type	Reference	Agricultural	NSW	Wet Pond	Reference+Agricultural	Total
Songbirds	10	10	8	2	11	11
Waterbirds	23	26	25	17	27	28
Total	33	36	33	19	38	39

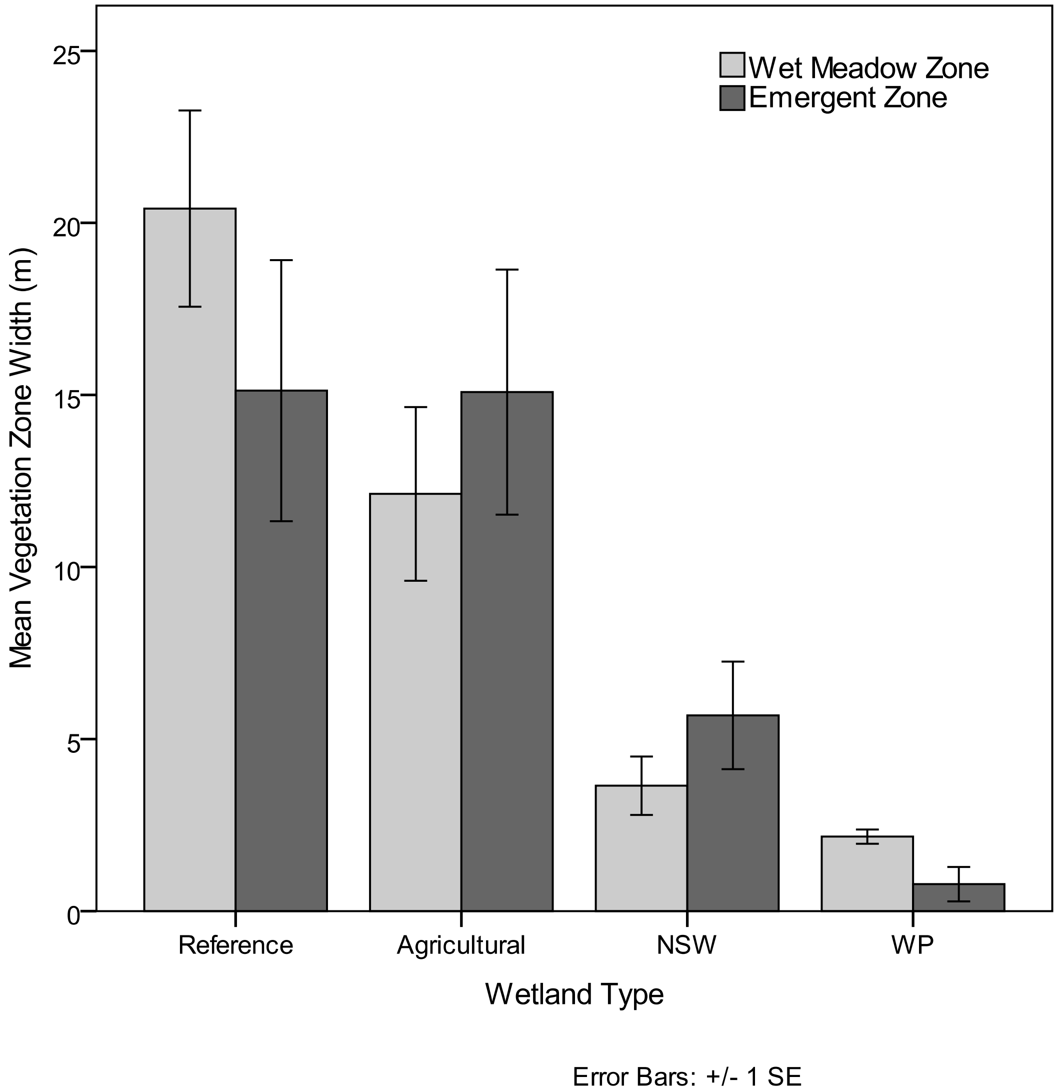
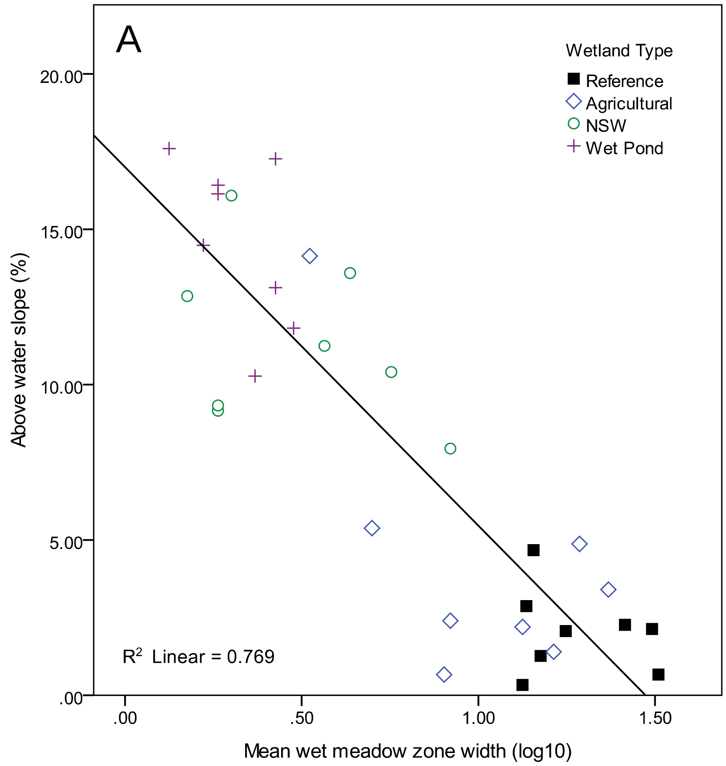


Figure 2-1. Width of emergent and wet meadow zones in 4 different wetland types. NSW = naturalized stormwater wetlands; WP = wet ponds.



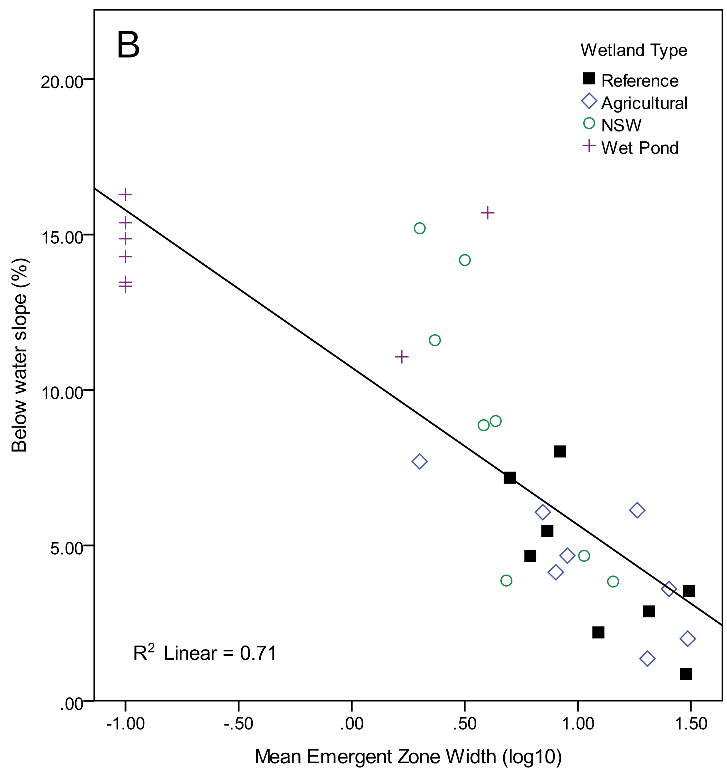


Figure 2-2. Correlation between the steepness of the shoreline slope and the width of wet meadow (Fig. 2-2a) and emergent (Fig. 2-2b) zones. A logarithmic transformation was applied to zone width to yield a linear relationship. NSW = naturalized stormwater wetland

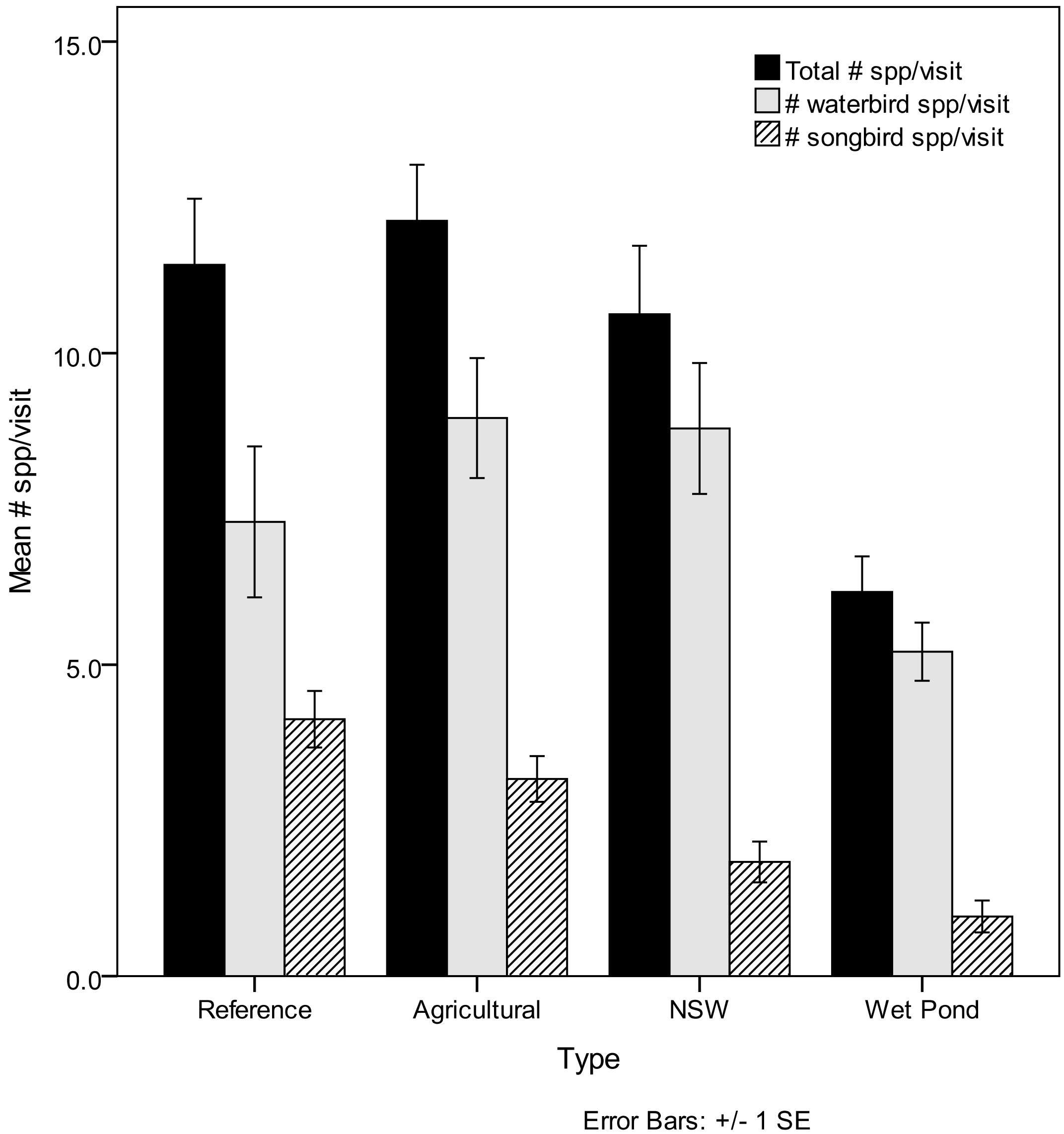


Figure 2-3. Average species richness per bird survey recorded at 4 different wetland types. NSW = naturalized stormwater wetland.

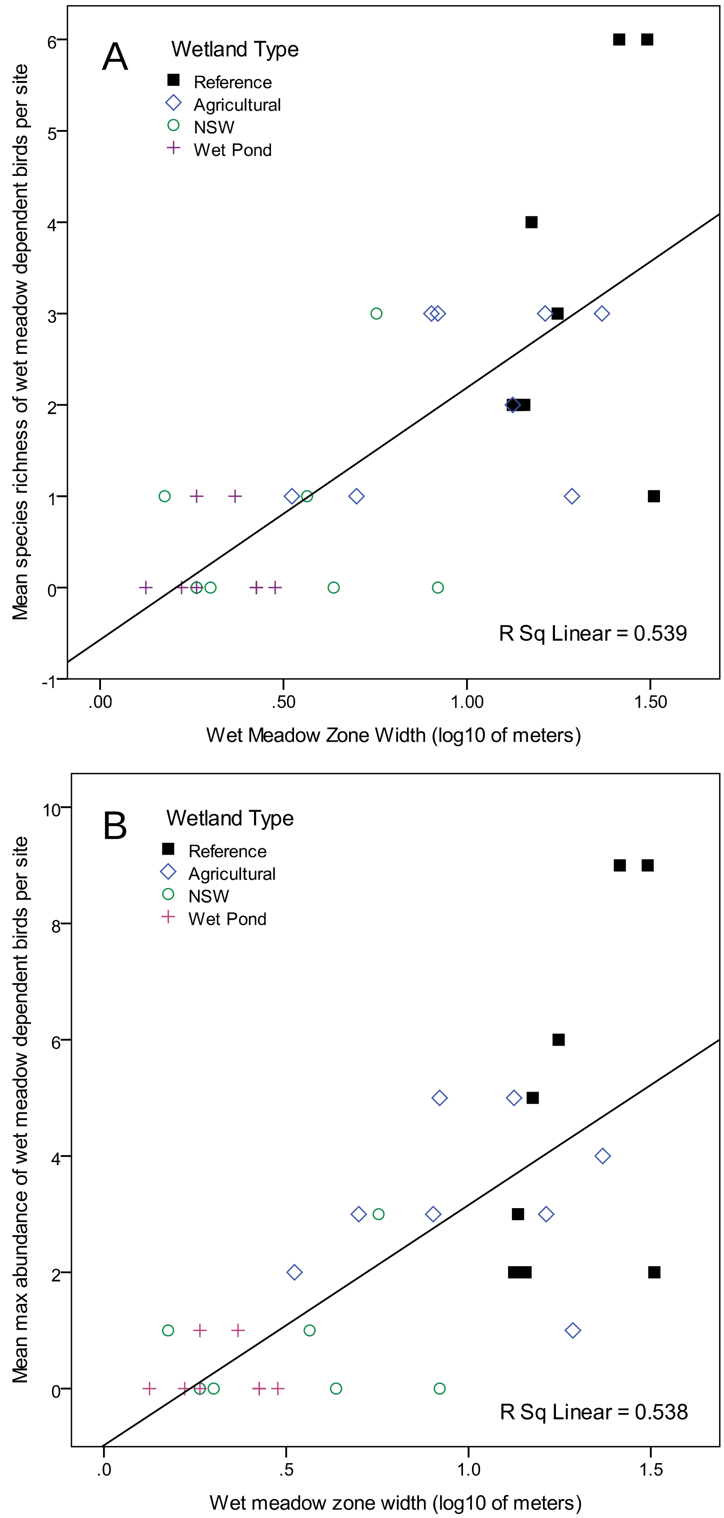


Figure 2-4. Correlation between the overall species richness (Fig. 2-4a) and maximum abundance (Fig 2-4b) of wet meadow dependent bird species with the width of the wet meadow zone. A logarithmic transformation was applied to zone width to yield a linear relationship.

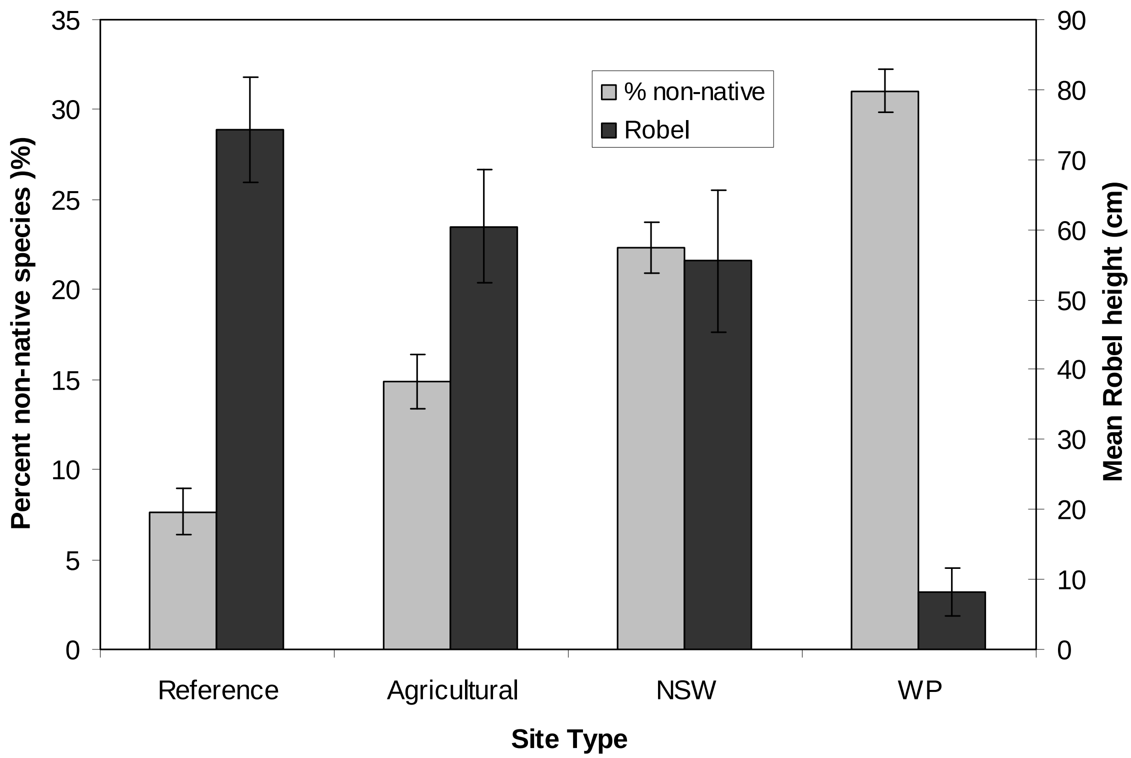


Figure 2-5. Mean proportion of non-native species in the plant community (shown in gray on the left axis) and mean Robel heights measured in the wet meadow zone (as a relative measure of aboveground biomass; shown in dark gray on the right axis) of four different wetland types. NSW = naturalized stormwater wetlands; WP = wet ponds.

CHAPTER 3

Development and evaluation of a floristic quality assessment system for marsh wetlands in Alberta's Parkland and Boreal Natural Regions

Introduction

Since amendments were made to the United States Clean Water Act in the late 1970's, compensatory wetland mitigation has become common across much of the United States (Spieles 2005). With these regulations in place, it has become necessary to quantify the success of wetland creation and restoration projects at mitigating the lost area and function of natural wetlands. There is still much debate over how to best assess wetland mitigation, however, various forms of wetland bioassessments have been used extensively to determine wetland condition and, in turn, the success of wetland mitigation. In these bioassessments, biological indicators are used as surrogates for complex ecological processes and wetland functions that are difficult to measure directly. Among the most well developed and commonly used biological indicators are wetland plants (US EPA 2002, Spieles 2005, Miller et al. 2006). Plants have been used as indicators in numerous regions, have well-documented response thresholds to wetland degradation, integrate disturbances at numerous biological scales (Ervin et al. 2006, Mack 2007), and are relatively easy to survey and monitor over time (Bowers and Boutin 2008). Among the many aspects of wetland plants and their associated communities that can be quantitatively measured (e.g., species richness, number of non-native species), one approach that has received much attention and has been found to be very useful is Floristic Quality Assessment (FQA).

The FQA approach was originally developed in the late 1970s as a method for quantifiable measurement of habitat quality in the Chicago area (Swink and Wilhelm 1979). The main concept behind the FQA approach is species conservatism. Through evolutionary processes, plant species develop unique life strategies and adaptations that result in specific tolerances to disturbance (whether it be natural or anthropogenic in nature) and differing fidelity to natural undisturbed habitats or communities (Milburn et al. 2007). The FQA approach uses a numerical rating system, called the coefficient of conservatism (hereafter C-value), to quantify a species tolerance to disturbance and affinity to natural habitats. C-values range from 0 to 10, where species with a C-value of 10 would exhibit very limited tolerance to disturbance and a high degree of fidelity to a narrow range of ecological parameters characteristic of undisturbed natural habitats. At the other end of the range, a C-value of 0 represents a non-native or invasive native species and low C-values represent generalist species capable of tolerating substantial levels of disturbance. Plant species that commonly occur in natural habitats, but that can also be found at both degraded and undegraded sites are assigned intermediate C-values. A species' level of conservatism includes a certain measure of relativity with respect to all other species in a region (i.e., the relative sensitivity of a highly sensitive species would decrease if there are even more highly sensitive species present). Thus, FQASs must be developed for a defined geographic area in which C-values are assigned by considering the level of conservatism of each species relative to all other species in that area.

Once a list of plant species for a specified geographic area have been assigned C-values, it is then possible to calculate various metrics that can quantify the overall floristic quality of a site. When compared to the same measures obtained from a reference site, these metrics can be used to assess the relative quality of a site. The most common FQA metrics are the mean C-value of all native species at a site and the Floristic Quality Index (FQI) which is the product of the site mean C-value and the square root of the total native species richness. The original approach to FQA did not include the contribution of non-native species because it was argued that non-native species had evolved separately from local, native plant communities and thus natural area assessments should avoid combining and comparing between non-native and native regional flora (Swink and Wilhelm 1979, Ervin et al. 2006). Since the initial development of the FQA approach, several authors have refined the original metric equations and presented them as improvements over the original (Cohen et al. 2004, Miller and Wardrop 2006, DeBerry 2006). Published refinements on the FQA have included approaches that include the contribution of non-native species to the plant community (e.g., Miller and Wardrop 2006) and consideration of relative abundance of species within a plant community (e.g., Cohen et al. 2004).

The FQA approach to site assessment has been repeatedly shown to be highly correlated to various independently developed disturbance ratings and other measures of wetland site quality. For depressional wetlands in Ohio, the FQI was strongly correlated with an a prior assigned disturbance rank and various measures of soil quality (Lopez and Fennessy 2002). The FQI was also highly correlated with disturbance gradients in riverine wetlands (Fennessy et al. 1998a) and in depressional marshes and swamps (Fennessy et al. 1998b). In North Dakota, Mushet et al. (2002) showed the utility of the FQA approach for monitoring the development of restored ecosystems over time. Considering the proven success of the FQA approach, several jurisdictions have applied C-values to their regional flora and adopted the FQA approach: Missouri (Ladd 1993), Ohio (Andreas et al. 2004), Michigan (Herman et al. 1997), Illinois (Taft et al. 1997, Swink and Wilhelm 1994), North Dakota (Northern Great Plains FQA Panel 2001), Wisconsin (Bernthal 2003), Indiana (Rothrock 2004), Pennsylvania (Beatty et al. 2002), Minnesota (Milburn et al. 2007), Florida (Cohen et al. 2004) and Mississippi (Herman et al. 2006). The only example of the FQA approach in Canada has come from southern Ontario (Oldham et al. 1995).

The primary objectives of this study were to develop a FQA system (FQAS) that would be applicable in Alberta and to evaluate its usefulness as a wetland condition assessment tool. Despite Alberta having a provincial policy that promotes wetland restoration and creation, there are no standardized approaches in the province to assess wetland condition, nor are there any standardized quantitative ecological performance standards that could be used in such an assessment. As part of the evaluation of the FQAS, this study also set out to compare relatively complex floristic quality metrics derived from the FQAS against more traditional and basic measures of plant community composition. Specifically, we wanted to investigate the influence of merging a non-native species component and measures of relative abundance into the calculation of FQA metrics.

Methods

Alberta Floristic Quality Assessment System Development

Geographic Extent

The first step in the development process was to define geographic criteria to limit the extent of the floristic quality assessment. Previous studies have outlined that the development of biological indicators must be regionally specific (Angermeier et al. 2000). FQASs are particularly sensitive to geographic differences and coefficients of conservatism (hereafter C-values) are typically assigned independently among different geographic areas to acknowledge the difference in species' tolerances across their range (Oldham et al. 1995, Bernthal 2003, Milburn et al. 2007). Species tolerances and habitat requirements can differ between the center and the periphery of species' ranges. Alberta is a very large (661,848 km^2), ecologically and geographically diverse province. We decided to restrict our initial efforts to only the boreal forest and parkland natural regions of Alberta (Natural Regions Committee 2006), two regions of the province that are currently facing issues related to land development and wetland loss. Accordingly, two sets of C-values were assigned to each species; one for the parkland region and one for the boreal. It has been suggested that the use of ecologically meaningful geographic units may improve the accuracy of FQASs (Bourdaghs et al. 2006).

<u>Wetland and Species Type</u>

Wetlands in Alberta are sub-divided into five wetland types: bog, fen, swamp, marsh and shallow open water (Alberta Water Council 2007). The development of the Alberta FQAS was further restricted to marsh and shallow open water wetlands. These wetland types were chosen because they are the most common wetland types in the parkland natural region (Natural Regions Committee 2006) and because doing so maintained consistency with other closely related research focusing on marsh wetlands in the parkland and boreal regions. Species unique to other wetland types, such as bogs and fens, were not included. Other wetland types will likely be included in future revisions of Alberta's FQAS.

Bryophytes were largely excluded from the list, although a few species commonly encountered in marshes were included. Plants included in the FQAS varied from submersed aquatic species and emergent species (i.e., obligate wetland and facultative wetland species), to species requiring less saturated conditions that could potentially occur in wetland-upland transitional habitats at the edge of wetlands (i.e., facultative species). This approach differs slightly from some other studies where species had to meet a certain level of wetland indicator status for inclusion in the wetland FQAS (Milburn et al. 2007).

Species List

Using the restrictions and guidelines we had set in place for the development of the FQAS, we compiled a list of 407 plant species (Appendix A). The list was initially gathered using species records from two wetland studies in the boreal region and one study in the parkland region. That initial list was then supplemented with species from the Alberta Natural Heritage Information Centre's (ANHIC) List of All Vascular Plant Elements (2006) that were identified as having the potential to occur in marsh wetland habitats in the boreal and/or parkland natural regions of Alberta based on accepted provincial distributions and species descriptions (Moss 1994, Lahring 2003).

Assignment of Coefficients of Conservatism

In jurisdictions where plant species C-values exist already, the assignment process has followed one of two common approaches or, in some cases, a combined approach. In all cases, the development of FQASs have relied on the expert opinion of botanists with knowledge of the flora for the designated geographic extent of the study, typically at the State level. Some studies have convened a panel of botanists and conducted a workshop in which C-values are assigned through discussion and a consensus decision approach (Bernthal 2003). In others, botanists were contacted independently and asked to assign C-values in isolation. In those cases the most common approach has been to average the

C-values assigned to each species by all participating botanists and take the mean value as the final C-value (Cohen et al. 2004, Herman et al. 2006). In some cases a combined approach has been used where the authors have assigned C-values themselves before distributing the values for external review and then refining the values as deemed necessary based on the external recommendations (Oldham et al. 1995, Ervin et al. 2006).

For the development of Alberta's FQAS we opted to contact botanists independently. This approach avoided the difficult scheduling issues inherent to gathering a meeting of professionals and university faculty from across a large geographic area. An independent approach can also yield results in a more timely manner, garner increased participation and eliminate the possible bias of single individuals in a workshop setting, as has been noted as a possible effect of the workshop approach by others (Herman et al. 2006). Eight botanists agreed to participate in the development of Alberta's FQAS. The list of participating botanists included university faculty, provincial and federal government personnel, professional consultants and well-respected, amateur botanists.

To establish a common basis of knowledge, each participant was sent a background document that explained the concepts of biological assessment, floristic quality assessment and species conservatism. Of particular importance, each participant was provided with the following breakdown of C-value scores that was used in the development of the Ohio FQAI (Andreas and Lichvar 1995).

0: all non-native (alien, exotic) species and native species that are opportunistic invaders of natural areas or those that are typically part of ruderal communities;

1-3: native species found in a wide variety of plant communities and very tolerant of disturbance;

4-6: native species typically associated with a specific plant community, but can tolerate moderate disturbance;

7-8: native species found in a narrow range of plant communities in advanced stages of succession, but can tolerate minor disturbance;

9-10: native species restricted to a narrow range of ecological conditions, with very low to no tolerance of disturbance.

Participants were asked to follow this same template in assigning C-values for the development of the Alberta FQAS. Participants were also provided with detailed process related instructions outlining the approach for assigning C-values. We asked that participants assign C-values only to species with which they felt experienced and comfortable, preferring to receive fewer confidently-assigned values instead of a greater number of speculativive values.

Once all responses had been received, we analyzed the disagreement/ variability among the assigned C-values across each species. In cases when there was high disagreement among participants, we used Peirce's criterion (Peirce 1852, Ross 2003) to eliminate C-value outliers before calculating and assigning final C-values. Despite being virtually unused in scientific literature in recent years, Peirce's criterion "is a rigorous method based on probability theory that can be used to eliminate data outliers or spurious data in a rational way" (Ross 2003). With the outliers removed, we calculated the median value of the remaining assigned C-values. Because C-values are reported explicitly as whole numbers, median values with a decimal place were rounded up to the nearest whole number. We chose to use the median value instead of the mean value in an attempt to arrive at C-values that were best representative of the "average" assigned value.

Assessment of Alberta's Floristic Quality Assessment System

One of the primary objectives of a FQAS is to develop a quantitative and standardized assessment of wetland condition. Although our approach was comparable to several other studies and the fact that FQA metrics such as mean C-value and the FQI have been repeatedly shown to be closely correlated with various measures of site disturbance in many other jurisdictions (Lopez and Fennessy 2002, Mushet et al. 2002), we wanted to ensure that the newly developed Alberta FQA would function in a similar manner. To achieve this validation, we relied on linear regression analyses between several FQA metrics calculated for a suite of 32 wetlands and an independently developed disturbance gradient, as has been done in previous studies (Lopez and Fennessy 2002, U.S.

EPA 2002, Ervin et al. 2006). Additionally, the metrics with the strongest correlations were regressed against each other to assess metric redundancy. We also used analysis of variance (ANOVA) to assess the efficacy of the same FQA metrics to differentiate among four wetland types determined a priori and chosen to span a broad range of anthropogenic disturbances.

Disturbance Gradient

The disturbance gradient used in the validation of the Alberta FQAS was developed using a suite of 32 wetlands from within the zone of relevance of the FQAS, including parts of both the parkland and boreal natural regions. The approach to the development of the disturbance gradient closely followed the objective approach recently developed by Rooney and Bayley (2010). Multiple water chemistry (e.g., total cations, total nitrogen), sediment (e.g., % water in sediment) and physical parameters (e.g., maximum depth, amplitude) were measured and then an optimal sub-set of eight variables was selected using the data's correlation structure as an objective approach to variable selection. Variables were standardized by percentile binning and then weighted such that the various categories were weighted equally. The approach used in the development of the disturbance gradient was initially created for use in testing biotic metrics for the development of a locally-relevant indices of biotic integrity (IBIs) and thus is appropriate to use as a correlate for FQA based metrics.

Wetland Sites used for Validation

Thirty-two permanent, open water marsh wetlands (Class V; Stewart and Kantrud 1971) wetlands from in and around the City of Edmonton, Alberta were selected for use as validation of the Alberta FQAS. The City of Edmonton (53.54°N latitude and 113.50°W longitude) is located in the parkland natural region (Natural Regions Committee 2006) at the northern extreme of the North American Great Plains Ecoregion (Commission for Environmental Cooperation 1997), but some of the wetlands were located in an area approximately 30km east of Edmonton that is actually a southern, disjunct part of the boreal forest natural region. The Edmonton area is characterized by a climate typical of northern temperate climates with warm summers and cold winters. The mean annual temperature is 3.9°C and the mean annual precipitation is 477mm (averaged 1971-2000; Environment Canada 2008). Marsh wetlands, located in undrained post-glacial depressions are the most common wetland type in east-central Alberta (Natural Regions Committee 2006). For these reasons, this region provides a suitable testing ground for FQAS developed using the 407 species records compiled from marsh wetlands across the northern half of Alberta.

The suite of 32 wetlands included both natural and constructed wetlands. Constructed wetlands (n=16) were one of two types of stormwater management facilities: wet ponds (n=8) or naturalized stormwater wetlands (n=8). Wet ponds (WPs) represent the old standard stormwater facility type that was typically constructed during the 1980's and 1990's that have no wetland naturalization and have stone riprap along their shorelines to protect against erosion.

Naturalized stormwater wetlands (NSWs) include specific design and construction features such as areas of shallow water, placement of salvaged wetland soils, undulating shorelines and planting of aquatic plant species, all intended to create more typical marsh wetland habitat. Because of their stormwater functions, all WPs and NSWs were situated in urban areas or in close proximity to provincial roadways.

Natural wetlands were sub-divided into two categories: agricultural wetlands (n=8) and undisturbed reference wetlands (n=8). Agricultural wetlands were defined as wetlands having at least 50% of the land within a 500m buffer under cultivation or pasture. Undisturbed reference wetlands were defined as wetlands located within a protected area (e.g., Elk Island National Park) and having less than 10% of the land within 500m under cultivation or pasture, with no agricultural land directly bordering the wetland.

Potential sites were identified using high resolution (0.25m pixels) 2007 digital aerial photography. Permanency of selected sites was confirmed through a review of aerial photographs from multiple years. All sites were ground truthed prior to the initiation of fieldwork to confirm suitability for inclusion in the study. Selected constructed wetlands were at least 3 years old. All wetlands were between 1 ha and 10 ha in size.

Although the four wetland types were defined categorically, we hypothesized that these four wetland types would also span a broad spectrum of human disturbance and effectively form a gradient from low disturbance to high disturbance. Figure 3-1 plots all 32 sites against the disturbance gradient and illustrates that the 32 sites do indeed form a consistent gradient.

Vegetation Surveys

Vegetation surveys were conducted at each wetland between 31 July, 2008 and 27 August, 2008; the optimal time for vegetation surveys in central Alberta. Vegetation surveys consisted of two components: a quadrat based survey followed by a time-restricted walkabout survey.

Prior to fieldwork, three survey transects perpendicular to the shoreline were randomly identified for each wetland using recent (2007) aerial photographs. At each transect, the number of distinct vegetation zones between the open water and upland-wetland interface was determined. Zones were differentiated based on the composition of dominant species. Within each zone, two 1 m^2 quadrats were placed at the mid-point of each zone; individual quadrats were separated by 5 m in a direction perpendicular to the transect. All plants within each quadrat were identified to species and percent cover was estimated using a modified Daubenmire (1959) cover-abundance scale.

Following the quadrat surveys, a time-restricted walkabout survey was completed by a pair of surveyors to ensure that a comprehensive inventory of all plant species at each wetland was obtained. The walkabout survey focused on identifying additional species that had not been recorded during the quadrat surveys. All previously identified vegetation zones were included in the survey. Any additional wetland plant communities not encompassed in the quadrat surveys were specifically targeted to maximize the detection of new species. The walkabout survey was completed when 15 minutes of active search time had been spent in each vegetation zone. In all cases, the search time was sufficient to reach a considerable point of diminishing return in the identification of new species.

Metrics

Our assessment of the Alberta FQAS focused on a range of traditional and alternative FQA metrics. We first calculated the mean C-value (eq. 1) for each site as:

$$(\mathbf{1})\ \text{mean } C_j = (\sum CC_{ij}) / N_j$$

where CC_{ij} is the C-value for species i at site j and N is the number of native species at site j. We then calculated the floristic quality index (FQI; eq. 2) in the conventional manner (Wilhelm and Masters 1995, Andreas and Lichvar 1995):

$$(\mathbf{2})\ FQI_j = \text{mean } C_j \times \sqrt{N_j}$$

where mean C_j is calculated as in eq. 1 and N is as described above . Both these traditional approaches exclude non-native plant species.

We also calculated variations of the mean C-value (eq. 3) and FQI (eq. 4) that included non-native species. These were:

$$(\mathbf{3})\ \text{mean } C_{Tj} = (\sum CC_{ij}) / T_j$$

$$(\mathbf{4})\ FQI_{Tj} = \text{mean } C_{Tj} \times \sqrt{T_j}$$

where T is the total species richness of site j including non-native species.

Among the most common critiques of the traditional FQI approach is the sensitivity of the index to species richness (Matthews 2003, Miller and Wardrop 2006). To develop a metric that eliminated this bias we calculated an adjusted FQI following Miller and Wardrop (2006). They devised an approach where an adjusted FQI value was determined as a percentage of the maximum attainable index score by assuming that the mean C-value of a site is 10 (i.e., the highest possible mean C-value) and that all species are native. The equation for this adjusted FQI (eq. 5) is calculated as:

$$(\mathbf{5})\ \mathrm{FQI}_{ADJ} = [\ (\text{ mean } \mathrm{C}_j / 10\ )(\ \sqrt{N_j} / \sqrt{T_j}\ )\ ] \times 100$$

where the mean C_j is as calculated in eq. 1 and excludes non-native species. *N* is the number of native species at site *j* and *T* is the total species richness of site *j* including non-native species.

Following the logic that species with higher relative frequency at a site should have a comparatively greater influence on a quantitative floristic quality score, various authors have proposed abundance or frequency weighted indices (Cohen et al. 2004, DeBerry 2006). We calculated a relative abundance weighted FQI (eq. 6) that closely followed the approach of DeBerry (2006):

$$(\mathbf{6})\ \mathrm{FQI}_{ABUND} = (\ \sum \mathrm{CC}_{ij}\, RA_{ij} / N_j\ )\ (\ \sqrt{N_j}\ )$$

where RA_{ij} is the relative abundance of species *i* at site *j* as computed by species *i* frequency divided by the sum of all native species frequencies at site *j*. Using this equation, the individual C-value of each native species is weighted by that species' relative abundance. The weighted C-values are then summed across a site and multiplied by the square root of the native plant species richness as with the traditional FQI approach. Because this equation required the calculation of relative abundance, only species recorded in quadrat surveys were included (i.e., those with percent cover values). All other equations use species lists compiled through quadrat surveys and supplemental walkabout data.

To contrast with the above FQA based metrics (i.e., those using C-values), we also calculated the proportion of non-native species for each site. This metric is commonly included among regulatory performance criteria in compliance studies (Spieles 2005) and is also commonly included in vegetation based IBIs (e.g., Mack 2007). This metric was simply calculated as:

$$\textbf{(7)}\ \%\mathrm{NN} = (\ NN\ /\ T\ ) \times 100$$

where *NN* is the number of non-native plant species at a site and *T* is the total plant species richness, calculated by summing the number of native and non-native plant species.

We also calculated two floristic quality metrics that we hypothesized would combine the strengths of incorporating a non-native species component and the extra information included through abundance weighting. Those two metrics were the summed percent cover of all non-native species (%cover NN) and the ratio of the summed percent of non-native species to the summed percent cover of native species (%cover NN : %cover N). Again, because these calculations incorporated measures of relative abundance, only species recorded in quadrat surveys were included.

$$(\mathbf{8})\ \%\text{cover NN} = \sum \%\text{coverNN}_{ij}$$

$$(\mathbf{9})\ \%\text{cover NN} : \%\text{cover N} = \left(\sum \%\text{coverNN}_{ij} \right) / \left(\sum \%\text{coverN}_{ij} \right)$$

where $\%\text{coverNN}_{ij}$ is the summed percent cover of all non-native species i at site j averaged across all quadrats. The component $\%\text{cover N}_{ij}$ is measured the same way for native species only.

Results

Coefficients of Conservatism

Of the 407 species included in the list, participants assigned C-values to 406 species in at least one of the parkland or boreal natural regions; only one species was unrated in both regions. For six species, C-values were assigned for only one of the two regions.

The C-values for all 406 species occurring in the Alberta FQAS are included in Appendix A. The mean C-value across all species included in the list is 4.3 in the parkland region and the mean in the boreal is 4.6. Of the species on the list, 43 species (11%) are designated as exotic (non-native) by the ANHIC and were assigned a C-value of zero. Within the parkland region, 5 native species were assigned a C-value of zero; 4 native species in the boreal were assigned a zero. Ignoring all non-native species, the mean C-value for species in the parkland region is 4.9 and the average in the boreal is 5.1 (n=364). The majority of plant species were assigned C-values of 5 or less, 70.6% (283 species) and 64.4% (261 species) for the parkland and boreal regions, respectively. Fifteen species were assigned a C-value of 9 in the parkland, while twelve species received the same value in the boreal. Only 1 species (*Isoetes echinospora*) received a C-value of 10 and that was only in the parkland region (it received a C-value of 9 in the boreal). For both regions, the frequency distribution of C-values is centered around a C-value of 5 (Figure 3-2).

The mean pairwise correlation between participants was 0.68 (range 0.55-0.85) for the parkland region and 0.66 (range 0.50-0.79) for the boreal. Mean C-value standard errors among participants were 0.66 and 0.61 for the parkland and boreal regions respectively. These values are very similar to those reported by Cohen et al. (2004) for a FQAS developed in Florida; one of the few published studies to report on the variability of assigned C-values. Maximum disagreement between participant assigned C-values averaged 3.6 for the parkland region and 4.2 for the boreal. The number of participants that assigned a C-value to a species averaged 5.5 and 7.0 out of a possible 8 participants for the parkland and boreal regions respectively. This suggests that our participants were more familiar with species in the boreal region compared to the parkland region.

Following the application of Peirce's criterion, mean C-value standard errors were reduced to 0.56 for the parkland region and 0.54 for the boreal region. The mean maximum disagreement among raw C-values was reduced to 2.9 and 3.6 for the parkland and boreal regions, respectively.

Assessment of FQAS

Regression Analysis

All tested metrics had significant regression relationships with the disturbance gradient (Table 3-1; P<0.030). Among the metrics tested, the strongest correlation with the disturbance gradient came from the proportion of non-native species (eq. 7) in the plant community (r^2=0.658, P<0.001; Figure 3-3). Among the C-value based metrics, the standard calculation of the mean C-value (eq. 1) and the FQI (eq. 2) had the two weakest relationships to the disturbance gradient (r^2=0.457 and 0.453, P<0.001). Incorporation of non-native species into the calculations of both mean C-value (eq. 3) and the FQI (eq. 4) strengthened the correlations with the disturbance gradient (r^2=0.655 and 0.534, P<0.001). The FQI_{ADJ} (eq. 5), which also considers the contribution of non-native species to total species richness, resulted in a comparably strong relationship (r^2=0.620, P<0.001). The FQI_{ABUND} (eq. 6), which weighted individual species C-values by their relative abundance, was correlated to the disturbance gradient to a similar degree as the traditional C-value and FQI approach (r^2=0.474, P<0.001). These frequency weighted results mimic those obtained by Cohen et al. (2004), with only marginal improvement in metric performance compared to non-weighted FQI metrics. Two metrics were tested that included a non-native component and abundance weighting, but no inclusion of C-values. Those two metrics (%cover NN and %NN : %N) yielded the two poorest significance values (although still all significant at α=0.05) and lowest correlation values ($r^2 \leq 0.169$).

Based on the strength of the regression analysis, the four most robust metrics in decreasing order were: %NN (proportion of non-native spp. in plant community), mean C_T (mean C-value using total species richness), FQI_{ADJ} (FQI score as a percentage of the maximum attainable index score) and FQI_T (FQI score using using total species richness).

Analysis of Variance (ANOVA)

Results of the ANOVA revealed significant differences ($P<0.001$) among wetland types for seven of the nine floristic metrics tested (Table 3-2). The metrics %cover NN ($p=0.196$) and %NN : %N ($p=0.053$) were the two non-significant metrics.

Among the seven metrics with significant results, Tukey pairwise comparisons revealed that the various metrics varied at differentiating among wetland types. Two of the seven metrics yielded significant differences ($p\leq0.013$) across all 6 of the possible pairwise comparisons among wetland types: %NN and mean C_T (Table 3-2). Between these two, the significance values for %NN were more strongly significant ($p\leq0.004$) compared to mean C_T ($p\leq0.013$), suggesting that %NN is a marginally better metric for differentiating among wetland types (Figure 3-4). Three metrics (FQI, FQI_T, FQI_{ADJ}) had significant comparisons ($p\leq0.026$) for 5 of the 6 possible comparisons. In each case, the only pairwise comparison that the metric could not effectively differentiate between was that of agricultural wetlands and naturalized stormwater wetlands, although in 2 of the 3 cases the results were only marginally non-significant ($p=0.052$ and 0.074). The abundance weighted calculation of the FQI was successful in differentiating between wetland types for 4 of the 6 pairwise comparisons. It performed poorly at differentiating between agricultural wetlands and reference wetlands ($p=0.283$) and naturalized stormwater wetlands ($p=0.141$). The traditional calculation of mean C-value was the weakest metric, with significant differences occurring between only 3 of the possible 6 pairwise comparisons. Not surprisingly, the only significant differences occurred between wetland types hypothesized as being the furthest apart along the disturbance gradient, specifically between natural reference wetlands and both naturalized stormwater wetlands ($p=0.008$) and wet ponds ($p<0.001$), and agricultural wetlands and wet ponds ($p=0.014$).

Based on the strength of the Tukey pairwise comparisons, the four most robust metrics in decreasing order were: %NN (proportion of non-native spp. in plant community), mean C_T (mean C-value using total spp. richness), FQI_T (FQI score using using total spp. richness) and FQI_{ADJ} (FQI score as a percentage of the maximum attainable index score). These four metrics also performed the strongest in the regression analysis, although in the regression analysis FQI_{ADJ} performed slightly better than FQI_T.

Discussion

There is still much debate over how to best assess wetland condition and new approaches to wetland assessment continue to emerge. Among the many approaches, FQA continues to receive much attention and represents perhaps the most standardized and commonly used approach. Despite having its criticisms, the underlying concept of species conservatism and reliance on expert opinion has resulted in the creation of an effective assessment tool in many different regions.

Alberta's Floristic Quality Assessment System

The development of the Alberta FQAS used an approach comparable to many other jurisdictions. Using an approach that relied on the independent assignment of C-values from a group of respected botanists, we were provided with expert opinion on a list of 406 wetland plant species in a timely, un-biased manner. Our treatment of the data using Peirce's criterion and use of the median value instead of the mean dealt effectively with the variability of the assigned C-values. Analysis of the C-values assigned during the Alberta FQAS development process suggested that the disagreement among participants was comparable to that obtained during other studies (Cohen et al. 2004). The observed distribution of Alberta's of C-values peaks around a value of 5 for both the parkland and boreal regions (Figure 3-2). Although this distribution is not left-skewed [e.g., North Dakota (Northern Great Plains FQA Panel 2001) and southern Ontario (Oldham et al. 1995)] or right-skewed [e.g., Mississippi (Herman et al. 2006)] as in most other jurisdictions, it is comparable to the distribution of C-values obtained in a study of streambank habitats in an agricultural landscape in southeastern Ontario, Canada (Bowers and Boutin 2008). The observed C-value distribution is also a reasonable result considering that marsh wetlands, regardless of their landscape position (pristine vs. highly disturbed), are inherently variable systems that have to handle extremes of natural variability including drought, flooding, fire, naturally elevated salinity and beaver influence. Accordingly, many native species characteristic of marsh wetlands are adapted to wide ranges of natural variability, providing them with some level of resilience to disturbance and ability to grow in a range of wetland habitat conditions. In Minnesota, shallow and deep marshes scored the lowest average FQI scores among twelve wetland plant communities (Milburn et al. 2007). At the other end of the spectrum were poor fen and bog communities which are characterized by a very low level of natural variability thanks to very stable hydrology extending over many, many years (Milburn et al. 2007).

The results of the validation process provide evidence that the Alberta FQAS can function as an effective wetland bioassessment tool. The significant relationships between the a priori determined disturbance gradient and the FQA metrics suggest that the FQA approach can yield several useful quantitative ecological metrics for assessing wetland ecological condition. The efficacy of the same FQA metrics at differentiating among wetland types with different disturbance levels further supports the Alberta FQAS as a valid and functional bioassessment tool.

Floristic Quality Assessment Metrics

Not all measured FQA metrics performed equally. The traditional approach to the calculation of mean C-value and the FQI (i.e., excluding non-native species) were the two weakest performing metrics of the six FQAS based metrics tested. The original FQA approach and the calculation of its related metrics have received some criticism for their bias towards increasing species richness (in calculating the FQI) and, by relation, site size/area (Matthews 2003). Another common critique has been the exclusion of non-native species (Cohen et al. 2004, Ervin et al. 2006). The bias towards species richness, although to some extent true, has been justified by several authors. Fennessy et al. (1998b) justify the inclusion of a species richness factor in the FQI by suggesting that a high level of species richness can indicate a more resilient and thus more valuable site. Some authors do not acknowledge this as a valid argument and, instead, prefer to focus on the calculation of mean C-value, which excludes any species richness bias (Miller and Wardrop 2006). Cohen et al. (2004) argued that the exclusion of non-native taxa from FQA metrics, as done in the original approach (Swink and Wilhelm 1979), requires empirical validation. To date, no studies have provided such validation. Ervin et al. (2006) state that until such validation exists, non-native species should be included as a component of any proposed method of floristically quantifying wetland condition.

The arguments supporting the inclusion of non-native species in floristic assessment are also numerous. In some cases, a single non-native species can have disastrous consequence for wetland health, regardless of the diversity and richness of the remaining native taxa that are present (Ervin et al. 2006). Alberta's depressional marsh wetlands are not threatened by any such singularly disastrous invasive species, although the potential impacts of non-native species is always possible. Once established, non-native species can inhibit the establishment or persistence of native species (Walker and Vitousek 1991) and increase the likelihood of invasion by additional non-native species (Parker et al. 2006).

Despite their argument against the inclusion of non-native taxa, Swink and Wilhelm (1979, 1994) acknowledged that the simple presence of non-native species can represent a disturbance. Expanding on this statement and the fact that the plant community integrates disturbances at numerous biological scales (Lopez and Fennessy 2000, Ervin et al. 2006), the level of disturbance at a site influences the suite of native taxa present. Through similar processes, disturbance can thus affect the number and extent of invasion by non-native species. The three fundamental factors said to be responsible for overall wetland disturbance are land cover change in the local landscape, the presence and condition of a vegetated buffer separating the wetland from adjacent land use, and the hydrologic conditions of the wetland itself (Lopez and Fennessy 2002). Each of these disturbances can be linked to one of four abiotic factors which in turn are linked to increased invasibility by non-native species (Matthews et al. 2009). Invasion by non-native species has been linked to increased resource availability, particularly nitrogen and phosphorous enrichment (Suding et al. 2005). In many landscapes, and especially human impacted regions like Alberta, nutrient enrichment is related to the proximity and proportion of agricultural land in the local landscape (Matthews et al. 2009, Campbell et al. 2009). The presence and quality of a vegetated buffer surrounding a wetland can influence the amount of nutrients reaching the wetland and thus influencing the invasibility of a site to non-native species. The level of invasion of non-natives should also increase as the extent of roads and urban areas in the vicinity of wetlands increases (Houlahan et al. 2006). Altered hydrology also has the potential to influence invasibility through mechanisms that include increased propagule delivery along drainage channels and increased availability of suitable establishment sites as a result of increased sedimentation or altered hydroperiod (e.g., unnatural flooding/drawdown regime). Considering these many direct and indirect relationships between disturbance and the presence or abundance of non-native species, non-native species composition constitutes a valid and valuable component of floristic quality assessment.

Results of this study provide further rationale for the inclusion of non-native species in FQA. In all cases, metrics that included some consideration of non-natives species performed better than those that did not. The improved ability of metrics to differentiate among wetland types and the strengthened correlation with an independent disturbance gradient suggest that the incorporation of information on non-native species results in a more refined assessment measure. In this regard, this study joins a growing collection of studies that reinforce the notion that non-native species must be included in floristic-based evaluations of wetland condition (Cohen et al. 2004, Ervin et al. 2006).

Traditional vs. FQA Metrics

Interestingly, our results show that despite the additional information contained within FQA metrics regarding species conservatism of native taxa, the simple measure of proportion of non-native species was the strongest performing metric when compared to the disturbance gradient. This metric was also best at differentiating among four wetland types selected to span a range of anthropogenic disturbance levels. These results agree with those of Bowers and Boutin (2008) where the proportion of non-native species was the most effective metric at expressing a disturbance gradient and those of Ervin et al. (2006) who stated that non-native species in and of themselves are an indicator worthy of further investigation. Others have warned that the simple measure of non-native species richness may not provide enough information on the plant community and that, instead, the dominance of non-native species relative to native species may have more relevance in terms of conservation and restoration goals (Matthews et al. 2009). Although we agree with the logic behind this principle, our results do not support this assertion. The two metrics we calculated that included both abundance weighting and non-native species (%cover NN and %NN : %N) performed much more poorly than the other metrics included in this study. Both abundance-weighted metrics were restricted to data collected at the quadrat level, while the other broader metrics used species data recorded at the site level. We hypothesize that the relative abundance of non-native species calculated using pooled quadrat data simply are not sufficient to represent the overall condition of the site. It is possible that relative abundance data, if collected in a more comprehensive manner such that it better represents the entire site (e.g., more quadrats), could result in more useful measurements than metrics based simply on elements of species richness.

In any discussion of floristic quality assessment and the process of determining which metrics or approach is best for the assessment of wetland ecological condition, there must be some consideration of the feasibility, ease and associated cost of implementing those measures being considered. Wetland managers pressured by time and fiscal constraints often desire assessment approaches that are quick and easy to implement (Johnston et al. 2009). Any inclusion of abundance weighting is inherently going to involve considerably more effort than the compilation of simple site species lists. Anecdotally, during our fieldwork the collection of all quadrat-based vegetation data at a site typically required 3 or 4 hours. In contrast, a walkabout survey lasting an hour or less could yield a comprehensive site species list. Increasing survey effort to overcome the issues of using quadrat or plot-based data only and their associated relative abundance measures, as discussed above, would only exacerbate the considerable effort required to compile such metrics. Our results, and those of other studies (Cohen et al. 2006), have shown that there is little improvement in metric performance when abundance weighting is included. Resources would, thus, be best used to sample additional sites instead of spending the time collecting detailed abundance data.

Despite the strength of a single metric such as proportion of non-native species, or the strength of an integrative floristic quality metric such as the mean C-value or FQI, caution should be taken in the interpretation of assessment results when they use only plant based metrics (Bernthals 2003, Spieles 2005). Focusing on plants alone ignores other wetland functions that may be of great value in determining a sites overall condition or ecological value. Although a

FQA based metric may integrate disturbance at the landscape level, from altered hydrology and other sources, it may not capture ecological functioning related to the provision of wildlife habitat or information on threatened and endangered species. Initiatives to develop indices of biotic integrity (IBIs) that integrate metrics from multiple components of the biotic wetland ecosystem are likely to be better at describing a site's overall condition. The biotic integrity concept is based on the premise that "healthy ecosystems support and maintain a balanced, adaptive community of organisms with species diversity, composition, and functional organization comparable to that of natural habitats within a given region" (Karr and Dudley 1981). The use of a FQAS, through the integration of a single or multiple FQA based metrics into an IBI as representation of the floristic component of wetlands, could result in a more comprehensive assessment approach than use of FQA metrics alone (DeBerry 2006).

Conclusions

The creation of an assessment tool that follows a standardized approach and that has precedence for use by multiple regulatory agencies in the United States provides the opportunity for many applications of the Alberta FQA. The FQA approach has potential uses in wetland condition assessment, determining wetland mitigation success and tracking wetland plant community establishment over time. The FQA approach provides standardized and quantitative metrics useful as quick and effective stand-alone measures of wetland condition, the first of their kind in western Canada. As the practice of wetland assessment continues to develop, particularly in Alberta and other regions of Canada, FQA will also be useful as a component of more comprehensive and all-inclusive ecological function assessment tools.

Literature Cited

Alberta Natural Heritage Information Centre. 2006. List of All Vascular Plant Elements. Parks and Protected Areas Division. Alberta Community Development. Edmonton, AB.

Andreas, B. K. and R. W. Lichvar. 1995. Floristic index for establishing assessment standards: a case study for northern Ohio. Wetlands Research Program Technical Report WRP-DE-8. US Army Corp of Engineers, Washington, DC.

Andreas, B. K. and J. J. Mack, J. S. McCormac. 2004. Floristic quality assessment index for vascular plants and mosses for the state of Ohio. Wetland Ecology Group, Division of Surface Water, Ohio Environmental Protection Agency, Columbus, Ohio. 217 p.

Angermeier, P.L., R.A. Smogor, and J.R. Stauffer. 2000. Regional frameworks and candidate metrics for assessing biotic integrity in mid-Atlantic highland streams. Transactions of the American Fisheries Society 129: 962-981.

Beatty, G. H., H. Henderson, C. Keener, and R. Gruver. 2002. Vascular plants of Centre County, Pennsylvania and their coefficients of conservatism: Plants of Upper Penns Creek Watershed, documented and probable. Pennsylvania

Native Plant Society and Pennsylvania Department of Environmental Protection. State College, PA, USA.

Bernthal, T. W. 2003. Development of a floristic quality assessment methodology for Wisconsin. Final Report to USEPA. Prepared by Wisconsin Department of Natural Resources. Madison, WI.

Bourdaghs, M., C. A. Johnston, and R. R. Regal. 2006. Properties and performance of the floristic quality index in Great Lakes coastal wetlands. Wetlands 26:718-735.

Bowers, K. and C. Boutin. 2008. Evaluating the relationship between floristic quality and measures of plant biodiversity along stream bank habitats. Ecological Indicators 8:466-475.

Campbell, B.D., R.J. Haro and W.B. Richardson. 2009. Effects of agricultural land use on chironomid communities: Comparisons among natural wetlands and farm ponds. Wetlands 29:1070-1080.

Cohen, M.J., S. Carstenn and C.R. Lane. 2004. Floristic quality indices for biotic assessment of depressional marsh condition in Florida. Ecological Applications 14: 784-794

Commission for Environmental Cooperation. 1997. Ecological regions of North America: toward a common perspective. Commission for Environmental Cooperation, Montreal, Quebec, Canada. 71pp.

DeBerry, D.A.. 2006. Floristic quality index: Ecological and management implications in created and natural wetlands. Ph.D. Dissertation. The College of William and Mary, Virginia.

Environment Canada. 2008. National Climate Data and Information Archive. Canadian Climate Normals or Averages 1971-2000. www.climate.weatheroffice.gc.ca. Accessed March 11, 2010.

Ervin, G.N., B.D. Herman, J.T. Bried and D.C. Holly. 2006. Evaluating non-native species and wetland indicator status as components of wetlands floristic assessment. Wetlands 26: 114-1129.

Fennessy, M. S., R. Geho, B. Elifritz, and R. Lopez. 1998a. Testing the floristic quality assessment index as an indicator of riparian wetland quality. Final report to U.S. EPA. Ohio Environmental Protection Agency, Division of Surface Water, Columbus, Ohio, USA.

Fennessy, M. S., M. Gray, R. D. Lopez, and J. Mack. 1998b. An ecological assessment of wetlands using reference sites. Final report to U.S. EPA. Ohio Environmental Protection Agency, Division of Surface Water, Columbus, Ohio, USA.

Herman, B.D., J.D. Madsen and G.N. Ervin. 2006. Development of coefficients of conservatism for wetland vascular flora of north and central Mississippi. GeoResources Institute Report 4001. Mississippi State, Mississippi.

Herman, K. D., L. A. Masters, M. R. Penskar, A. A. Reznicek, G. S. Wilhelm, and W. W. Brodowicz. 1997. Floristic quality assessment: Development and application in the state of Michigan (USA). Natural Areas Journal 17: 265-279.

Houlahan, J.E., P.A. Keddy, K. Makkay and C.S. Findlay. 2006. The effects of adjacent land use on wetland species richness and community composition. Wetlands 25: 79-96.

Johnston, C.A., J.B. Zedler, M.G. Tulbure, C.B. Frieswyk, B.L. Bedford and L.Vaccaro. 2009. A unifying approach for evaluating the condition of wetland plant communities and identifying related stressors. Ecological Applications 19: 1739-1757.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspectives on water quality goals. Environmental Management 5:55-68.

Ladd, D. 1993. The Missouri floristic quality assessment system. The Nature Conservancy. St. Louis, MO, USA.

Lahring, Heinjo. 2003. Water and Wetland Plants of the Prairie Provinces. Canadian Plains Research Center. Regina, SK, Canada.

Lopez, R. D. and M. S. Fennessy. 2002. Testing the floristic quality assessment index as an indicator of wetland condition. Ecological Applications 12:487–497.

Mack, J. J. 2007. Developing a wetland IBI with statewide application after multiple testing iterations. Ecological Indicators 7:864–81.

Matthews, J. W. 2003. Assessment of the floristic quality index for use in Illinois, USA, wetlands. Natural Areas Journal 23:53–60.

Matthews, J.W., A.L. Peralta, A. Soni, P. Baldwin, A.D. Kent and A.G. Endress. 2009. Local and landscape correlates of non-native species invasion in restored wetlands. Ecography 32: 1031-1039.

Milburn, S. A., M. Bourdaghs and J.J. Husveth. 2007. Floristic quality assessment for Minnesota wetlands. Minnesota Pollution Control Agency. St. Paul, Minnesota.

Miller, S.J. and D.H. Wardrop. 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecological Indicators 6: 313-326.

Miller, S.J., D.H. Wardrop, W.M. Mahaney and R.P. Brooks. 2006. A plant-based index of biotic integrity (IBI) fro headwater wetlands in central Pennsylvania. Ecological Indicators 6: 290-312.

Moss, E.H. 1983. Flora of Alberta. University of Toronto Press. Toronto. 687pp.

Mushet, D.M., N.E. Euliss Jr. and T.L. Shaffer. 2002. Floristic quality assessment of one natural and three restored wetland complexes in North Dakota, USA. Wetlands 22: 126-138.

Natural Regions Committee 2006. Natural Regions and Subregions of Alberta. Compiled by D.J. Downing and W.W. Pettapiece. Government of Alberta. Pub. No. T/852.

Northern Great Plains Floristic Quality Assessment Panel. 2001. Floristic quality assessment for plant communities of North Dakota, South Dakota (excluding the Black Hills), and adjacent grasslands. Jamestown, ND: Northern Prairie Wildlife Research Center Online. http://www.npwrc.usgs.gov/resource/plants/fqa/index.htm (Version 26JAN2001).

Oldham, M.J., W.D. Bakowsky and D.A. Sutherland. 1995. Floristic quality assessment system for southern Ontario. Natural Heritage Information Centre. Peterborough, Ontario.

Parker, J.D., D.E. Burkepile and M.E. Hay. 2006. Opposing effects of native and exotic herbivores on plant invasions. Science 311 (5766): 1459-1461.

Peirce, B. 1852. Criterion for the rejection of doubtful observations. Astronomical Journal II 45: 161-163.

Rooney, R.C. and S.E. Bayley. 2010. Quantifying a stress gradient: An objective approach to variable selection, standardization and weighting in ecosystem assessment. Ecological Indicators 10: 1174-1183.

Ross, S.M.. 2003. Peirce's criterion for the elimination of suspect experimental data. Journal of Engineering Technology 20: 38-41.

Rothrock, P. E. 2004. Floristic quality assessment in Indiana: the concept, use, and development of coefficients of conservatism. Indiana Department of Environmental Management, Office of Water Quality. Indianapolis, IN, USA. Final report for ARN A305-4-53, EPA Wetland Program Development Grant CD975586-01.

Spieles, D.J. 2005. Vegetation development in created, restored and enhanced mitigation wetland banks of the United States. Wetlands 25: 51-63.

Stewart, R.E. and H.A. Kantrud. 1971. Classification of natural ponds and lakes in the glaciated prairie region. Resource Publication 92, Bureau of Sport Fisheries and Wildlife, U.S. Fish and Wildlife Service, Washington, D.C. 57pp

Suding, K.N., S.L Collins, L. Gough, C. Clark, E.E. Cleland, K.L. Gross, D.G. Milchunas and S. Pennings. 2005. Functional- and abundance-based mechanisms explain diversity loss due to N fertilization. Proceedings of the National Academy of Sciences of the United States of America 102: 4387-4392.

Swink, F. A. and G. S. Wilhelm. 1994. Plants of the Chicago Region, fourth edition. Morton Arboretum. Lisle, IL, USA.

Taft, J.B., G.S. Wilhelm, D.M. Ladd and L.A. Masters. 1997. Floristic quality assessment for vegetation in Illinois: A method for assessing vegetation integrity. Erigenia 15: 3–95.

Walker, L.R. and P.M. Vitousek. 1991. An invader alters germination and growth of a native dominant tree in Hawaii. Ecology 72: 1449-1455.

Wilhelm, G.S. and L.A. Masters. 1995. Floristic quality assessment in the Chicago region and application computer programs. The Morton Arboretum. Lisle, Illinois.

United States Environmental Protection Agency. 2002. Methods for evaluating wetland condition: Using vegetation to assess environment condition in wetlands. Office of Water, U.S. Environmental Protection Agency, Washington, DC. EPA-822-R-02-02-020.

Table 3-1. Comparison of regression models between different floristic quality metrics and wetland disturbance scores for 32 wetland in central Alberta, Canada.

Metric	Description	*P*	r^2
	Floristic Quality Assessment Metrics (C-value based)		
mean C	mean C-value of native spp. only	<0.001	0.457
FQI	FQI score using native spp. only	<0.001	0.453
mean C_T	mean C-value using total spp. richness	<0.001	0.655
FQI_T	FQI score using using total spp. richness	<0.001	0.534
FQI_{ADJ}	FQI score as a percentage of the maximum attainable index score	<0.001	0.620
FQI_{ABUND}	relative abundance weighted FQI score; native spp. only	<0.001	0.474
	'Traditional' Metrics (non C-value based)		
%NN	proportion of non-native spp. in plant community	<0.001	0.658
%cover NN	total percent cover of non-native spp.	0.030	0.148
%cover NN : %cover N	ratio of total percent cover of non-native spp. to native spp.	0.020	0.169

Table 3-2. Comparison of Tukey HSD pairwise comparison significance values among four wetland types in central Alberta, Canada for the seven floristic quality metrics with significant ANOVA results when comparing mean metric values among the wetland types.

	Metric						
Pairwise Comparison	mean C	FQI	mean C_T	FQI_T	FQI_{ADJ}	FQI_{ABUND}	%NN
Agricultural vs. NSW [1]	0.448	0.279	0.013*	0.074	0.052	0.141	0.003*
Agricultural vs. Reference	0.219	0.016*	0.004*	0.002*	0.019*	0.283	0.004*
Agricultural vs. Wet pond	0.014*	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*
NSW vs. Reference	0.008*	<0.001*	<0.001*	<0.001*	<0.001*	0.002*	<0.001*
NSW vs. Wet pond	0.304	0.026*	0.006*	0.007*	0.021*	0.037*	0.001*
Reference vs. Wet pond	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*
# of significant comparisons/metric	3	5	6	5	5	4	6

[1] NSW = naturalized stormwater wetland

* significant at α=0.05

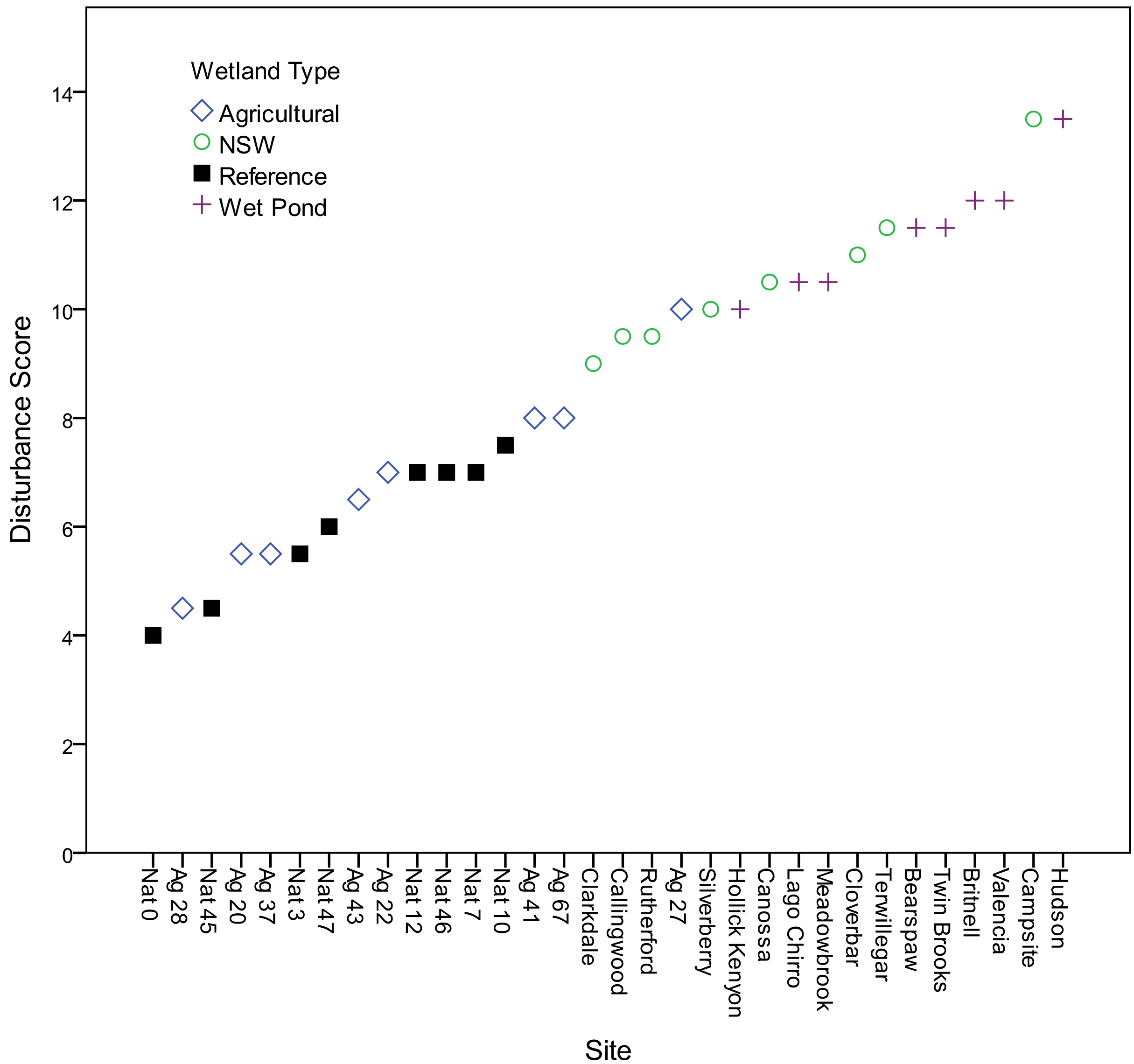


Figure 3-1. Plot of disturbance scores calculated using an objective approach following Rooney and Bayley (2010) for 32 wetland sites in central Alberta, Canada demonstrating a consistent gradient from low disturbance to high disturbance across four different wetland types. NSW = naturalized stormwater wetland.

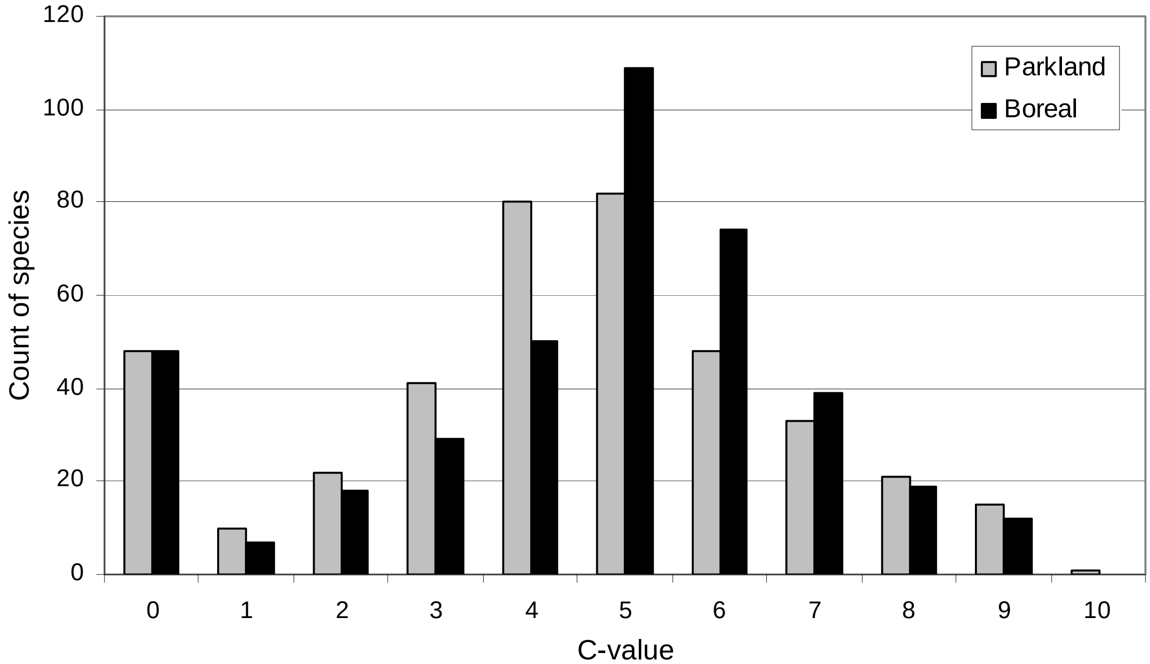


Figure 3-2. Frequency distribution of 406 coefficients of conservatism (i.e., C-values) for marsh wetland species in the parkland and boreal natural regions of Alberta, Canada

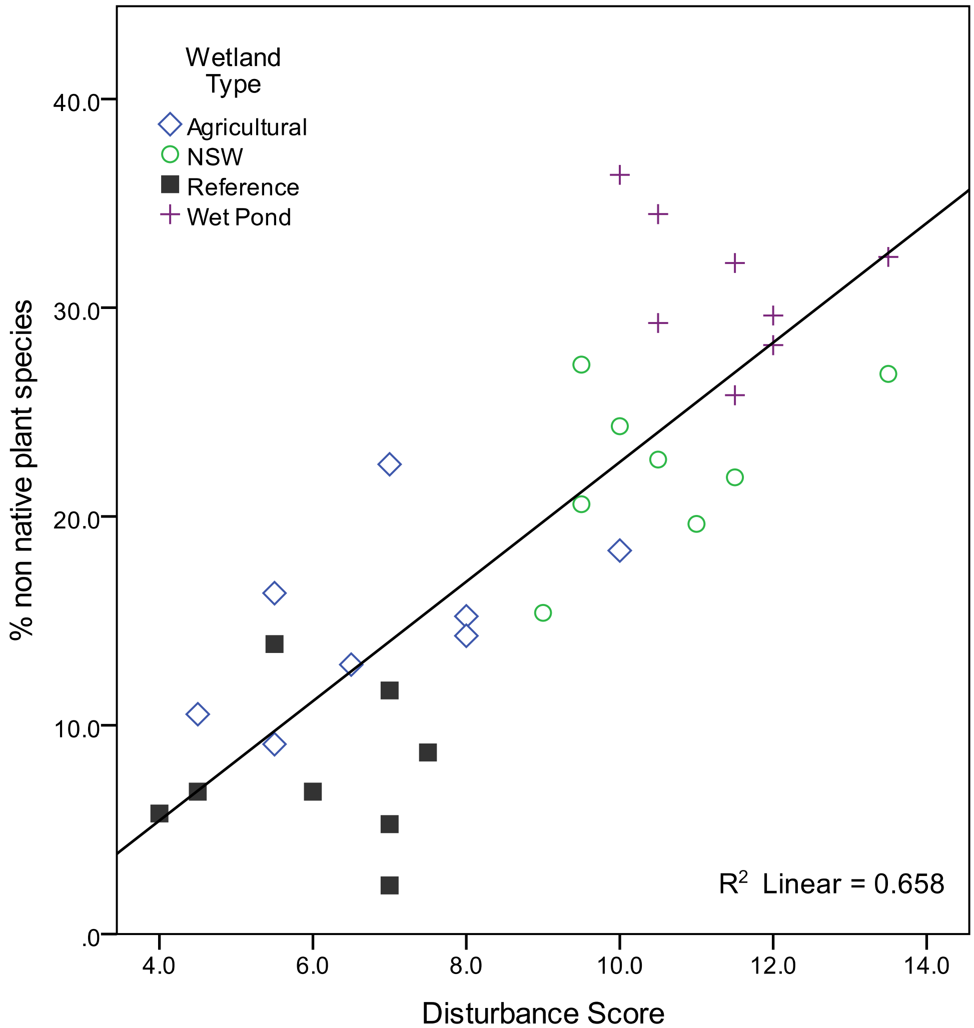


Figure 3-3. Correlation between the % non-native plant species and an objectively calculated wetland disturbance score (r^2=0.658, P<0.001) for 32 wetlands of 4 different types in central Alberta, Canada. NSW = naturalized stormwater wetland.

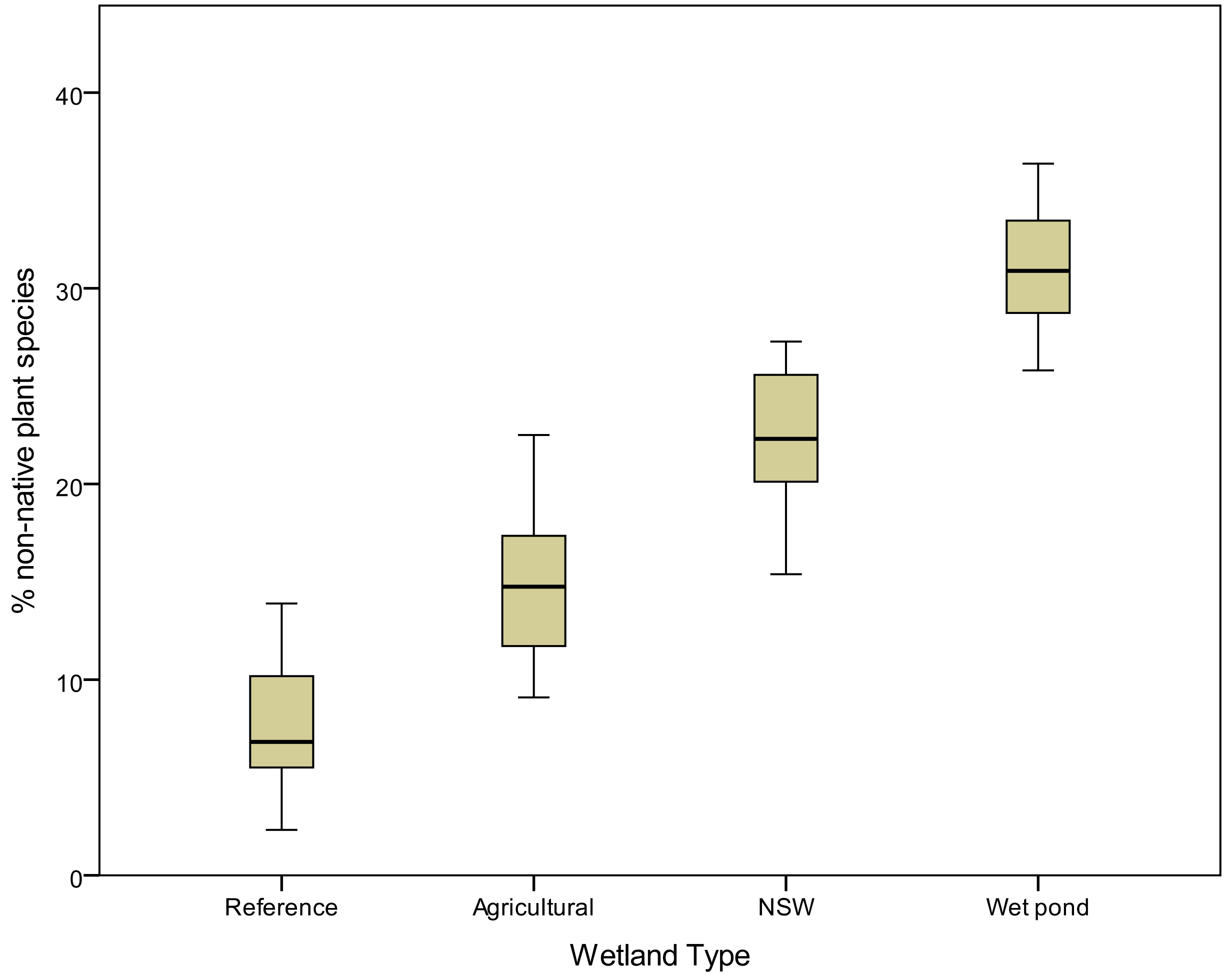


Figure 3-4. Box plot diagram comparing "% non-native plant species" among 4 wetland types. The bottom of the box and lower whisker represent the first quartile and lower data limit, respectively, while the top of the box and upper whisker are the third quartile and upper data limit. The median is represented by the solid bar bisecting the box. NSW = naturalized stormwater wetland

CHAPTER 4

General Discussion and Conclusions

In Alberta, most created wetlands accepted as compensation have been naturalized stormwater management facilities (NSWMFs). These facilities involve an effort to create wetland habitat (e.g., creation of shallow water areas for emergent vegetation, riparian shrub plantings) and, because of this, these facilities are often accepted by the Province as a form of wetland compensation. Despite this, the primary function of these facilities remains the provision of stormwater storage. A facility's capacity for storage exists in the area above normal water level. Because of this, and because there are financial motivations to minimize the area needed for the development of stormwater facilities, thus maximizing the area available for more profitable land uses (e.g., residential development), stormwater facilities are designed as steep-sided basins. Current government construction standards (Alberta Environment Protection 1999, City of Edmonton 2008) require side slopes of created stormwater wetlands to measure between 5:1 (horizontal run [H]:vertical rise [V]) to 7:1. Although the shoreline slopes we calculated for created wetlands were slightly shallower than this (i.e., 7:1 to 10:1), they were significantly steeper than the very shallow 20:1 to 50:1 shoreline slopes that we found characterized natural marsh wetlands.

This fundamental difference in basin morphology between created and natural wetlands resulted in distinctly different wetland zonation, with the steeper slopes of created wetlands resulting in fewer, narrower wetland vegetation zones. As is typical of natural open marsh wetlands, natural wetlands in our study area included multiple vegetation zones which typically comprised, from wettest to driest, a wide emergent common cattail (*Typha latifolia*) zone, a heavily dominated sedge (*Carex* spp.) zone and a mixed sedge and wetland grass (*Calamagrostis* spp.) zone. In contrast, created wetlands averaged less than two vegetation zones and were most often represented by a narrow emergent zone and an even narrower, weedy wet-meadow zone. Zampella and Laidig (2003) and Jenkins and Greenway (2007) also found restricted wetland zonation at created wetlands. Our study, however, extends the relationship between shoreline slope and impacts to wetland zonation to include impacts to wetland dependent songbirds. The reduction in habitat diversity caused by fewer wetland zones and the reduction in available habitat caused by the narrower wetland zones combined to result in reduced species richness and abundance of wet-meadow dependent songbirds among our created wetlands. Despite differences in emergent zone width between natural and created wetlands, there was surprisingly no difference in terms of species richness of waterfowl and other waterbirds between these wetland types.

Our research also yielded some interesting differences between created and natural wetlands in terms of the proportion of non-native species present in the plant community. Created wetlands had more non-native species than did natural wetlands. Although there are likely a number of factors that combine to result in the high non-native species presence in created wetlands, we argue that the steeper shoreline slope of created wetlands are likely a strong determining factor on non-native species presence through its influence on such processes as inundation time and soil moisture levels. Indeed, the vast majority of the non-native species were upland species that had established in the wet meadow zone, a pattern also seen by others (e.g., Bowers and Boutin 2008).

These differences in non-native species between created and natural wetlands reported in the second chapter of this thesis provide an example of the usefulness of floristic metrics at quantifying wetland condition. In Chapter 3 we greatly expand on this concept and present the Floristic Quality Assessment (FQA) approach as an effective wetland assessment tool and discuss the development of this approach in Alberta.

Despite having a provincial policy that promotes wetland restoration and creation, there are no standardized approaches in Alberta for the assessment of wetland condition, nor are there any standardized quantitative ecological performance standards that could be used in such an assessment. The FQA approach uses a numerical rating system, called the coefficient of conservatism (hereafter C-value), to quantify a plant species' tolerance to disturbance and affinity to natural habitats. With C-values assigned to a list of plant species in a given area, it is then possible to calculate various community and site level metrics that can quantify the overall floristic quality of a site. When compared to the same measures obtained from a reference site, these metrics can be used to indicate the relative quality of a site. The most commonly used FQA metrics are the mean C-value of all native species at a site and the Floristic Quality Index (FQI) which is the product of the site mean C-value and the square root of the total native species richness. Early work on the FQA stated that non-native species should be excluded from FQA (Swink and Wilhelm 1979, Ervin et al. 2006). Our results, however, support a growing body of scientific literature that shows that the performance of FQA metrics improve when they include consideration of non-native species (Cohen et al. 2004, Miller and Wardrop 2006). When related to a gradient generated using disturbance scores calculated for the suite of 32 wetlands that formed the focus of Chapter 2, FQA metrics that included the contribution of non-native species to the plant community showed improved correlations. Surprisingly, however, despite the additional information contained within FQA metrics regarding species conservatism of native taxa, the simple measure of proportion of non-native species was the strongest performing metric when compared to the disturbance gradient. Although this result agrees with that of others (Ervin et al. 2006, Bowers and Boutin 2008), it should be cautioned that the simple measure of non-native species richness may not provide enough information on the plant community and that, instead, the dominance of non-native species relative to native species may have more relevance in terms of conservation and restoration goals (Matthews et al. 2009).

Regardless of the strength of any single floristic metric, we agree with others that have cautioned against wetland assessment approaches based only on plant metrics (Bernthal 2003, Spieles 2005). Focusing on plants alone ignores other wetland functions that may be of great value in determining a sites overall condition or ecological value. The biotic integrity concept is based on the premise that “healthy ecosystems support and maintain a balanced, adaptive community of organisms with species diversity, composition, and functional organization comparable to that of natural habitats within a given region" (Karr and Dudley 1981). We feel that initiatives to develop indices of biotic integrity (IBIs) that integrate multiple biological metrics are likely to be the best at describing a wetland site's overall condition.

Conclusion

The definitive benchmark for successful wetland compensation is often described as achieving a similarity between compensation (i.e., restored or created) wetlands and natural wetlands in the surrounding area (Galatowitsch and Van der Valk 1994). Ideally, created wetlands should mimic natural wetlands in terms of both structure and function. In this regard, I have shown that created stormwater wetlands are significantly different from their natural surrogates in terms of one key structural parameter: shoreline slope. And, following the notion that improper wetland structure will negatively affect the functional capabilities of the wetland (Hoeltje and Cole 2009), I have shown that the fundamental difference in slope is highly correlated to wetland zonation, wetland songbird richness and abundance, and likely plays a role influencing various aspects of the plant community including the presence of non-native species and the height and density of plants.

It has been argued that it is unreasonable to expect similar levels of ecological functioning to be sustained by created wetlands, particularly in highly urbanized settings, relative to wetlands in a natural landscape position (Brinson and Rheinhardt 1996). Our data show that created stormwater wetlands do indeed fall short of successfully replicating natural wetlands across many of the parameters we measured. Current design and construction standards for created wetlands, which are largely guided by the need to provide stormwater storage, are acting as a major impediment to the creation of natural wetland vegetation zonation at stormwater wetlands. Steep shoreline slopes effectively limit the ability of created wetlands to replace the full suite of habitat functions provided by natural wetlands. However, if naturalized stormwater wetlands were designed and constructed having slopes similar to natural wetlands, we feel that the differences we observed between natural and created wetlands would be reduced.

We've also presented the FQA as an effective, standardized approach to the assessment of wetland condition, particularly when non-native species are included in metric calculation. However, as our approach in Chapter 2 highlights, we feel there is merit in analysis of not only the plant community, but other aspects of the wetland ecosystem. Ultimately, an assessment approach that incorporates measures of multiple taxa and best parallels the concept of biotic integrity will likely be the best at describing a site's overall ecological condition. It is important that academics and wetland managers alike continue to ponder these questions with an ultimate objective of developing widely standardized approaches to the practice of wetland assessment. Effective wetland assessment will remain an integral part of wetland compensation policy, but is also required to inform an adaptive management process through which our abilities to restore and create wetlands must continue to improve.

Literature Cited

Bernthal, T. W. 2003. Development of a floristic quality assessment methodology for Wisconsin. Final Report to USEPA. Prepared by Wisconsin Department of Natural Resources. Madison, WI.

Bowers, K. and C. Boutin. 2008. Evaluating the relationship between floristic quality and measures of plant biodiversity along stream bank habitats. Ecological Indicators 8:466-475.

Brinson, M.M. and R. Rheinhardt. 1996. The role of reference wetlands in functional assessment and mitigation. Ecological Applications 6: 69-76.

Cohen, M.J., S. Carstenn and C.R. Lane. 2004. Floristic quality indices for biotic assessment of depressional marsh condition in Florida. Ecological Applications 14: 784-794.

Ervin, G.N., B.D. Herman, J.T. Bried and D.C. Holly. 2006. Evaluating non-native species and wetland indicator status as components of wetlands floristic assessment. Wetlands 26(4): 114-1129.

Galatowitsch, S.M. and A.G. van der Valk. 1994. Restoring prairie wetlands. Ames, Iowa. Iowa State University Press. 246 pp.

Hoeltje, S.M. and C.A. Cole. 2009. Comparison of function of created wetlands of two age classes in central Pennsylvania. Environmental Management 43: 597-608.

Jenkins, G.A. and M. Greenway. 2007. Restoration of a constructed stormwater wetland to improve its ecological and hydrological performance. Water Science and Technology 56: 109-116.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspectives on water quality goals. Environmental Management 5:55-68.

Matthews, J.W., A.L. Peralta, A. Soni, P. Baldwin, A.D. Kent and A.G. Endress. 2009. Local and landscape correlates of non-native species invasion in restored wetlands. Ecography 32: 1031-1039.

Miller, S.J. and D.H. Wardrop. 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecological Indicators 6: 313-326.

Spieles, D.J. 2005. Vegetation development in created, restored and enhanced mitigation wetland banks of the United States. Wetlands 25: 51-63.

Swink, F. A. and G. S. Wilhelm. 1994. Plants of the Chicago region, fourth edition. Morton Arboretum. Lisle, IL, USA. 900 pp.

Zampella, R. A. and K. J. Laidig. 2003. Functional equivalency of natural and excavated coastal plain ponds. Wetlands 23:860-876.

APPENDICES

Appendix A. List of 406 marsh wetland plant species and coefficients of conservatism (C-values; possible values = 0 to 10) assigned for the Parkland and Boreal natural regions of Alberta, Canada. Blanks indicate that the species was not assigned a C-value and likely does not regularly occur within that region. See pages 57 through 59 for a detailed explanation of the meaning and origin of C-values.

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
1	*Achillea sibirica*	many-flowered yarrow		3	5
2	*Acorus americanus*	sweet flag	Acorus calamus	8	8
3	*Actaea rubra*	red and white baneberry		4	6
4	*Agrimonia striata*	agrimony		5	5
5	*Agrostis scabra*	rough hair grass		2	2
6	*Agrostis stolonifera* *	redtop		0	0
7	*Alisma plantago-aquatica*	broad-leaved water-plantain	Alisma triviale	4	4
8	*Allium schoenoprasum*	wild chives		4	6
9	*Alnus incana*	river alder		4	5
10	*Alnus viridis*	green alder	Alnus crispa	3	4
11	*Alopecurus aequalis*	short-awned foxtail		4	4
12	*Alopecurus pratensis* *	meadow foxtail		0	0
13	*Amelanchier alnifolia*	saskatoon		3	4
14	*Anemone riparia*	tall anemone		6	7
15	*Arctostaphylos uva-ursi*	common bearberry		5	5
16	*Arnica chamissonis*	leafy arnica		5	5
17	*Artemisia biennis*	biennial wormwood		2	2
18	*Aster borealis*	marsh aster	Aster junceus	6	6
19	*Aster brachyactis*	rayless aster		4	4
20	*Aster ciliolatus*	Lindley's aster		3	4
21	*Aster conspicuous*	showy aster		4	4
22	*Aster ericoides*	tufted white prairie aster		5	4
23	*Aster falcatus*	creeping white prairie aster		5	5
24	*Aster hesperius*	western willow aster		6	6
25	*Aster lanceolatus*	panicled aster		3	4
26	*Aster modestus*	large northern aster		4	5
27	*Aster pauciflorus*	few-flowered aster		6	6
28	*Aster puniceus*	purple-stemmed aster		5	5
29	*Astragalus Canadensis*	Canadian milk vetch		5	5
30	*Astragalus dasyglottis*	purple milk vetch		5	5
31	*Atriplex heterosperma* *	saltbush		0	0
32	*Atriplex prostrata* *	prostrate saltbush	Atriplex patula	0	0
33	*Atriplex subspicata*	spearscale saltbush		4	5
34	*Barbarea orthoceras*	American winter cress		4	6
35	*Beckmannia syzigachne*	slough grass		2	3
36	*Betula glandulosa*	bog birch		5	6
37	*Betula neoalaskana*	Alaska birch		6	6
38	*Betula occidentalis*	water birch		5	6
39	*Betula papyrifera*	white birch		6	5
40	*Betula pumila*	dwarf birch		5	6
41	*Bidens cernua*	nodding beggarticks		4	4
42	*Bidens tripartite* *	tall beggarticks		0	0
43	*Bromus ciliatus*	fringed brome		4	5
44	*Bromus inermis*	awnless brome		0	0
45	*Calamagrostis canadensis*	bluejoint		3	2

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
46	*Calamagrostis inexpansa*	northern reed grass		4	4
47	*Calamagrostis stricta*	narrow reed grass	Calamagrostis neglecta	4	4
48	*Calla palustris*	water arum		7	8
49	*Callitriche hermaphroditica*	northern water-starwort		5	6
50	*Callitriche verna*	vernal water-starwort		4	5
51	*Caltha natans*	floating marsh-marigold		6	6
52	*Caltha palustris*	marsh-marigold		6	6
53	*Capsella bursa-pastoris* *	shepherd's-purse		0	0
54	*Cardamine pensylvanica*	bitter cress		3	5
55	*Cardamine pratensis*	meadow bitter cress		7	7
56	*Carex aquatilis*	water sedge		4	3
57	*Carex atherodes*	awned sedge		5	5
58	*Carex aurea*	golden sedge		3	3
59	*Carex bebbii*	Bebb's sedge		3	4
60	*Carex brunnescens*	brownish sedge		5	5
61	*Carex buxbaumii*	brown sedge		9	9
62	*Carex canescens*	short sedge	Carex curta	7	6
63	*Carex capillaries*	hair-like sedge		6	6
64	*Carex chordorrhiza*	prostrate sedge		9	7
65	*Carex crawfordii*	Crawford's sedge			5
66	*Carex diandra*	two-stamened sedge		5	6
67	*Carex disperma*	two-seeded sedge		8	6
68	*Carex eburnean*	bristle-leaved sedge		6	7
69	*Carex gynocrates*	northern bog sedge		8	8
70	*Carex heleonastes*	Hudson Bay sedge		9	9
71	*Carex interior*	inland sedge		5	6
72	*Carex lacustris*	lakeshore sedge		7	8
73	*Carex lasiocarpa*	hairy-fruited sedge		7	6
74	*Carex limosa*	mud sedge		9	7
75	*Carex livida*	livid sedge		9	8
76	*Carex loliacea*	rye-grass sedge		7	7
77	*Carex oligosperma*	few-fruited sedge		8	9
78	*Carex pauciflora*	few-flowered sedge		9	9
79	*Carex paupercula*			8	8
80	*Carex pellita*	woolly sedge	Carex lanuginosa	5	6
81	*Carex praegracilis*	graceful sedge		4	5
82	*Carex prairea*	prairie sedge		7	7
83	*Carex praticola*	meadow sedge		6	5
84	*Carex retrorsa*	turned sedge		4	5
85	*Carex rostrata*	beaked sedge		8	8
86	*Carex sartwellii*	Sartwell's sedge		5	5
87	*Carex saxatilis*	rocky-ground sedge		6	6
88	*Carex stipata*	awl-fruited sedge		4	5
89	*Carex sychnocephala*	long-beaked sedge		5	5
90	*Carex tenera*	broad-fruited sedge		5	5
91	*Carex tenuiflora*	thin-flowered sedge		8	8
92	*Carex torreyi*	Torrey's sedge		5	5
93	*Carex trisperma*	three-seeded sedge		9	8
94	*Carex utriculata*	small bottle sedge		5	5
95	*Carex vaginata*	sheathed sedge		7	5
96	*Carex viridula*	green sedge		7	7
97	*Carex vulpinoidea*	fox sedge		8	8
98	*Castilleja raupii*	purple paintbrush		5	5
99	*Cerastium arvense*	field mouse-ear chickweed		3	4
100	*Ceratophyllum demersum*	hornwort		4	5
101	*Chenopodium album* *	lamb's-quarters		0	0
102	*Chenopodium capitatum*	strawberry blite		1	2
103	*Chenopodium rubrum*	red goosefoot		4	4

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
104	*Chenopodium salinum*	oak-leaved goosefoot		3	3
105	*Chrysosplenium iowense*	golden saxifrage		5	5
106	*Chrysosplenium tetrandrum*	green saxifrage		6	6
107	*Cicuta bulbifera*	bulb-bearing water-hemlock		5	6
108	*Cicuta maculata*	water-hemlock	Cicuta douglasii	4	5
109	*Cicuta virosa*	narrow-leaved water-hemlock		6	6
110	*Cinna latifolia*	drooping wood-reed		6	7
111	*Cirsium arvense* *	creeping thistle		0	0
112	*Coeloglossum viride*	bracted bog orchid	Habenaria viridis	7	7
113	*Conium maculatum* *	poison hemlock		0	0
114	*Coptis trifolia*	goldthread		8	9
115	*Cornus stolonifera*	red-osier dogwood		2	3
116	*Corydalis aurea*	scrambled eggs		1	2
117	*Crepis tectorum* *	annual hawk's-beard		0	0
118	*Cypripedium parviflorum*			7	8
119	*Delphinium glaucum*	Sierra larkspur		5	5
120	*Deschampsia caspitosa*	tufted hair grass		4	4
121	*Descurainia sophia* *	flixweed		0	0
122	*Distichlis stricta*	salt grass	Distichlis spicata	7	7
123	*Dodecatheon pulchellum*	saline shooting star	Dodecatheon pauciflorum	6	7
124	*Dracocephalum parviflorum*	American dragonhead		1	0
125	*Drepanocladus aduncus*	Drepanocladus aduncus		2	4
126	*Elatine triandra*	waterwort		9	8
127	*Eleocharis acicularis*	needle spike-rush		4	4
128	*Eleocharis palustris*	creeping spike-rush		4	5
129	*Eleocharis quinqueflora*	few-flowered spike-rush		8	7
130	*Elodea Canadensis*			5	6
131	*Elymus trachycaulus*	slender wheat grass	Agropyron trachycaulum	2	3
132	*Epilobium angustifolium*	common fireweed		1	1
133	*Epilobium ciliatum*	northern willowherb	Epilobium glandulosum	2	2
134	*Epilobium leptophyllum*	narrow-leaved willowherb		6	7
135	*Epilobium palustre*	marsh willowherb		3	5
136	*Equisetum arvense*	common horsetail		1	1
137	*Equisetum fluviatile*	swamp horsetail		5	5
138	*Equisetum hyemale*	common scouring-rush		4	4
139	*Equisetum laevigatum*	smooth scouring-rush		4	5
140	*Equisetum palustre*	marsh horsetail		5	5
141	*Equisetum pratense*	meadow horsetail		5	4
142	*Equisetum scirpoides*	dwarf scouring-rush		5	5
143	*Equisetum sylvaticum*	woodland horsetail		5	6
144	*Equisetum variegatum*	variegated horsetail		5	6
145	*Erigeron acris*	northern daisy fleabane	Erigeron angulosus	3	3
146	*Erigeron elatus*	tall fleabane		7	7
147	*Erigeron lonchophyllus*			5	4
148	*Erigeron philadelphicus*	Philadelphia fleabane		2	3
149	*Eriophorum brachyantherum*	close-sheathed cotton grass		4	7
150	*Eriophorum chamissonis*	russett cotton grass		5	7
151	*Eriophorum gracile*	slender cotton grass		7	7
152	*Eriophorum polystachion*	tall cotton grass		6	6
153	*Eriophorum scheuchzeri*	one-spike cotton grass		8	7
154	*Eriophorum viridi-carinatum*	thin-leaved cotton grass		6	6
155	*Erysimum cheiranthoides*	wormseed mustard		0	1
156	*Eupatorium maculatum*	spotted Joe-pye weed		9	6
157	*Fragaria vesca*	woodland strawberry		4	4
158	*Fragaria virginiana*	wild strawberry		1	2
159	*Galeopsis tetrahit* *	hemp-nettle		0	0
160	*Galium labradoricum*	Labrador bedstraw		5	7
161	*Galium trifidum*	small bedstraw		5	5

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
162	*Galium triflorum*	sweet-scented bedstraw		4	5
163	*Gentianopsis detonsa*	northern fringed gentian		7	8
164	*Geum aleppicum*	yellow avens		3	3
165	*Geum macrophyllum*	large-leaved yellow avens		3	4
166	*Geum rivale*	purple avens		6	6
167	*Glaux maritime*	sea milkwort		6	6
168	*Glyceria borealis*	northern manna grass		5	6
169	*Glyceria grandis*	common tall manna grass		6	5
170	*Glyceria pulchella*	graceful manna grass		4	6
171	*Glyceria striata*	fowl manna grass		4	4
172	*Glycyrrhiza lepidota*	wild licorice		5	5
173	*Gratiola neglecta*	clammy hedgehyssop		4	5
174	*Helenium autumnale*	sneezeweed		5	5
175	*Helianthus nuttallii*	Nuttall's sunflower		3	4
176	*Heracleum lanatum*	cow parsnip		4	4
177	*Hieracium umbellatum*	narrow-leaved hawkweed		2	3
178	*Hierochloe hirta*			4	5
179	*Hippuris vulgaris*	common mare's-tail		5	5
180	*Hordeum jubatum*	foxtail barley		1	1
181	*Hypericum majus*	large Canada St. John's-wort			6
182	*Impatiens capensis*	spotted touch-me-not		3	4
183	*Impatiens noli-tangere*	western jewelweed		4	6
184	*Isoetes echinospora*	northern quillwort		10	9
185	*Iva axillaris*			3	3
186	*Juncus alpinoarticulatus*	alpine rush	Juncus alpinus	4	4
187	*Juncus balticus*	wire rush	Juncus arcticus, Juncus ater	3	3
188	*Juncus brevicaudatus*	short-tail rush		7	6
189	*Juncus bufonius*	toad rush		2	2
190	*Juncus filiformis*	thread rush		6	6
191	*Juncus longistylis*	long-styled rush		5	5
192	*Juncus nodosus*	knotted rush		4	4
193	*Juncus tenuis*	slender rush	Juncus dudleyi	3	3
194	*Juncus vaseyi*	big-head rush		5	5
195	*Lactuca pulchella*	common blue lettuce		4	4
196	*Lactuca serriola* *	prickly lettuce		0	0
197	*Larix laricina*	tamarack		6	6
198	*Ledum groenlandicum*	common Labrador tea		6	6
199	*Lemna minor*	common duckweed		4	3
200	*Lemna trisulca*	ivy-leaved duckweed		4	4
201	*Lepidium densiflorum*	common pepper-grass		0	0
202	*Limosella aquatica*	mudwort		3	2
203	*Linaria vulgaris* *	Toadflax		0	0
204	*Listera borealis*	northern twayblade		9	9
205	*Lobelia dortmanna*	water lobelia		9	9
206	*Lobelia kalmii*	Kalm's lobelia		9	9
207	*Lomatogonium rotatum*	marsh felwort		7	8
208	*Lotus corniculatus* *	bird's-foot trefoil		0	0
209	*Lycopus asper*	western water-horehound		4	5
210	*Lycopus uniflorus*	northern water-horehound		3	6
211	*Lysimachia lanceolata*	lanceleaf loosestrife		5	
212	*Lysimachia thyrsiflora*	tufted loosestrife		6	6
213	*Lythrum salicaria* *	purple loosestrife		0	0
214	*Marchantia polymorpha*			2	1
215	*Matricaria matricarioides* *	pineappleweed		0	0
216	*Matricaria perforate* *	scentless chamomile	matricaria maritima	0	0
217	*Melilotus alba* *	white sweet-clover		0	0
218	*Melilotus officinalis* *	yellow sweet-clover		0	0
219	*Mentha arvensis*	wild mint	Mentha canadensis	4	4

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
220	*Mentha spicata* *	spearmint		0	0
221	*Menyanthes trifoliata*	buck-bean		8	8
222	*Monolepis nuttalliana*	spear-leaved goosefoot		1	1
223	*Muhlenbergia glomerata*	bog muhly		7	8
224	*Muhlenbergia richardsonis*	mat muhly		6	6
225	*Myrica gale*	sweet gale		7	7
226	*Myriophyllum exalbescens*	spiked water-milfoil	Myriophyllum sibiricum	4	5
227	*Myriophyllum verticillatum*	water-milfoil		4	5
228	*Najas flexilis*	slender naiad		8	7
229	*Nasturtium officinale* *	water cress		0	0
230	*Nuphar lutea*	yellow pond-lily		5	6
231	*Nymphaea tetragona*	white water-lily		9	8
232	*Parnassia palustris*	northern grass-of-parnassus		5	6
233	*Pedicularis groenlandica*	elephant's-head		7	5
234	*Pedicularis parviflora*	swamp lousewort		7	6
235	*Petasites frigidus var frigidus*	sweet coltsfoot		6	5
236	*Petasites frigidus var palmatus*	palmate-leaved coltsfoot		4	4
237	*Petasites frigidus var sagittatus*	arrow-leaved coltsfoot		4	5
238	*Petasites frigidus var x vitifolius*	vine-leaved coltsfoot		5	5
239	*Phalaris arundinacea*	reed canary grass		2	2
240	*Phalaris canariensis* *	canary grass		0	0
241	*Phleum pratense* *	timothy		0	0
242	*Phragmites australis*	reed	Phragmites communis	4	5
243	*Physostegia parviflora*	false dragonhead		6	5
244	*Picea mariana*	black spruce		6	5
245	*Plagiobothrys scouleri*	Scouler's popcornflower		2	3
246	*Plantago eriopoda*	saline plantain		5	4
247	*Plantago major* *	common plantain		0	0
248	*Plantago maritima*	sea-side plantain			6
249	*Platanthera hyperborea*	northern green bog orchid	Habenaria hyperborea	5	5
250	*Poa palustris*	fowl bluegrass		3	3
251	*Poa pratensis*	Kentucky bluegrass		0	0
252	*Polemonium acutiflorum*	tall Jacob's-ladder			7
253	*Polygonum amphibium*	water smartweed	Polygonum natans	4	4
254	*Polygonum arenastrum* *	common knotweed	Polygonum aviculare	0	0
255	*Polygonum coccineum*	water smartweed		4	5
256	*Polygonum erectum*	striate knotweed		1	2
257	*Polygonum lapathifolium*	pale persicaria	Polygonum scabrum	2	2
258	*Polygonum persicaria* *	lady's-thumb		0	0
259	*Polygonum ramosissimum*	bushy knotweed		3	3
260	*Polygonum viviparum*	alpine bistort		4	7
261	*Potamogeton alpinus*	alpine pondweed		7	6
262	*Potamogeton crispus* *	crisp-leaved pondweed		0	0
263	*Potamogeton filiformis*	thread-leaved pondweed		5	5
264	*Potamogeton foliosus*	leafy pondweed		4	5
265	*Potamogeton friesii*	Fries' pondweed		5	6
266	*Potamogeton gramineus*	various-leaved pondweed		3	5
267	*Potamogeton natans*	floating-leaf pondweed		5	5
268	*Potamogeton obtusifolius*	blunt-leaved pondweed		7	6
269	*Potamogeton pectinatus*	sago pondweed		3	4
270	*Potamogeton praelongus*	white-stem pondweed		6	5
271	*Potamogeton pusillus*	small-leaf pondweed	Potamogeton berchtoldii	4	5
272	*Potamogeton richardsonii*	clasping-leaf pondweed		4	4
273	*Potamogeton strictifolius*	linear-leaved pondweed		5	5
274	*Potamogeton vaginatus*	large-sheath pondweed		5	5
275	*Potamogeton zosteriformis*	flat-stemmed pondweed		5	5
276	*Potentilla anserina*	silverweed		3	3
277	*Potentilla gracilis*	graceful cinquefoil		5	5

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
278	*Potentilla norvegica*	rough cinquefoil		2	2
279	*Potentilla palustris*	marsh cinquefoil		7	7
280	*Potentilla rivalis*	brook cinquefoil		4	5
281	*Primula incana*	mealy primrose		7	6
282	*Puccinellia distans* *	slender salt-meadow grass		0	0
283	*Puccinellia nuttalliana*	Nuttall's salt-meadow grass		5	5
284	*Ranunculus abortivus*	small-flowered buttercup		5	5
285	*Ranunculus acris* *	tall buttercup		0	0
286	*Ranunculus aquatilis*	large-leaved white water crowfoot	Ranunculus trichphyllus	5	5
287	*Ranunculus cymbalaria*	seaside buttercup		4	4
288	*Ranunculus gmelinii*	yellow water crowfoot	Ranunculus purshii	4	5
289	*Ranunculus hyperboreus*	boreal buttercup		5	7
290	*Ranunculus lapponicus*	Lapland buttercup		7	7
291	*Ranunculus longirostris*	longbeak buttercup		4	4
292	*Ranunculus macounii*	Macoun's buttercup		5	5
293	*Ranunculus pensylvanicus*	bristly buttercup		5	5
294	*Ranunculus reptans*	creeping spearwort		5	6
295	*Ranunculus sceleratus*	celery-leaved buttercup		3	3
296	*Rhinanthus minor* *	yellow rattle	Rhinanthus borealis	4	3
297	*Ribes americanum*	wild black currant		6	7
298	*Ribes glandulosum*	skunk currant		6	6
299	*Ribes hudsonianum*	northern black currant		7	7
300	*Ribes lacustre*	bristly black currant		6	6
301	*Ribes oxyacanthoides*	northern gooseberry		3	4
302	*Ribes triste*	wild red currant		5	5
303	*Rorippa curvipes*	bluntleaf yellowcress		5	3
304	*Rorippa palustris*	marsh yellow cress	Rorippa islandica	4	4
305	*Rosa acicularis*	prickly rose		3	2
306	*Rosa woodsii*	common wild rose		4	4
307	*Rubus arcticus*	dwarf raspberry	Rubus acaulis	5	6
308	*Rubus idaeus*	wild red raspberry	Rubus strigosus	1	1
309	*Rubus pubescens*	dewberry		5	5
310	*Rumex crispus* *	curled dock		0	0
311	*Rumex maritimus*	golden dock		2	4
312	*Rumex occidentalis*	western dock	Rumex fenestratus	4	5
313	*Rumex orbiculatus*	water dock	Rumex britannica	4	5
314	*Rumex triangulivalvis*	narrow-leaved dock	Rumex mexicanus, Rumex salcifolius	3	3
315	*Ruppia cirrhosa*	widgeon-grass	Ruppia occidentalis	9	6
316	*Sagittaria cuneata*	arum-leaved arrowhead		5	5
317	*Sagittaria latifolia*	broad-leaved arrowhead		6	5
318	*Salicornia rubra*	samphire	Salicornia europaea	4	6
319	*Salix arbusculoides*	shrubby willow		4	5
320	*Salix bebbiana*	beaked willow		2	2
321	*Salix candida*	hoary willow		8	6
322	*Salix discolor*	pussy willow		2	2
323	*Salix exigua*	sandbar willow	Salix interior	2	3
324	*Salix glauca*	smooth willow		3	3
325	*Salix lucida*	shining willow	Salix lasiandra	5	6
326	*Salix lutea*	yellow willow		4	5
327	*Salix maccalliana*	velvet-fruited willow		3	5
328	*Salix myrtillifolia*	myrtle-leaved willow		5	6
329	*Salix petiolaris*	basket willow		4	4
330	*Salix planifolia*	flat-leaved willow		4	4
331	*Salix prolixa*	Mackenzie's willow	Salix rigida	5	6
332	*Salix pseudomonticola*	false mountain willow		4	5
333	*Salix pyrifolia*	balsam willow		6	6
334	*Salix scouleriana*	Scouler's willow		3	5
335	*Salix serissima*	autumn willow		6	6

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
336	*Schoenoplectus acutus*	great bulrush		5	5
337	*Schoenoplectus tabernaemontani*	common great bulrush	Scirpus acutus, Scirpus validus	4	4
338	*Scirpus cespitosus*	tufted bulrush		7	6
339	*Scirpus hudsonianus*	Hudson Bay bulrush		6	7
340	*Scirpus microcarpus*	small-fruited bulrush		3	3
341	*Scirpus paludosus*	prairie bulrush		6	6
342	*Scirpus pungens*	three-square rush		6	6
343	*Scolochloa festucacea*	spangletop	Fluminia festucacea	4	5
344	*Scutellaria galericulata*	marsh skullcap	Scutellaria epilobiifolia	6	5
345	*Senecio congestus*	marsh ragwort		3	3
346	*Senecio eremophilus*	cut-leaved ragwort		4	5
347	*Sisyrinchium montanum*	common blue-eyed grass		5	5
348	*Sium suave*	water parsnip		5	5
349	*Smilacina stellata*	star-flowered Solomon's-seal		5	5
350	*Smilacina trifolia*	three-leaved Solomon's-seal		8	8
351	*Solidago canadensis*	Canada goldenrod		2	3
352	*Solidago gigantea*	late goldenrod		4	5
353	*Solidago graminifolia*	flat-topped goldenrod		5	5
354	*Sonchus arvensis* *	perennial sow-thistle		0	0
355	*Sonchus asper* *	prickly annual sow-thistle		0	0
356	*Sonchus uliginosus* *	smooth perennial sow-thistle		0	0
357	*Sparganium angustifolium*	narrow-leaved bur-reed		6	6
358	*Sparganium eurycarpum*	giant bur-reed		5	5
359	*Sparganium minimum*	slender bur-reed		7	6
360	*Spartina gracilis*	alkali cord grass		6	7
361	*Spartina pectinata*	prairie cord grass			9
362	*Spergularia salina*	salt-marsh sand spurry		8	9
363	*Spiraea alba*	white meadowsweet		5	5
364	*Spiranthes romanzoffiana*	hooded ladies'-tresses		8	8
365	*Spirodela polyrhiza*	larger duckweed		4	5
366	*Stachys palustris*	marsh hedge-nettle		4	4
367	*Stellaria calycantha*	northern stitchwort		4	4
368	*Stellaria crassifolia*	fleshy stitchwort		6	6
369	*Stellaria longifolia*	long-leaved chickweed		4	5
370	*Stellaria longipes*	long-stalked chickweed		3	5
371	*Stuckenia filiformis*			5	5
372	*Stuckenia pectinata*			2	2
373	*Suaeda calceoliformis*	western sea-blite		4	5
374	*Tanacetum vulgare* *	common tansy		0	0
375	*Taraxacum laevigatum* *	red-seeded dandelion		0	0
376	*Taraxacum officinale* *	common dandelion		0	0
377	*Thlaspi arvense* *	stinkweed		0	0
378	*Tofieldia glutinosa*	sticky false asphodel		7	7
379	*Trichophorum clintonii*	Clinton's bulrush	Scirpus clintonii	9	7
380	*Trifolium hybridum* *	alsike clover		0	0
381	*Trifolium pretense* *	red clover		0	0
382	*Trifolium repens* *	white clover		0	0
383	*Triglochin maritime*	seaside arrow-grass		5	5
384	*Triglochin palustris*	slender arrow-grass		6	6
385	*Typha latifolia*	common cattail		2	2
386	*Urtica dioica*	common nettle		3	3
387	*Urtica urens* *	small nettle	Urtica gracilis	0	0
388	*Utricularia cornuta*	horned bladderwort		8	9
389	*Utricularia intermedia*	flat-leaved bladderwort		7	7
390	*Utricularia minor*	small bladderwort		7	7
391	*Utricularia vulgaris*	common bladderwort		4	5
392	*Vaccinium vitis-idaea*	bog cranberry		5	6
393	*Valeriana dioica*	northern valerian		6	6

ID	Species name	Common Name	Synonyms	Parkland C	Boreal C
394	*Veronica americana*	American brooklime		4	5
395	*Veronica anagallis-aquatica*	water speedwell		0	0
396	*Veronica peregrina*	hairy speedwell		3	4
397	*Veronica scutellata*	marsh speedwell		3	4
398	*Vicia americana*	wild vetch		3	3
399	*Viola macloskeyi*			8	7
400	*Viola nephrophylla*	bog violet		7	7
401	*Viola palustris*	marsh violet		6	5
402	*Wolffia borealis*	northern ducksmeal		8	6
403	*Wolffia columbiana*	watermeal		8	7
404	*Zannichellia palustris*	horned pondweed		5	6
405	*Zizania aquatica* *	wild rice		0	0
406	*Zizia aptera*	heart-leaved Alexanders		6	7

* Denotes species designated as non-native (i.e., exotic) by the Alberta Natural Heritage Information Centre (ANHIC)

Appendix B. Summary of the 4 wetland types and 32 wetland sites used during this study. UTM locations represent approximate central points for each wetland.

Wetland Type	Wetland Sub-Type	Site Name	UTM Location Easting	UTM Location Northing	Controlling Jurisdiction
Natural	Reference	Nat 0	380446	5952470	Parks Canada (Elk Island National Park)
		Nat 10	375414	5900017	Parks Canada (Elk Island National Park)
		Nat 12	377299	5945220	Parks Canada (Elk Island National Park)
		Nat 3	370997	5929317	Alberta Parks (Cooking Lake Blackfoot PRA)
		Nat 45	372045	5932180	Parks Canada (Elk Island National Park)
		Nat 46	377369	5937827	Parks Canada (Elk Island National Park)
		Nat 47	371273	5939673	Parks Canada (Elk Island National Park)
		Nat 7	381816	5926686	Alberta Parks (Cooking Lake Blackfoot PRA)
	Agricultural	Ag 20	366453	5943370	Private
		Ag 22	365238	5914928	Private
		Ag 27	355570	5924961	Private
		Ag 28	350481	5906592	Private
		Ag 34	386957	5937153	Private
		Ag 37	353091	5939153	Private
		Ag 41	359478	5940379	Private
		Ag 67	385640	5927930	Private
Created	Naturalized Stormwater Wetland	Callingwood	323844	5930324	Alberta Transportation
		Campsite	306022	5939922	Alberta Transportation
		Canossa	333023	5945774	City of Edmonton
		Clarkdale	350582	5935898	Strathcona County
		Cloverbar	348908	5936431	Strathcona County
		Rutherford	331567	5922418	City of Edmonton
		Silverberry	341392	5926247	City of Edmonton
		Terwillegar	327451	5925234	City of Edmonton
	Wet Pond	Bearspaw	333687	5924559	City of Edmonton
		Brintnell	341005	5943544	City of Edmonton
		Lago Chirro	336052	5943841	City of Edmonton
		Hollick Kenyon	339496	5943789	City of Edmonton
		Hudson	331495	5942592	City of Edmonton
		Meadowbrook	341230	5927463	City of Edmonton
		Twin brooks	331865	5924446	City of Edmonton
		Valencia	335835	5946220	City of Edmonton

Appendix C. Wetland dependent bird species observed during surveys at 32 wetlands of 4 different types. Primary detection method identifies the primary survey type used to detect the species (V=visual, A=auditory). The habitation association column indicates species identified as having a specific association with either the emergent vegetation zone (EM) or the wet meadow (WM) zone; species without a listed habitat association are either dependent on the whole wetland system or nest in surrounding upland habitats. Guild defines species as a waterbird (WB) or songbird (SB). NSW = naturalized stormwater wetlands; WP = wet ponds.

					Mean maximum abundance per site [1]							
Common name	**Scientific name**	**Primary detection method**	**Habitat association**	**Guild**	**Reference**		**Agricultural**		**NSW**		**WP**	
					Mean	(SE)	Mean	(SE)	Mean	(SE)	Mean	(SE)
Canada Goose	*Branta canadensis*	V		WB	0.63	(0.26)	1.88	(0.81)	1.25	(0.37)	1.88*	(0.67)
Gadwall	*Anas strepera*	V		WB	0.75	(0.37)	2.00	(0.46)	0.38	(0.18)	0.13*	(0.13)
American Wigeon	*Anas americana*	V		WB	1.00	(0.60)	0.38	(0.26)	0.25*	(0.16)	0.00	(0.00)
Mallard	*Anas platyrhynchos*	V		WB	3.75	(1.71)	13.63*	(11.07)	6.00*	(1.71)	6.50*	(3.09)
Blue-winged Teal	*Anas discors*	V		WB	6.00*	(1.72)	4.00	(0.68)	4.00	(0.76)	1.88	(1.46)
Northern Shoveler	*Anas clypeata*	V		WB	0.38	(0.18)	1.75	(0.82)	0.63	(0.32)	0.00	(0.00)
Green-winged Teal [2]	*Anas crecca*	V		WB	2.00*	(1.45)	0.75	(0.31)	0.25	(0.16)	0.00	(0.00)
Canvasback	*Aythya valisineria*	V	EM	WB	0.25	(0.25)	0.25	(0.16)	0.00	(0.00)	0.00	(0.00)
Redhead	*Aythya americana*	V	EM	WB	0.50	(0.27)	1.75*	(0.59)	0.63	(0.38)	0.13	(0.13)
Ring-necked Duck	*Aythya collaris*	V	EM	WB	0.38*	(0.18)	0.50	(0.50)	0.00	(0.00)	0.00	(0.00)
Lesser Scaup [2]	*Aythya affinis*	V		WB	1.00	(0.33)	1.00*	(0.57)	1.38	(0.53)	0.75	(0.49)
Bufflehead	*Bucephala albeola*	V		WB	0.50	(0.19)	0.38*	(0.26)	0.13	(0.13)	0.00	(0.00)
Common Goldeneye	*Bucephala clangula*	V		WB	0.00	(0.00)	0.00	(0.00)	0.13	(0.13)	0.25*	(0.16)
Ruddy Duck	*Oxyura jamaicensis*	V	EM	WB	0.75*	(0.37)	1.63	(0.50)	0.88	(0.30)	0.13	(0.13)
Pied-billed Grebe [2]	*Podilymbus podiceps*	V	EM	WB	0.25	(0.16)	0.25	(0.16)	0.63	(0.26)	0.13	(0.13)
Horned Grebe [2]	*Podiceps auritus*	V	EM	WB	0.25*	(0.16)	0.25	(0.16)	0.25	(0.25)	0.00	(0.00)
Red-necked Grebe	*Podiceps grisegena*	V		WB	0.25	(0.16)	0.50	(0.27)	0.63	(0.26)	1.25*	(0.16)
Eared Grebe	*Podiceps nigricollis*	V	EM	WB	0.00	(0.00)	0.13	(0.13)	0.00	(0.00)	0.00	(0.00)
Great Blue Heron [2]	*Ardea herodias*	V		WB	0.25	(0.25)	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)
Sora [2]	*Porzana carolina*	A	EM	WB	1.25	(0.25)	0.50	(0.33)	0.00	(0.00)	1.50	(0.27)

Common name	Scientific name	Primary detection method	Habitat association	Guild	Mean maximum abundance per site [1] Reference Mean (SE)		Agricultural Mean (SE)		NSW Mean (SE)		WP Mean (SE)	
American Coot	*Fulica americana*	V	EM	WB	3.38*	(1.22)	6.63	(3.02)	4.13*	(1.43)	0.75*	(0.49)
Killdeer	*Charadrius vociferus*	V		WB	0.00	(0.00)	0.13	(0.13)	0.00	(0.00)	0.63	(0.38)
Spotted Sandpiper	*Actitis macularius*	V		WB	0.00	(0.00)	0.00	(0.00)	0.88	(0.30)	0.63	(0.26)
Wilson's Snipe	*Gallinago delicata*	A	WM	WB	0.50	(0.27)	0.13	(0.13)	0.00	(0.00)	0.00	(0.00)
Wilson's Phalarope	*Phalaropus tricolor*	V		WB	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)	0.25	(0.25)
Franklin's Gull	*Leucophaeus pipixcan*			WB	0.00	(0.00)	0.00	(0.00)	0.13	(0.13)	0.13	(0.13)
Ring-billed Gull	*Larus delawarensis*	V		WB	0.13	(0.13)	1.25	(1.00)	2.00	(0.65)	1.00	(0.50)
Black Tern [2]	*Chlidonias niger*	V	EM	WB	2.38	(1.21)	2.63	(2.09)	0.75	(0.31)	0.00	(0.00)
Alder Flycatcher	*Empidonax alnorum*	A		SB	0.38	(0.26)	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)
Eastern Phoebe [2]	*Sayornis phoebe*	A		SB	0.13	(0.13)	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)
Marsh Wren	*Cistothorus palustris*	A	EM	SB	0.50	(0.50)	0.38	(0.38)	0.00	(0.00)	0.13	(0.13)
Common Yellowthroat [2]	*Geothlypis trichas*	A	WM	SB	1.00	(0.38)	0.75	(0.31)	0.00	(0.00)	0.00	(0.00)
Le Conte's Sparrow	*Ammodramus leconteii*	A	WM	SB	1.38	(0.42)	1.00	(0.38)	0.00	(0.00)	0.13	(0.13)
Nelson's Sparrow	*Ammodramus nelsoni*	A	WM	SB	0.13	(0.13)	0.00	(0.00)	0.00	(0.00)	0.13	(0.13)
Song Sparrow	*Melospiza melodia*	A	WM	SB	1.13	(0.23)	1.25	(0.37)	0.25	(0.16)	0.13	(0.13)
Lincoln's Sparrow	*Melospiza lincolnii*	A	WM	SB	0.50	(0.19)	0.00	(0.00)	0.00	(0.00)	0.25	(0.16)
Swamp Sparrow	*Melospiza georgiana*	A	WM	SB	0.13	(0.13)	0.13	(0.13)	0.00	(0.00)	0.00	(0.00)
Red-winged Blackbird	*Agelaius phoeniceus*	V	EM	SB	2.13*	(0.35)	6.00*	(1.12)	3.00*	(0.55)	4.25*	(0.45)
Yellow-headed Blackbird	*Xanthocephalus xanthocephalus*	V	EM	SB	0.00	(0.00)	0.75	(0.62)	0.00	(0.00)	0.25	(0.16)

[1] maximum abundance for waterbirds calculated using Indicated Breeding Pairs closely following xxxx (); maximum abundance for songbirds calculated using singing males
[2] species listed as 'Sensitive' by Alberta Sustainable Resource Development (2005)
* species confirmed as nesting at the site through observation of a nest and/or behavioral observations including nest building and active courting displays in suitable habitat. All other species can be considered as 'probable breeders'.

Appendix D. Plant species observed during quadrat surveys at 32 wetlands of 4 different types in central Alberta, Canada during the summer of 2010. Plant nomenclature closely follows the Integrated Taxonomic Information System (ITIS; www.itis.gov).

		# of sites where species was present in each wetland type (of a maximum of 8)					
Species name	**Common name**	**Agricultural**	**Reference**	**Naturalized stormwater wetland**	**Wet pond**	**Total # of sites where present (of 32)**	**Proportion of sites where present (%)**
Achillea millefolium	common yarrow	1	1	1		3	9.4
Achillea siberica	Siberian yarrow	1				1	3.1
Elymus repens	quackgrass				1	1	3.1
Agrostis scabra	rough hair grass	2	2	1	1	6	18.8
Alisma plantago-aquatica	broad-leaved water-plantain	1	2	1		4	12.5
Alopecurus aequalis	short-awned foxtail	2	4	1	3	10	31.3
Amelanchier alnifolia	saskatoon		1			1	3.1
Aster hesperius	western willow aster	2	2	1	2	7	21.9
Aster puniceus	purple-stemmed aster	4	6	1	1	12	37.5
Beckmannia syzigachne	slough grass	5	2	7	5	19	59.4
Bidens cernua	nodding beggarticks	2	5	2	1	10	31.3
Calamagrostis canadensis	bluejoint	5	8	2		15	46.9
Calamagrostis inexpansa	northern reed grass	5	7	4	2	18	56.3
Caltha palustris	marsh-marigold		1			1	3.1
Carex aquatilis	water sedge	5	7	5	2	19	59.4
Carex atherodes	awned sedge	8	8	6	2	24	75.0
Carex bebbii	Bebb's sedge	2		2		4	12.5
Carex diandra	two-stamened sedge		4	2		6	18.8
Carex utriculata	small bottle sedge	7	7	5	3	22	68.8
Chenopodium album *	lamb's-quarters	3	3	2	1	9	28.1
Cicuta bulbifera	bulb-bearing water-hemlock	4	3	3		10	31.3
Cicuta maculata	water-hemlock	2				2	6.3
Cirsium arvense *	creeping thistle	7	8	8	4	27	84.4
Cornus stolonifera	red-osier dogwood			1		1	3.1
Descurainia sophia *	flixweed			1	1	2	6.3

		# of sites where species was present in each wetland type (of a maximum of 8)					
Species name	**Common name**	**Agricultural**	**Reference**	**Naturalized stormwater wetland**	**Wet pond**	**Total # of sites where present (of 32)**	**Proportion of sites where present (%)**
Eleocharis palustris	creeping spike-rush	1	3	4	4	12	37.5
Epilobium angusifolium	fireweed		2			2	6.3
Epilobium ciliatum	northern willowherb	6	3	3	4	16	50.0
Epilobium palustre	marsh willowherb	1	2	1		4	12.5
Equisetum arvense	common horsetail	1	2	3	3	9	28.1
Equisetum fluviatile	swamp horsetail	2	2	1		5	15.6
Equisetum pratense	meadow horsetail	1	1	2		4	12.5
Equisetum sylvaticum	woodland horsetail		1			1	3.1
Erigeron philadelphicus	Philadelphia fleabane	1				1	3.1
Fragaria vesca	woodland strawberry		1			1	3.1
Fragaria virginiana	wild strawberry	2	1	1	1	5	15.6
Galeopsis tetrahit *	hemp-nettle	5	4	1	2	12	37.5
Galium trifidum	small bedstraw	7	7	3		17	53.1
Geum aleppicum	yellow avens	1				1	3.1
Geum rivale	purple avens	1	3	1		5	15.6
Glyceria grandis	common tall manna grass	4	3	4	1	12	37.5
Hieracium umbellatum	narrow-leaved hawkweed		1	1		2	6.3
Hordeum jubatum	foxtail barley	4	2	5	7	18	56.3
Impatiens capensis	spotted touch-me-not		2			2	6.3
Impatiens noli-tangere	western jewelweed	1				1	3.1
Iva axillaris				1		1	3.1
Juncus alpinoarticulatus	alpine rush		1	4	1	6	18.8
Juncus balticus	wire rush			1	3	4	12.5
Juncus bufonius	toad rush			3	1	4	12.5
Juncus nodosus	knotted rush	1				1	3.1
Lycopus asper	western water-horehound	3	4	1	1	9	28.1
Lycopus uniflorus	northern water-horehound	3	3			6	18.8
Lysimachia thyrsiflora	tufted loosestrife	1	1			2	6.3

		# of sites where species was present in each wetland type (of a maximum of 8)					
Species name	**Common name**	**Agricultural**	**Reference**	**Naturalized stormwater wetland**	**Wet pond**	**Total # of sites where present (of 32)**	**Proportion of sites where present (%)**
Marchantia polymorpha		1		1	1	3	9.4
Matricaria matricarioides *	pineappleweed			1	2	3	9.4
Matricaria perforate *	scentless chamomile				2	2	6.3
Melilotus alba *	white sweet-clover			1	2	3	9.4
Melilotus officinalis *	yellow sweet-clover	6	7	4	2	19	59.4
Mentha arvensis	wild mint		1			1	3.1
Petasites frigidus var palmatus	palmate-leaved coltsfoot		2			2	6.3
Petasites frigidus var sagittatus	arrow-leaved coltsfoot	6	4	8	1	19	59.4
Phalaris arundinacea	reed canary grass				1	1	3.1
Phleum pratense *	timothy			1		1	3.1
Plantago major *	common plantain	4	3	6	5	18	56.3
Poa palustris	fowl bluegrass	5	5	6	5	21	65.6
Poa pratensis	Kentucky bluegrass	1				1	3.1
Polygonum amphibium	water smartweed	1			1	2	6.3
Polygonum coccineum	water smartweed	2	4	3		9	28.1
Polygonum lapathifolium	pale persicaria	1	2	2		5	15.6
Potentilla anserina	silverweed	1	3	3	3	10	31.3
Potentilla norvegica	rough cinquefoil	3	2	2	1	8	25.0
Potentilla palustris	marsh cinquefoil	2	4	1		7	21.9
Ranunculus abortivus	small-flowered buttercup	1			2	3	9.4
Ranunculus cymbalaria	seaside buttercup	2	4	2	1	9	28.1
Ranunculus macounii	Macoun's buttercup	1			3	4	12.5
Ranunculus sceleratus	celery-leaved buttercup	1	3			4	12.5
Rorippa palustris	marsh yellow cress	2	2		1	5	15.6
Rosa woodsii	common wild rose		1			1	3.1
Rubus arcticus	dwarf raspberry		1			1	3.1
Rubus idaeus	wild red raspberry		1			1	3.1
Rubus pubescens	dewberry		1			1	3.1

		# of sites where species was present in each wetland type (of a maximum of 8)					
Species name	**Common name**	**Agricultural**	**Reference**	**Naturalized stormwater wetland**	**Wet pond**	**Total # of sites where present (of 32)**	**Proportion of sites where present (%)**
Rumex crispus *	curled dock		1			1	3.1
Rumex maritimus	golden dock				2	2	6.3
Rumex occidentalis	western dock	5	7	2	1	15	46.9
Sagittaria cuneata	arum-leaved arrowhead	3	4	2		9	28.1
Scolochloa festucacea	spangletop	1	4			5	15.6
Schoenoplectus tabernaemontani	common great bulrush	3	1	2		6	18.8
Scirpus microcarpus	small-fruited bulrush	2	2	1		5	15.6
Scutellaria galericulata	marsh skullcap	5		4	1	10	31.3
Senecio congestus	marsh ragwort	4	8	1		13	40.6
Sium suave	water parsnip	1	2	1		4	12.5
Smilacina stellata	star-flowered Solomon's-seal	2	5	2	1	10	31.3
Solidago canadensis	Canada goldenrod		1			1	3.1
Sonchus arvensis *	perennial sow-thistle		2			2	6.3
Sonchus asper *	prickly annual sow-thistle	6	4	8	6	24	75.0
Sparganium eurycarpum	giant bur-reed				1	1	3.1
Stachys palustris	marsh hedge-nettle	2	4	2		8	25.0
Stellaria calycantha	northern stitchwort	1	2	1	1	5	15.6
Stellaria longifolia	long-leaved chickweed		1			1	3.1
Taraxacum officinale *	common dandelion	2	4	1	1	8	25.0
Thlaspi arvense *	stinkweed	4	2	5	8	19	59.4
Trifolium hybridum *	alsike clover	3	1	6	8	18	56.3
Typha latifolia	common cattail	5	6	8	5	24	75.0
Urtica dioica	common nettle	5	5			10	31.3
Vicia americana	wild vetch	1	3		1	5	15.6
Total species richness		**73**	**79**	**67**	**53**	**--**	**--**

* Denotes species designated as non-native (i.e., exotic) by the Alberta Natural Heritage Information Centre (ANHIC)

Appendix E. Hydrograph illustrating the change in water depth of a stormwater wetland (top line) and a natural wetland (bottom line) over the course of the summer in 2008 (end of May through the end of September).