

Effects of Water-level Management on the Abundance and Habitat Use of
Waterfowl and Marsh Birds in the Saskatchewan River Delta, Manitoba,
Canada

by

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Abstract

Waterbird habitat in the Saskatchewan River Delta (SRD) has been altered by numerous upstream and downstream anthropogenic developments. Wetland water-level management has been used in an attempt to mitigate these changes, but its effects on the local waterbird community are unknown. Using an experimental approach, I examined the effects of wetland water-level management on waterfowl and marsh birds in the SRD.

In 2007, three wetland basins in the Summerberry Marsh Complex, Manitoba were partially drawn down to an average open water depth of 54 cm. Three additional wetlands in the complex were managed with high water levels, with an average open water depth of 89 cm. In 2008 and 2009, I surveyed waterfowl within the study wetlands during different life stages using point counts and aerial surveys. Breeding marsh birds, including American Bitterns (*Botaurus lentiginosus*), Least Bitterns (*Ixobrychus exilis*), Sora (*Porzana carolina*), Virginia Rails (*Rallus limicola*), Yellow Rails (*Coturnicops noveboracensis*), American Coots (*Fulica americana*), and Pied-billed Grebes (*Podilymbus podiceps*) were also surveyed in the study wetlands during the spring using call-response surveys. Generalized linear and generalized linear mixed models were used to determine the relationships between relative bird abundances to the physical characteristics, vegetation characteristics, and forage fish and invertebrate abundances within the wetlands.

The partial water-level drawdowns benefitted dabbler species of waterfowl, but negatively impacted diver species of waterfowl. Dabblers preferred the shallow water of

the partial drawdown wetlands during nearly all life-stages, except brooding. Diver species preferred the deeper water in the non-drawdown wetlands, but only during the spring-breeding and fall-staging periods. Invertebrate abundance affected wetland use by dabbling and diver species of waterfowl, particularly in the spring. Subsequently, wetlands with high forage fish abundances were avoided. The partial water-level drawdowns did not affect wetland selection by dabbling or diver broods, but invertebrate abundance was important. During the post-breeding period, dabbling species of waterfowl preferred the partial drawdown wetlands, whereas diver species tended to emigrate from the study wetlands. In the fall, dabbling densities on the partial drawdown wetlands increased from an average of 0.4 birds per hectare to 1.5 birds per hectare. This influx was likely due to the accessibility and availability of submerged aquatic vegetation, including the genera *Potamogeton* and *Stuckenia*.

Densities of most marsh bird species were relatively high in the study wetlands, compared to wetlands in southern Manitoba and elsewhere in North America. However, no Least Bitterns were detected, and only a single Yellow Rail was encountered.

The partial water-level drawdowns did not benefit any species of marsh birds examined. American Bitterns, American Coots, and Pied-billed Grebes chose the deeper, non-drawdown wetlands, due to high abundances of forage fish. Vegetation interspersion, particularly that offered by *Schoenoplectus* was preferred by these species, likely due to reducing interspecific competition for nesting and foraging sites. The Sora and Virginia Rail were not affected by water depth, or vegetation characteristics but were influenced

by invertebrate abundances. Subsequently, the Sora avoided wetlands with high fish abundance.

Due to the wide variety of waterbird species in the SRD, wetland water-level management should be focused on creating a diverse range of habitats to meet the varied species' requirements. I suggest this be accomplished by managing at the wetland-complex scale, using partial water-level drawdowns with a depth of approximately 50 cm as shallow-wetland habitat and wetlands at Full Supply Level with a depth of approximately 100 cm as deep-wetland habitat. This would provide a wide range of habitats in relatively close proximity, promoting avian biodiversity.

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Chapter 1 – Past and present wetland management within the Saskatchewan River Delta, Manitoba

1.1 Background

Wetland loss is a severe problem in the prairies of Canada and the United States. Up to 71% of wetlands have been lost in the prairie provinces and about 20 million ha in Canada (The Atlas of Canada 2004). In the United States the lower 48 states have lost about 53% of their wetlands, about 42 million ha (Dahl 1990). This habitat loss may be responsible for population declines in some species of waterfowl and marsh birds and illustrates the need to prevent further wetland loss, and to preserve the existing wetland habitats (Eddleman et al. 1988; Conway et al. 1994; Patterson 1994).

The continued loss of wetland habitat in Canada and the United States may place more pressure on wetlands in other regions, as displaced waterfowl and marsh birds search for suitable habitat. One such area may be the Saskatchewan River Delta (SRD). Wetlands in the SRD are less susceptible to drought and are known to provide a refuge to waterfowl when potholes in the prairies are dry (Figure 1.1; United States Fish and Wildlife Service and Manitoba Department of Mines and Natural Resources 1961; Uchtmann 1983).

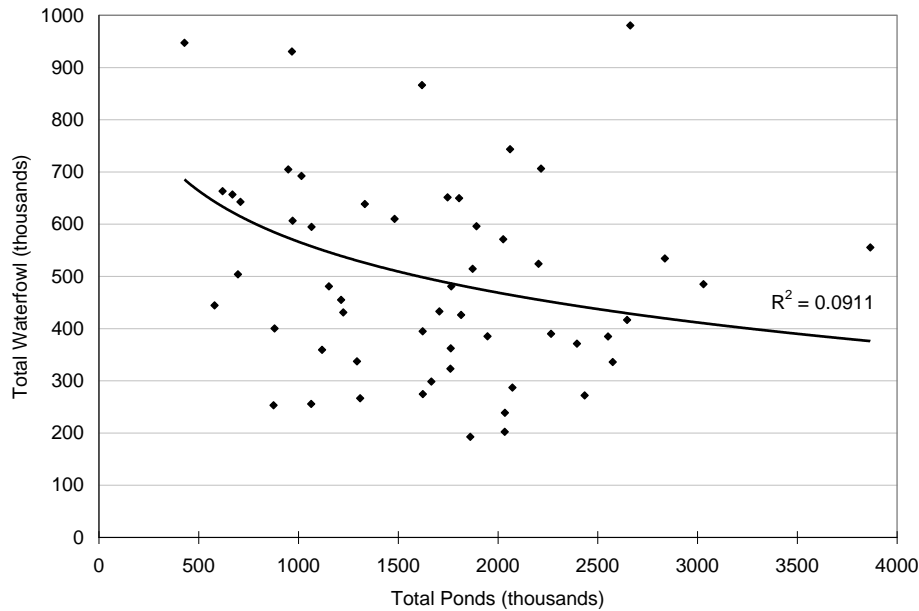


Figure 1.1 - Relationship between total number of prairie ponds and waterfowl in the SRD (log trend line). Total ponds calculated for Manitoba and Saskatchewan from 1955-1970 (Strata 32-35, 38-40), compared against total waterfowl for the SRD (Stratum 25) (United States Fish and Wildlife Service Migratory Bird Data Center 2009).

The SRD is the largest freshwater-inland delta in North America. It covers approximately 9,000 km² across Manitoba and Saskatchewan (Figure 1.2). The region has been recognized internationally as a Canadian Important Bird Area due to the globally significant waterfowl numbers it supports (IBA Canada 2009) and may be a candidate for designation as a Ramsar site (Lindgren 2001). The SRD provides important nesting and migration habitat for waterfowl, contributing significantly to continental waterfowl populations (Harper 1975; Uchtmann 1983). However, long-term trend data from aerial surveys conducted by the United States Fish and Wildlife Service show a decrease in the total numbers of ducks counted within the SRD during annual spring counts (Figure 1.3). The cause of this downward trend is unknown. However, in recent decades the Saskatchewan River has undergone extensive anthropogenic developments that have

altered the hydrological cycle in the delta. These changes may result in a modification of wetlands' wet-dry cycle, creating a more terrestrial environment, and reducing the amount of habitat available for waterfowl and other wetland-dependent species of birds (Clay 1978; Smith 2003). The wet-dry cycle in wetlands ensures there is spatial and temporal variation in avian habitat. This variability is important as species' requirements vary, as do individual requirements during different life cycle stages (Murkin et al. 1997). Therefore, a need exists for research to better understand how wetlands in the SRD can be managed to create productive habitat for waterfowl and marsh birds influenced by an altered hydrological cycle.

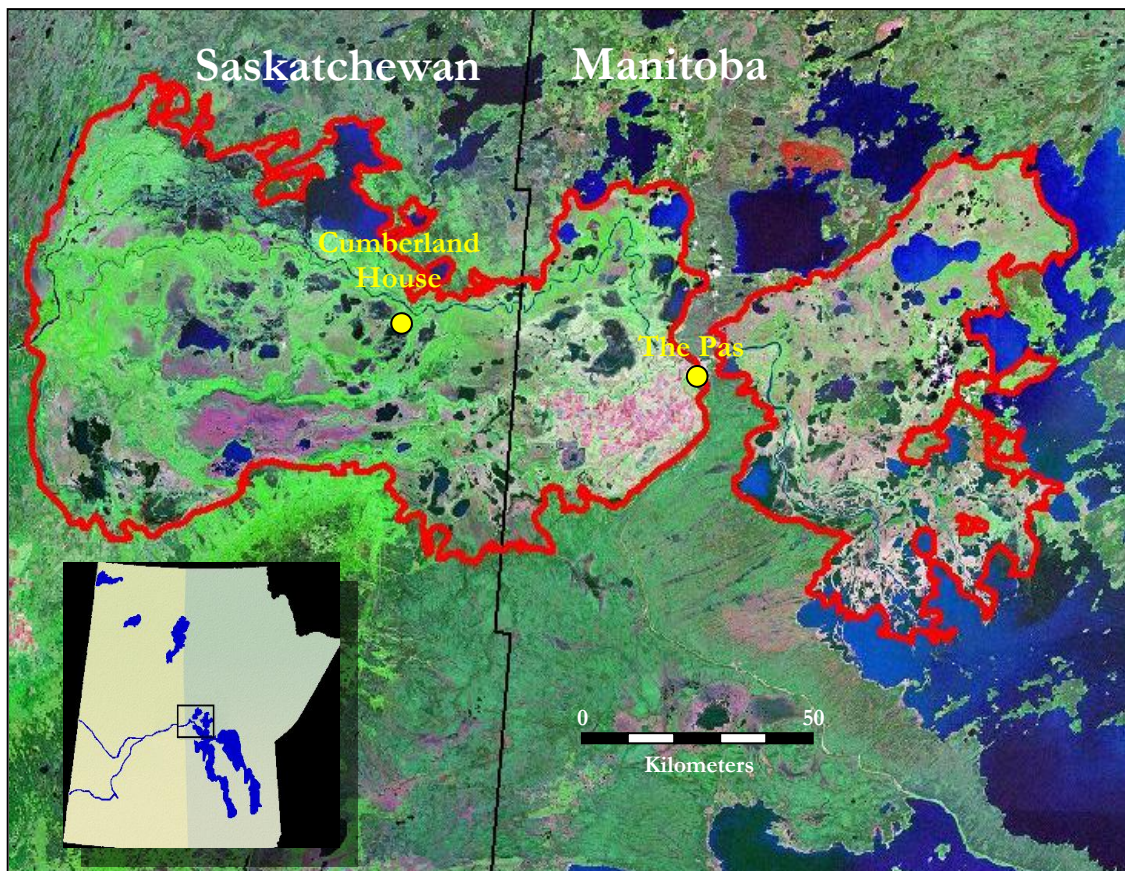


Figure 1.2- Location (inset) and extent of the Saskatchewan River Delta (red), in Saskatchewan and Manitoba (Ducks Unlimited Canada 2006a).

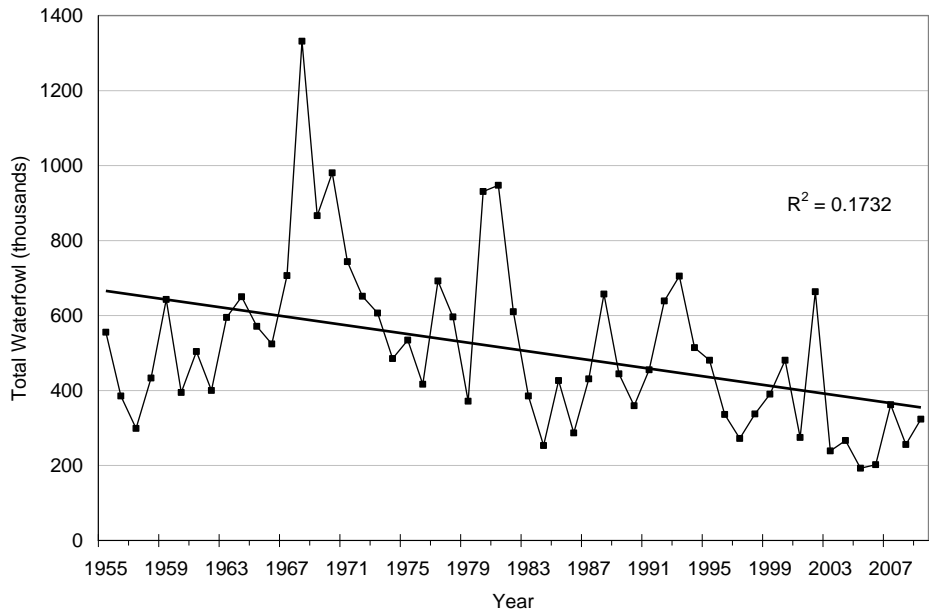


Figure 1.3- Total number of breeding ducks counted in the Saskatchewan River Delta (Stratum 25) from 1955-2009. Trend line indicates an overall decrease in waterfowl numbers (United States Fish and Wildlife Service Migratory Bird Data Center 2009).

1.1.1 Anthropogenic Impacts in the Delta

Historically, the Saskatchewan River commonly had two water-flow peaks, one in early spring from regional runoff, and a second in early summer resulting from snow melt in the Rocky Mountains. These water flow peaks, if sufficiently high, would cause the Saskatchewan River to overbank, flooding the wetlands isolated from the main channels. The periodic flooding or “flood pulse” (Junk et al. 1989) provided by the river is important for recharging wetlands in the SRD and generating a dynamic state needed to keep wetlands productive (Clay 1978; Prowse and Conly 2000). Flooding deposits nutrients and sediments, helps to carry away detritus, and changes water depth in wetlands, influencing the productivity and biota within (Smith 1981; Mitsch and Gosselink 2000). Periodic flooding would also create a diverse range of wetland habitat conditions in the delta. The location, extent, and duration of the flooding would likely

vary according to the amount of water flow and potential ice jams in the river or active, main channels (Prowse and Conly 2000). This variation in flooding could cause spatial variation in wetland habitat by affecting wetlands in parts of the delta and not others, It could also influence habitat diversity by creating new shallow lakes, oxbow lakes, and channels through sediment deposition and river avulsions. Over time, these wetlands may eventually become filled with peat and develop into marshes, fens, and eventually bogs (National Wetlands Working Group 1988).

However, in the last century the SRD has been impacted by upstream and downstream anthropogenic developments, resulting in more stabilized water levels in the Saskatchewan River, and less frequent and less intense flooding of wetlands along the river (Clay 1978; Smith 1981; Partners FOR the Saskatchewan River Basin 2008).

Upstream of the delta, nineteen hydro-electric dams have been constructed on the Saskatchewan River and its tributaries, including the E. B. Campbell dam, which was built less than 100 km upstream of the SRD (Figure 1.4; Partners FOR the Saskatchewan River Basin 2008). The upstream dams have reduced the water flow peaks in the spring and summer by retaining much of the water for sustained power generation throughout the year (Figure 1.5). Water flow in the Saskatchewan River is further reduced by water consumption for uses such as irrigation and petroleum production/refining. This water consumption, measured in The Pas, Manitoba, is approximately 10-20% of the naturalized flow, which is the flow that would have occurred in the absence of these developments (Partners FOR the Saskatchewan River Basin 2008). Prior to these developments overbank flooding of the Saskatchewan River occurred on average once

every 12 years, whereas post-development flooding is estimated to occur on average once every 45 years (KGS Group 2003; Utchmann 1983). It is believed that this altered water regime, caused by upstream hydro-dams and water consumption, has resulted in drier conditions within the SRD (Smith 2003).



Figure 1.4- Contributing and non-contributing drainage areas of the Saskatchewan River Basin with locations of two major hydro-electrical developments affecting the Saskatchewan River Delta (Modified from Partners FOR the Saskatchewan River Basin 2009).

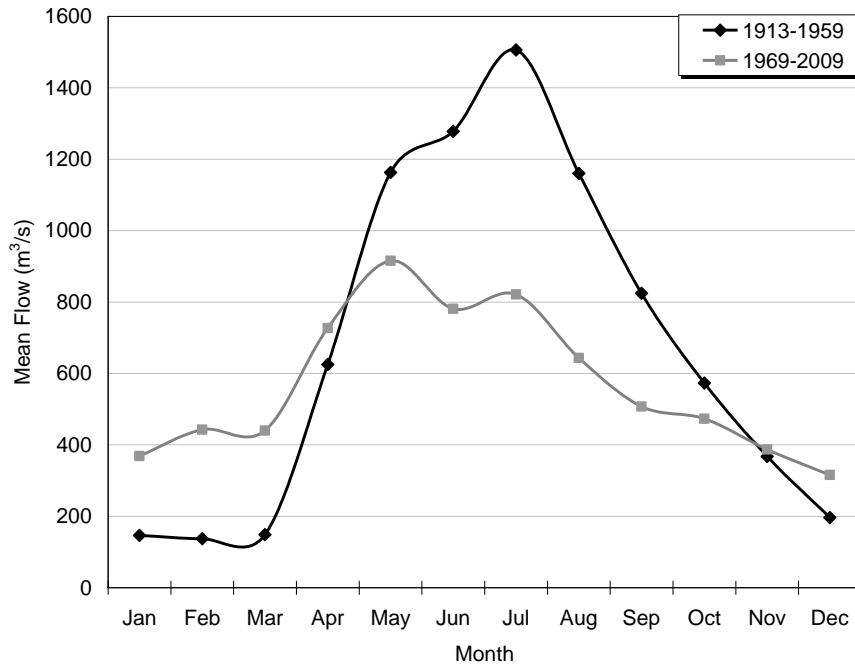


Figure 1.5- Mean monthly flow (m^3/s) of the Saskatchewan River, measured at The Pas, Manitoba prior to hydro-electrical development (1913-1959) and post hydro-electrical development (1969-2009) (Environment Canada Wateroffice 2009).

Downstream of the SRD, the creation of the reservoir for the Grand Rapids Dam permanently flooded about 100,000 ha of wetlands and associated upland habitat in the lower delta of the SRD (United States Fish and Wildlife Service 1961; Uchtmann 1983). The effects of this habitat loss on the avian community are largely unknown. However, it is believed that this was the most productive part of the delta as the Saskatchewan River actively deposited sediment there (Lindgren 2001). Large numbers of Greater White-fronted Geese (*Anser albifrons*) and shorebirds staged on the gravel bars in the lower delta during fall migration. However, due to the destruction of this habitat, these birds have been displaced (Lindgren 2001). The substantial loss of wetland habitat likely had a

negative impact on other avian species in the area. However, due to a lack of avian surveys before and after the flooding, it has not been possible to quantify these impacts.

Additional wetland loss has also occurred in the Carrot River Valley area in the SRD (Figure 1.6). Over 30,000 ha of wetlands were drained for agricultural development, and 7,000 ha of wetlands within the Helldiver complex were deteriorated due to the diversion of the Pasquia River into the Carrot River (Uchtmann 1983). Again, the impacts on the avian community are largely unknown due to a lack of quantitative avian surveys before and after the developments. It is likely that this has had a negative effect on numerous species of wetland dependent birds in the area.

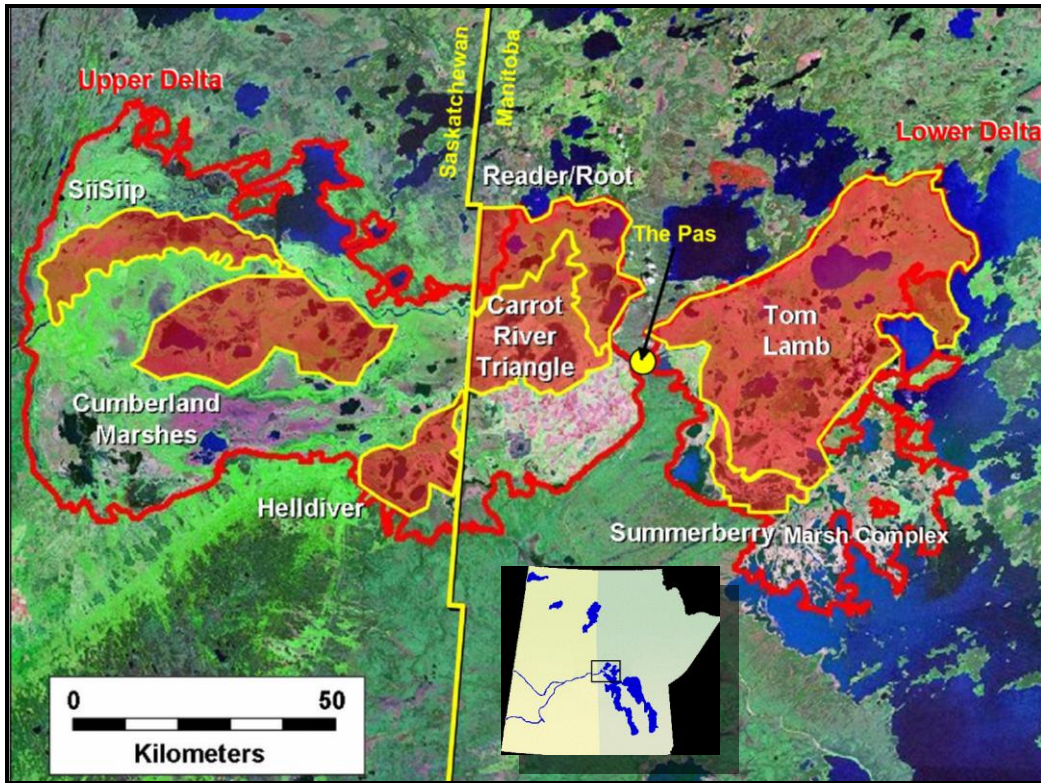


Figure 1.6- Major marsh complexes managed by Ducks Unlimited Canada within the Saskatchewan River Delta (Ducks Unlimited Canada 2006b).

1.2 Historical Water Management

Water-level management in the delta began in the 1930s. The Provincial governments of Manitoba and Saskatchewan, the Federal government, and several private investors, including Tom Lamb and Hudson's Bay Company, attempted to stabilize water levels in SRD wetlands to increase production of muskrats for trapping. The importance of the SRD to waterfowl was recognized by Ducks Unlimited Canada (DUC), which began managing and developing wetlands in the 1940s. Today, DUC operates 111 water control structures, and manages approximately 200,000 ha of wetlands in Manitoba and Saskatchewan (Figure 1.6). In the past, water control structures were used to remove excess water from local precipitation/runoff, and stabilize water levels through the summer to increase waterfowl production. More recently, water control structures have been operated using the drawdown/refill technique, in an attempt to emulate the historical, natural cycle of the wetlands. The drawdown/refill technique is often used in wetland management within the prairie pothole region of North America to develop the hemi-marsh condition favoured by waterfowl (Murkin et al. 1997). However, complications may arise when this technique is applied to wetlands in the SRD. Vegetation responds differently than predicted; drawdowns can produce dense, monotypic stands of emergent, and/or floating mats of vegetation, both of which can persist for long periods and create poor waterfowl habitat (Smith 1986, 1987). Smith (1987) suggested that emergent vegetation diversity is reduced during drawdowns when the substrate is exposed only for short periods or when it is exposed for long periods and allowed to dry excessively. Smith (1987) also suggested that floating vegetation may be a

result of reflooding the basins before vegetation had sufficient time to develop an ample root system to keep it anchored in the substrate.

1.2 Research Requirements

Since the conventional drawdown/refill technique may produce different results in the SRD compared to prairie wetlands, it has not been possible to accurately predict effects of water-level management in the SRD. Through trial and error management, it was learned that complete drawdowns should be avoided, but partial drawdowns might be used to prevent the growth of monotypic communities of vegetation. It was also discovered that vegetation must be given time to develop an ample root system prior to refilling, to prevent floating mats of vegetation from developing (Smith 1987).

The inability to predict the effects of water levels in wetlands of the SRD, and the financial and ecological costs associated with trial and error management, illustrated a need for experimental data examining the effects of commonly used marsh management practices in the SRD, such as water-level management. From these needs the Summerberry Marsh Research Project was developed. This project was to use an experimental approach using multiple, isolated wetlands undergoing different water-level treatments, to examine the effects of water-level management on wetland vegetation and water quality, muskrat population dynamics, and waterfowl and marsh bird populations. This document is limited to the topic of waterfowl and marsh birds; for discussion of wetland vegetation and water quality see Watchorn (2010), and for muskrat population dynamics see Ervin (2010). An experimental approach provided the benefit of having treatments and controls with replication. Replication and controls help to avoid spurious conclusions caused by spatial and temporal differences that may be present in wetlands

due to their dynamic nature and diverse biotic community (Hulbert 1984; Vivian-Smith 1997; but also see Oksanen 2001).

By using an experimental approach to managing water levels in the SRD, it is hoped that the effects of wetland water-level management on waterfowl and marsh birds in the area can be determined. While the effects of water depth on waterfowl are relatively well understood, they warrant examination in the SRD as the area has been shown to be unique. Marsh birds have never been studied in this area, so very little is known about their populations or use of the wetlands in the area.

1.3 Study Site

My study took place during the spring and summers of 2008 and 2009, in the managed wetlands of the Summerberry Marsh Complex (SMC). The SMC is located in the SRD, approximately 25 km southeast of The Pas, Manitoba (Figures 1.6 & 1.7). The marsh complex covers approximately 15,000 ha, primarily between the Summerberry and Saskatchewan Rivers, consisting of a series of permanent marshes and shallow lakes formed by the two bordering rivers (Clay 1978). Ducks Unlimited Canada manages water levels in the marsh using 28 water control structures on individual basins. In addition to the water control structures, DUC also constructed 48 nesting islands and established 18,000 m of deep-water channels in the SMC in an attempt to create better waterfowl habitat and increase productivity. In the past, water levels in the complex could be controlled using two, 10,000 U.S. gallon/minute pumps located at the south end of Ravensnest Lake along the Saskatchewan River. These pumps could move water from Ravensnest Lake or the Saskatchewan River eastward to wetlands in the lower Ravensnest zone, and into the backchannel, where it could be distributed to wetlands in the Central Zone (Figure 1.8). However, due to high operating costs, and regulations introduced in 1976 by the Fisheries Act, specifically Section 30 (Fisheries Act 2010), pumping is no longer an economically feasible method of water-level management. Water control structures in individual basins are now used to drawdown wetlands by releasing water into the Saskatchewan River during low periods, or refilling wetlands by allowing water to enter from the river during high periods or by retaining precipitation.

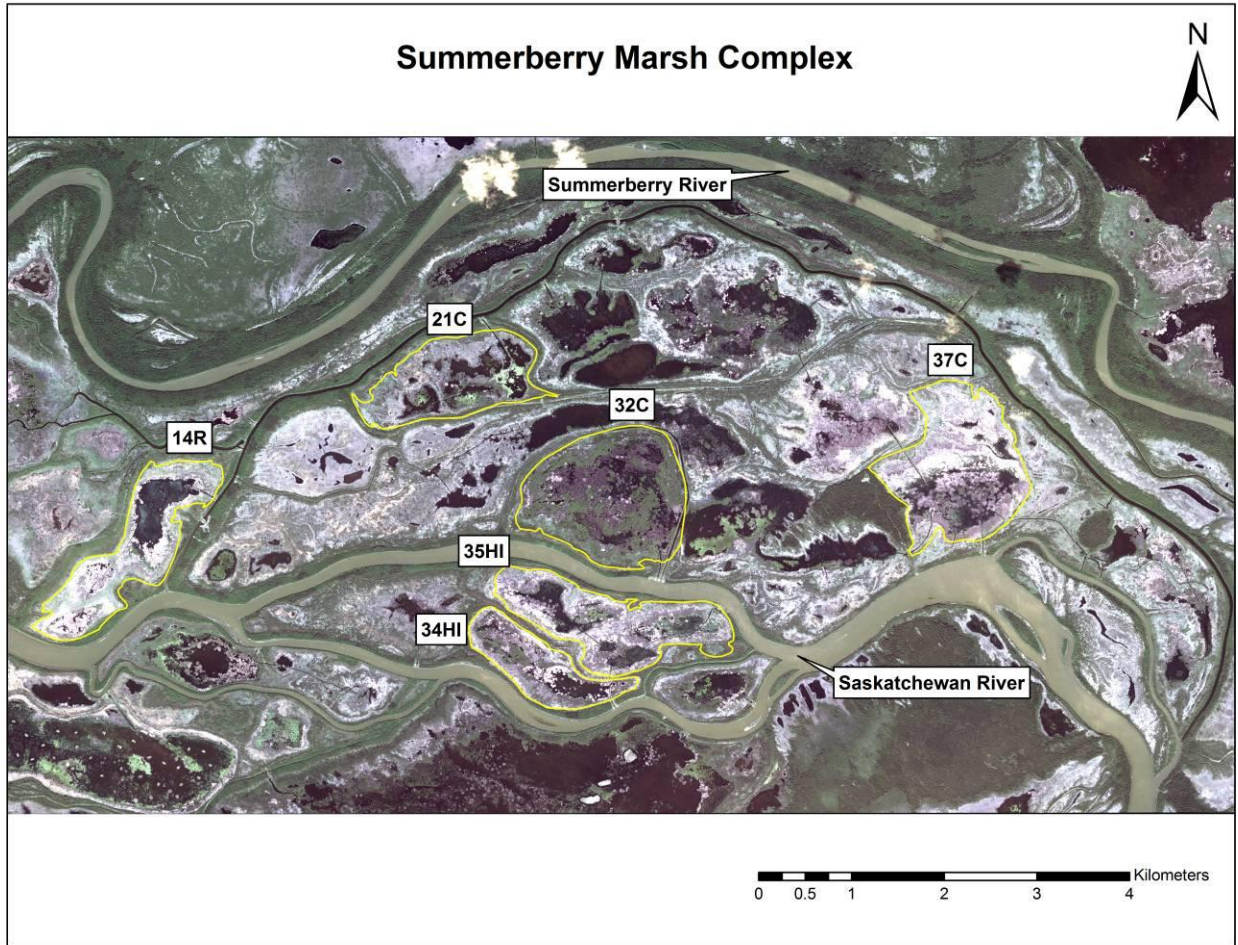


Figure 1.7- Overview of the Summerberry Marsh Complex, Manitoba in 2009. Study wetlands are outlined in yellow.

1.4 Objectives

The objectives of the study were to:

- 1) Determine the densities of different waterfowl species during the breeding, post-breeding, brooding, and staging/migrating life stages to determine what waterfowl species and life stages are supported by the Summerberry Marsh Complex.
- 2) Evaluate effects of the partial water-level drawdown management on waterfowl (Family *Anatidae*) from breeding through fall staging, by modeling the relationship between waterfowl relative abundance and cover and food resources, such as water depth, emergent and submergent vegetation characteristics, and invertebrate abundances.
- 3) Provide wetland managers with information on the species diversity, relative densities, and breeding period chronology of marsh birds in the SRD to direct management decisions.
- 4) Evaluate the effects of the partial water-level drawdown management on marsh birds during the breeding season by modeling the relationship between marsh bird relative abundance and cover and food resources, such as water depth, emergent and submergent vegetation characteristics, and invertebrate abundances.

1.5 Experimental Design

Six wetlands in the SMC were selected for the experimental design: 14R, 21C, 32C, 34HI, 35HI, and 37C (Figures 1.8 & 1.9, Table 1.1). Each chosen wetland basin is equipped with a water control structure (Figure 1.9) that opens into the Saskatchewan River, and water levels can be lowered via gravity flow into the river and can be raised via flow from the river during high water periods or by the retention of precipitation. Three of the wetlands: 14R, 37C, and 35HI were partially drawn down to emulate natural drawdown conditions and initiate a vegetation response. The partial water-level drawdowns were initiated on 10 August, 2007 and were completed by September 2007, water levels were maintained at target levels through 2010 (Table 1.1). Water levels were lowered to approximately 60 cm in the deepest part of the basin, enough to allow under-ice movement by muskrats, but not enough to expose mudflats. The remaining three wetlands: 21C, 32C, and 34HI were managed to maintain their high water levels and act as controls. The water levels in these basins remained at approximately 100 cm in the deepest part of the basins.

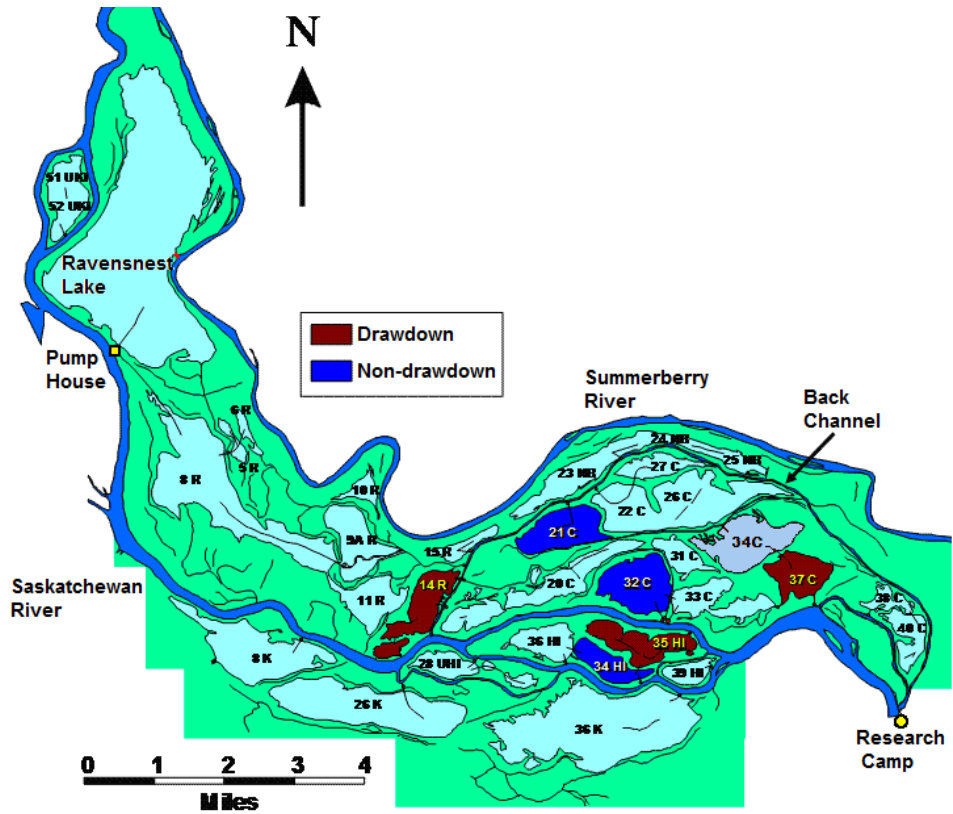


Figure 1.8- Drawdown and non-drawdown (control) wetlands within the Summerberry Marsh Complex in the Saskatchewan River Delta, Manitoba.



Figure 1.9- Example of water control structure placement and design (inset) within wetland 14R in the Summerberry Marsh Complex, Manitoba.

Table 1.1- Characteristics of study wetlands in the Summerberry Marsh Complex, Manitoba.

Wetland	Treatment	Location		Basin Area (ha)	Water Level Prior to Drawdown (fasl)	Target Water Level (fasl)	Avg. Open Water Depth (cm)
		Easting	Northing				
14R	Partial Drawdown	374978	5943434	143	845.32	844.00	46
21C	Non-drawdown	378318	5945213	137	844.00	844.00	81
32C	Non-drawdown	380036	5943918	207	843.00	843.00	99
34HI	Non-drawdown	379310	5942055	84	844.00	844.00	87
35HI	Partial Drawdown	380048	5942502	156	844.23	843.00	63
37C	Partial Drawdown	383848	5944209	195	843.70	842.50	53

1.6 Hypotheses and Predictions

1) Partial water-level drawdowns would allow dabbling species of waterfowl and their broods (Genera *Anas*, *Aix*) to access the wetland bottom more easily while foraging. This would increase the abundance of dabbling waterfowl and broods during the spring-breeding, post-breeding, brooding, and fall-staging periods. Other wetland characteristics, including increasing amounts of water/vegetation interspersion, submergent vegetation biomass, and invertebrate abundances would provide greater amounts of cover and food available to dabblers and their broods and would also increase abundances.

2) Partial water-level drawdowns would reduce the amount of deep-water foraging habitat and escape cover available for diving species of waterfowl and their broods (Genera *Aythya*, *Bucephala*, *Fulica*). This would result in greater abundances of diving waterfowl and broods on non-drawdown wetlands during the spring-breeding, post-breeding, brooding, and fall-staging periods. Non-drawdown wetlands would provide areas of open, deep water that would enable divers and their broods to forage and escape more easily. Other wetland characteristics, including increasing amounts of open water area, submergent vegetation biomass, and invertebrate abundances would provide greater amounts of cover and food available to divers and their broods and would also increase abundances.

3) Partial water-level drawdowns would provide walking species of marsh birds such as the Sora (*Porzana carolina*), Virginia Rail (*Rallus limicola*), and American Bittern (*Botaurus lentiginosus*) with larger areas of emergent vegetation in shallow water, for

breeding, foraging, and nesting. This would increase the abundance of these species in the drawdown wetlands. Other wetland characteristics, including increasing amounts of emergent vegetation and invertebrate abundances would provide greater amounts of cover and food available to walking species of marsh birds and would also increase abundances.

4) Partial water-level drawdowns would reduce the amount of deep-water foraging habitat and escape cover available for diving/swimming species of marsh birds such as the American Coot (*Fulica americana*) and Pied-billed Grebe (*Podilymbus podiceps*). This would result in greater abundances of these species on the non-drawdown wetlands relative to drawn-down wetlands. Other wetland characteristics, including increasing amounts of open water, invertebrate abundances, and forage fish abundance would provide greater amounts of cover and food available to diving/swimming species of marsh birds and would also increase abundances.

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Appendix 1.1- Water-level History of the Study Wetlands in the Summerberry Marsh Complex, Manitoba

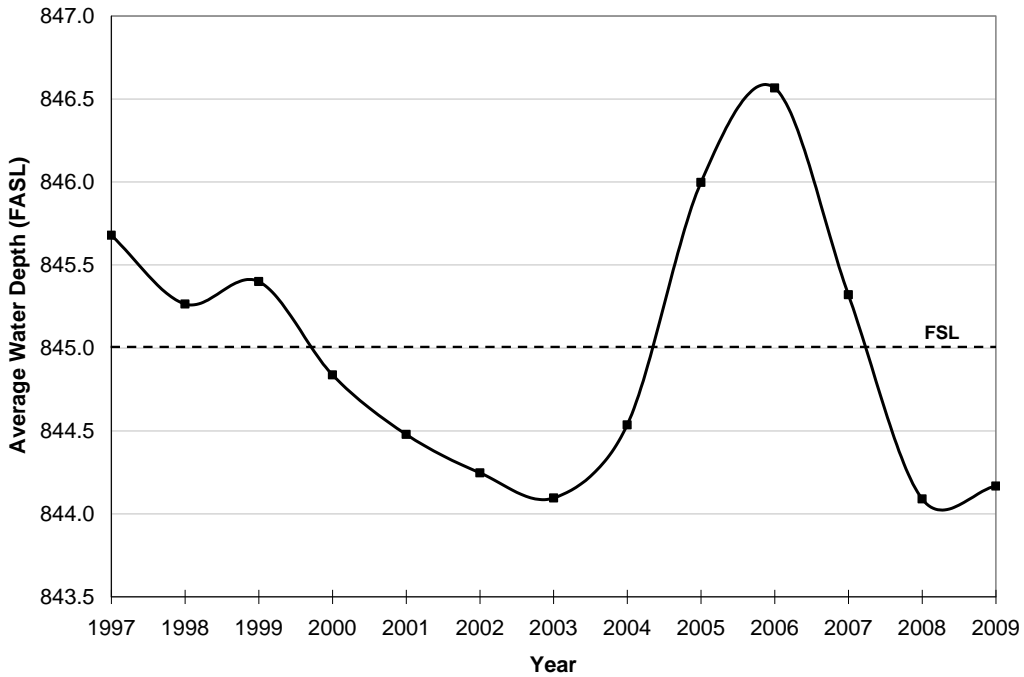


Figure 1.10- Average water depth (feet above sea level) from 1997-2009 for wetland 14R in the Summerberry Marsh Complex, Manitoba.

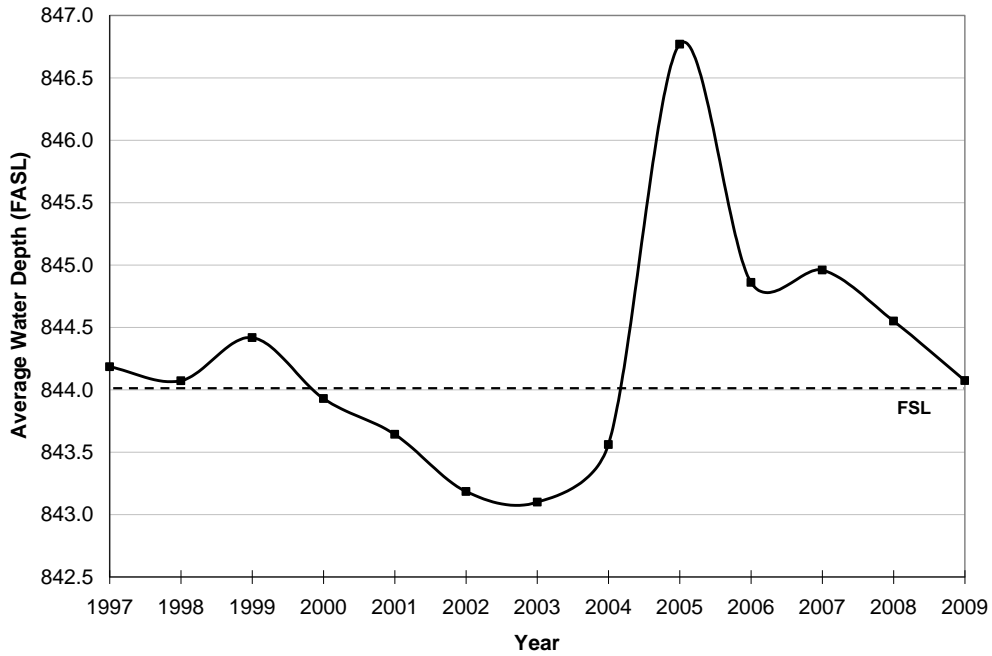


Figure 1.11- Average water depth (feet above sea level) for wetland 21C in the Summerberry Marsh Complex, Manitoba.

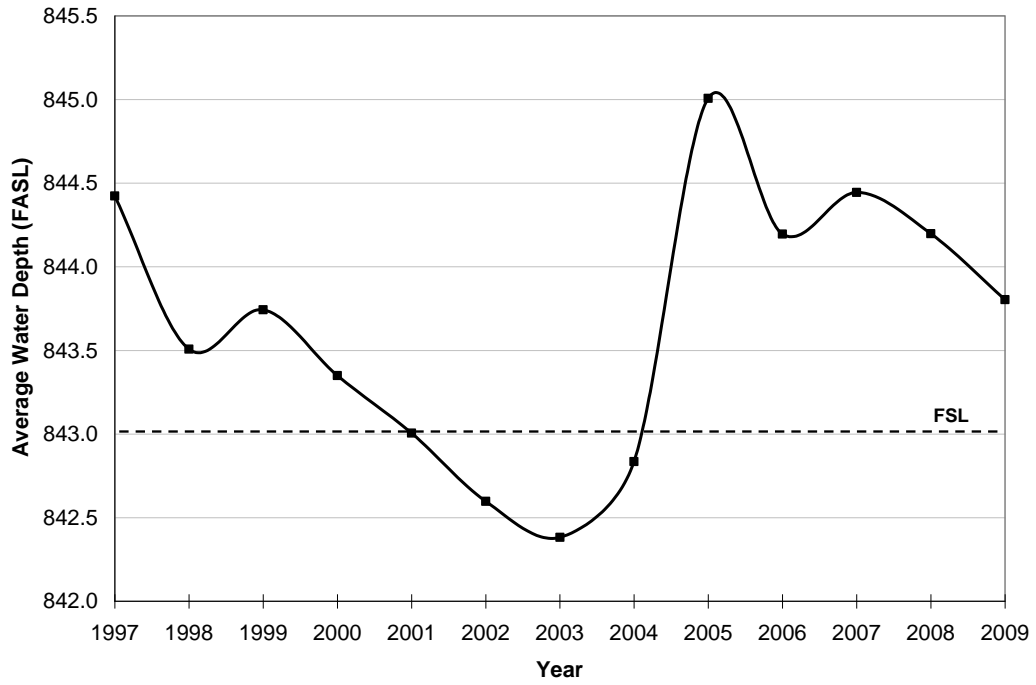


Figure 1.12- Average water depth (feet above sea level) for wetland 32C in the Summerberry Marsh Complex, Manitoba.

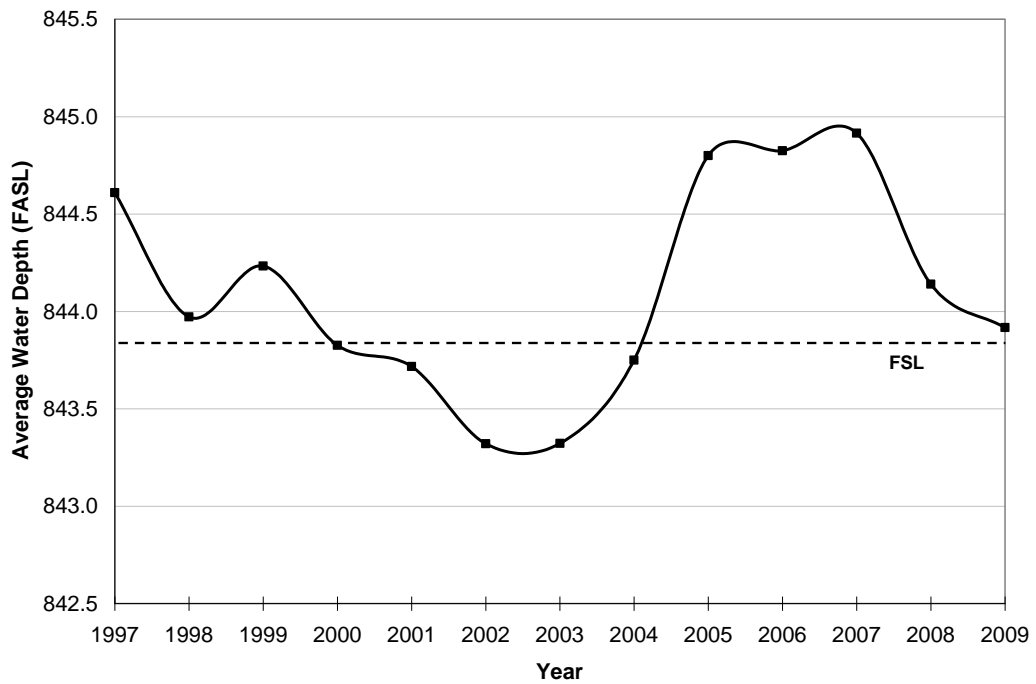


Figure 1.13- Average water depth (feet above sea level) for wetland 34HI in the Summerberry Marsh Complex, Manitoba.

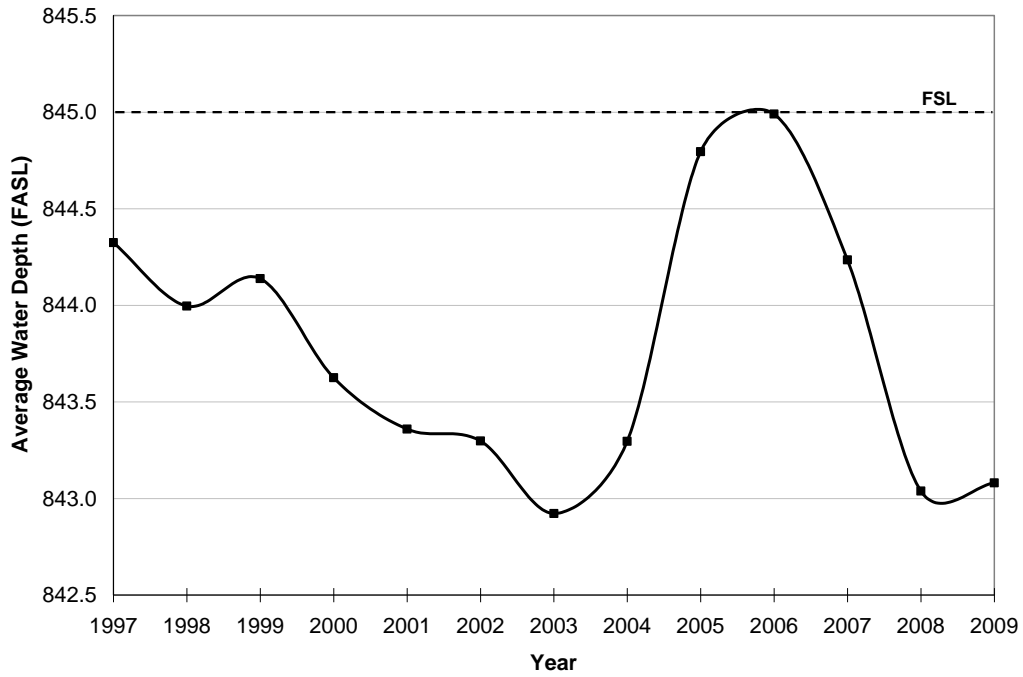


Figure 1.14- Average water depth (feet above sea level) from 1997-2009 for wetland 35HI in the Summerberry Marsh Complex, Manitoba.

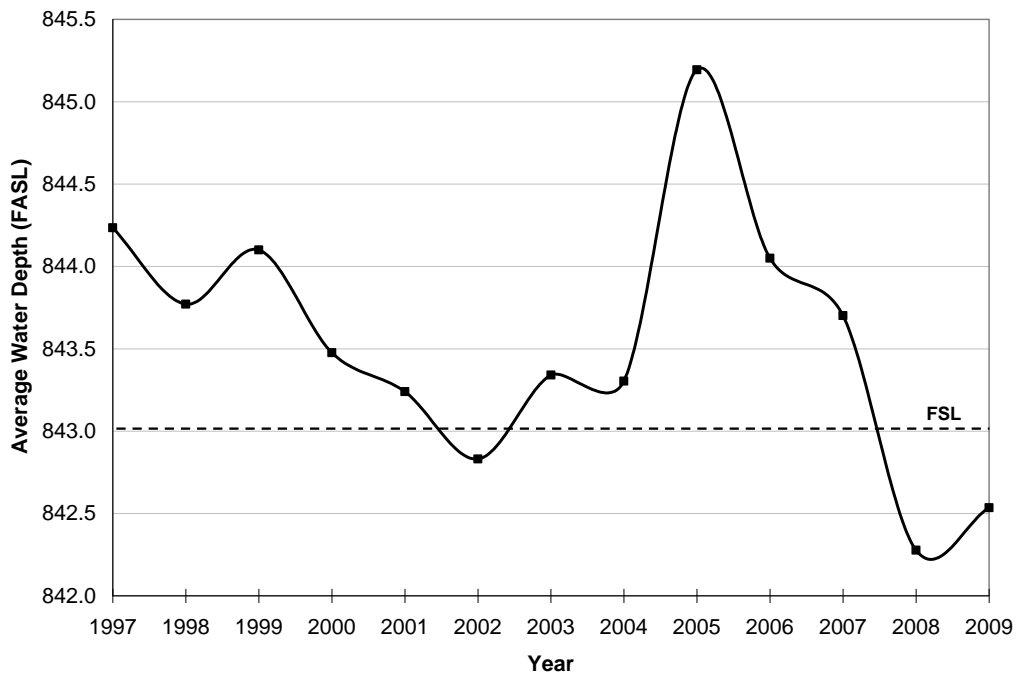


Figure 1.15 Average water depth (feet above sea level) for wetland 37C in the Summerberry Marsh Complex, Manitoba.

Chapter 2 - Effects of a partial water-level drawdown on marsh birds in the Saskatchewan River Delta, Manitoba

2.1 Introduction

The Saskatchewan River Delta (SRD) supports a large diversity of marsh birds that are dependent upon wetlands for all or part of their life cycle. Numerous species of marsh birds such as rails, grebes, and bitterns are known to breed and nest in the area (Lindgren 2001). However, no research on marsh bird populations in the SRD has ever been performed. Many of the species common in the SRD are believed to be declining throughout North America as a result of wetland loss (Eddleman et al. 1988; Conway et al. 1994), but due to their secretive nature and low detectability by traditional bird surveys, such as the Breeding Bird Survey, the extent of this decline is not known. With continued loss of wetland habitat, areas such as the SRD may become increasingly important for breeding marsh birds. But without knowledge of the community of marsh birds in the SRD, it is impossible to quantify the ecological value of this area, or to determine how it should be managed for these species.

Current wetland management in the SRD is focused on waterfowl and muskrat (*Ondatra zibethicus*) production. Management has primarily been focused on controlling wetland water levels in an attempt to create productive habitat for these taxa i.e., diverse vegetation species in a hemi-marsh state (Smith and Jones 1982). Wetland management for waterfowl can create habitat that is also suitable for some species of marsh birds. Johnson and Dinsmore (1986) found that wetland management practices used to promote use by waterfowl also provided suitable habitat for Soras (*Porzana carolina*) and Virginia

Rails (*Rallus limicola*). The hemi-marsh state, preferred by waterfowl and wetland managers, may also benefit the American Bittern (*Botaurus lentiginosus*), as the increased amount of vegetation/water edge may provide more sites for foraging (Rehm and Baldassarre 2007). Other species of marsh birds, such as the American Coot (*Fulica americana*) and Pied-billed Grebe (*Podilymbus podiceps*), prefer wetlands with large expanses of open water for foraging and escape from predators (Muller and Storer 1999; Brisbin and Mowbray 2002), similar to many diving species of waterfowl (Murkin et al. 1997; Murkin et al. 2000). Some work has been done in the SRD examining the effects of wetland management on the abundance of waterfowl, but to the best of my knowledge this study was the first to consider marsh birds in the SRD.

Because water-level manipulation is a commonly applied management technique in the SRD it is important to understand the habitat requirements of marsh birds in the area and the potential effects of water-level manipulations on them. Water depth in a wetland will ultimately determine the habitat available for marsh birds by influencing the vegetation composition for nesting and thermal cover, and food availability and accessibility (Murkin et al. 2000).

Marsh bird use of habitat is strongly influenced by water depth. Morphological adaptations such as leg length, bill length and shape, and webbed feet will determine the depths that can be exploited by species, and reduce interspecific competition through niche separation (Baker 1979; Poysa 1983). Species such as the Sora, Virginia Rail, and Yellow Rail (*Coturnicops noveboracensis*) prefer wetlands with shallower water (<20

cm) because their main means of locomotion is walking, and they forage by gleaning or probing the surface of the water and/or substrate (Zimmerman et al. 2002a, 2002b).

However, these species may use deeper water areas providing there is ample floating, residual or submerged vegetation to allow them walk above the surface of the water (Johnson 1985). Species such as the American Coot or Pied-billed Grebe that forage by diving prefer deeper water as it provides them with greater mobility (Johnson and Dinsmore 1986; Murkin et al. 1997; Murkin et al. 2000).

Water depth is also an important determinant of the emergent vegetation community within a wetland. Murkin et al. (2000) illustrated that changes in water levels can have drastic impacts on the flora of wetlands through the wet-dry cycle. Vegetation structure and species composition affects the types and abundance of cover, which will impact wetland use by marsh birds. For example, smaller rail species require greater thermal protection during the early breeding season (Weller 1999). Therefore, species such as the Sora and Virginia Rail prefer dense emergent vegetation, particularly *Typha* species (Bookhout 1995; Melvin and Gibbs 1996). Dense emergent vegetation also creates structure that allows these species to move over water and access deeper water areas (Johnson 1985). The American Bittern is less of a habitat specialist, but prefers interspersed emergent vegetation as a result of its tendency to forage along water edges (Lor 2007; Rehm and Baldassarre 2007; Lowther et al. 2009). The American Coot and Pied-billed Grebe use a wider variety of wetland types. These species require some dense emergent vegetation, interspersed with water, in which to build their floating-platform

nests, commonly supported by *Typha* spp. and/or *Schoenoplectus* spp. (Muller and Storer 1999; Brisbin and Mowbray 2002).

In addition to affecting available cover resources, water depth also plays an important role in determining food availability. Emergent vegetation species grow along coenoclines, which are determined by water depth and basin topography (Whittaker 1967; Seabloom et al. 1998). These coenoclines will determine the abundance of emergent vegetation species such as *Carex* spp. and *Schoenoplectus* spp.; seeds of these species provide an important food source for the Sora and Yellow Rail (Whittaker 1967; Bookhout 1995; Melvin and Gibbs 1996). Water depth can also affect the amount of light penetration in the water column, influencing submerged aquatic vegetation species richness and density (Anderson 1978; Barko et al. 1986). Submerged aquatic vegetation is an important food source for marsh birds such as the American Coot (Brisbin et al. 2002) and provides valuable habitat for invertebrates (Krull 1970; Longcore et al. 2006). Invertebrates provide an important protein-rich food source for several species marsh birds, including the Sora, Virginia Rail, and Pied-billed Grebe (Horak 1970; Krull 1970; Rundle and Sayre 1983; Muller and Storer 1999; Longcore et al. 2006).

Water depth will influence water temperature and concentrations of dissolved oxygen, which will determine the abundances and diversity of forage fish present in wetlands (Sargent and Galat 2006). Deeper water depths within wetlands may prevent anoxic conditions from developing during the heat of the summer, or ice to form to the wetland substrate in winter, allowing forage fish to survive (Danylchuk and Tonn 2003). forage

fish provide an important food source for several piscivorous marsh bird species, including the American Bittern and Pied-billed Grebe (Brisbin et al. 2002; Lowther et al. 2009). However, large numbers of small fish have been shown to reduce the abundance of invertebrates, directly competing with waterfowl, and presumably marsh birds, for this food source (Bouffard and Hanson 1997; Zimmer et al. 2000; Hornung and Foote 2006).

Because water depth plays such an important role in determining the quality and quantity of marsh bird habitat, I evaluated effects of wetland water-level manipulations on marsh birds in the SRD at multiple scales. Examining relationships of marsh birds to wetland characteristics at multiple scales increased the likelihood of determining scale-dependent patterns that could potentially exist in a highly variable wetland habitat (Weins 1989; Orians and Wittenberger 1991; Paracuellos 2006). This reduced the chance of making erroneous conclusions regarding the water-level treatment in my study. My study focused on the third and fourth levels of avian selection, as defined by Johnson (1980). Johnson (1980) defines the third level as the use of specific habitat within the home range. In the context of my study, this is the selection of the study wetlands by marsh birds. Using this scale allowed me to examine the effects of water-level management on the settling patterns of marsh birds. Understanding the patterns of settlement is important as many species of marsh birds do not leave a selected wetland until brood rearing is complete (Johnson and Dinsmore 1985). The fourth level of selection is defined by Johnson (1980) as the microhabitat selection within the specific habitat. In context of my study this is the establishment of territories within the study wetlands by marsh birds. Since marsh bird breeding territories can be small, 45-55-m² (Kaufmann 1989), examining marsh birds

relationships at this scale allowed me to investigate finer grain wetland characteristics affecting marsh bird use of the study wetlands.

2.2 Objectives

- 1) Provide wetland managers with novel information on the species diversity, relative densities, and breeding period chronology of marsh birds in the SRD to direct management decisions.

- 2) Evaluate the effects of the partial water-level drawdown management on marsh birds during the breeding season by modeling the relationship between marsh bird relative abundance and cover and food resources, such as water depth, emergent and submergent vegetation characteristics, and invertebrate abundances.

2.3 Study Area and Experimental Design

My study took place during 2008 and 2009 in the managed wetlands of Summerberry Marsh Complex (SMC). The SMC is located within the SRD, approximately 25 km southeast of The Pas, Manitoba. A partial water-level drawdown was initiated in three wetlands in 2007 and continued through 2010. Three additional wetlands were consecutively managed with high water levels to act as control in the experiment. See Chapter 1 for a more detailed description.

2.4 Methods

2.4.1 Call-response Surveys

Survey protocol followed that of Conway (2005). In 2008, surveys were conducted at 35 points distributed within the six study wetlands. In 2009, 33 of the original 35 points were surveyed. Two points in a drawdown wetland were relocated due to problems caused by beaver flooding. In addition to the original 33 points, a single point was added to three of the six wetlands, and one wetland received an additional two survey points, totaling 40 points within the six study wetlands.

Survey points were selected randomly by first selecting a single point from all possible locations along the open water/emergent vegetation interface within a wetland using QuickBird satellite imagery. Subsequent survey points, a minimum distance of 400 m away, were mapped along the open water/emergent vegetation interface, and from these one was chosen randomly. This process was repeated until the wetland could not accommodate more survey points. The Universal Transverse Mercator (UTM) coordinates of the survey points were taken from the aerial imagery and entered into a hand-held Geographic Position System (GPS) unit. Most survey points were located along the open water/emergent vegetation interface. However, several of these points were inaccessible prior to the start of surveys and were subsequently relocated along the upland/wetland interface. Two survey points in 2008, and four survey points in 2009, respectively, were located along the upland/wetland interface.

Surveys were conducted from a canoe, with the exception of the points along the upland/wetland interface. Surveys began half-hour before sunrise, and lasted until all points within a wetland had been surveyed, but no later than 1000 hours. A portable audio system and MP3 player were used to broadcast territorial calls of the target species at a volume of 80-90 db measured 1 m from the speakers to elicit vocal responses from target species. Speakers were positioned above the hull of the canoe, and faced the open water of the wetland as per Conway (2005). At points along the upland/wetland interface the speakers were placed on top of a 0.5-m tall, folding stool.

The broadcast sequence consisted of 5 minutes of silence/passive listening followed by 30 seconds of call from one species and 30 seconds of silence, then 30 seconds of call of another species and 30 seconds of silence, repeated until all species' calls had been broadcast. In 2008, the broadcast sequence consisted of territorial calls of the target species in the following order: Least Bittern (*Ixobrychus exilis*), American Bittern, Yellow Rail, Sora, Virginia Rail, and Pied-billed Grebe. In 2009, the broadcast sequence consisted of the territorial calls of target species in the following order: Sora, Virginia Rail, American Bittern, American Coot, and Pied-billed Grebe. The Least Bittern and Yellow Rail were dropped from the call-response sequence in 2009 due to a lack of detections in 2008.

In 2008, one observer recorded bird vocalizations at each point, recording when each individual responded during the survey sequence, species, type of call given, and distance to the source. Recording was limited to the boundary of the individual wetland, estimated

by the tree line. The tree line begins approximately at the Full Supply Level (FSL), which is the highest elevation at which water levels are managed. Distance was divided into six categories: 0-10 m, 10-25 m, 25-50 m, 50-100 m, >100 m, >200 m. Direction of the individual was also recorded to reduce the risk of double counting. In 2009, the same information was recorded using a double-observer approach (Nichols et al. 2000). Two observers sat at opposite ends of the canoe during surveys and independently recorded data.

These surveys were conducted four times in each study wetland from 27 May-2 July, 2008 and four times in each study wetland from 15 May-25 June, 2009, at approximately 10 day intervals. The surveys were not conducted during rain or if wind speed exceeded 20 km/h.

2.4.2 Invertebrate Sampling

In 2009, activity traps and sticky traps were used to sample nektonic and flying invertebrates, respectively, within the study wetlands. Activity traps followed the design of Riley and Bookhout (1990). The top, beveled section of a two litre, clear plastic, pop bottle was removed, inverted, and reattached to form the funnel and the remainder formed the catch portion of the trap. Wire was crossed vertically and horizontally at the narrow opening of the funnel to prevent entry by small, predaceous fish. Activity traps were set in pairs, one mounted horizontally and the other mounted vertically from a “t” frame made of wood that was driven into the substrate (Figure 2.1). Traps were set approximately 30 cm below the surface of the water. Samples from the activity traps

were sorted using 5000, 2000, and 1000- μm size sieves, sorted, and counted to determine abundance of class sizes.

The sticky traps followed the design of King and Wrubleski (1998). A cylinder, 6.25 cm in diameter, and 25 cm long, was wrapped with a transparent, plastic sheet that was coated with Tanglefoot (The Tanglefoot Co. Grand Rapids, MI). This cylinder was placed on the top of the “t” frame that held the activity traps, immediately above the surface of the water (Figure 2.1). After the sampling period, the plastic sheets were removed and the invertebrates stuck to the sheet were counted.

Two “t” frames holding a horizontal and vertical activity trap, and sticky trap, were deployed at random locations within a 200-m radius of each call-response/waterfowl survey point. Traps were placed in the open water areas of the wetlands (>3 m from emergent vegetation) for 24 hours. Sampling was done concurrently in one drawdown and one non-drawdown wetland from 15 May-25 June, 2009. Each wetland was sampled four times. The average number of invertebrates in the nektonic and sticky traps used in the plot-scale and wetland-scale analyses was calculated using the numbers of invertebrates from the first two surveys from the respective traps.

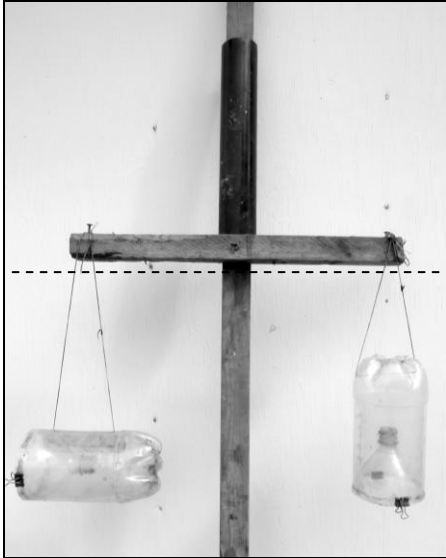


Figure 2.1- “t” frame invertebrate sampling apparatus with sticky trap on top, horizontal activity trap on the left and vertical activity trap on the right. Dashed line indicates the approximate location of the water level when the trap was deployed.

2.4.3 Forage Fish Sampling

In 2008 and 2009 a Beamish trap was used to sample forage fish in the study wetlands. In late August 2008, two drawdown wetlands and two non-drawdown wetlands were sampled using two traps set for 24 hours. One trap was placed randomly in the wetland, and the other was set in the channel connecting the wetland to its control structure on the Saskatchewan River or Back Channel (see Figure 1.7). Two wetlands 35HI, a drawdown wetland, and 34HI, a non-drawdown wetland were not sampled in 2008 due to a lack of time. Data were substituted for these wetlands with data from 2009 as the numbers of fish caught did not significantly differ between 2008 and 2009 in the other wetlands, according to a paired *t*-test ($p = 0.19$). In 2009, sampling was conducted in late August through September. The trapping effort and placement of traps differed in 2009 compared to 2008. Each wetland was sampled using two traps that were set for two 24-hour

periods. One trap was placed randomly in the open water of the wetland, and the other was placed randomly near vegetation. In both years, captured fish were identified to species [Brook Stickleback (*Culaea inconstans*), Central Mudminnow (*Umbra limi*), Fathead Minnow (*Pimephales promelas*), and Northern Pike (*Esox lucius*)], counted, and released back into the wetland. Fish relative abundance was estimated using the index of number of fish caught per trap hour, not including the Northern Pike, to account for differing sampling efforts among wetlands and years.

2.4.4 Submerged Aquatic Vegetation Sampling

Submerged aquatic vegetation (SAV) was sampled in open water at three random clip-plots taken within a 200-m radius of each call-response/waterfowl survey point.

Consequently, the number of samples taken depended upon the number of call-response/waterfowl survey points in the wetland. The bottom of a 55 gallon plastic drum was removed to form 0.25-m² cylinder. This cylinder was pushed into the substrate at the sampling point and a small rake was used to remove above-ground vegetation from within the cylinder.

The vegetation was dried at approximately 80°C for 48 hours, and weighed to determine biomass. Biomass was averaged at each point for the plot-scale analyses using the data from the three random clip-plots taken within the 200-m radius of each call-response/waterfowl survey point. For the wetland-scale analyses, data from all clip-plots in the respective wetland were used. Samples were taken on 4 and 5 August, 2009 when the standing biomass was approximately at its maximum.

2.4.5 Emergent Vegetation Mapping

Emergent vegetation maps were created for each study wetland for 2008 and 2009. The purpose of these maps was to delineate and quantify the major cover types within each wetland and at each call-response point. Cover types identified by these maps included Sparganium (*Sparganium* spp.), Carex (*Carex* spp.), Equisetum (*Equisetum* spp.), Scirpus (*Schoenoplectus tabernaemontani*), Typha (*Typha* spp.), Phragmites (*Phragmites australis*), Water (open water), and Trees (dominated by *Salix* spp., but also included *Populus* spp., *Acer negundo*, *Fraxinus pennsylvannica*, and other species). For a detailed explanation of this procedure see Appendix 4.0.

2.5 Data Analysis

2.5.1 Observer Detection Probability

Prior to data analysis, observer detection probabilities were calculated for the 2009 call-response data to determine if the observed number of birds required adjustment due to observer bias. The program DOBSERV (Hines 2000) was used to estimate detection probabilities for observers and each species. Detections from all four surveys, and individual birds that were detected at multiple points during a survey were included in the analysis. To determine if both observers detected the same bird, the species, distance, time of first detection, and call type of each detection was compared from the recorded data of the two observers. Data were summarized by calculating the numbers of each target species at each point detected by the primary observer, secondary observer, and those not detected by the primary observer but by the secondary observer. Since the independent observer method was used, the primary and secondary observer roles were determined *a posteriori* by randomly picking a primary observer for the first point and alternating roles at subsequent points.

I used program DOBSERV with the independent observer setting and examined each target species as an individual group (Hines 2000). See Nichols et al. (2000) for a detailed explanation of DOBSERV program statistics.

The DOBSERV program generates detection probability estimates under several models (Table 2.1). The best model was selected by comparing Akaike's Information Criterion (AIC) values and selecting the model with the lowest value. Detection probabilities of

observers were calculated using this model. I also included the results of the P(s,.) model to calculate the detection probabilities of individual species to compare these probabilities among species.

Table 2.1- Model definitions for the DOBSERV program (Hines 2000).

Model	Detection Probability (p)
P (.,.)	Same for all species and both observers
P (s,.)	Different for each species but equal among observers
P (.,i)	Equal among species but different between observers
P (g,.)	Equal within groups of species and equal among observers
P (g,i)	Equal within groups of species but different between observers
P (s,i)	Different for each species and different between observers

2.5.2 Analysis of Density and Chronology

The peak of the marsh bird breeding season was identified by comparing the total number of detections in each survey round. The first two surveys in 2008 and 2009 contained the greatest number of detections and are hereafter referred to as the “peak”. Subsequent analyses of the call-response survey data were performed on the peak number of detections for 2008 and 2009.

The findings of Hay (2006), who performed call-response surveys in wetlands across southern Manitoba for breeding marsh birds in 2003 and 2004, were included to act as reference data for the densities and chronology of breeding marsh birds in the SMC. The average numbers of birds per point were calculated from Hay (2006) using the total number of detections from 2003 and 2004 and dividing it by the total number of sites surveyed in 2003 and 2004 during the peak of the breeding season, 3 May-12 June, 2003

and 2004. A statistical comparison of my findings with Hay (2006) was not performed due to a lack of detailed avian survey data in Hay (2006).

I also included abundance estimates of marsh birds from several wetland complexes located in North America as reference data for the SMC. There is a lack of published data for marsh bird abundance estimates, particularly for Canada. Therefore, the nearest wetlands with available marsh bird data included for reference were located within Minnesota State, USA. Data for these wetland complexes, excluding the Summerberry Marsh Complex, were modified from Conway (2005). The average number of detections per point per year for each of these wetlands, including the Summerberry Marsh, was calculated by dividing the total number of birds detected in each wetland by the number of years surveyed, then the number of replicates in a year, and finally the average number of points in the survey between years. The call-response surveys used in all wetland complexes consistently followed those described in Conway (2005). Despite similar methods, caution should be used when comparing these results to those of the Summerberry Marsh Complex, Manitoba due to disparities in timing, location, habitat structure, climate, and absence of statistical tests etc. Statistical tests were not performed on these data due to a lack of detailed results in Conway (2005).

2.5.3 Habitat Selection - Wetland-scale Analyses

To examine the effects of the partial water-level drawdowns and other habitat characteristics on the settling patterns of marsh birds, habitat selection models were constructed at the wetland scale.

Using ArcMap 9.1 the FSL of each wetland was used to clip the wetland vegetation map for 2008 and 2009. The Patch Analyst 4.2 extension in ArcMap 9.1 was used to measure various vegetation characteristics within the wetlands, including areas of vegetation and water patches. Average open water depth in the wetlands was calculated by averaging all water depths taken during ground-truthing in open water areas (see Appendix 4.0 for detailed methods on water depth measurement).

To obtain a more accurate representation of the numbers of marsh birds present within the wetland, all detections made within the FSL of the wetland, were included in the analyses, and repeat detections were eliminated from the analyses. To standardize the numbers of birds in the various sized wetlands, the numbers of birds detected during the peak were divided by the area of the wetland's FSL to obtain an estimate of birds/ha. .

The GENMOD procedure in SAS 9.2 was used to model the density (birds/ha) of each species. Models were fit using either the negative binomial or Poisson distribution, depending on which fit best, with a log-link function. Importance of independent variables in the models was assessed using hypothesis testing with an $\alpha = 0.1$. If models were overdispersed (Pearson's chi-square/df > 1.67) or underdispersed (Pearson's chi-square/df < 0.95) the "pscale" option was used to multiply the covariance matrix by the Pearson's chi-square/df value to account for the dispersion. Due to relatively small number of wetlands in the study the degrees of freedom available for the analyses was small. Therefore, each variable listed below was run as a separate model in an attempt to

explain the densities of different marsh bird species relative to each of these potentially biologically independent mechanisms. The models/variables used in the analyses were:

1) Average open water depth. Water depth is known to affect marsh bird use of wetlands, and is determined by morphology and feeding habits. Species that forage by diving, such as the Pied-billed Grebe and American Coot, show preference for deeper water wetlands (Krapu et al. 1970; Otto 1983; Muller and Storer 1999; Murkin et al. 2000; Brisbin et al. 2002), whereas species such as the Sora and Virginia Rail prefer shallower water wetlands as their main method of locomotion is walking (Sayre and Rundle 1984; Johnson and Dinsmore 1986; Conway 1995; Melvin and Gibbs 1996).

2) Area of open water. Open water provides cover for species such as the Pied-billed Grebe and American Coot that dive when threatened. However, it may have negative effects on walking species such as the Sora and Virginia Rail as it reduces the amount of habitat available to them (Johnson and Dinsmore 1986; Muller and Storer 1999; Murkin et al. 2000; Brisbin et al. 2002).

3) Area of emergent vegetation. This included the areas of different vegetation classes excluding Trees, calculated from the vegetation maps (Appendix 4.0). Emergent vegetation provides thermal and nesting cover for marsh birds and increased amounts of emergent vegetation with wetlands has been shown to be positively related to several marsh bird species such as the American Bittern, Sora, and Virginia Rail (Bookhout 1995; Melvin and Gibbs 1996; Lowther et al. 2009).

4) Forage fish abundance. Fish are an important food source for piscivorous species such as the America Bittern and Pied-billed Grebe (Muller and Storer 1999; Lowther et al. 2009). Fish abundance also negatively affects invertebrate abundances (Bouffard and Hanson 1997; Zimmer et al. 2000; Hornung and Foote 2006), which are a major food source for several marsh bird species (Horak 1970; Conway 1995; Melvin and Gibbs 1996; Muller and Storer 1999; Lowther et al. 2009).

5) Invertebrate abundance from activity and sticky traps (2009 only). Invertebrates may be an important food source for gleaning species such as the Sora and Virginia Rail (Horak 1970; Conway 1995; Melvin and Gibbs 1996). Larger invertebrates may also be an important food source for species such as the Pied-billed Grebe and American Bittern (Muller and Storer 1999; Lowther et al. 2009).

2.5.4 Habitat Selection - Plot-scale Analyses

To examine the effects of the partial water-level drawdowns and other habitat characteristics on microhabitat selection and territory establishment of the target species of marsh bird , multivariate statistical models were also constructed using plot-scale habitat variables.

Using ArcMap 9.1 the locations of the call-response points for 2008 and 2009 were added to the respective wetland vegetation maps. Plots were created by establishing a 200-m radius, circular buffer around each call-response point, and extracting the 200-m

radius plots from the wetland vegetation maps. The FSL boundary of each wetland was overlaid and the buffered points were clipped to remove areas outside the boundary of the wetland, making plots variable in size. The Patch Analyst 4.2 extension in ArcMap 9.1 was used to measure vegetation characteristics within the plots, including areas of vegetation and water, numbers of vegetation and water patches, and the lengths of vegetation and water edge. To measure the correct amount of edge, excluding the arbitrary edge of 200 m boundary, the polygons were converted to polylines using ET Geo Wizard 9.9 extension in ArcMap 9.1 and clipped with a 199.9-m radius buffer to remove the outer edge. The lengths of the polylines were then used as a measure of vegetation and water edge within the plots. Vegetation and water edge density, calculated as the length of edge in kilometres per hectare, were also determined from these measurements. Average water depth around each point was calculated by averaging all water depths taken during ground-truthing, in all vegetation types and open water areas, inside the 200 m buffers (see Appendix 4.0 for detailed methods on water depth measurement).

To standardize the numbers of birds in the various sized plots, the numbers of birds detected during the peak were divided by the size of the respective plot. Only initial detections within 200 m of the point were used; repeat detections and detections further than 200 m were eliminated from the analyses.

The distribution of the data was examined using Q-Q plots and the GENMOD procedure in SAS 9.2. The data did not follow any of the common distributions (normal, Poisson,

negative binomial), so were changed to a binary format, indicating presence or absence at each point. The GLIMMIX procedure in SAS 9.2 was then used to model the binary data. I included “Wetland” as a random variable to account for possible variation in plots among wetlands. Models were fit using a binomial distribution and a log-link function. Importance of independent variables in the models were assessed using hypothesis testing with an $\alpha = 0.1$. I used combinations of variables from the vegetation mapping, invertebrate, submergent vegetation, and fish sampling to develop biologically justifiable models *a priori* in an attempt to explain the presence of different marsh bird species. The models used in the analyses were:

1) “Treatment”. This model included average water depth within the plots. Water depth is known to affect marsh bird use of wetlands, as determined by morphology and feeding habits. Species that forage by diving, such as the Pied-billed Grebe and American Coot, prefer deeper wetlands (Krapu et al. 1970; Otto 1983; Muller and Storer 1999; Murkin et al. 2000; Brisbin et al. 2002), whereas species such as the Sora and Virginia Rail prefer shallower wetlands as their main method of locomotion is walking (Sayre and Rundle 1984; Johnson and Dinsmore 1986; Conway 1995; Melvin and Gibbs 1996).

2) “Cover”. This model included measurements from the wetland vegetation maps. Variables included water edge density, as a measure of vegetation/water interspersion, and area of open water. Higher amounts of interspersion will result in increased water edge densities. Interspersion has been shown to be important for marsh bird species; it may increase invertebrate abundances, an important food source, and provides cover for

several species of marsh birds (Voigts 1976; Kaminski and Prince 1981; Johnson and Dinsmore 1986; Conway 1995; Brisbin et al. 2002; Rehm and Baldassarre 2007). Areas of open water provide cover for species that dive when threatened, such as the Pied-billed Grebe and American Coot. However, it may have negative effects on walking species such as the Sora and Virginia Rail as it reduces the amount of habitat available to them (Johnson and Dinsmore 1986; Muller and Storer 1999; Murkin et al. 2000; Brisbin et al. 2002).

3) “Veg. Composition”. This model included the areas of different vegetation classes, excluding Trees, derived from the vegetation maps. Different vegetation species offer different cover types (*Schoenoplectus* vs. *Phragmites*), nesting cover (*Schoenoplectus* for overwater nesting vs. *Carex* for upland nesting), food (seeds of different species), and residual vegetation (*Typha* vs. *Schoenoplectus*). Many species of marsh birds show preferences for different species of emergent vegetation (Bookhout 1995; Melvin and Gibbs 1996; Lowther et al. 2009).

4) “Food”. This model was only performed with the 2009 data as habitat data were not collected in 2008. The model included invertebrate abundances from activity and sticky traps and submerged aquatic vegetation biomass. Invertebrates are an important food source for gleaning species such as the Sora and Virginia Rail (Horak 1970; Conway 1995; Melvin and Gibbs 1996). Larger invertebrates may be an important food source for species such as the Pied-billed Grebe and American Bittern (Muller and Storer 1999; Lowther et al. 2009). Submergent vegetation may be an important food source for species

such as the American Coot (Brisbin 2002), and has been shown to affect invertebrate abundance (Krull 1970; Voigts 1976; Zimmer et al. 2000).

2.6 Results

2.6.1 Observer Detection Probability

The models P (.,.) and P (g,.) were identical for predicting the detection probabilities for the 2009 call-response data. Both suggest that detection probability is equal among species and among observers (Table 2.2).

Table 2.2- Detection probability models created by the DOBSERV program for the 2009 call-response data from the Summerberry Marsh Complex, Manitoba, 2009.

Model	Likelihood	DF	AIC	G-O-F	AICc	QAIC	QAICc
P (.,.)	-23.11	11	48.226	0.9809	48.24	48.23	48.24
P (g,.)	-23.11	11	48.226	0.9809	48.24	48.23	48.24
P (.,i)	-23.11	10	50.220	0.9649	50.25	50.22	50.25
P (g,i)	-23.11	10	50.220	0.9649	50.25	50.22	50.25
P (s,.)	-22.05	7	54.095	0.9844	54.25	54.09	54.25
P (s,i)	-21.33	1	62.657	1.000	63.22	62.66	63.22

The models indicated that the probability of one of the two observers detecting a bird was greater than 94%. This resulted in only a small change from the number of individuals detected ($X_{..}$) to the adjusted number of individuals (N) (Table 2.3).

Table 2.3- Estimates of N using the P (.,.) model for the 2009 call-response data from the Summerberry Marsh Complex, Manitoba.

Species	$X_{..}$	p	$SE(p)$	N	$SE(N)$	Lower 95% CI	Upper 95% CI
Sora	128	0.9403	0.0092	136.12	3.23	131.84	145.19
Virginia Rail	61	0.9403	0.0092	64.87	2.13	62.41	71.59
American Bittern	32	0.9403	0.0092	34.03	1.51	32.55	39.44
American Coot	66	0.9403	0.0092	70.19	2.22	67.58	77.10
Pied-billed Grebe	118	0.9403	0.0092	125.49	3.08	121.45	134.23

$X_{..}$ – Observed number of individuals.

p – Detection probability.

N – Estimated number of individuals ($N = X_{..}/p$)

The detection probability of individual species was calculated using the P (s,.) model.

The American Bittern, American Coot, and Virginia Rail had the lowest detection probabilities around 92%. The Sora and Pied-billed Grebe had higher detection probabilities at 95% and 94%, respectively (Table 2.4).

Due to the high probability of detection no corrections were made to bird numbers prior to data analysis.

Table 2.4- Estimates of detection probability for individual species using the P (s,.) model for the 2009 call-response data from the Summerberry Marsh Complex, Manitoba.

Species	X..	<i>p</i>	SE(<i>p</i>)	<i>N</i>	SE(<i>N</i>)	Lower 95% CI	Upper 95% CI
Sora	128	0.9545	0.0133	134.10	3.14	130.36	143.78
Virginia Rail	61	0.9266	0.0277	65.83	3.01	62.57	75.86
American Bittern	32	0.9216	0.0402	34.72	2.29	32.65	43.43
American Coot	66	0.9207	0.0283	71.68	3.32	67.97	82.43
Pied-billed Grebe	118	0.9445	0.0161	124.93	3.45	120.76	135.41

X.. – Observed number of individuals.

p – Detection probability.

N – Estimated number of individuals ($N = X../p$)

2.6.2 Chronology

In 2008 the highest number of detections occurred during the first survey round, 27 May-2 June (Figure 2.2). The number of detections for each species gradually declined as the spring progressed, with the exception of the Virginia Rail, which had a relatively constant number of detections throughout the spring. No Least Bitterns were detected during the surveys. A single Yellow Rail was detected in drawdown wetland 14R during the third survey round and two other occurrences of this species were noted in the area in 2008, but not detected during the surveys.

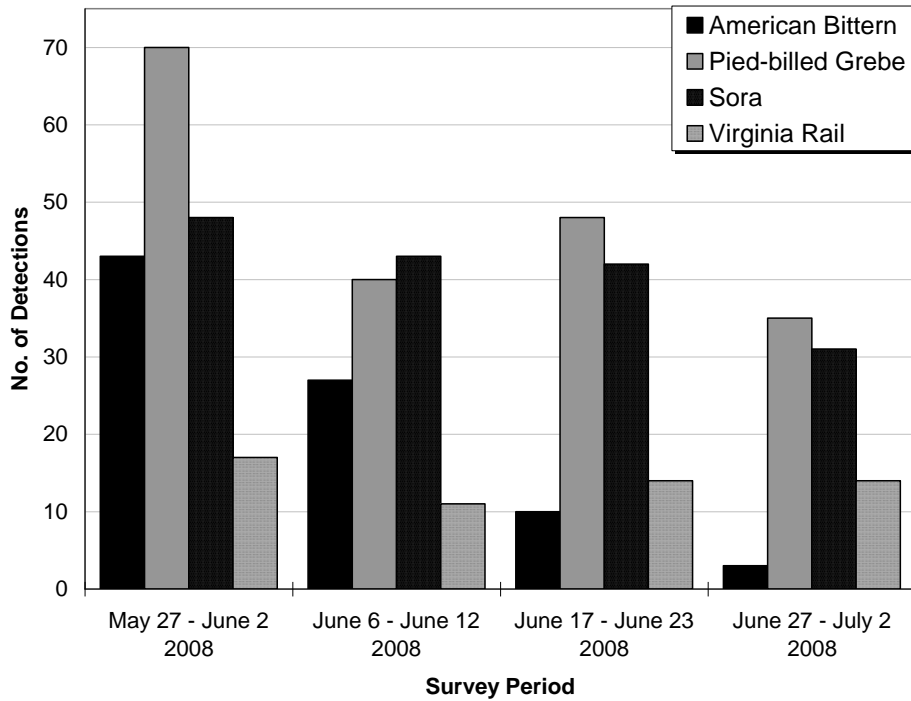


Figure 2.2- Number of target species detected in each survey period in the Summerberry Marsh Complex, Manitoba, 2008. Note: Least Bittern and Yellow Rail not included due to absence and very low abundance, respectively.

In 2009 the highest number of detections occurred during the first survey round, 15 May-23 May (Figure 2.3). The number of detections for each species declined steadily as the spring progressed, with the exception of the Virginia Rail, which had a relatively constant number of detections throughout the spring.

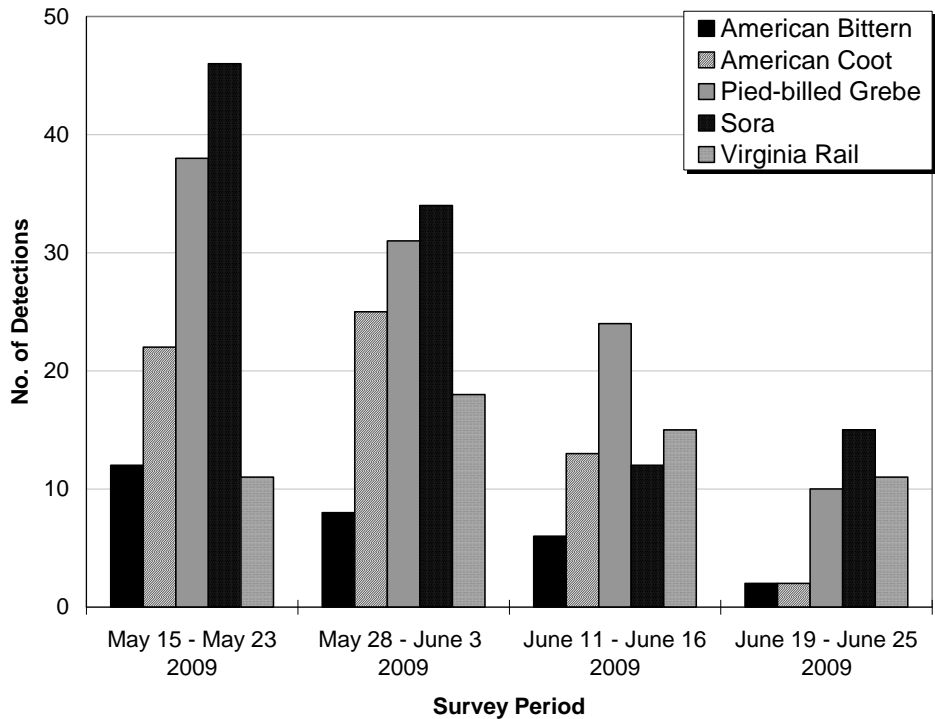


Figure 2.3- Number of target species detected in each survey period in the Summerberry Marsh Complex, Manitoba, 2009.

2.6.3 Densities

In 2008 and 2009 the densities of the Pied-billed Grebe, Sora, and Virginia Rail were higher in the study wetlands of the SMC compared to densities in wetlands in southern Manitoba (Hay 2006) (Tables 2.5 & 2.6). The densities of American Bitterns were also higher in the SMC in 2008 compared to Hay (2006), but were lower in 2009 (Tables 2.5 & 2.6).

The average number of Pied-billed Grebes, Soras, and Virginia Rails detected per point per year were higher in the SMC compared to other areas in Alberta, Canada and Minnesota State, USA (Table 2.7). The average number of American Bitterns and American Coots in the SMC were second highest, being below the numbers in Agassiz

National Wildlife Refuge, Minnesota and Alberta BCR11, Alberta, respectively (Table 2.7).

Table 2.5- Number of birds per point during the peak of the breeding season in the Summerberry Marsh Complex, Manitoba, 27 May-12 June 2008 and the average number of birds detected by Hay (2006) during the peak of the breeding season across southern Manitoba, 3 May-18 June, 2003-04. Standard deviation could not be calculated for the Hay (2006) data due to the lack of detailed survey data.

	Avg. All Wetlands	Std. Dev	Avg. Drawdown	Std. Dev	Avg. Non-drawdown	Std. Dev	Avg. Hay (2006)
American Bittern	0.96	0.50	0.79	0.11	1.13	0.72	0.54
Pied-billed Grebe	1.55	1.06	1.04	0.69	2.07	1.25	0.43
Sora	1.27	0.57	1.03	0.63	1.50	0.50	1.16
Virginia Rail	0.40	0.26	0.34	0.23	0.46	0.33	0.18
Yellow Rail	0.00	0.00	0.00	0.00	0.00	0.00	NA
Least Bittern	0.00	0.00	0.00	0.00	0.00	0.00	0.07

Table 2.6- Number of birds per point during the peak of the breeding season in the Summerberry Marsh Complex, Manitoba, 15 May-3 June 2009 and the average number of birds detected by Hay (2006) during the peak of the breeding season across southern Manitoba, 3 May-18 June, 2003-04. Standard deviation could not be calculated for the Hay (2006) data due to the lack of detailed survey data.

	Avg./Point	Std. Dev	Avg. Drawdown	Std. Dev	Avg. Non-drawdown	Std. Dev	Avg. Hay (2006)
American Bittern	0.27	0.41	0.07	0.07	0.46	0.54	0.54
American Coot	0.61	0.74	0.22	0.10	1.00	0.96	NA
Pied-billed Grebe	0.92	1.18	0.32	0.38	1.52	1.49	0.43
Sora	0.98	0.42	1.14	0.25	0.81	0.54	1.16
Virginia Rail	0.35	0.28	0.37	0.24	0.32	0.36	0.18

Table 2.7- Average number of birds detected per point per year in the Summerberry Marsh Complex, Manitoba and other wetland complexes throughout North America. Data for the Summerberry Marsh Complex obtained from this study and the remaining data was modified from Conway (2005).

Name	Location	Year(s) Surveyed	Avg. No. Replicates /Year	Avg. No. Survey Pts/Year	American Bittern	American Coot	Pied-billed Grebe	Sora	Virginia Rail
Summerberry Marsh Complex	Central Manitoba	2008 2009	4	38	0.37	0.41	0.97	0.89	0.37
Alberta BCR11	Central Alberta	2002	2	320	0.00	0.57	0.01	0.25	0.00
Agassiz National Wildlife Refuge	Northwest Minnesota	2000 2001 2002	3	44	0.54	0.11	0.61	0.70	0.25
Litchfield Wetland Management District	Southcentral Minnesota	2002 2003	3	31	0.04	0.00	0.56	0.40	0.21
Prairie Potholes Region	Across Minnesota	2002 2003	3	132	0.08	0.00	0.10	0.36	0.15
Red Lake Band of Chippewa Indians	Northcentral Minnesota	2002 2003	3	36	0.30	0.38	0.45	0.50	0.12

2.6.4 Habitat Selection Models - Wetland-scale Results

The American Bittern, American Coot, and Pied-billed Grebe preferred deeper water and settled in greater abundances in the non-drawdown wetlands (Table 2.8). This relationship was strongest for the American Bittern and Pied-billed Grebe in 2009. In 2009, the model predicted that a 10 cm increase in water depth would result in a 0.2 bird/ha increase of the American Bittern and Pied-billed Grebe during a single survey (Appendix 2.1, Tables 2.11 & 2.14). Sora and Virginia Rail abundance did not appear to be affected by the partial water-level drawdowns (Table 2.8).

The area of open water in the wetlands had variable effects among species and between years. American Bitterns were the only species that appeared to prefer wetlands with greater areas of open water (Table 2.8). However, this relationship only occurred in 2008 and was quite weak, suggesting an increase of approximately 0.02 bitterns/ha for every

hectare increase in open water (Appendix 2.1, Table 2.10). American Coot, Pied-billed Grebe, and Virginia Rail abundances were all negatively affected by open water area (Table 2.8). These relationships were also all quite weak, suggesting a decrease of approximately 0.05 birds/ha for every hectare increase in open water (Appendix 2.1, Tables 2.12, 2.14, 2.17).

Increased vegetation area within the wetlands decreased the abundances of Sora and Virginia Rail in 2008 (Table 2.8). Again, the strength of this relationship was low, suggesting a decrease of approximately 0.005 Soras/ha and 0.01 Virginia Rails/ha, during a single survey, with every hectare increase in vegetation (Appendix 2.1, Tables 2.15 & 2.17).

The abundance of forage fish increased abundances of piscivorous species, including the American Bittern and Pied-billed Grebe, but decreased the abundance of the Sora (Table 2.8). For every 10 fish caught per hour there was an increase of approximately 0.3 bitterns and grebes per hectare and a decrease of about 0.2 Soras per hectare (Appendix 2.1, Tables 2.11, 2.14, 2.16).

Conversely, Sora abundance increased in areas with high abundances of nektonic and flying invertebrates. Virginia Rail abundance was also higher within wetlands supporting high abundances of flying invertebrates (Table 2.8).

Table 2.8- Summary of wetland-scale analyses indicating the significant variables in each of the statistical models of breeding marsh birds in the Summerberry Marsh Complex, Manitoba, 2008-09.

Species	Variables				2009 Only	
	Avg. Water Depth (cm)	Water Area (ha)	Vegetation Area (ha)	No. Fish/ Trap Hour	Avg. Inverts./ Activity Trap	Avg. Inverts./ Sticky Trap
American Bittern	08/09 (+)	08 (+)	---	09 (+)	--	--
Pied-billed Grebe	08/09 (+)	09 (-)	---	09 (+)	--	--
Sora	--	--	08 (-)	09 (-)	09 (+)	09 (+)
Virginia Rail	--	08 (-)	08 (-)	--	--	09 (+)
American Coot (2009 only)	09 (+)	09 (-)	--	--	--	--

08: Variable significant in 2008.

09: Variable significant in 2009.

08/09: Variable significant in 2008 and 2009.

(+): Significant variable produces positive parameter estimate.

(-): Significant variable produces negative parameter estimate.

-- : Not significant in 2008 or 2009.

2.6.5 Habitat Selection Models - Plot-scale Results

The “Treatment” model indicated the presence of the American Bittern, American Coot, and Pied-billed Grebe were all positively related to increasing water depth within the wetlands. The presence of Sora and Virginia Rail did not appear to be affected by water depth in this study (Table 2.9). The relationship to water depth was strongest for the Pied-billed Grebe; probability of occurrence of the Pied-billed Grebe increased per survey by approximately 0.4 with a 10 cm increase in water depth (Appendix 2.2, Tables 2.22 & 2.23).

The “Cover” model indicated that the presence of the American Bittern, American Coot, and Pied-billed Grebe were higher in areas within the wetlands with a greater amount of

interspersion. Again, the relationship was strongest for the Pied-billed Grebe (Table 2.9). The probability of occurrence of the Pied-billed Grebe increased by approximately 6.0 for every km/ha increase of interspersion (Appendix 2.2, Tables 2.22 & 2.23). The Virginia Rail was the only species negatively related to water area. This relationship was quite strong, suggesting a 0.5 decrease in probability of occurrence for every hectare increase in water area (Appendix 2.2, Table 2.26). The Sora was not affected by any cover characteristics (Table 2.9).

The “Veg. Composition” models showed variable responses to vegetation species between years. American Bittern presence increased in areas with *Carex* and *Scirpus*, but only in 2008. However, bitterns also appeared to avoid areas with *Equisetum* (Table 2.9). The relationship between bittern presence and *Equisetum* was strong, indicating a probability of occurrence decrease of approximately 2.0 in one survey for every hectare increase of *Equisetum* (Appendix 2.2, Tables 2.19 & 2.20). Areas of *Scirpus* also appeared to increase the presence of the Pied-billed, but only in 2009, and areas of *Typha* appeared to reduce the presence of the American Coot. Vegetation species composition did not appear to affect the presence of the Sora or Virginia Rail (Table 2.9).

The “Food” model indicated that the Virginia Rail was the only species influenced by invertebrate abundances. Virginia Rails were present in areas with greater amounts of nektonic invertebrates. The biomass of SAV or the abundance of flying invertebrates did not affect the presence of any marsh bird species in this study (Table 2.9).

Table 2.9- Summary of plot-scale analyses indicating the significant variables in each of the statistical models of breeding marsh birds in the Summerberry Marsh Complex, Manitoba, 2008-09.

Species	Treatment	Cover		Veg. Composition						Food (2009 only)		
	Average Water Depth (cm)	Water Area (ha)	Water Edge Density (km/ha)	<i>Sparganium</i> Area (ha)	<i>Carex</i> Area (ha)	<i>Equisetum</i> Area (ha)	<i>Phragmites</i> Area (ha)	<i>Scirpus</i> Area (ha)	<i>Typha</i> Area (ha)	Avg. SAV Biomass/Pt (g)	Avg. Inverts/Sticky Trap	Avg. Inverts/Activity Trap
American Bittern	08/09 (+)	--	08/09 (+)	--	08 (+)	08 (-)	--	08 (+)	--	--	--	--
Pied-billed Grebe	08/09 (+)	--	08/09 (+)	--	--	--	--	09 (+)	--	--	--	--
Sora	--	--	--	--	--	--	--	--	--	--	--	--
Virginia Rail	--	08 (-)	--	--	--	--	--	--	--	--	--	09 (+)
American Coot (2009 only)	09 (+)	--	09 (+)	--	--	--	--	--	09 (-)	--	--	--

08: Variable significant in 2008.

09: Variable significant in 2009.

08/09: Variable significant in 2008 and 2009.

(+): Significant variable produces positive parameter estimate.

(-): Significant variable produces negative parameter estimate.

-- : Not significant in 2008 or 2009.

2.7 Discussion

2.7.1 Chronology

The peak of the breeding season for the target species of marsh birds in the SMC appears to occur in late-May to early-June (Figures 2.2 & 2.3). I found the number of detections for all species were greatest during the first two survey periods: 27 May-12 June, 2008 and 15 May-3 June, 2009. This appears to be consistent with the findings of Hay (2006) who performed call-response surveys for marsh birds across southern Manitoba. Hay (2006) found the greatest number of detections for the American Bittern, Least Bittern, Pied-billed Grebe, and Sora, from 6 May-18 June, 2003-04. However, the time frame during which Hay (2006) observed the peak of vocalizations is quite large. Therefore, any subtle differences in marsh bird breeding chronology caused by difference in latitude between these studies may be masked.

In the SMC, vocalizations decreased steadily for most species as spring progressed with the exception of the Virginia Rail. This finding is consistent with Hay (2006) and Johnson and Dinsmore (1986) who also found a constant number of detections of Virginia Rails as spring progressed. These findings suggest that to obtain the peak number of breeding marsh birds in the SMC or SRD, future studies should commence as soon as wetlands can be accessed. Ideally, this would occur in the beginning of May, but this depends on the weather and timing of ice breakup on the Saskatchewan River.

2.7.2 Densities

The relatively high densities of marsh bird species compared to wetlands in southern Manitoba and elsewhere in North America suggest that the wetlands in the SRD provide important breeding habitat. Most species were common, with relatively comparable densities to other managed wetlands. Unfortunately, due to a lack of published data, particularly in boreal regions, no statistical comparisons of marsh bird densities could be made between wetland complexes. Therefore, conclusions from these data should be made with caution due to differences in habitat, latitude, etc. There appears to be some difference between the SMC and wetlands in southern Manitoba.. The Least Bittern was not detected in the SMC but it was detected at several locations in southern Manitoba (Hay 2006). The Least Bittern is not known to occur in the SRD, but does breed in the extreme southern part of Manitoba (Cornell Lab of Ornithology 2009a).

The Yellow Rail was not detected by Hay (2006), but we did have a single detection of a Yellow Rail during one of our surveys in 2008 and on two other occasions not during call-response surveys. The Yellow Rail is known to breed across Manitoba (Cornell Lab of Ornithology 2009b). Therefore, the low number of individuals detected within our study area may be due to our surveys being performed in the morning, whereas males tend to call primarily at night (Bookhout 1995). If the aim of future studies is to obtain accurate population estimates of the Yellow Rail, then night surveys should be performed.

2.7.3 Habitat Selection by Marsh Birds

Overall, the partial water-level drawdowns in the study wetlands did not benefit any species of marsh bird examined at either scale. The American Bittern, American Coot, and Pied-billed Grebe all preferred deeper water and had higher densities within the non-drawdown wetlands. Soras and Virginia Rails did not appear to be affected by the partial water-level drawdowns.

The preference for deeper water by the American Coot and Pied-billed Grebe was expected as these species require deep water to allow mobility during foraging and escape (Krapu et al. 1970; Otto 1983; Muller and Storer 1999; Murkin et al. 2000; Brisbin et al. 2002). The preference for deeper water by the American Bittern was not expected. I hypothesized that partial water-level drawdowns would create more areas of emergent vegetation in shallow water that would provide the American Bittern with more cover and structure for overwater movement. However, it appeared that American Bitterns preferred the deeper wetlands due to the relatively high abundance of small fish in one of the non-drawdown wetlands, 32C. Small fish provide an important food source for bitterns (Lowther et al. 2009) and likely influenced the high abundance of American Bitterns in wetland 32C. Similarly, the abundance of the Pied-billed Grebe was also highest in this wetland, likely as a result of its piscivory (Muller and Storer 1999). Water depth did not physically limit the use of wetlands by American Bitterns, and likely played an important role in determining the abundance of forage fish. Deeper water in wetland 32C may have provided favourable habitat for small fish, including sufficient dissolved oxygen, and prevention of freeze-out during winter (Danylchuk and Tonn

2003). As a result, it may have increased the abundances of the American Bittern and Pied-billed Grebe by providing them with greater prey abundance.

Alternatively, wetlands with higher forage fish abundance were avoided by insectivorous species, including the Sora. In other studies, fish have been shown to directly compete with waterfowl for invertebrates as food, reducing wetland use by waterfowl (Bouffard and Hanson 1997; Hornung and Foote 2006). In my study, Soras, similar to waterfowl, may have avoided wetlands with high fish abundance due to reduced invertebrate prey availability and increased competition.

Invertebrate abundances appeared to influence the presence and abundance of the Sora and Virginia Rail. Invertebrates are known to be an important food source for Soras and Virginia Rails, particularly during the spring when protein needs are high as a result of egg laying (Horak 1970; Rundle and Sayre 1983). Therefore, wetland selection by Soras and Virginia Rails may be more closely tied to invertebrate prey availability rather than other habitat characteristics. Because invertebrates can be highly variable, spatially and temporally (King and Wrubleski 1998), Soras and Virginia Rails may have used a broad range of vegetation and structure types, influencing their appearance as “habitat generalists” in this study and potentially other studies (Johnson and Dinsmore 1986; Hay 2006).

In addition to prey abundance, vegetation structure and composition within the wetlands also appeared to influence several species of marsh birds. The influence of open water

area was variable for several species at the two spatial scales. American Coots, Pied-billed Grebes, and Virginia Rails selected wetlands with less open water, whereas the American Bittern selected wetlands with more open water. Within the wetlands, at the plot scale, only the Virginia Rail was impacted by water area, appearing to prefer more vegetated areas. These results were expected for the Virginia Rail and American Bittern, but not for the American Coot and Pied-billed Grebe. The Virginia Rail is known to prefer dense, emergent vegetation that provides concealment and structure for overwater movement (Johnson 1985; Conway 1995) and the American Bittern has been shown to prefer foraging sites along water openings, which are related to open water area (Lor 2007). However, the American Coot and Pied-billed Grebe are swimming/diving species of marsh birds that typically prefer large expanses of open water for foraging (Desrochers and Ankney 1986; Muller and Storer 1999; Brisbin et al. 2002; Rehm and Baldassarre 2007). Therefore, American Coots and Pied-billed Grebes would be expected to select wetlands with greater amounts of open water. The disparity of my findings from other studies may be due to grebes and coots selecting areas within the wetlands containing greater vegetation interspersions rather than open water. The presence of both the American Coot and Pied-billed Grebe increased at sites with greater interspersions, but were unaffected by water area. American Coots and Pied-billed Grebes are known to nest along emergent vegetation edges in close proximity to open water (Gullion 1954; Desrochers and Ankney 1986; Muller and Storer 1999; Brisbin et al. 2002). Increased interspersions would provide coots with a greater amount of vegetation/water edge for nest sites. Therefore, it appears that interspersions within the wetlands may play a more important role in determining the presence of the American Coot and Pied-billed Grebe

in the SMC compared to water area. Interspersion may reduce interspecific competition for nesting sites, resulting in increased presence and densities of the species.

Interspersion also affected the presence of the American Bittern. Bitterns commonly forage along the vegetation/water interface, concealing themselves in the vegetation and ambushing passing prey in the open water (Lor 2007; Rehm and Baldassarre 2007; Lowther et al. 2009) Increased interspersion created greater amounts of vegetation/water edge, increasing the number of foraging sites available, reducing interspecific competition for these sites, and allowing birds to diffuse throughout the wetland. Soras and Virginia Rails were not affected by interspersion. This is in contrast to Conway (1995) who suggests Virginia Rails may use interspersion as a cue to select wetlands with high abundances of invertebrates.

Vegetation composition within the wetlands influenced the presence of several marsh bird species. *Scirpus (Schoenoplectus)* appeared to be favoured by American Bitterns and Pied-billed Grebes. This species of emergent vegetation is known to occur in open water at deeper water depths compared to other species (Welling et al. 1988; van der Valk et al. 2000). This provides bitterns with increased amounts of vegetation/water edge from which to forage, and provides grebes with overwater nesting habitat that is sufficiently deep to allow them to escape off their nests.

Bittern presence was also greater at sites with increased amounts of *Carex* spp.. The concealment offered by *Carex* spp. and the presence of small mammal prey, such as

Meadow Voles (*Microtus pennsylvanicus*), which are common in upland habitat in wetlands (Harris 1953), likely contributed to this relationship. In addition to providing concealment during foraging, *Carex* spp. also provides bitterns with dry, upland habitat for nesting (Duebbert and Lokemon 1977; Lowther et al. 2009). Conversely, *Equisetum* spp., which also occur in more upland areas of wetlands, were avoided by bitterns. *Equisetum* did not occur in the same densities as *Carex* (personal observation) and may have been avoided because it did not offer sufficient cover for nesting or foraging

2.8 Conclusions

The call-response surveys were effective for surveying most marsh bird species in the SMC. I did not detect any Least Bitterns, and encountered only a single Yellow Rail. This was likely due to surveying out of the Least Bittern's typical range, and Yellow Rails' tendency to call at night, whereas our surveys took place in the morning. Other marsh bird species, including American Bitterns, American Coots, Pied-billed Grebes, Soras, and Virginia Rails had relatively high densities in the SMC, which appears to provide important breeding habitat.

The partial water-level drawdowns did not benefit any species of marsh bird examined. American Bitterns, American Coots, and Pied-billed Grebe all showed a preference for deeper water at both the wetland and plot-scale. Soras and Virginia Rails did not appear to be influenced by water depth. The findings for the walking species of marsh birds, including American Bitterns, Soras, and Virginia Rails were not expected. I hypothesized these species would benefit from the partial water-level drawdowns as it would increase the amount of shallow flooded emergent vegetation used for breeding, nesting, and foraging. However, American Bitterns preferred the deeper water of the non-drawdown wetlands likely due to the high abundance of forage fish, particularly in wetland 32C. Both Soras and Virginia Rails did not show a preference for any water depth, but were positively related to high abundances of flying and nektonic invertebrates, as predicted. Subsequently, Soras were negatively influenced by forage fish abundance, potentially due to prey competition, which was not expected.

The partial water-level drawdowns did reduce the use by diving species of marsh birds, including American Coots and Pied-billed Grebes as predicted. Both species preferred the deep water in the non-drawdown wetlands likely due to the greater mobility for diving. Pied-billed Grebes also appeared to benefit from the high abundance of forage fish within the non-drawdown wetlands, but were not affected by invertebrate abundances. The preference for open water by diving species was also not supported by my findings. American Coots and Pied-billed Grebes were positively related to vegetation interspersion, particularly that offered by *Schoenoplectus*. This may be a result of reducing interspecific competition for nesting sites.

Due to the wide variety of marsh bird species in the SMC, a diverse range of wetland habitats are required to support them. I suggest this be accomplished by active water-level management that staggers water levels within the SMC (see Chapter 4 for details).

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Appendix 2.1- Wetland-scale Results of Habitat Models for 2008 and 2009

Table 2.10- Wetland-scale habitat model results for the relative density of the American Bittern in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

American Bittern						Wald 95% Confidence Limits		Wald Chi-Square	<i>p</i>
Models/Variables	Pearson chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower		
Average Open Water Depth (cm)	1.63	Negative Binomial	1	0.0119	0.0062	-0.0003	0.0241	3.66	0.0559
Water Area (ha)	1.61	Negative Binomial	1	0.0411	0.0248	-0.0074	0.0897	2.76	0.0968
No. Fish/ Trap Hour	0.96	Poisson	1	0.0215	0.0141	-0.0060	0.0491	2.35	0.1252
Vegetation Area (ha)	1.47	Negative Binomial	1	0.0019	0.0042	-0.0063	0.0101	0.21	0.6464

Table 2.11- Wetland-scale habitat model results for the relative density of the American Bittern in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

American Bittern						Wald 95% Confidence Limits		Wald Chi-Square	<i>p</i>
Models/Variables	Pearson's chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower		
Average Water Depth (cm)	0.93	Poisson	1	0.0490	0.0187	0.0123	0.0857	6.84	0.0089
Water Area (ha)	1.53	Negative Binomial	1	-0.0378	0.0641	-0.1635	0.0879	0.35	0.5552
No. Fish/Trap Hour	0.68	Poisson	1	0.0634	0.0177	0.0287	0.0982	12.79	0.0003
Vegetation Area (ha)	1.11	Negative Binomial	1	0.0128	0.0102	-0.0072	0.0328	1.57	0.2100
Avg. Inverts./ Activity Trap	1.24	Negative Binomial	1	-0.0743	0.0841	-0.2391	0.0906	0.78	0.3771
Avg. Inverts./ Sticky Trap	1.60	Negative Binomial	1	0.0033	0.0650	-0.1242	0.1307	0.00	0.9597

Table 2.12- Wetland-scale habitat model results for the relative density of the American Coot in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

American Coot						Wald 95% Confidence Limits		Wald Chi-Square	<i>p</i>
Models/Variables	Pearson's chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower		
Average Water Depth (cm)	0.83	Negative Binomial	1	0.0327	0.0185	-0.0035	0.0689	3.14	0.0763
Water Area (ha)	1.57	Negative Binomial	1	-0.1263	0.0500	-0.2243	-0.0282	6.37	0.0116
No. Fish/ Trap Hour	1.82	Negative Binomial	1	0.0501	0.0507	-0.0494	0.1495	0.97	0.3235
Vegetation Area (ha)	1.26	Negative Binomial	1	-0.0025	0.0101	-0.0223	0.0173	0.06	0.8048
Avg. Inverts./ Activity Trap	1.45	Negative Binomial	1	0.0459	0.0683	-0.0879	0.1797	0.45	0.5014
Avg. Inverts. / Sticky Trap	1.94	Negative Binomial	1	0.1036	0.0823	-0.0576	0.2648	1.59	0.2080

Table 2.13- Wetland-scale habitat model results for the relative density of the Pied-billed Grebe in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

Pied-billed Grebe						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.35	Poisson	1	0.0242	0.0067	0.0111	0.0374	13.13	0.0003
Water Area (ha)	1.22	Negative Binomial	1	0.0061	0.0291	-0.0509	0.0632	0.04	0.8335
No. Fish/Trap Hour	1.38	Negative Binomial	1	0.0278	0.0181	-0.0076	0.0632	2.37	0.1235
Vegetation Area (ha)	1.26	Negative Binomial	1	-0.0003	0.006	-0.0120	0.0114	0	0.9632

Table 2.14- Wetland-scale habitat model results for the relative density of the Pied-billed Grebe in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Pied-billed Grebe						Wald 95% Confidence Limits			
Models/Variables	Pearson's chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Water Depth (cm)	1.31	Negative Binomial	1	0.0404	0.0142	0.0125	0.0683	8.08	0.0045
Water Area (ha)	1.56	Negative Binomial	1	-0.1004	0.0560	-0.2102	0.0094	3.21	0.0732
No. Fish/ Trap Hour	1.67	Negative Binomial	1	0.0560	0.0271	0.0028	0.1092	4.26	0.0390
Vegetation Area (ha)	1.36	Negative Binomial	1	0.0007	0.0094	-0.0177	0.0190	0.01	0.9416
Avg. Inverts./ Activity Trap	1.46	Negative Binomial	1	0.0098	0.0627	-0.1132	0.1328	0.02	0.8757
Avg. Inverts./ Sticky Trap	1.64	Negative Binomial	1	0.0563	0.0594	-0.0602	0.1727	0.90	0.3435

Table 2.15- Wetland-scale habitat model results for the relative density of the Sora in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

Sora						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.21	Negative Binomial	1	0.0149	0.0130	-0.0106	0.0403	1.31	0.2525
Water Area (ha)	1.42	Negative Binomial	1	-0.0315	0.0299	-0.0902	0.0271	1.11	0.2917
No. Fish/ Trap Hour	1.17	Negative Binomial	1	-0.0282	0.0265	-0.0801	0.0236	1.14	0.2859
Vegetation Area (ha)	1.48	Negative Binomial	1	-0.0137	0.0042	-0.0220	-0.0054	10.44	0.0012

Table 2.16- Wetland-scale habitat model results for the relative density of the Sora in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Sora						Wald 95% Confidence Limits			
Models/Variables	Pearson's chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Water Depth (cm)	1.62	Negative Binomial	1	-0.0126	0.0094	-0.0311	0.0059	1.79	0.1812
Water Area (ha)	1.39	Negative Binomial	1	-0.0282	0.0245	-0.0763	0.0198	1.32	0.2497
No. Fish/ Trap Hour	1.35	Negative Binomial	1	-0.0378	0.0220	-0.0808	0.0052	2.96	0.0851
Vegetation Area (ha)	1.58	Negative Binomial	1	-0.0067	0.0045	-0.0156	0.0021	2.22	0.1365
Avg. Inverts./ Activity Trap	1.77	Negative Binomial	1	0.0561	0.0321	-0.0068	0.1190	3.05	0.0806
Avg. Inverts./ Sticky Trap	1.57	Negative Binomial	1	0.0431	0.0186	0.0065	0.0796	5.34	0.0209

Table 2.17- Wetland-scale habitat model results for the relative density of the Virginia Rail in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

Virginia Rail						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.13	Negative Binomial	1	0.0137	0.0180	-0.0216	0.049	0.58	0.4475
Water Area (ha)	1.16	Poisson	1	-0.0754	0.0260	-0.1263	-0.0244	8.41	0.0037
No. Fish/ Trap Hour	1.30	Negative Binomial	1	-0.0864	0.0609	-0.2057	0.0329	2.02	0.1558
Vegetation Area (ha)	1.17	Poisson	1	-0.0182	0.0069	-0.0319	-0.0046	6.89	0.0087

Table 2.18- Wetland-scale habitat model results for the relative density of the Virginia Rail in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Virginia Rail						Wald 95% Confidence Limits			
Models/Variables	Pearson's chi-square/df	Distribution	DF	Estimate	Std Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Water Depth (cm)	1.43	Negative Binomial	1	-0.0150	0.0201	-0.0544	0.0245	0.55	0.4570
Water Area (ha)	1.69	Negative Binomial	1	-0.0727	0.0502	-0.1711	0.0257	2.10	0.1477
No. Fish/Trap Hour	1.09	Negative Binomial	1	-0.0287	0.0438	-0.1145	0.0570	0.43	0.5113
Vegetation Area (ha)	1.12	Negative Binomial	1	-0.0101	0.0089	-0.0275	0.0073	1.29	0.2561
Avg. Inverts./ Activity Trap	1.81	Negative Binomial	1	0.0933	0.0645	-0.0331	0.2198	2.09	0.1480
Avg. Inverts./ Sticky Trap	1.57	Poisson	1	0.0884	0.0291	0.0313	0.1454	9.21	0.0024

Appendix 2.2 - Plot-scale Results of Habitat Models for 2008 and 2009

Table 2.19- Plot-scale habitat model results for the presence/absence of the American Bittern in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	1.11	Average Water Depth (cm)	0.0466	0.0205	27	2.27	0.0315
Cover	1.07	Water Area (ha)	-0.0467	0.2414	26	-0.19	0.8481
		Water Edge Density (km/ha)	3.6783	2.0037	26	1.84	0.0778
Veg. Composition	1.08	Sparganium Area (ha)	-7.2863	5.2201	22	-1.40	0.1767
		Carex Area (ha)	0.9918	0.5082	22	1.95	0.0638
		Equisetum Area (ha)	-4.6058	2.6481	22	-1.74	0.0960
		Phragmites Area (ha)	0.3106	0.3991	22	0.78	0.4447
		Scirpus Area (ha)	0.7170	0.3737	22	1.92	0.0681
		Typha Area (ha)	-0.2272	0.3740	22	-0.61	0.5498

Table 2.20- Plot-scale habitat model results for the presence/absence of the American Bittern in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	1.33	Average Water Depth (cm)	0.0592	0.0276	33	2.14	0.0396
Cover	1.19	Water Area (ha)	-0.0865	0.3858	32	-0.22	0.8240
		Water Edge Density (km/ha)	5.3163	2.7715	32	1.92	0.0640
Veg. Composition	1.17	Sparganium Area (ha)	-0.4233	3.4021	28	-0.12	0.9019
		Carex Area (ha)	-0.0747	0.5650	28	-0.13	0.8958
		Equisetum Area (ha)	-0.0639	1.0215	28	-0.06	0.9506
		Phragmites Area (ha)	0.0106	0.4812	28	0.02	0.9826
		Scirpus Area (ha)	0.2307	0.2804	28	0.82	0.4175
		Typha Area (ha)	-0.4081	0.4446	28	-0.92	0.3665
Food	0.89	Average SAV Biomass/Point	-0.0191	0.0431	31	-0.44	0.6613
		Average Inverts/Sticky Trap	-0.0045	0.0079	31	-0.56	0.5765
		Average Inverts/Activity Trap	-0.0206	0.0391	31	-0.53	0.6021

Table 2.21- Plot-scale habitat model results for the presence/absence of the American Coot in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	1.06	Average Water Depth (cm)	0.0430	0.0184	33	2.34	0.0252
Cover	0.99	Water Area (ha)	0.3765	0.2390	32	1.58	0.1251
		Water Edge Density (km/ha)	4.9310	2.2109	32	2.23	0.0329
Veg. Composition	1.09	Sparganium Area (ha)	-3.2568	3.1722	28	-1.03	0.3134
		Carex Area (ha)	0.0344	0.4120	28	0.08	0.9341
		Equisetum Area (ha)	-2.7639	1.7472	28	-1.58	0.1249
		Phragmites Area (ha)	-0.2059	0.3711	28	-0.55	0.5834
		Scirpus Area (ha)	0.4237	0.3148	28	1.35	0.1891
		Typha Area (ha)	-0.5726	0.3064	28	-1.87	0.0722
Food	1.09	Average SAV Biomass/Point	0.0237	0.0393	31	0.60	0.5511
		Average Inverts/Sticky Trap	0.0116	0.0074	31	1.58	0.1253
		Average Inverts/Activity Trap	-0.0102	0.0203	31	-0.50	0.6197

Table 2.22- Plot-scale habitat model results for the presence/absence of the Pied-billed Grebe in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	0.90	Average Water Depth (cm)	0.0796	0.0297	27	2.68	0.0123
Cover	1.20	Water Area (ha)	0.1736	0.2803	26	0.62	0.5409
		Water Edge Density (km/ha)	6.9001	3.8480	26	1.79	0.0846
Veg. Composition	1.43	Sparganium Area (ha)	-7.7709	5.6403	22	-1.38	0.1821
		Carex Area (ha)	0.2415	0.5035	22	0.48	0.6363
		Equisetum Area (ha)	-2.2427	2.5249	22	-0.89	0.3840
		Phragmites Area (ha)	0.4405	0.5071	22	0.87	0.3945
		Scirpus Area (ha)	0.5122	0.5269	22	0.97	0.3416
		Typha Area (ha)	-0.0495	0.4617	22	-0.11	0.9157

Table 2.23- Plot-scale habitat model results for the presence/absence of the Pied-billed Grebe in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	0.99	Average Water Depth (cm)	0.0918	0.0284	33	3.23	0.0028
Cover	0.98	Water Area (ha)	0.2753	0.2358	32	1.17	0.2518
		Water Edge Density (km/ha)	6.3455	2.4632	32	2.58	0.0148
Veg. Composition	1.29	Sparganium Area (ha)	0.6532	4.6749	28	0.14	0.8899
		Carex Area (ha)	0.5859	0.4998	28	1.17	0.2509
		Equisetum Area (ha)	-3.4652	2.2292	28	-1.55	0.1313
		Phragmites Area (ha)	0.2522	0.4054	28	0.62	0.5389
		Scirpus Area (ha)	0.8181	0.4165	28	1.96	0.0595
		Typha Area (ha)	-0.4824	0.3156	28	-1.53	0.1376
Food	1.11	Average SAV Biomass/Point	0.0460	0.0393	31	1.17	0.2506
		Average Inverts/Sticky Trap	0.0017	0.0065	31	0.26	0.7971
		Average Inverts/Activity Trap	0.0120	0.0213	31	0.56	0.5777

Table 2.24- Plot-scale habitat model results for the presence/absence of the Sora in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	1.15	Average Water Depth (cm)	0.0189	0.0277	27	0.68	0.5002
Cover	1.12	Water Area (ha)	0.4373	0.3701	26	1.18	0.2481
		Water Edge Density (km/ha)	2.0969	3.6211	26	0.58	0.5675
Veg. Composition	0.87	Sparganium Area (ha)	-3.2789	4.7187	22	-0.69	0.4944
		Carex Area (ha)	-1.0954	0.7545	22	-1.45	0.1607
		Equisetum Area (ha)	-2.0943	2.5609	22	-0.82	0.4223
		Phragmites Area (ha)	-0.7577	0.7034	22	-1.08	0.2931
		Scirpus Area (ha)	-0.6588	0.4542	22	-1.45	0.1611
		Typha Area (ha)	-0.1929	0.4788	22	-0.40	0.6909

Table 2.25- Plot-scale habitat model results for the presence/absence of the Sora in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	1.02	Average Water Depth (cm)	-0.0328	0.0213	33	-1.54	0.1333
Cover	1.08	Water Area (ha)	-0.0344	0.3052	32	-0.11	0.9109
		Water Edge Density (km/ha)	-0.6390	2.3399	32	-0.27	0.7865
Veg. Composition	0.84	Sparganium Area (ha)	36.7174	33.9017	28	1.08	0.2880
		Carex Area (ha)	0.3593	0.5305	28	0.68	0.5037
		Equisetum Area (ha)	0.6717	1.4710	28	0.46	0.6515
		Phragmites Area (ha)	-0.0678	0.4015	28	-0.17	0.8671
		Scirpus Area (ha)	-0.0250	0.2534	28	-0.10	0.9220
		Typha Area (ha)	0.2906	0.3364	28	0.86	0.3951
Food	0.88	Average SAV Biomass/Point	-0.0155	0.0524	31	-0.29	0.7703
		Average Inverts/Sticky Trap	0.0011	0.0103	31	0.11	0.9132
		Average Inverts/Activity Trap	0.1913	0.1185	31	1.61	0.1165

Table 2.26- Plot-scale habitat model results for the presence/absence of the Virginia Rail in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	1.06	Average Water Depth (cm)	-0.0177	0.0166	27	-1.07	0.2941
Cover	1.24	Water Area (ha)	-0.4985	0.2858	26	-1.74	0.0930
		Water Edge Density (km/ha)	-2.5008	2.2077	26	-1.13	0.2676
Veg. Composition	1.24	Sparganium Area (ha)	0.0552	3.8686	22	0.01	0.9887
		Carex Area (ha)	-0.0183	0.4043	22	-0.05	0.9643
		Equisetum Area (ha)	3.8768	2.3466	22	1.65	0.1127
		Phragmites Area (ha)	0.0989	0.3544	22	0.28	0.7828
		Scirpus Area (ha)	-0.0984	0.2593	22	-0.38	0.7079
		Typha Area (ha)	0.0700	0.3480	22	0.20	0.8424

Table 2.27- Plot-scale habitat model results for the presence/absence of the Virginia Rail in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate significant variables ($\alpha = 0.1$).

Model	Chi-square/df	Variables	Estimate	Std Error	DF	t-value	p
Treatment	1.05	Average Water Depth (cm)	-0.0113	0.0155	33	-0.73	0.4714
Cover	1.08	Water Area (ha)	0.2873	0.2239	32	1.28	0.2087
		Water Edge Density (km/ha)	-0.6485	1.7545	32	-0.37	0.7141
Veg. Composition	1.21	Sparganium Area (ha)	0.1449	3.0120	28	0.05	0.9620
		Carex Area (ha)	0.3128	0.4052	28	0.77	0.4466
		Equisetum Area (ha)	-1.1432	1.0730	28	-1.07	0.2958
		Phragmites Area (ha)	-0.1420	0.3654	28	-0.39	0.7005
		Scirpus Area (ha)	-0.1540	0.2471	28	-0.62	0.5382
		Typha Area (ha)	-0.1726	0.2358	28	-0.73	0.4703
Food	1.04	Average SAV Biomass/Point	0.0181	0.0383	31	0.47	0.6394
		Average Inverts/Sticky Trap	-0.0012	0.0066	31	-0.17	0.8624
		Average Inverts/Activity Trap	0.1030	0.0584	31	1.76	0.0878

Appendix 2.3 – Results of Forage Fish, Invertebrate, and Submerged Aquatic Vegetation Sampling

Table 2.28- Number of fish caught per trap hour in 2008 and 2009. Data are absent for wetlands 34HI and 35HI in 2008. Note: in wetland 32C in 2009 the numbers of fish caught were too numerous to identify and were therefore labeled “Mixed Species”, these were a combination of Brook Stickleback and Fathead Minnow.

Wetland	Brook Stickleback		Central Mudminnow		Fathead Minnow		Northern Pike		Mixed Species		Total	
	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009
14R	2.06	1.45	0	0	0	0	0	0	--	--	2.06	1.45
35HI	--	0	--	0	--	0.00	--	0.04	--	--	--	0.04
37C	0.21	0.01	0	0.10	0.02	0.00	0.02	0.01	--	--	0.25	0.12
21C	0.02	0.66	0.25	0.82	1.40	0.13	0	0	--	--	1.67	1.61
32C	14.56	0	0.15	0.11	10.75	0	0	0	--	27.24	25.46	27.35
34HI	--	0.30	--	0.23	--	0	--	0	--	--	--	0.53

Table 2.29- Average number and standard deviation of nektonic invertebrates caught per activity trap during each survey round in 2009 in the Summerberry Marsh Complex, Manitoba. Shaded columns indicate drawdown wetlands.

	14R		35HI		37C		21C		32C		34HI	
	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev
May 15 - May 23 2009	2	0.22	4	0.38	2	0.15	1	0.07	1	0.16	4	0.31
May 15 - May 23 2009	7	0.36	10	0.77	5	0.41	33	3.28	2	0.24	5	0.52
May 15 - May 23 2009	22	1.63	10	0.38	11	0.71	23	1.14	3	0.25	13	1.02
May 15 - May 23 2009	9	0.53	25	4.04	9	0.57	21	1.62	6	0.61	19	2.09

Table 2.30- Average number and standard deviation of flying invertebrates caught per sticky trap during each survey round in 2009 in the Summerberry Marsh Complex, Manitoba. Shaded columns indicate drawdown wetlands.

	14R		35HI		37C		21C		32C		34HI	
	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev	Avg./ Pt	Std Dev
May 15 - May 23 2009	3	0.17	25	0.69	20	1.73	11	0.50	44	1.63	32	2.00
May 15 - May 23 2009	109	7.34	74	3.03	78	4.19	122	4.65	62	2.59	62	3.62
May 15 - May 23 2009	177	6.52	66	2.05	116	3.22	112	5.13	63	2.10	58	2.35
May 15 - May 23 2009	80	3.13	83	3.16	50	1.81	119	4.62	67	3.45	67	5.55

Table 2.31- Average weight (g) and standard deviation of submerged aquatic vegetation genera in 2009 from the Summerberry Marsh Complex, Manitoba. Shaded columns indicate drawdown wetlands.

Wetland	14R		35HI		37C		21C		32C		34HI	
	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.
<i>Ceratophyllum</i>	3.90	0.40	1.64	0.18	0.92	0.10	6.76	0.33	0.33	0.06	3.53	0.52
<i>Chara</i>	0.00	0.00	2.29	0.31	0.38	0.07	4.24	0.65	0.77	0.11	0.11	0.03
<i>Elodea</i>	0.33	0.05	2.59	0.52	1.38	0.18	0.00	0.00	0.01	0.00	4.00	0.53
<i>Fontinalis</i>	0.00	0.00	1.41	0.19	0.05	0.01	0.00	0.00	0.03	0.01	0.00	0.00
<i>Myriophyllum</i>	1.60	0.21	1.51	0.14	0.59	0.11	1.71	0.25	4.13	0.38	1.55	0.14
<i>Potamogeton</i>	3.89	0.28	5.54	0.56	4.95	0.41	0.93	0.07	1.44	0.13	3.32	0.27
<i>Sagittaria</i>	0.17	0.04	0.01	0.00	0.73	0.07	0.00	0.00	0.15	0.03	0.00	0.00
<i>Sparganium</i>	0.20	0.04	4.89	0.45	1.27	0.15	5.33	0.66	5.29	0.62	0.02	0.01
<i>Stuckenia</i>	0.30	0.05	3.05	0.14	2.49	0.18	0.30	0.05	3.44	0.40	0.27	0.06
<i>Utricularia</i>	0.00	0.00	0.00	0.00	0.09	0.01	0.00	0.00	0.36	0.06	0.00	0.00

Chapter 3 - Effects of a partial water-level drawdown on waterfowl in the Saskatchewan River Delta, Manitoba

3.1 Introduction

The Saskatchewan River Delta (SRD) provides important nesting and migrating habitat for numerous species of waterfowl and significantly contributes to continental waterfowl populations (Harper 1975; Uchtmann 1983). As a result, the delta has been recognized internationally as a Canadian Important Bird Area (IBA Canada 2009). The delta is particularly important to waterfowl during drought years in the prairie region of Canada and the USA, when pothole wetlands are dry. The wetlands in the delta are more stable and not as prone to dry conditions, providing a refuge for waterfowl during these times (Figure 1.1; United States Fish and Wildlife Service and Manitoba Department of Mines and Natural Resources 1961; Uchtmann 1983). Further, regions such as the SRD offer important habitat for displaced waterfowl as wetlands are lost in the prairies due to human activities. However, there has been a downward trend in spring waterfowl numbers in the SRD (Figure 1.3; United States Fish and Wildlife Service Migratory Bird Data Center 2009). The cause of this trend is unknown, but the delta has been impacted by several anthropogenic developments in recent decades. Large areas of wetlands have been flooded or converted to crops, and the hydrological cycle of the Saskatchewan River has been altered by hydro-electrical developments, creating more terrestrial habitat (Clay 1978; Smith 2003). The loss and continued deterioration of wetland habitat in the delta highlights the need for effective wetland management to maintain the productivity of the area.

In an attempt to maintain waterfowl productivity, Ducks Unlimited Canada has created extensive infrastructure throughout the SRD. This has included the construction of nesting islands and deep-water channels in wetlands in an attempt to create upland nesting habitat and promote accessibility to upland nesting habitat. Water-control structures have also been constructed in wetland basins to provide control over water levels and allow water drawdown/refill capabilities. These developments have been successful in prairie wetlands (Giroux 1981; Duebbert 1982; Broschart and Linder 1986), but there have been unforeseen consequences with these developments in the SRD. The spoil piles created by the deep-water channeling promoted rapid growth and expansion of Willow (*Salix* spp.) and *Phragmites* spp., which overgrew much of the channels (C. E. Smith, Ducks Unlimited Canada, personal communication; personal observation). The nesting islands created habitat sinks for waterfowl, as many of the nests on the islands were destroyed by American Mink (*Neovison vison*) (C. E. Smith, personal communication). Water-level manipulations intended to mimic the wet-dry cycle, commonly used in prairie wetlands, resulted in monotypic communities of emergent vegetation and large floating mats of vegetation that persisted for long periods of time (Smith 1986; Smith 1987).

Current wetland management in the SRD typically involves manipulating wetland water levels to produce diverse vegetation species in a hemi-marsh state (Smith and Jones 1982). Partial water-level drawdowns are used instead of complete dewatering drawdowns to prevent the development of monotypic vegetation communities and floating vegetation mats. However, quantitative data on the effects of this management on

waterfowl in the SRD are lacking. Coarse-scale waterfowl surveys conducted by the U.S. Fish and Wildlife Service are insufficient for detecting management effects at a basin scale. Ducks Unlimited Canada has performed some spring population trend surveys in the past, but these were not focused on the effects of water-level management (Smith and Jones 1982). Therefore, my study was aimed at gathering fine-scale, quantitative data on the effects of water-level management on waterfowl in a relatively understudied, boreal-wetland ecosystem. These data were collected over the spring, summer, and fall to identify the impacts at several stages of the waterfowl life cycle.

Water depth is the most important driver of habitat suitability for waterfowl, as it influences vegetation composition for nesting and thermal cover, and food availability and accessibility (Murkin et al. 1997). The development of coenoclines, composition, and density of emergent vegetation within wetlands is largely determined by water depth and basin topography (Whittaker 1967; Seabloom et al. 1998). The characteristics of wetland vegetation play an important role in influencing wetland use by waterfowl. For example, vegetation structure, such as the hemi-marsh state, is preferred by numerous species of waterfowl as it provides visual isolation between pairs in the breeding season, thermal cover, and increased invertebrate abundances (Kaminski and Prince 1981; Murkin et al. 1997). The species of emergent vegetation may also play a role in waterfowl use of a wetland. Vegetation species such as *Schoenoplectus* spp. provides increased overwater nesting sites, as this species occurs in deeper water compared to other species. Alternatively, *Carex* spp. may provide increased upland nesting habitat (Welling et al. 1988; van der Valk et al. 2000).

Food accessibility and availability are also influenced by water depth. The accessibility of food at different depths is dictated by morphological features of waterfowl, such as body and neck length, bill morphology and feeding strategy, i.e. dabbling or diving (Poysa 1983). The availability of important food sources, such as submerged aquatic vegetation (SAV), is determined by water depth, which limits light penetration in the water column (Robel 1961; Anderson 1978; Barko et al. 1986). Submerged aquatic vegetation is the predominant food source for American Wigeon (*Anas americana*) and Canvasback (*Aythya affinis*) (Mowbray 1999; Mowbray 2002). It is also fed on extensively by waterfowl during fall to provide the energetic requirements of migration (Dirschl 1969; Anderson and Low 1976; Sugden and Driver 1980; Murkin and Kadlec 1986; Baldassare and Bolen 2006). In addition to being an important food source for waterfowl, SAV also provides habitat for invertebrates and can increase invertebrate abundance (Voigts 1976; van den Berg et al. 1997; Zimmer et al. 2000). Invertebrates provide an important food source for waterfowl, particularly for egg-laying hens and broods (Murkin and Kadlec 1986; Cox et al. 1998).

Water depth also influences invertebrate populations by controlling oxygen levels in the water and the depth of freezing of the water column and substrate during the winter (Murkin and Ross 2000). Similarly, oxygen levels and freezing depth also affect the presence and abundance of fish in wetlands (Sargent and Galat 2006). The presence of fish can reduce the amount of invertebrates available to waterfowl and directly compete

with them for this food resource (Bouffard and Hanson 1997; Zimmer et al. 2000; Hornung and Foote 2006).

While water depth plays an important role in determining the quality and quantity waterfowl habitat, the patterns of habitat selection are largely dependent upon the spatial scales at which they are studied (Wiens 1989; Orians and Wittenberger 1991; Paracuellos 2006). Therefore, I examined the relationships of waterfowl to wetland characteristics at multiple scales to increase the likelihood of determining scale-dependent patterns that could exist in a highly variable wetland environment and reduce the chance of making erroneous conclusions regarding water-level treatment in my study (Weins 1989; Orians and Wittenberger 1991; Paracuellos 2006). Numerous studies have examined the effects of landscape variables on avian species, but this study focuses on habitat selection at a wetland scale and micro-scale. These spatial scales were chosen based on the spatial extent of the effects of the water-level manipulations, and the *a priori* knowledge of important habitat variables of waterfowl.

Wetland-scale habitat selection analyses are common in waterfowl studies. Habitat selection at this scale has been attributed to water depth (Murkin et al. 1997; Colwell and Taft 2000; Isola et al. 2000), vegetation characteristics (Murkin et al. 1997; Stafford et al. 2007; Webb et al. 2010), food abundance (Joyner 1980; Murkin and Kadlec 1986; Longcore 2006), and fish presence (Hill et al. 1987; Hornung and Foote 2006). However, studies examining habitat selection by waterfowl at a finer, micro-scale are uncommon, although microhabitat influences habitat selection by waterfowl; factors such as local

vegetation characteristics (Kaminski and Prince 1981), submerged vegetation abundance (Anderson and Low 1976; Monda and Ratti 1988), and water depth (Monda and Ratti 1988), may all influence microhabitat suitability. This study examined habitat selection of spring-breeding waterfowl at the wetland and micro-scale. Habitat selection of waterfowl through the summer and fall was examined at the wetland scale to identify how the water-level manipulations impacted the waterfowl during different life stages. By examining habitat selection at the wetland and micro-scale, and throughout the year, a better understanding of the effects of water-level manipulations on waterfowl settling patterns, territory establishment, and during different life-cycle stages can hopefully be achieved to guide future management.

3.2 Objectives

- 1) Determine the densities of different waterfowl species during the breeding, post-breeding, brooding, and staging/migrating life stages to determine what waterfowl species and life stages are supported by the Summerberry Marsh Complex.

- 2) Evaluate effects of the partial water-level drawdown management on waterfowl from breeding through fall staging, by modeling the relationship between waterfowl relative abundance and cover and food resources, such as water depth, emergent and submergent vegetation characteristics, and invertebrate abundances.

3.3 Study Area and Experimental Design

My study took place during 2008 and 2009 in the managed wetlands of Summerberry Marsh Complex (SMC). The SMC is located within the SRD, approximately 25 km southeast of The Pas, Manitoba. A partial water-level drawdown was initiated in three wetlands in 2007 and continued through 2010. Three additional wetlands were managed with high water levels to act as controls in the experiment. See Chapter 1 for a more detailed description.

3.4 Methods

3.4.1 Plot-scale Surveys - Spring-breeding Waterfowl

In 2008, waterfowl spring-breeding surveys were conducted from a canoe travelling along transects in each of the study wetlands. Transects consisted of the route taken between randomly selected points of a consecutive call-response survey for marsh birds (see Chapter 2 for methods on the selection of points). Transects were approximately 400 m long, but varied according to how direct the route was between points ($SD = 107$ m). The number of transects within a wetland was dependent on the number of call-response points it contained. The three drawdown wetlands contained a total of 6,325 m of transects, which was not significantly different from the 7,103 m of transects in the three non-drawdown wetlands ($p = 0.92$). A hand-held Global Positioning System (GPS) unit was used to record the transect routes to ensure the same routes were travelled in all survey rounds.

Transect surveys began half-hour before sunrise and lasted until all points in the call-response survey were visited, but no later than 1000 hours. In 2008, two dependent observers, using 8x42 power binoculars when necessary, counted all visible waterfowl (Family Anatidae), recording species, sex, and flock arrangement along each transect. These surveys were conducted four times in each study wetland from 27 May-2 July, 2008, at approximately 10-day intervals, but not during rain or if wind speed exceeded 20 km/h.

In 2009, spring waterfowl surveys were changed to a 200-m radius limited point count, using a double-observer approach (Nichols et al. 2000). These surveys were conducted from a canoe, and done consecutively with a call-response survey for marsh birds (Chapter 2). Surveys began half-hour before sunrise and lasted until all call-response points were visited, but no later than 1000 hours. Two dependent observers, using 8x42 power binoculars when necessary, counted all visible waterfowl, recording species, sex, and flock arrangement within a 200-m radius at each call-response point for a period of 10 minutes. The primary observer recorded all birds that were detected and indicated these to the secondary observer. The secondary observer recorded all birds detected by the primary observer as well as any additional birds that were not detected by the primary observer. The primary and secondary observer roles were alternated between points. These surveys were conducted at 40 points, with 6-8 points in each of the six study wetlands from 15 May-25 June, 2009. Surveys were repeated in each study wetland four times at approximately 10 day intervals. Surveys were not performed during rain or if wind speed exceeded 20 km/h. For an explanation of methods on the selection of survey points see Chapter 2.

3.4.2 Wetland-scale Surveys - Spring-breeding Waterfowl

In 2009, waterfowl spring-breeding surveys were conducted using a helicopter. Surveys were flown at a height of approximately 30 m, with a speed of approximately 50 km/h. The helicopter travelled along the periphery of the open water/emergent vegetation interface of each wetland basin to allow the two observers, sitting on the same side of the aircraft, to overlook the wetland. A single pass around the wetland was done and an

additional pass over the center of some of the larger, more circular wetlands (32C and 37C) was required to ensure complete coverage. A double-observer approach with a primary and secondary observer was used to account for species and observer detection probabilities (Nichols et al. 2000). Two dependent observers, using voice recorders, counted and recorded all visible waterfowl which were identified to species and flock arrangement. The primary observer recorded all birds that he detected and indicated these, via the radio system, to the secondary observer. The secondary observer recorded any birds that were not detected by the primary observer. Surveys began at approximately 0800 hours, and were conducted twice: 22 May and 10 June, 2009. The surveys were not conducted during rain or if wind speed exceeded 20 km/h.

3.4.3 Wetland-scale Surveys - Broods and Post-breeding Waterfowl

In 2008, brood surveys were conducted from an airboat travelling along parallel transects 100 m apart until the entire wetland basin was surveyed. Two dependent observers counted all visible broods. Where possible, broods were observed from a distance using 8x42 binoculars to minimize disturbance. Broods were aged according to Gollop and Marshall (1954). Surveys began one hour after sunrise and were done three times on each study wetland from 2 July-25 July, 2008, but not during rain or if wind speed exceeded 20 km/h.

In 2009, broods and post-breeding waterfowl were counted using aerial surveys. These surveys used the same methods as the spring-breeding waterfowl aerial surveys. When a brood was detected by the primary or secondary observer, the observers coordinated in

aging and counting the number of ducklings to ensure accuracy. Broods were aged according to Gollop and Marshall (1954) and species and number of ducklings were recorded. These surveys began at approximately 0800 hours, and were conducted three times: 3 July, 28 July, and 14 August, 2009, at approximately three-week intervals.

3.4.4 Wetland-scale Surveys - Fall-staging Waterfowl

In 2008, fall-staging waterfowl were counted using aerial surveys. These surveys used similar methods as the spring-breeding and brood/post-breeding surveys. These surveys also used two observers, but data were recorded independently, rather than dependently by the two observers. Waterfowl were only identified as being dabbling species (Genera *Anas*, *Aix*) or diver species (Genera *Aythya*, *Bucephala*, or *Fulica*). One survey was conducted on 2 October 2008.

In 2009, fall-staging waterfowl were counted using aerial surveys. The surveys used the same methods as the 2009 spring-breeding and brood/post-breeding waterfowl aerial surveys. Surveys began at approximately 0800 hours, and were conducted three times: 4 September, 18 September, and 8 October 2009.

3.4.5 Observer Detection Probability

Prior to data analysis, observer detection probabilities were calculated for the 2009 waterfowl survey data at the plot and wetland scale to determine if the observed number of birds required adjustment due to observer bias. The program DOBSERV (Hines 2000) was used to estimate detection probabilities for observers and each species. For the plot-

scale data, detections from all four surveys were included in the analysis. Data were summarized by calculating the numbers of each species at each point detected by the primary observer, and those not detected by the primary observer but by the secondary observer. For the wetland-scale data, detections from all seven aerial surveys were included in the analysis. Data were summarized by calculating the numbers of each species in each wetland detected by the primary observer, and those not detected by the primary observer but by the secondary observer.

For the plot-scale and wetland-scale analyses I used program DOBSERV (Hines 2000) with the dependent observer setting and examined each target species as an individual group. See Nichols et al. (2000) for a detailed explanation of DOBSERV program statistics.

The DOBSERV program generates detection probability estimates under several models (Table 3.1). The best model was selected by comparing Akaike's Information Criterion (AIC) values and selecting the model with the lowest value.

Table 3.1- Model definitions for the DOBSERV program (Hines 2000).

Model	Detection Probability (p)
P (.,.)	Same for all species and both observers
P (s,.)	Different for each species but equal among observers
P (.,i)	Equal among species but different between observers
P (g,.)	Equal within groups of species and equal among observers
P (g,i)	Equal within groups of species but different between observers
P (s,i)	Different for each species and different between observers

3.4.6 Invertebrate Sampling

In 2009, activity traps and sticky traps were used to sample for nektonic and flying invertebrates, respectively, within the study wetlands. Activity traps followed the design of Riley and Bookhout (1990). The top, beveled section of a two litre, clear plastic, pop bottle was removed, inverted, and reattached to form the funnel and the remainder formed the catch portion of the trap. Wire was crossed vertically and horizontally at the narrow opening of the funnel to prevent entry by small, predaceous fish. Activity traps were set in pairs, one mounted horizontally and the other mounted vertically from a “t” frame made of wood that was driven into the substrate (Figure 3.1). Traps were set approximately 30 cm below the surface of the water. Samples from the activity traps were filtered through 5000, 2000, and 1000 μm mesh sieves, sorted, and counted to determine abundance of class sizes.

The sticky traps followed the design of King and Wrubleski (1998). A cylinder, 6.25 cm in diameter, and 25 cm long, was wrapped with a transparent, plastic sheet that was coated with Tanglefoot (The Tanglefoot Co. Grand Rapids, MI). This cylinder was placed on the top of the “t” frame that held the activity traps, immediately above the surface of the water (Figure 3.1). After the sampling period, the plastic sheets were removed and the invertebrates stuck to the sheet were counted.

Two “t” frames holding a horizontal and vertical activity trap, and sticky trap, were deployed at random locations within a 200-m radius of each call-response/waterfowl survey point. Traps were placed in the open water areas of the wetlands (>3 m from

emergent vegetation) for 24 hours. Sampling was done concurrently in one drawdown and one non-drawdown wetland from 15 May to 25 June, 2009. Each wetland was sampled four times. The total number of invertebrates in the activity and sticky traps used in the plot-scale analyses were calculated by summing the number of invertebrates counted in each respective trap. The average number of invertebrates in the nektonic and sticky traps used in the wetland-scale analyses was calculated using the numbers of invertebrates from all four surveys from the respective traps.

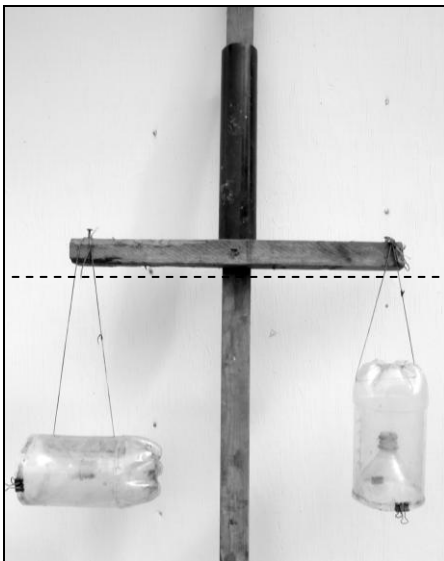


Figure 3.1- “t” frame invertebrate sampling apparatus with sticky-trap on top, horizontal activity trap on the left and vertical activity trap on the right. The dashed line indicates the approximate location of the water level when the trap was deployed.

3.4.7 Forage Fish Sampling

In 2008 and 2009, a Beamish trap was used to sample forage fish in the study wetlands. In late August 2008, two drawdown wetlands and two non-drawdown wetlands were sampled using two traps set for 24 hours. One trap was placed randomly in the wetland, and the other was set in the channel connecting the wetland to its control structure on the

Saskatchewan River or Back Channel (see Figure 1.7). Two wetlands: 35HI, a drawdown wetland, and 34HI, a non-drawdown wetland, were not sampled in 2008 due to a lack of time. Data were substituted for these wetlands with data from 2009 as the numbers of fish caught did not significantly differ between 2008 and 2009 in the other wetlands, according to a paired *t*-test ($p = 0.19$). In 2009, sampling was conducted in late August through September. The trapping effort and placement of traps differed in 2009 compared to 2008. Each wetland was sampled using two traps that were set for two, 24-hour periods. One trap was placed randomly in the open water of the wetland, and the other was placed randomly near vegetation. In both years, captured fish were identified to species [Brook Stickleback (*Culaea inconstans*), Central Mudminnow (*Umbra limi*), Fathead Minnow (*Pimephales promelas*), and Northern Pike (*Esox lucius*)], counted, and released back into the wetland. Fish relative abundance was estimated using the index of number of fish caught per trap hour, not including the Northern Pike, to account for differing sampling efforts among wetlands and years.

3.4.8 Submerged Aquatic Vegetation Sampling

Submerged aquatic vegetation (SAV) was sampled in open water at three random clip-plots taken within a 200-m radius of each call-response/waterfowl survey point. Consequently, the number of samples taken depended upon the number of call-response/waterfowl survey points in the wetland. The bottom of a 55 gallon plastic drum was removed to form 0.25-m² cylinder. This cylinder was pushed into the substrate at the sampling point and a small rake was used to remove above-ground vegetation, from within the cylinder. Samples were taken on 4 and 5 August, 2009 when the standing

biomass was approximately at its maximum. The above-ground vegetation was sorted to genus, dried at 80°C for 48 hours, and weighed to determine biomass. Vegetation genera, which are commonly used by waterfowl as food, were used as an estimate of vegetation biomass in the wetland and plot-scale analyses of waterfowl. These vegetation genera, included *Ceratophyllum*, *Chara*, *Elodea*, *Myriophyllum*, *Potamogeton*, *Sagittaria*, *Sparganium*, and *Stuckenia* (Dirschl 1969; Krull 1970; Anderson and Low 1976; Sugden and Driver 1980; Baldassare and Bolen 2006). Biomass was averaged at each point for the plot-scale analyses using the data from the three random clip-plots taken within the 200-m radius of each call-response/waterfowl survey point. For the wetland-scale analyses, data from all clip-plots in the respective wetland were used to calculate the average biomass.

3.4.9 Emergent Vegetation Mapping

Emergent vegetation maps were created for each study wetland for 2008 and 2009. The purpose of these maps was to delineate and quantify the major cover types within each wetland and at each waterfowl survey point. Cover types identified by these maps included *Sparganium* (*Sparganium* spp.), *Carex* (*Carex* spp.), *Equisetum* (*Equisetum* spp.), *Scirpus* (*Schoenoplectus tabernaemontani*), *Typha* (*Typha* spp.), *Phragmites* (*Phragmites australis*), Water (open water), and Trees (dominated by *Salix* spp., but also included *Populus* spp., *Acer negundo*, *Fraxinus pennsylvannica*, and other species). For a detailed explanation of this procedure see Appendix 4.0.

3.4.10 Habitat Selection - Wetland-scale Analyses

To examine the effects of the partial water-level drawdowns and other habitat characteristics on the settling patterns of waterfowl during different life stages, habitat selection models were constructed at the wetland scale.

Using ArcMap 9.1 the Full Supply Level (FSL) of each wetland was used to clip the wetland vegetation map for 2008 and 2009. The FSL is the approximate beginning of the tree line, as this is the highest elevation at which water levels are managed. The Patch Analyst 4.2 extension in ArcMap 9.1 was used to measure vegetation characteristics within the wetlands, including areas of vegetation and water patches. Average open water depth in the wetlands was calculated by averaging all water depths taken during ground-truthing in open water areas (see Appendix 4.0 for detailed methods on water depth measurement).

Densities of the three most common waterfowl species, Blue-winged Teal (*Anas discors*), Mallard (*Anas platyrhynchos*), and Ring-necked Duck (*Aythya collaris*), as well as grouped dabbling species (*Anas*, *Aix*) and grouped diver species (*Aythya*, *Bucephala*, *Fulica*) were calculated using the total number of waterfowl observed by both observers during each respective aerial survey in 2009 and dividing this by the area of each wetland's FSL. This included using waterfowl totals from the two spring-breeding surveys, three post-breeding surveys, and two fall-staging surveys in 2009. For the 2008 fall-staging survey densities of grouped dabbling species (*Anas*, *Aix*) and grouped diver species (*Aythya*, *Bucephala*, *Fulica*) were calculated using the average number of

waterfowl observed by the two observers during the single survey and dividing this by the area of each wetland's FSL. This was done because observers used an independent recording technique, therefore there was no way to determine if birds were missed or the differences were due to differences in enumeration of flocks. Brood densities were calculated by determining the number of unique broods and dividing this by the area of each wetland's FSL. Unique broods were determined by comparing the species, brood size, and age class of ducklings seen during each survey. Broods of the same species, size (± 2 ducklings), and age class (± 1 age class) were determined to be repeats if the change in age class (± 2 days from the class midpoint defined by Gollop and Marshall 1954) was equal to the elapsed time between surveys. Brood movement was not taken into account.

The GENMOD procedure in SAS 9.2 was used to model the density (birds/ha) of each species. Models were fit using either the negative binomial or Poisson distribution, according to which fit best, with a log-link function. Importance of independent variables in the models was assessed using hypothesis testing with an $\alpha = 0.1$. If models were overdispersed (Pearson's chi-square/df > 1.67) or underdispersed (Pearson's chi-square/df < 0.95) the "pscale" option was used to multiply the covariance matrix by the Pearson's chi-square/df value to account for the dispersion. Due to relatively small number of wetlands in the study the degrees of freedom available for the analyses was small.

Therefore, each variable listed below was run as a separate model in an attempt to explain variation in the densities of different waterfowl groups and species during different stages of the life cycle. The models/variables used in the analyses were:

- 1) Average open water depth. Water depth is known to affect waterfowl use of wetlands, and is determined by morphology and feeding habits (Poysa 1983).

- 2) Area of open water. Open water offers cover for diving species such as the Ring-necked Duck and Canvasback, which dive when threatened (Hohman and Eberhardt 1998; Mowbray 2002).

- 3) Area of emergent vegetation. This included the areas of different vegetation classes, excluding Trees, calculated from the vegetation maps. Emergent vegetation provides thermal and nesting cover for waterfowl (Kaminski and Prince 1981; Murkin et al. 1997).

- 4) Invertebrate abundance from activity and sticky traps (2009 only). Invertebrates are an important food source for waterfowl, particularly breeding hens and broods (Cox et al. 1998; Murkin and Kadlec 1986).

- 5) Forage fish abundance. Fish have been shown to negatively affect invertebrate abundances (Bouffard and Hanson 1997; Zimmer et al. 2000; Hornung and Foote 2006), which are a major food source for waterfowl, particularly for breeding hens and broods that have high protein demands (Cox et al. 1998; Murkin and Kadlec 1986).

3.4.11 Habitat Selection - Plot-scale Analyses

To examine the effects of the partial water-level drawdowns and other habitat characteristics on the microhabitat selection and territory establishment of waterfowl, multivariate statistical models were also constructed using plot-scale habitat variables.

Using ArcMap 9.1, the 2008 survey transects and 2009 survey points were mapped on the respective wetland vegetation maps. Plots were delineated by creating a 200-m radius buffer around each transect (2008) and point (2009), and extracting the areas of the vegetation maps under these buffers. The FSL boundary of each wetland was overlaid and the buffered points were clipped to remove areas outside the boundary of the wetland, making plots variable in size. The Patch Analyst 4.2 extension in ArcMap 9.1 was used to measure various vegetation characteristics within the plots, including areas of vegetation and water patches, numbers of vegetation and water patches, and the amounts of vegetation and water edge. To measure the correct amount of edge, excluding the arbitrary edge of the 200 m boundary, the polygons were converted to polylines using ET Geo Wizard 9.9 extension in ArcMAP 9.1 and clipped with a 199.9-m radius buffer to remove the outer edge. The lengths of the polylines were then used as a measure of vegetation and water edge within the plots. Vegetation and water edge density, calculated as the length of edge in kilometres per hectare, were also calculated from these measurements. Average water depth within plots was calculated by averaging all water depths taken during ground-truthing, in all vegetation types and open water areas, inside the 200 m buffers (see Appendix 4.0 for detailed methods on water depth measurement).

Waterfowl densities were calculated per unit area within each 200 m buffer using the total number of waterfowl from all four survey rounds in 2008 and 2009. Densities of the three most common species, Blue-winged Teal, Mallard, and Ring-necked Duck, as well as grouped dabbling species (Genus *Anas*, *Aix*) and grouped diver species (Genus *Aythya*, *Bucephala*, *Fulica*) were used for modeling.

I ensured that statistical assumptions were met by first examining the distribution of the waterfowl densities. I used Q-Q plots of the residuals and the GEMOD procedure in SAS 9.2 to compare the fit of the common distributions (normal, Poisson, negative binomial). The procedure indicated that the normal distribution fit the data well. Generalized linear mixed-effects models in the GLIMMIX procedure in SAS 9.2 were used to model the relationship between waterfowl relative abundance and cover and food resources at the plot scale. I included “Wetland” as a random variable to account for possible variation between the different wetlands. Models were fit using a normal distribution and a log-link function. Importance of independent variables in the models was assessed using hypothesis testing with an $\alpha = 0.1$. I used combinations of variables from the vegetation mapping, invertebrate, submergent vegetation, and fish sampling to develop biologically justifiable models *a priori* in an attempt to explain the presence of different waterfowl groups and species. The models used in the analyses were:

- 1) “Treatment”. This model included average water depth within the plots. Water depth is known to affect waterfowl use of wetlands, and is determined by morphology and feeding habits (Poysa 1983; Murkin et al. 1997).

2) “Cover”. This model included measurements from the wetland vegetation maps. Variables included water edge density, as a measure of vegetation/water interspersion, and area of open water. Higher amounts of interspersion will result in increased water edge densities.

Interspersion has been shown to be important for waterfowl, as it may increase invertebrate abundance, an important food source for waterfowl, as well as provide visual isolation during breeding, escape cover, and thermal cover (Kaminski and Prince 1981; Murkin et al. 1997).

3) “Veg. Composition”. This model included the areas of different vegetation classes, excluding Trees, derived from the wetland vegetation maps. Different vegetation species offer different cover types (*Schoenoplectus* vs. *Phragmites*), nesting cover (*Schoenoplectus* for overwater nesting vs. *Carex* for upland nesting), and food (seeds of different species).

4) “Food”. This model was only performed with the 2009 data as habitat data were not collected in 2008. The model included invertebrate abundances from activity and sticky traps and SAV biomass. Invertebrates are an important food source for waterfowl, particularly breeding hens and broods (Cox et al. 1998; Murkin and Kadlec 1986). Submerged aquatic vegetation is an important food source for many species of waterfowl (Bartonek and Hickey 1969; Anderson and Low 1976; Brisbin et al. 2002), and may to

affect invertebrate abundance (Voigts 1976; van den Berg et al. 1997; Zimmer et al. 2000).

3.5 Results

3.5.1 Wetland-scale Observer Detection Probability

Detection probabilities of waterfowl during the aerial surveys at the wetland-scale were best predicted by the P (s,i) model (Table 3.2). This model suggests that detection probabilities differed between observers and between species.

Table 3.2- Detection probability models created by the DOBSERV program for the wetland-scale waterfowl survey data from the Summerberry Marsh Complex, Manitoba, 2009.

Model	Likelihood	DF	AIC	G-O-F	AICc	QAIC	QAICc
P (s,i)	-29.16	0	90.31	*****	90.52	90.31	90.52
P (s,.)	-54.20	8	124.40	0.00	124.5	124.4	124.5
P (g,i)	-92.58	14	189.16	0.00	189.2	189.2	189.2
P (.,i)	-92.58	14	189.16	0.00	189.2	189.2	189.2
P (g,.)	-98.74	15	199.48	0.00	199.5	199.5	199.5
P (.,.)	-98.74	15	199.48	0.00	199.5	199.5	199.5

The output of the P (s, i) model (Table 3.3) shows that most species observed had a high detection probability (>90%) with the exception of the American Wigeon (AMWI), Canada Goose (CAGO, *Branta canadensis*), and American Coot (AMCO). Due to the majority of species having high detection probabilities, particularly the most common waterfowl species, i.e. Blue-winged Teal (BWTE), Mallard (MALL), and Ring-necked Duck (RNDU), data analyses were performed on the uncorrected observed numbers (X..) of waterfowl.

Table 3.3- Estimates of N using the $P(s,i)$ model from the seven aerial surveys conducted at the wetland scale in the Summerberry Marsh Complex, Manitoba, 2009. See Appendix 1 for definitions of species notation.

Species	$X_{..}$	p	$SE(p)$	N	$SE(N)$	Lower 95%	Upper 95%
AMWI	37	0.7818	0.1776	47.33	11.35	38.81	96.07
BWTE	485	0.9736	0.0069	498.16	5.11	491.31	512.42
CAGO	95	0.8381	0.107	113.36	15.21	99.45	170.7
GWTE	9	1		9			
LESC	16	1		16			
MALL	1458	0.9853	0.0025	1479.69	6	1470.74	1494.92
NOPI	2	1		2			
NOSH	10	1		10			
RNDU	436	0.9944	0.0022	438.45	1.86	436.65	445.17
SWGO	2	1		2			
AMCO	19	0.6571	0.33	28.91	15.03	20.16	103.36
CANV	27	0.9135	0.0807	29.56	3.1	27.4	43.49
BUFF	7	1		7			
GADW	1	1		1			
REDH	3	1		3			

$X_{..}$ – Observed number of individuals.

p – Detection probability.

N – Estimated number of individuals ($N = X_{..}/p$)

3.5.2 Plot-scale Observer Detection Probability

Detection probabilities of waterfowl at the plot scale were best predicted by the $P(s,.)$ model (Table 3.4). This model suggests that detection probability is different for each species but the same between observers.

Table 3.4- Detection probability models created by the DOBSERV program for the plot-scale waterfowl survey data from the Summerberry Marsh Complex, Manitoba, 2009.

Model	Likelihood	DF	AIC	G-O-F	AICc	QAIC	QAICc
$P(s,.)$	-9.69	10	39.38	0.499	39.67	39.38	39.67
$P(g,.)$	-20.01	19	42.02	0.052	42.03	42.02	42.03
$P(.,.)$	-20.01	19	42.02	0.052	42.03	42.02	42.03
$P(g,i)$	-19.20	18	42.39	0.056	42.41	42.39	42.41
$P(.,i)$	-19.20	18	42.39	0.056	42.41	42.39	42.41
$P(s,i)$	-5.02	0	50.03	1.000	51.17	50.03	51.17

The output of the P(s,.) model shows that detection probabilities are high (>99%) for all species. Only the Ringed-necked Duck (RNDU) and Blue-winged Teal (BWTE) show detection probabilities below 100%, but are still very high at 99.96% and 99.48%, respectively (Table 3.5).

Due to the high detection probabilities of all waterfowl species, the uncorrected observed numbers of waterfowl (X..) were used for plot-scale data analyses.

Table 3.5- Estimates of *N* using the P(s,.) model from the four surveys conducted at the plot-scale in the Summerberry Marsh Complex, Manitoba, 2009. See Appendix 1 for definitions of species notation.

Species	X..	p	SE(p)	<i>N</i>	SE(<i>N</i>)	Lower 95% CI	Upper 95% CI
CAGO	25	1		25			
CANV	54	1		54			
MALL	145	1		145			
RNDU	339	0.9996	0.0004	339.15	0.41	339.01	341.61
BWTE	119	0.9948	0.0038	119.62	0.91	119.08	124.09
LESC	17	1		17			
BUFF	14	1		14			
NOSH	6	1		6			
GWTE	7	1		7			
GOEY	3	1		3			
AMWI	13	1		13			
AMCO	3	1		3			
REDH	13	1		13			
WODU	2	1		2			

X.. – Observed number of individuals.

p – Detection probability.

N – Estimated number of individuals ($N = X../p$)

3.5.3 Plot-scale Surveys - Spring-breeding Waterfowl

In 2008, a total of 857 waterfowl were observed along transects in the study wetlands.

Three species composed 75% of the total observations; Blue-winged Teal (13.6% of total), Mallard (31.9%), and Ring-necked Duck (29.4%) (Table 3.6). In 2009, a total of

762 waterfowl were observed at the survey points in the study wetlands. Similar to 2008, three species made up approximately 79% of the total observations: Blue-winged Teal (15.9% of total), Mallard (19.0%), and Ring-necked Duck (44.5%) (Table 3.7).

Table 3.6- Total waterfowl counted within the 200-m buffered transects during the four surveys in the six study wetlands in the Summerberry Marsh Complex, Manitoba, 2008. Shaded rows indicate drawdown wetlands. See Appendix 1 for definitions of species notation.

Wetland	AMWI	BUFF	BWTE	CANV	GADW	GWTE	LESC	MALL	NOPI	NOSH	REDH	RNDU	WODU	Wetland Total	Grand Total
14R	3	1	24	4	3	8	6	103	3	8	2	48	0	213	594
37C	10	2	32	35	3	12	0	102	0	15	4	11	4	230	
35HI	3	0	23	32	0	3	2	44	4	6	2	32	0	151	
21C	0	0	17	0	0	0	33	13	0	0	0	61	0	124	266
32C	0	0	12	2	0	0	0	5	0	0	0	78	0	97	
34HI	0	0	9	0	0	0	4	7	0	0	2	23	0	45	
Species Total	16	3	117	73	6	23	45	274	7	29	10	253	4		860

Table 3.7- Total waterfowl counted within the 200-m radius buffers during the four surveys in the six study wetlands in the Summerberry Marsh Complex, Manitoba, 2009. Shaded rows indicate drawdown wetlands. See Appendix 1 for definitions of species notation.

Wetland	AMCO	AMWI	BUFF	BWTE	CAGO	CANV	GOEY	GWTE	LESC	MALL	NOSH	REDH	RNDU	WODU	Wetland Total	Grand Total
14R	0	0	4	38	9	6	1	2	2	28	4	0	81	0	175	473
35HI	0	8	4	25	4	15	0	5	6	37	2	0	64	2	172	
37C	0	3	6	24	8	10	1	0	2	46	0	4	22	0	126	
21C	0	2	0	23	4	12	1	0	7	18	0	0	102	0	169	274
32C	3	0	0	2	0	9	0	0	0	9	0	9	50	0	82	
34HI	0	0	0	1	0	2	0	0	0	7	0	0	13	0	23	
Species Total	3	13	14	113	25	54	3	7	17	145	6	13	332	2		747

We did not observe a peak in the total numbers of waterfowl in the drawdown wetlands at the plot-scale. In 2008, waterfowl abundance was highest in late May and early June and declined steadily until late June and early July (Figure 3.2). In 2009, the trend in waterfowl numbers was opposite; waterfowl abundance was lowest in late May and increased steadily until late June (Figure 3.3).

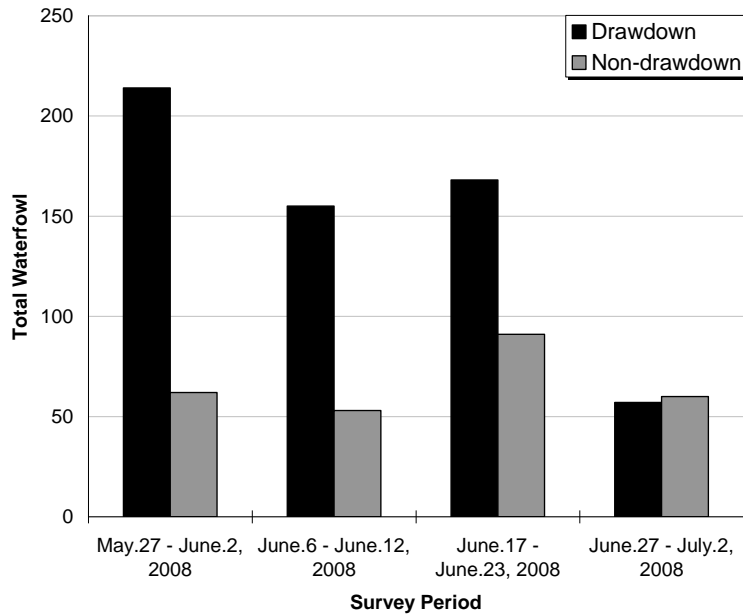


Figure 3.2- Total numbers of waterfowl observed in 2008 on the drawdown and non-drawdown wetlands during each survey round in the Summerberry Marsh Complex, Manitoba.

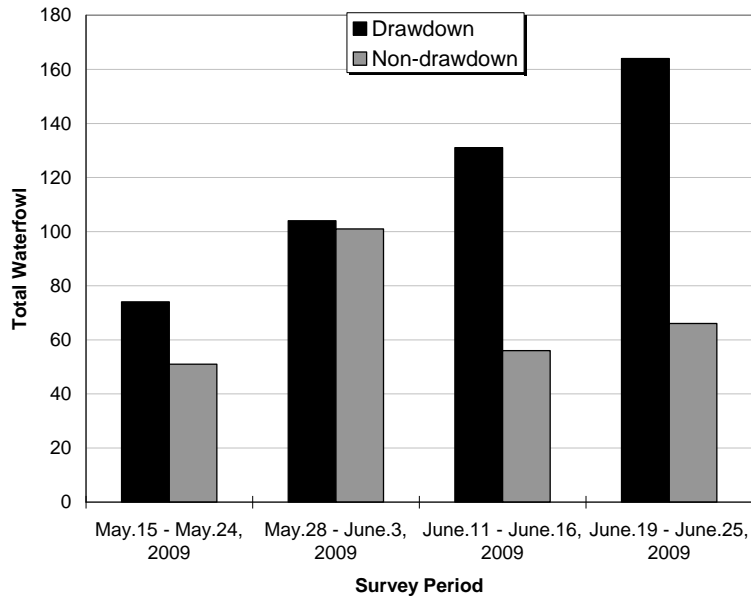


Figure 3.3- Total numbers of waterfowl observed in 2009 on the drawdown and non-drawdown wetlands during each survey round in the Summerberry Marsh Complex, Manitoba.

3.5.4 Wetland-scale - Spring-breeding Surveys

A total of 716 waterfowl were counted during the two aerial, spring-breeding surveys.

Three species produced 79.4% of the total observations: Blue-winged Teal (24.2% of total), Mallard (18.7%), and Ring-necked Duck (36.5%) (Table 3.8).

Table 3.8- Total number of waterfowl counted during the spring aerial surveys, conducted 22 May and 10 June 2009, in the Summerberry Marsh Complex, Manitoba. Shaded rows indicate drawdown wetlands. See Appendix 1 for definitions of species notation.

Wetland	AMCO	AMWI	BUFF	BWTE	CAGO	CANV	GADW	GWTE	LESC	MALL	NOPI	NOSH	REDH	RNDU	SNGO	Wetland Total	Grand Total
14R	2	5	0	62	10	4	1	3	2	42	2	8	0	30	2	173	368
35HI	0	0	4	37	5	7	0	0	0	26	0	0	0	32	0	111	
37C	0	4	3	12	20	3	0	0	0	21	0	0	0	21	0	84	
21C	3	0	3	38	5	0	0	5	12	23	0	0	0	81	0	170	348
32C	7	0	0	10	5	6	0	1	0	6	0	0	0	70	0	105	
34HI	3	0	0	14	5	7	0	0	0	16	0	0	1	27	0	73	
Species Total	15	9	10	173	50	27	1	9	14	134	2	8	1	261	2	716	

3.5.5 Wetland-scale - Post-breeding Surveys

A total of 676 waterfowl were counted during the three aerial post-breeding surveys.

Nearly all of the observations, 98.1% of the total, consisted of three species: Blue-winged Teal (31.2% of total), Mallard (46.0%), and Ring-necked Duck (20.9%) (Table 3.9).

Table 3.9– Total number of waterfowl counted during the aerial post-breeding surveys, conducted 3 July, 28 July, and 14 August 2009, in the Summerberry Marsh Complex, Manitoba. Shaded rows indicate drawdown wetlands.

Wetland	AMCO	AMWI	BWTE	MALL	RNDU	Wetland Total	Grand Total
14R	0	4	48	54	27	133	388
35HI	0	0	18	76	37	131	
37C	1	1	12	105	5	124	
21C	0	0	24	25	35	84	288
32C	3	4	88	33	11	139	
34HI	0	0	21	18	26	65	
Species Total	4	9	211	311	141		676

3.5.6 Wetland-scale Surveys - Broods

In 2008, we counted 73 unique broods. The majority of the broods, 56.2%, were American Coot or Ring-necked Duck, both over-water nesting species. Broods of these species were most common on wetland 21C, a non-drawdown wetland. Upland nesting species, including Blue-winged Teal and Mallard consisted of 17.8% and 6.9% of the total number of broods observed, respectively (Table 3.10). In 2009, we counted only 41 unique broods. Upland nesting species, including Blue winged Teal and Mallard made up the majority of the observations, 73.2%, while the American Coot and Ring-necked Duck made up only 2.4% and 24.4% of the observations that year (Table 3.11).

Table 3.10- Total number of unique broods counted during the three airboat surveys performed from 2 July to 25 July 2008 in the Summerberry Marsh Complex, Manitoba. Shaded rows indicate drawdown wetlands. See Appendix 1 for definitions of species notation.

Wetland	AMCO	AMWI	BUFF	BWTE	CANV	GADW	LESC	MALL	REDH	RNDU	UNKN	Wetland Total	Grand Total
14R	0	1	0	3	0	0	0	2	0	2	0	8	29
35HI	4	0	2	4	0	0	0	0	0	2	0	12	
37C	0	0	2	2	1	0	0	2	0	2	0	9	
21C	8	0	1	3	1	1	1	1	1	16	0	33	44
32C	5	0	0	1	0	0	0	0	0	1	0	7	
34HI	0	0	1	0	0	0	0	0	0	1	2	4	
Species Total	17	1	6	13	2	1	1	5	1	24	2		73

Table 3.11- Total number of unique broods counted during the three aerial surveys conducted 3 July, 28 July, and 14 August 2009 in the Summerberry Marsh Complex, Manitoba. Shaded rows indicate drawdown wetlands.

Wetland	AMCO	BWTE	MALL	RNDU	Wetland Total	Grand Total
14R	0	3	3	3	9	23
35HI	0	3	7	2	12	
37C	0	1	1	0	2	
21C	0	7	2	3	12	18
32C	1	1	0	1	3	
34HI	0	2	0	1	3	
Species Total	1	17	13	10		41

3.5.7 Wetland-scale - Fall-staging Surveys

In 2008, an average of 600 waterfowl was counted by the two observers during the single, aerial, fall-staging survey. Dabbler species were predominant, consisting of 84% of the observations (Table 3.12). In 2009, 1219 waterfowl were counted during the three aerial fall-staging surveys. Mallards were the predominant species, consisting of 83% of the total observations. Blue-winged Teal had the second highest number of observations with 8.3% of the total, and the third most common species was the Canada Goose with

3.7% of the total. The Ring-necked Duck was the fourth most common species with 2.9% of the total observations (Table 3.13).

Table 3.12- The average number of dabbling species, diver species, and Canada Geese observed by the two observers during the aerial, fall-staging survey, conducted 2 October 2008, in the Summerberry Marsh Complex, Manitoba. Shaded rows indicate drawdown wetlands. See Appendix 1 for definitions of species notation.

Wetland	Avg. Dabbling	Avg. Diver	Avg. CAGO	Total Wetland Avg.	Grand Total
14R	86	0	4	90	528
37C	188	1	21	210	
35HI	197	7	25	229	
21C	12	1	0	13	72
32C	12	35	0	47	
34HI	9	1	4	13	
Species Total	504	45	54		600

Table 3.13- Total number of waterfowl observed during two aerial, fall-staging surveys, conducted on 4 September and 8 October 2009, in the Summerberry Marsh Complex, Manitoba. Data from the survey conducted 18 September, 2009 is not included due to a recorder malfunction. Shaded rows indicate drawdown wetlands.

Wetland	AMCO	AMWI	BWTE	CAGO	GESC	MALL	REDH	RNDU	Wetland Total	Grand Total
14R	0	11	2	0	0	310	0	0	323	952
35HI	0	0	20	10	0	223	0	0	253	
37C	1	2	11	30	0	331	0	1	376	
21C	0	0	10	0	0	38	2	0	50	267
32C	0	0	44	5	0	59	0	12	120	
34HI	0	8	14	0	2	51	0	22	97	
Species Total	1	21	101	45	2	1012	2	35		1219

Waterfowl densities were higher during the first two surveys in the spring compared to the surveys conducted in the summer (July 3 & 28). Waterfowl densities were lowest during the summer and gradually increased into early fall. In late fall, there was a large increase of waterfowl densities in the drawdown wetlands, while the densities in the non-drawdown wetlands remained relatively low (Figure 3.4).

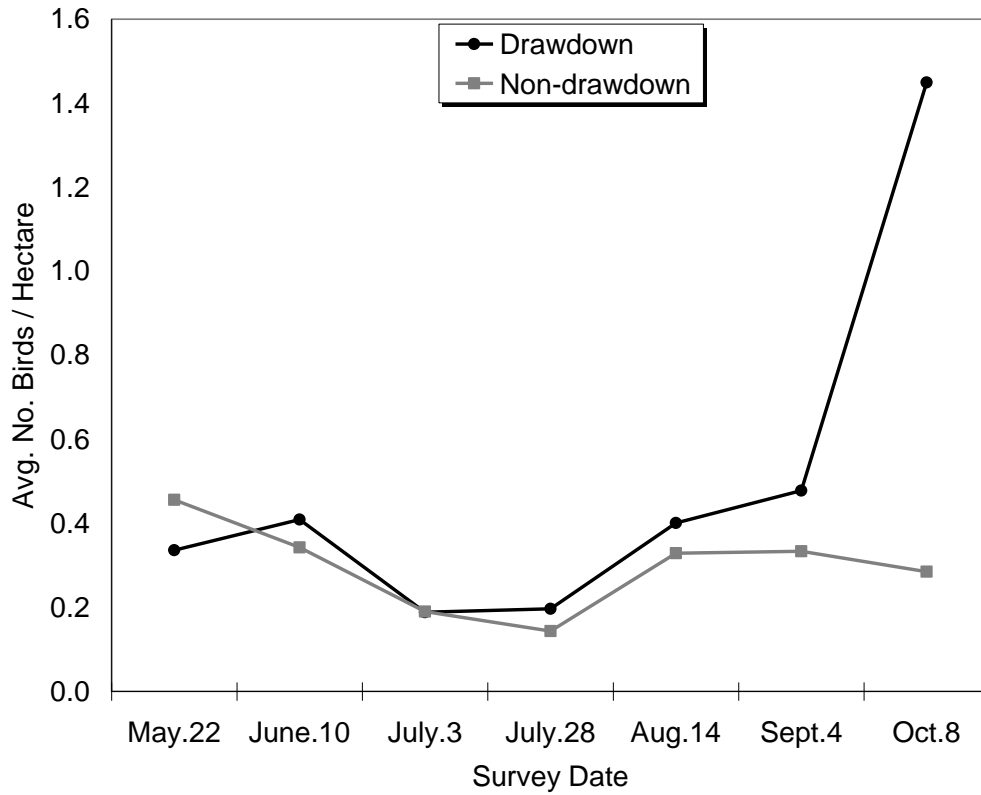


Figure 3.4- Average number of waterfowl per hectare on the drawdown and non-drawdown wetlands during each survey period in the Summerberry Marsh Complex, Manitoba, 2009.

3.5.8 Habitat Selection Models - Wetland-scale Results

The partial water-level drawdowns appeared to increase the densities of dabbling species, including Mallards, during the spring-breeding, post-breeding, and fall-staging periods. In contrast, the drawdowns decreased the densities of diving species, including the Ring-necked Duck, during the spring and fall (Table 3.14). For both dabblers and divers the strength of the relationship was strongest during the fall-staging period. Fall-staging models predicted a 0.6 dabbling duck per hectare decrease and 0.5 diving duck per hectare increase for every 10 cm increase in water depth in 2008 (Appendix 3.5, Tables 3.31 & 3.32). This relationship was slightly weaker for dabblers in 2009, suggesting an increase of 0.15 ducks per hectare with a 10 cm increase in water depth (Appendix 3.5, Table 3.33). Blue-winged Teal were unaffected by the drawdowns until the fall, when they showed a preference for the deeper water of the non-drawdown wetlands. Broods did not appear to be affected consistently by the water level manipulations (Table 3.14).

Water area affected both dabbling and diving densities. Dabblers selected for wetlands with increased water area in the summer, and divers were negatively related to water area in the fall (Table 3.14).

Vegetation area was positively related to dabbling densities in the fall, but negatively related to Blue-winged Teal and Mallard abundances in the spring. Diving species, including the Ring-necked Duck, were also negatively related to vegetation area in the summer (Table 3.14).

In the spring and summer, nektonic invertebrates were positively related to dabbler brood densities, and to densities of divers and Ring-necked Ducks. However, nektonic invertebrates were also negatively correlated with relative abundance of dabblers in the summer. Flying invertebrates were positively related to dabblers, including Blue-winged Teal and Mallard, in the spring. Conversely, these waterfowl species were negatively related to flying invertebrates in the fall. Broods of diver species were also positively related to flying invertebrates. However, diver species, including the Ring-necked Duck, were negatively correlated with abundance of flying invertebrates during the fall (Table 3.14).

Fish abundance had a negative effect on dabbler broods, dabbler species, Mallard, and Blue-winged Teal during the spring, summer, and fall. The Ring-necked Duck also had a negative relationship with forage fish abundance in the summer (Table 3.14).

Submerged aquatic vegetation biomass was positively related to dabbler brood abundance, and negatively related to divers, including the Ring-necked Duck in the fall (Table 3.14).

Table 3.14- Summary of significant variables at the wetland-scale for waterfowl during the spring-breeding and post-breeding periods in 2009, and the brooding and fall-staging periods in 2008-09 in the Summerberry Marsh Complex. In 2008, only wetland-scale brood and fall-staging surveys were conducted. Brood results are limited to dabbling and diver categories in 2008 and 2009.

		Variables				2009 Only		
		Average Open Water Depth (cm)	Water Area (ha)	Vegetation Area (ha)	No. Fish/ Trap Hour	Avg. SAV biomass/ Point	Avg. Inverts./ Activity Trap	Avg. Inverts./ Sticky Trap
Dabblers	2008	Fa(-)	--	Fa(+)	Br(-) Fa(-)	--	--	--
	2009	Sp(-) Su(-) Fa(-)	Su(+)	--	Sp(-) Fa(-)	Br(+)	Su(-) Br(+)	Sp(+)
Divers	2008	Fa(+)	--	--	Fa(+)	--	--	--
	2009	Sp(+) Fa(+)	Fa(-)	Su(-)	--	Fa(-)	Sp(+) Su(+)	Br(+) Fa(-)
Blue-winged Teal		Fa(+)	--	Sp(-)	Sp(-) Su(+)	--	--	Sp(+) Fa(-)
Mallard		Sp(-) Su(-) Fa(-)	--	Sp(-)	Sp(-) Su(-) Fa(-)	--	--	Sp(+)
Ring-necked Duck		Sp(+) Fa(+)	--	Su(-)	Su(-)	Fa(-)	Sp(+) Su(+)	Fa(-)

Sp: Variable significant during the spring-breeding period.

Su: Variable significant during the post-breeding period.

Br: Variable significant for broods (Total Dabbling and Total Diver categories only).

Fa: Variable significant during the fall-staging period.

(+): Significant variable produces positive parameter estimate.

(-): Significant variable produces negative parameter estimate.

--: Variable not significant.

3.5.9 Habitat Selection Models - Plot-scale Results

The partial water-level drawdowns appeared to increase dabbler and Blue-winged Teal densities. The effects of the drawdown were not apparent in 2008, but in 2009 densities of dabbler species and Blue-winged Teal were negatively correlated with increasing water depth (Table 3.15). However, this relationship was relatively weak, indicating that an increase of 10 cm in depth would decrease densities of dabblers and Blue-winged Teal by 0.04 and 0.07 individuals/ha, respectively (Appendix 3.6, Tables 3.43 & 3.45).

The “Cover” models indicated that densities of dabbler and diver species, including the Mallard and Ring-necked Duck, were positively related to water area. Water edge density, or interspersion, was also negatively related to divers and the Ring-necked Duck in 2009, indicating a preference for more open water areas within the wetlands (Table 3.15). This relationship was strong among divers and Ring-necked Ducks, with a decrease of approximately 0.5 ducks per hectare for every kilometre per hectare increase in interspersion (Appendix 3.6, Tables 3.44 & 3.47).

The “Veg. Composition” models showed inconsistent relationships between vegetation species and waterfowl between years. Densities of dabbler species, Mallards, and Ring-necked Duck were positively related to the amount of *Sparganium*. Dabblers and Mallards had a strong relationship with *Sparganium* (Table 3.15). The models predicted a one hectare increase of *Sparganium* would result in a 2.2 and 0.9 duck per hectare increase of dabblers and Mallards, respectively (Appendix 3.6, Tables 3.38, 3.41, 3.43, 3.46). Dabblers were also found to have a strong positive relationship with *Equisetum* in

2008, but this relationship was absent in 2009. The only other positive relationships occurred between divers, the Ring-necked Duck, and *Carex* (Table 3.15). However, these relationships were only found in 2009 and were relatively weak, suggesting an increase of approximately 0.08 ducks per hectare for every one hectare increase of *Carex* (Appendix 3.6, Tables 3.44 & 3.47). The remainder of vegetation species all had negative relationships with at least one group or species of waterfowl. *Typha*, in particular, had a negative relationship to several species of waterfowl. Divers, including the Ring-necked Duck were negatively related to *Typha* in both 2008 and 2009 (Table 3.15). Dabblers were also strongly negatively related to *Typha* area in 2008, with a decrease of 0.8 ducks per hectare for every hectare increase of *Typha* (Appendix 3.6, Table 3.38).

The “Food” model indicated that the abundance of flying invertebrates was positively related to relative abundance of dabbler species, including Blue-winged Teal, and diver species, including Ring-necked Duck. Nektonic invertebrates were positively related to diver species and the Ring-necked Duck. Submerged aquatic vegetation biomass was negatively related to Blue-winged Teal density and positively related to Ring-necked Duck density (Table 3.15).

Table 3.15- Summary of plot-scale analyses indicating the significant variables in each of the statistical models of spring-breeding waterfowl in the Summerberry Marsh Complex, Manitoba, 2008-09.

Species	Treatment	Cover		Veg. Composition						Food (2009 only)		
	Average Water Depth (cm)	Water Area (ha)	Water Edge Density (km/ha)	<i>Sparganium</i> Area (ha)	<i>Carex</i> Area (ha)	<i>Equisetum</i> Area (ha)	<i>Phragmites</i> Area (ha)	<i>Scirpus</i> Area (ha)	<i>Typha</i> Area (ha)	Avg. SAV Biomass/ Pt (g)	Avg. Inverts/ Activity Trap	Avg. Inverts/ Sticky Trap
Dabblers	09 (-)	08/09 (+)	--	08/09 (+)	08 (-)	08 (+)	09 (-)	08 (-)	08 (-)	--	--	09 (+)
Divers	--	08/09 (+)	09 (-)	--	09 (+)	09 (-)	--	--	08/09 (-)	--	09 (+)	09 (+)
Blue-winged Teal	09 (-)	--	--	--	--	09 (-)	09 (-)	09 (-)	--	09 (-)	--	09 (+)
Mallard	--	08 (+)	--	08 (+)	08 (-)	--	08 (-)	08 (-)	08 (-)	--	--	--
Ring-necked Duck	--	08/09 (+)	09 (-)	08 (+)	09 (+)	09 (-)	--	--	08/09 (-)	09 (+)	09 (+)	09 (+)

08: Variable significant in 2008.

09: Variable significant in 2009.

08/09: Variable significant in 2008 and 2009.

(+): Significant variable produces positive parameter estimate.

(-): Significant variable produces negative parameter estimate.

-- : Not significant in 2008 or 2009.

3.6 Discussion

The predominant species I observed during my spring surveys at both the plot and wetland scale were not consistent with the annual spring surveys conducted by the U.S. Fish and Wildlife service. According to their findings, from 2003-2009 the three most common species of waterfowl observed in the SRD were the Mallard, Generic Scaup (Greater and Lesser Scaup [*Aythya marila* and *A. affinis*]), and Generic Goldeneye (Common and Barrow's Goldeneye [*Bucephala clangula* and *B. islandica*]). The Ring-necked Duck and Blue-winged Teal ranked fourth and fifth most common in their surveys, respectively (United States Fish and Wildlife Service Migratory Bird Data Center 2009). The reason for the discrepancy between the species observed between the waterfowl surveys is likely due to the relatively small amount of habitat surveyed in my study compared to the U.S. Fish and Wildlife surveys. My surveys were limited to the habitat within six wetland basins, whereas the U.S. Fish and Wildlife surveys take place across the entire SRD, reflecting waterfowl abundances across highly variable habitat types.

3.6.1 Chronology of Waterfowl Use in the Summerberry Marsh Complex

The plot-scale surveys suggested dissimilarity in the timing of spring migration for breeding/staging waterfowl between years. This dissimilarity was likely due to differences of weather; in 2008 the spring was warmer and drier compared to 2009 (Table 3.16). The warmer and drier conditions would have favoured an earlier migration of waterfowl (Blokpoel and Richardson 1978; Hammond and Johnson 1984; Murphy-

Klassen et al. 2005), resulting in the earlier arrival of waterfowl at the Summerberry Marsh Complex.

Table 3.16- Mean daily temperature, average maximum temperature, average minimum temperature, and total precipitation measured in The Pas, Manitoba in the spring of 2008 and 2009 (Environment Canada 2010).

Date	Mean Daily Temp. (°C)	Avg. Max. Temp. (°C)	Avg. Min. Temp. (°C)	Total Precip. (mm)
May 2008	7.2	14.0	0.4	10.1
May 2009	5.6	12.2	-1.0	36.7
June 2008	15.5	21.9	9.1	40.4
June 2009	14.2	19.6	8.7	49.8

The colder, wetter spring in 2009 may have also decreased productivity, resulting in the low number of broods detected that year. Hammond and Johnson (1984) found that brood productivity was higher in North Dakota, USA during years with a warmer spring. Low temperatures may cause an increased risk of hens and ducklings dying from exposure, increase vulnerability to predators, and reduce feeding opportunities (Hammond and Johnson 1984).

In early summer (July) a moult migration out of the wetlands of the SMC appears to take place. Moulting imposes a flightless state on waterfowl for approximately three to five weeks, increasing their vulnerability to predation (Hohman et al. 1992). Some species undergo a moult migration to find habitat that provides them with cover and adequate food (Baldassarre and Bolen 2006a). Early summer migration away from the SMC seemed to be greatest among the diver species of waterfowl. During the three summer aerial surveys, the Ring-necked Duck was the only waterfowl diver species observed (with the exception of the American Coot). Numbers of Ring-necked Ducks decreased

from spring to summer, indicating movement away from wetlands in the SMC. Movement was likely to larger, more open areas of deeper water, such as nearby Lake Winnipegosis, which is an important moulting habitat for the Redhead (*Aythya americana*) (Bailey and Titman 1985), or the mouth of the delta at Cedar Lake (personal observation). This finding is in contrast to the findings of the Pasquia Project, which took place in the SRD from 2001-03. This project suggested that waterfowl numbers in the SRD increased during the moulting period (Smith et al. 2002; Smith and Friedt 2003; Smith and Friedt 2004). The reason for this discrepancy is likely due to the differences in the size of wetlands surveyed in the projects. In the Pasquia Project, wetlands chosen for post-breeding surveys were 300-1000 ha in size (Smith et al. 2002), whereas the wetlands in my study were only 83-207 ha. Because waterfowl prefer larger basins and bodies of water for moulting (Oring 1964; Hohman et al. 1992), the wetlands in the Pasquia Project were likely preferred by waterfowl.

In the fall, the SMC saw a large influx of waterfowl, particularly in the drawdown wetlands. It appears that the drawdown wetlands provide improved habitat for staging waterfowl, potentially due to increased cover and food resource availability and accessibility. This result is consistent with observations made during the Pasquia Project. Although we did not see a peak of staging waterfowl in 2009, Smith and Friedt (2003) suggest that migration and fall staging in the SRD may depend on weather.

3.6.2 Habitat Selection of Spring-breeding Waterfowl

Habitat use by waterfowl in the spring breeding season was variable among species. Dabbler species of waterfowl, including Mallard and Blue-winged Teal, used the shallower drawdown wetlands and shallower areas within the wetlands; in contrast, diver species and Ring-necked Duck chose the deeper, non-drawdown wetlands, but did not select for water depth within the wetlands. The findings at the wetland scale were predicted and are consistent with other studies (White and James 1978; Savard et al. 1994; Murkin et al. 1997). These results are likely a result of food accessibility and nest site availability. Dabbler species of waterfowl are limited to using shallower water for feeding due to their morphology and feeding tactics (Poysa 1983). The shallower water in the drawdown wetlands provided dabbler species access to submerged aquatic vegetation and invertebrates on and among the vegetation. The maximum depth retained in the drawdown wetlands of approximately 60 cm of water was deeper than is ideal for dabblers, resulting in the selection of shallower areas within the basin; Fredrickson and Reid (1988) suggest that dabblers forage most efficiently at depths of 2-10 inches (~5-25 cm).

The deeper water of the non-drawdown wetlands was likely preferred by diving species of waterfowl as it provided a greater amount of mobility during feeding, and escape from threats. The deeper water of the non-drawdown wetlands also likely provided diver species with more potential over-water nesting sites compared to the drawdown wetlands (Appendix 4.0; Hohman and Eberhardt 1998; Mowbray 2002). The non-drawdown wetlands contained more flooded emergent vegetation and open water areas for nesting

and foraging (Appendix 4.0). The drawdown wetlands may have been avoided due to the growth of dense submerged aquatic vegetation to the surface of the water, which may impede the mobility of diving ducks and their broods (Poysa 1983a; Monda and Ratti 1988).

The absence of plot scale selection by diver species within the non-drawdown wetlands may be due to this group of species being predominantly Ring-necked Ducks, which have more general habitat preferences compared to other *Aythya* species. White and James (1978) reported that habitat used by wintering Ringed-necked Ducks was less exclusive in terms of emergent/submergent vegetation and water depth. Ring-necked Ducks preferred moderate depths of 88-114 cm, but also used shallower areas with 30-88 cm of water. This was in contrast to other *Aythya* species, including Canvasback and Lesser Scaup, which preferred deeper waters from 114-213 cm. In this study, drawdown wetlands were not deep enough to meet the requirements of Ring-necked Ducks, but non-drawdown wetlands apparently provided the broad range of depths preferred by the species.

The effects of the partial water-level drawdowns on dabbling species were only apparent during the second year of the drawdown, suggesting a delay in dabbling use of the wetlands. This delay may be caused by the gradual response of the submerged aquatic vegetation community and/or the invertebrate community. The partial water-level drawdowns may have increased the temperature of the water and/or the amount of light penetration in the water column. This promotes the growth of submerged aquatic

vegetation, increasing the abundance of this food source, while simultaneously providing invertebrate habitat, further increasing food abundance (Barko et al. 1986; van den Berg et al. 1997; Zimmer et al. 2000). Invertebrate abundance may have also been influenced by the growth of algae, which is an important food source for invertebrates (Murkin and Ross 2000). The shallower water of the drawdown wetlands allowed more sediment re-suspension, which increased the available nutrients in the water column and consequently algae abundance (E. Watchorn, 2010, University of Manitoba, Thesis). The response of submerged aquatic vegetation, algae, and invertebrates in the drawdown wetlands likely required more than a single growing season to respond. Therefore, dabbler use of the wetlands was delayed until more favourable habitat conditions developed.

Higher invertebrate abundances tended to increase the relative abundances of all groups and species of waterfowl, as predicted. This may be a result of the particularly high demands for protein by waterfowl for egg development (Murkin and Kadlec 1986). This is consistent with other studies that have shown dabbler species of waterfowl select habitats with high invertebrate abundances during the spring-breeding season (Joyner 1980; Murkin and Kadlec 1986). However, in contrast to the studies by Joyner (1980) and Murkin and Kadlec (1986), I observed that flying invertebrates were important to breeding dabbler species of waterfowl, but nektonic invertebrates were not. This may have been due to prey availability. Dirschl (1969) showed that Blue-winged Teal and Lesser Scaup diets changed significantly in proportion to available food items in the SRD. In my study, nektonic invertebrates were nearly absent during the first sampling rounds in the spring; I found an average of approximately two and seven invertebrates per

trap in the drawdown wetlands during the first and second survey rounds, respectively. However, flying invertebrates were abundant in the first and second rounds of sampling. These traps averaged approximately 18 to 85 invertebrates per trap during the first and second survey rounds, respectively (Appendix 7). Therefore, it appears that flying invertebrates provided a more reliable food source early in the spring. This finding is supported by the work of Swanson and Sargeant (1972), who found that dabbling species feeding at night predominantly fed on emerging midges and mayflies at the surface of the water. Alternatively, dabblers, Blue-winged Teal, and Mallard abundances were all negatively related to abundance of forage fish, perhaps as a result of the impacts of fish in the invertebrate community; the presence of fish can have detrimental effects on the abundance of nektonic invertebrates, directly competing against waterfowl for this food source (Bouffard and Hanson 1997; Zimmer et al. 2000; Hornung and Foote 2006).

Unlike other studies, vegetation interspersion did not appear to play an important role in determining habitat selection by dabbling species in the SMC (Kaminski and Prince 1981; Murkin et al. 1997). Blue-winged Teal and Mallards selected wetlands with less vegetation area and appeared to prefer open water areas within the wetlands. This may have been due to the strong preference by dabbling species for *Sparganium*. *Sparganium* is an important food source for waterfowl in the SRD (Dirschl 1969). Due to *Sparganium* occurring in the central, deeper parts of the wetlands basins (Appendix 4.0), this likely influenced the relationship between dabblers and open water. Diver species of waterfowl and Ring-necked Duck also preferred open water. This finding is consistent with other studies and is due to the foraging and escape behaviours of the species (Oring 1964;

Monda and Ratti 1988; Murkin et al. 1997). Alternatively, *Typha* was avoided by dabblers and divers, perhaps due to high stem densities and large amounts of residual litter. Dense emergent vegetation can impede the movement of marsh birds (Conway 1995), and presumably, could impede access by waterfowl. Therefore, waterfowl may have avoided *Typha* as it did not provide favourable cover for thermal protection or nesting.

3.6.3 Habitat Selection by Post-breeding Waterfowl

In summer, dabbler species, including Mallards, preferred for the drawdown wetlands. These findings are consistent with Murkin et al. (1997), who found that dabbler species, including Mallards, preferred shallower water during the summer. However, Murkin et al. (1997) attributed this preference to cover provided by the dense emergent vegetation. In this study, dabblers appeared to prefer wetlands that provided more open water. The preference for shallow, open water may be a response of dabblers to the energetic demands of moulting. Waterfowl can undergo nutritional stress, where one or more physiological functions are impaired due to nutrient demands exceeding nutrient intake (Hohman et al. 1992). To meet the energy demands, dabblers may feed in shallow water of the drawdown wetlands as it enables them to efficiently forage on submerged aquatic vegetation and invertebrates (Poysa 1983). The preference for open water areas is likely due to the luxuriant growth of submerged aquatic vegetation in these areas (personal observation), and the avoidance of terrestrial predators (Oring 1964).

No water depth preferences of diver species were found. This is in contrast to the findings of Murkin et al. (1997) who found that diving ducks preferred deeper water that allows them to dive and escape threats during the post-breeding period. The discrepancy between Murkin et al. (1997) and my study may be due to the majority of diving ducks leaving my study wetlands to moult. As previously suggested, diver species may undergo a moult migration, likely to larger, deeper bodies of water.

3.6.4 Habitat Selection by Broods

Dabbler and diver waterfowl broods did not show a preference for any water depth. Monda and Ratti (1988) showed that Mallard and Blue-winged Teal broods used depths from <0.3 m to 0.9 m, which suggests the water depths in both our drawdown and non-drawdown wetlands may have provided sufficient brood-rearing habitat for these species. The lack of influence of water depth on diver broods may have been due to the majority of diver broods I observed being Ring-necked Duck. Broods of this species prefer habitat similar to dabbler species (Monda and Ratti 1988), which use a broad range of water depths that were provided in both the drawdown and non-drawdown wetlands.

As predicted, both dabbler and diver broods selected wetlands that provided abundant invertebrates. Invertebrates provide an important food source for growing ducklings (Cox et al. 1998) and broods have been shown to select for wetlands with high invertebrate populations (Poysa and Virtanen 1994; Fast et al. 2004). Consequently, wetlands containing fish were avoided by broods. Fish may directly compete with ducklings for invertebrates, decreasing their growth and survival rates (Eriksson 1978; Hill et al 1987;

Bouffard and Hanson 1997; Cox et al. 1998). Fish reduce the number of invertebrates available to foraging broods, resulting in less efficient foraging and decreased growth rates (Hill et al. 1987; Cox et al. 1998). Decreased foraging efficiency requires ducklings to forage longer and move more often, increasing energy expenditure and the chances of predation (Hill et al. 1987; Bouffard and Hanson 1997).

3.6.5 Habitat Selection by Fall-staging Waterfowl

The drastic increase of waterfowl in the fall suggests that wetlands in the SMC provide important fall staging habitat. This is consistent with the findings of the Pasquia Project (Smith et al. 2002; Smith and Friedt 2003; Smith and Friedt 2004). The shallower water depths in drawdown wetlands were preferred by dabbling species, including the Mallard, but diving species, the Ring-necked Duck, and Blue-winged Teal selected for the deeper water of the non-drawdown wetlands. These results are consistent with those of Murkin et al. (1997) who also found that dabbling species preferred shallower water in the fall compared to diver species, and Blue-winged Teal shifted to deeper water.

The strong preference for the drawdown wetlands by dabbling species and Mallards may be due a shift in diet from invertebrates to aquatic vegetation. In the fall, invertebrates did not appear to influence habitat selection by waterfowl. This was expected as diets of dabbling species have been shown to shift from invertebrates to aquatic vegetation to provide a more carbohydrate-rich diet, necessary to supply the energy needed for migration (Dirschl 1969; Anderson and Low 1976; Sugden and Driver 1980; Murkin and Kadlec 1986; Baldassare and Bolen 2006). Even though I did not find that dabbling

species selected sites with greater amounts of submerged aquatic vegetation, this may be due to waterfowl selectively foraging on certain types of vegetation. The genera *Potamogeton* and *Stuckenia* are a preferred food source for many species of waterfowl (Krull 1970; Anderson and Low 1976). In this study, I found that *Potamogeton* and *Stuckenia* had a combined average biomass of 7.44 ± 6.91 g/point in the drawdown wetlands, which was significantly higher compared to the 5.31 ± 3.31 g/point in non-drawdown wetlands, according to a paired *t*-test ($p = 0.002$) (Appendix 8). Therefore, dabbling species may have preferred the drawdown wetlands as the water depth promoted the growth of these plants (Anderson 1978), and allowed dabbling species to efficiently forage on them.

Diver species and Ring-necked Ducks appeared to avoid areas with more submerged aquatic vegetation. Similar to dabbling species, this may have also been due to a shift in diet. Benthic invertebrates are an important food source for Ring-necked Ducks, particularly during the fall (Hohman and Eberhardt 1998; Thompson 1973). Therefore, dense submerged aquatic vegetation may impede foraging dives and were avoided (Monda and Ratti 1988).

Cover also appeared to be important to waterfowl at this time of year. This is also consistent with Murkin et al. (1997) who suggested that waterfowl select for areas with emergent vegetation to provide thermal protection during the fall migration.

3.7 Conclusions

Point counts and aerial surveys were effective for counting waterfowl at the plot and wetland scale, respectively. The double-observer method added reliability to the waterfowl abundance estimates by providing estimates of detection for observers and species.

The effects of the partial water-level drawdowns on waterfowl were variable depending on the species and time of year. Spring-breeding dabbler species, including Mallards, preferred the shallower water of the partial drawdown wetlands at both the wetland and plot scale as predicted. However, the relationship was not apparent until the second year of the drawdown, potentially caused by gradual response in the submerged aquatic vegetation and/or invertebrate communities. The preference for the shallow water was likely due to increased food accessibility and availability. As hypothesized, submerged aquatic vegetation, including *Sparganium*, as well as flying invertebrates were important food sources for dabblers. However, the avoidance of wetlands by dabblers with high abundances of fish was not expected.

As predicted, spring-breeding diver species, including Ring-necked Ducks, preferred the deeper water of the non-drawdown wetlands. The preference for deeper water was likely a result of providing greater mobility for foraging and escape. As expected, both flying and nektonic invertebrates were an important food source for diver species, due to the high protein demands of the birds at this time of year. Within the wetlands, open water areas were preferred, likely for escape cover.

During the post-breeding period, dabbling species of waterfowl, including Mallards, again showed a preference for the shallower water of the drawdown wetlands as predicted. However, the preference for wetlands with more open water by dabblers was not expected at this time of year. Diver species of waterfowl did not have a preference for any water depth during the post-breeding period. This was unexpected, and may have been due to most diver species of waterfowl emigrating from the study wetlands to find more suitable moulting habitat.

Broods of dabbling and diver species of waterfowl did not exhibit a preference for any water depth. This was not anticipated and suggests the depths within both the drawdown and non-drawdown wetlands met the habitat requirements of these species. As hypothesized, both dabbling and diver broods selected for wetlands with abundant invertebrates to provide a protein-rich food source for ducklings. Subsequently, wetlands with high abundances of fish were avoided, likely due to prey competition.

In the fall, the density of dabbling species on the drawdown wetlands increased, as predicted, due to improved accessibility to food and the high abundance of *Potamogeton* and *Stuckenia*. Alternatively, Blue-winged Teal and diver species, including Ring-necked Ducks preferred the deeper water of the non-drawdown wetlands. This relationship was expected for Ring-necked Ducks, but not for Blue-winged Teals. The deeper water was likely preferred by diver species as it may have provided them with greater mobility for

foraging on nektonic invertebrates, which are an important food source at this time of year.

Overall, the partial water-level drawdowns were preferred by dabbling species of waterfowl in nearly all life stages, excluding brooding. Alternatively, diving species of waterfowl preferred the deeper water of the non-drawdown wetlands. These findings highlight the need for active-water level management in the SMC in order to provide habitat for both dabbling and diving species of waterfowl. I suggest this be accomplished by staggering water levels within the marsh complex using wetlands undergoing a partial water-level drawdown and others held at Full Supply Level (see Chapter 4 for details).

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Appendix 3.1- Definitions of Waterfowl Species Notation

AMCO - American coot (*Fulica americana*)

AMWI - American wigeon (*Anas americana*)

BUFF - Bufflehead (*Bucephala albeola*)

BWTE - Blue-winged teal (*Anas discors*)

CAGO – Canada Goose (*Branta canadensis*)

CANV - Canvasback (*Aythya valisineria*)

GADW - Gadwall (*Anas strepera*)

GESC - Generic Scaup, includes Lesser Scaup (*Aythya affinis*) and Greater Scaup (*Aythya marila*)

GOEY – Common Goldeneye (*Bucephala clangula*)

GWTE - Green-winged teal (*Anas crecca*)

LESC - Lesser scaup (*Aythya affinis*)

MALL - Mallard (*Anas platyrhynchos*)

NOPI - Northern pintail (*Anas acuta*)

NOSH - Northern shoveler (*Anas clypeata*)

REDH - Redhead (*Aythya americana*)

RNDU - Ring-necked duck (*Aythya collaris*)

SNGO - Snow Goose (*Chen caerulescnes*)

UNKN - Unknown

WODU - Wood duck (*Aix sponsa*)

Appendix 3.2- Wetland-scale Habitat Model Results for Spring-breeding Waterfowl in 2009

Table 3.17- Wetland-scale habitat model results for the relative density of spring-breeding dabbling species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Dabblers						Wald 95% Confidence Limits		Wald Chi-Square	p
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower		
Average Open Water Depth (cm)	1.33	Negative Binomial	1	-0.0236	0.0104	-0.0439	-0.0033	5.18	0.0229
Water Area (ha)	1.44	Negative Binomial	1	-0.0101	0.0443	-0.0968	0.0767	0.05	0.8196
Vegetation Area (ha)	1.55	Negative Binomial	1	-0.0112	0.0072	-0.0254	0.0029	2.41	0.1205
Avg. SAV biomass/Point	1.29	Negative Binomial	1	-0.0060	0.0235	-0.0521	0.0401	0.06	0.7991
Avg. Inverts./ Activity Trap	1.91	Negative Binomial	1	0.0734	0.0792	-0.0817	0.2286	0.86	0.3536
Avg. Inverts./ Sticky Trap	1.23	Negative Binomial	1	0.0271	0.0115	0.0045	0.0497	5.52	0.0188
No. Fish/Trap Hour	1.75	Negative Binomial	1	-0.0615	0.0273	-0.1151	-0.0078	5.05	0.0246

Table 3.18- Wetland-scale habitat model results for the relative density of spring-breeding diver species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Divers						Wald 95% Confidence Limits		Wald Chi-Square	p
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower		
Average Open Water Depth (cm)	1.63	Negative Binomial	1	0.0190	0.0086	0.0021	0.0358	4.86	0.0274
Water Area (ha)	1.54	Negative Binomial	1	0.0124	0.0282	-0.0428	0.0677	0.20	0.6586
Vegetation Area (ha)	1.63	Negative Binomial	1	-0.0057	0.0052	-0.0160	0.0046	1.19	0.2758
Avg. SAV biomass/Point	1.44	Negative Binomial	1	0.0195	0.0213	-0.0222	0.0612	0.84	0.3596
Avg. Inverts./ Activity Trap	1.51	Negative Binomial	1	0.0521	0.0281	-0.0031	0.1072	3.42	0.0643
Avg. Inverts./ Sticky Trap	1.31	Negative Binomial	1	0.0100	0.0120	-0.0136	0.0335	0.69	0.4063
No. Fish/Trap Hour	1.66	Negative Binomial	1	0.0065	0.0205	-0.0337	0.0466	0.1	0.7525

Table 3.19- Wetland-scale habitat model results for the relative density of spring-breeding Blue-winged Teal in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Blue-winged Teal						Wald 95% Confidence Limits		Wald Chi-Square	<i>p</i>
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower		
Average Open Water Depth (cm)	1.08	Negative Binomial	1	-0.0204	0.0139	-0.0476	0.0068	2.16	0.1420
Water Area (ha)	1.29	Negative Binomial	1	0.0096	0.0472	-0.0830	0.1021	0.04	0.8394
Vegetation Area (ha)	1.39	Negative Binomial	1	-0.0161	0.0084	-0.0326	0.0003	3.70	0.0545
Avg. SAV biomass/Point	1.44	Negative Binomial	1	0.0074	0.0267	-0.0450	0.0598	0.08	0.7807
Avg. Inverts./ Activity Trap	1.97	Negative Binomial	1	0.0990	0.0828	-0.0633	0.2614	1.43	0.2319
Avg. Inverts./ Sticky Trap	1.38	Negative Binomial	1	0.0294	0.0132	0.0035	0.0554	4.94	0.0262
No. Fish/Trap Hour	1.26	Negative Binomial	1	-0.0556	0.0279	-0.1103	-0.0009	3.97	0.0462

Table 3.20- Wetland-scale habitat model results for the relative density of spring-breeding Mallard in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Mallard						Wald 95% Confidence Limits		Wald Chi-Square	<i>p</i>
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower		
Average Open Water Depth (cm)	1.43	Negative Binomial	1	-0.0212	0.0100	-0.0409	-0.0016	4.50	0.0340
Water Area (ha)	1.36	Negative Binomial	1	-0.0229	0.0398	-0.1009	0.0552	0.33	0.5660
Vegetation Area (ha)	1.53	Negative Binomial	1	-0.0109	0.0064	-0.0235	0.0016	2.91	0.0880
Avg. SAV biomass/Point	1.15	Negative Binomial	1	-0.0063	0.0229	-0.0513	0.0387	0.08	0.7837
Avg. Inverts./ Activity Trap	1.63	Negative Binomial	1	0.0645	0.0543	-0.0418	0.1709	1.42	0.2342
Avg. Inverts./ Sticky Trap	1.29	Negative Binomial	1	0.0215	0.0122	-0.0024	0.0455	3.11	0.0780
No. Fish/Trap Hour	1.52	Negative Binomial	1	-0.0692	0.0262	-0.1205	-0.0179	6.99	0.0082

Table 3.21- Wetland-scale habitat model results for the relative density of spring-breeding Ring-necked Duck in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Ring-necked Duck						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.67	Negative Binomial	1	0.0194	0.0085	0.0027	0.0361	5.17	0.0230
Water Area (ha)	1.56	Negative Binomial	1	0.0154	0.0285	-0.0404	0.0712	0.29	0.5894
Vegetation Area (ha)	1.65	Negative Binomial	1	-0.0048	0.0054	-0.0154	0.0058	0.79	0.3735
Avg. SAV biomass/Point	1.45	Negative Binomial	1	0.0177	0.0220	-0.0253	0.0608	0.65	0.4199
Avg. Inverts./Activity Trap	1.55	Negative Binomial	1	0.0488	0.0291	-0.0082	0.1058	2.81	0.0936
Avg. Inverts./Sticky Trap	1.35	Negative Binomial	1	0.0107	0.0121	-0.0130	0.0343	0.78	0.3763
No. Fish/Trap Hour	1.75	Negative Binomial	1	0.0094	0.027	-0.0435	0.0623	0.12	0.7268

Appendix 3.3- Wetland-scale Habitat Model Results for Post-breeding Waterfowl in 2009

Table 3.22- Wetland-scale habitat model results for the relative density of post-breeding dabbling species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Dabblers						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.55	Negative Binomial	1	-0.0064	0.0037	-0.0136	0.0009	2.97	0.0846
Water Area (ha)	1.41	Negative Binomial	1	0.0073	0.0134	-0.0190	0.0336	0.30	0.5866
Vegetation Area (ha)	1.53	Negative Binomial	1	0.0026	0.0022	-0.0018	0.0070	1.36	0.2438
Avg. SAV biomass/Point	1.44	Negative Binomial	1	-0.0051	0.0084	-0.0216	0.0113	0.38	0.5387
Avg. Inverts./ Activity Trap	1.55	Negative Binomial	1	-0.0282	0.0146	-0.0567	0.0004	3.73	0.0534
Avg. Inverts./ Sticky Trap	1.41	Negative Binomial	1	0.0005	0.0059	-0.0110	0.0120	0.01	0.9322
No. Fish/Trap Hour	1.45	Negative Binomial	1	0.0028	0.0087	-0.0144	0.0199	0.1	0.7523

Table 3.23- Wetland-scale habitat model results for the relative density of post-breeding diver species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Divers						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	0.99	Negative Binomial	1	0.0054	0.0157	-0.0254	0.0361	0.12	0.7319
Water Area (ha)	0.98	Negative Binomial	1	-0.0242	0.0365	-0.0958	0.0474	0.44	0.5076
Vegetation Area (ha)	1.63	Negative Binomial	1	-0.0161	0.0040	-0.0239	-0.0083	16.39	<.0001
Avg. SAV biomass/Point	1.21	Negative Binomial	1	0.0168	0.0252	-0.0326	0.0663	0.45	0.5042
Avg. Inverts./ Activity Trap	1.37	Negative Binomial	1	0.0922	0.0494	-0.0046	0.1889	3.49	0.0619
Avg. Inverts./ Sticky Trap	1.07	Negative Binomial	1	0.0032	0.0165	-0.0291	0.0356	0.04	0.8453
No. Fish/Trap Hour	0.83	Negative Binomial	1	-0.0381	0.0238	-0.0848	0.0085	2.57	0.1092

Table 3.24- Wetland-scale habitat model results for the relative density of post-breeding Blue-winged Teal in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Blue-winged Teal						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.56	Negative Binomial	1	0.0109	0.0104	-0.0095	0.0313	1.10	0.2943
Water Area (ha)	1.26	Negative Binomial	1	0.0093	0.0334	-0.0561	0.0747	0.08	0.7802
Vegetation Area (ha)	1.28	Negative Binomial	1	0.0005	0.0063	-0.0119	0.0129	0.01	0.9334
Avg. SAV biomass/Point	1.12	Negative Binomial	1	-0.0269	0.0236	-0.0731	0.0194	1.30	0.2547
Avg. Inverts./Activity Trap	1.04	Negative Binomial	1	-0.0407	0.0422	-0.1235	0.0420	0.93	0.3350
Avg. Inverts./Sticky Trap	1.33	Negative Binomial	1	0.0009	0.0148	-0.0280	0.0299	0.00	0.9498
No. Fish/Trap Hour	1.43	Negative Binomial	1	0.0325	0.0188	-0.0043	0.0693	3	0.0833

Table 3.25- Wetland-scale habitat model results for the relative density of post-breeding Mallard in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Mallard						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.27	Negative Binomial	1	-0.0223	0.0056	-0.0333	-0.0114	15.99	<.0001
Water Area (ha)	1.44	Negative Binomial	1	0.0040	0.0318	-0.0584	0.0663	0.02	0.9007
Vegetation Area (ha)	1.28	Negative Binomial	1	0.0042	0.0053	-0.0063	0.0147	0.62	0.4308
Avg. SAV biomass/Point	1.47	Negative Binomial	1	0.0058	0.0169	-0.0273	0.0389	0.12	0.7300
Avg. Inverts./Activity Trap	1.40	Negative Binomial	1	-0.0154	0.0432	-0.1000	0.0693	0.13	0.7220
Avg. Inverts./Sticky Trap	1.45	Negative Binomial	1	-0.0007	0.0132	-0.0267	0.0252	0.00	0.9552
No. Fish/Trap Hour	1.32	Negative Binomial	1	-0.0323	0.0164	-0.0644	-0.0003	3.91	0.0481

Table 3.26- Wetland-scale habitat model results for the relative density of post-breeding Ring-necked Duck in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Ring-necked Duck						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	0.93	Negative Binomial	1	0.0042	0.0171	-0.0293	0.0377	0.06	0.8046
Water Area (ha)	0.95	Negative Binomial	1	-0.0267	0.0397	-0.1044	0.0510	0.45	0.5009
Vegetation Area (ha)	1.54	Negative Binomial	1	-0.0187	0.0048	-0.0281	-0.0093	15.30	<.0001
Avg. SAV biomass/Point	1.14	Negative Binomial	1	0.0177	0.0279	-0.0370	0.0724	0.40	0.5251
Avg. Inverts./Activity Trap	1.38	Negative Binomial	1	0.1093	0.0573	-0.0030	0.2216	3.64	0.0565
Avg. Inverts./Sticky Trap	1.03	Negative Binomial	1	0.0041	0.0182	-0.0315	0.0398	0.05	0.8198
No. Fish/Trap Hour	0.76	Negative Binomial	1	-0.0495	0.0252	-0.0989	0	3.85	0.0498

Appendix 3.4- Wetland-scale Habitat Model Results for Waterfowl Broods in 2008 and 2009

Table 3.27- Wetland-scale habitat model results for the relative density of dabbler broods in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Dabbler Broods						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	p
Average Open Water Depth (cm)	0.65	Poisson	1	-0.0151	0.0112	-0.0371	0.0069	1.82	0.1777
Water Area (ha)	0.84	Poisson	1	-0.0107	0.0328	-0.0750	0.0536	0.11	0.7450
No. Fish/Trap Hour	0.41	Poisson	1	-0.0623	0.0329	-0.1269	0.0022	3.58	0.0585
Vegetation Area (ha)	0.63	Poisson	1	-0.0079	0.0059	-0.0194	0.0036	1.81	0.1783

Table 3.28- Wetland-scale habitat model results for the relative density of diver broods in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Diver Broods						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	p
Average Open Water Depth (cm)	1.73	Negative Binomial	1	0.0241	0.0320	-0.0386	0.0868	0.57	0.4516
Water Area (ha)	2.05	Negative Binomial	1	0.0513	0.0900	-0.1250	0.2277	0.33	0.5684
No. Fish/Trap Hour	2.01	Negative Binomial	1	-0.0285	0.0640	-0.1539	0.0969	0.20	0.6558
Vegetation Area (ha)	1.71	Negative Binomial	1	-0.0108	0.0165	-0.0431	0.0216	0.43	0.5141

Table 3.29- Wetland-scale habitat model results for the relative density of dabbler broods in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Dabbler Broods						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	p
Average Open Water Depth (cm)	1.41	Negative Binomial	1	-0.0106	0.0168	-0.0436	0.0223	0.40	0.5274
Water Area (ha)	1.13	Negative Binomial	1	0.0349	0.0474	-0.0581	0.1278	0.54	0.4623
Vegetation Area (ha)	1.28	Negative Binomial	1	-0.0124	0.0094	-0.0309	0.0060	1.74	0.1875
Avg. SAV biomass/Point	1.44	Poisson	1	0.0427	0.0191	0.0053	0.0801	5.01	0.0253
Avg. Inverts./Activity Trap	1.03	Poisson	1	0.1092	0.0405	0.0298	0.1885	7.28	0.0070
Avg. Inverts./Sticky Trap	1.76	Negative Binomial	1	0.0229	0.0209	-0.0180	0.0639	1.20	0.2728
No. Fish/Trap Hour	1.22	Negative Binomial	1	-0.073	0.0507	-0.1723	0.0264	2.07	0.1500

Table 3.30- Wetland-scale habitat model results for the relative density of diver broods in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Diver Broods						Wald 95% Confidence Limits			
Models/Variables	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	p
Average Open Water Depth (cm)	0.64	Poisson	1	0.0007	0.0151	-0.0289	0.0303	0.00	0.9636
Water Area (ha)	0.61	Poisson	1	0.0175	0.0452	-0.0711	0.1061	0.15	0.6991
Vegetation Area (ha)	0.50	Poisson	1	-0.0094	0.0072	-0.0235	0.0048	1.68	0.1944
Avg. SAV biomass/Point	0.64	Poisson	1	0.0127	0.0269	-0.0401	0.0655	0.22	0.6363
Avg. Inverts./ Activity Trap	0.44	Poisson	1	0.0715	0.0442	-0.0152	0.1582	2.61	0.1061
Avg. Inverts./ Sticky Trap	0.35	Poisson	1	0.0251	0.0135	-0.0013	0.0515	3.47	0.0625
No. Fish/Trap Hour	1.54	Negative Binomial	1	-0.0099	0.0267	-0.0622	0.0425	0.14	0.7121

Appendix 3.5- Wetland-scale Habitat Model Results for Fall-staging Waterfowl in 2008 and 2009

Table 3.31- Wetland-scale habitat model results for the relative density of fall-staging dabbling species in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Dabblers						Wald 95% Confidence Limits			
Model	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.62	Negative Binomial	1	-0.0623	0.015	-0.0916	-0.0329	17.27	<.0001
Water Area (ha)	0.85	Negative Binomial	1	0.0833	0.064	-0.0421	0.2088	1.69	0.1930
Vegetation Area (ha)	1.16	Negative Binomial	1	0.0124	0.0164	-0.0198	0.0446	0.57	0.4504
No. Fish/Trap Hour	0.92	Negative Binomial	1	-0.0975	0.0378	-0.1717	-0.0234	6.64	0.0099

Table 3.32- Wetland-scale habitat model results for the relative density of fall-staging diver species in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Divers						Wald 95% Confidence Limits			
Model	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.36	Negative Binomial	1	0.0546	0.0240	0.0076	0.1017	5.18	0.0229
Water Area (ha)	1.22	Negative Binomial	1	0.1223	0.0789	-0.0323	0.2769	2.41	0.1209
Vegetation Area (ha)	0.83	Negative Binomial	1	0.0257	0.0125	0.0012	0.0501	4.24	0.0394
No. Fish/Trap Hour	2.60	Poisson	1	0.1004	0.0295	0.0426	0.1581	11.61	0.0007

Table 3.33- Wetland-scale habitat model results for the relative density of fall-staging dabbling species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Dabblers						Wald 95% Confidence Limits			
Model	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.14	Negative Binomial	1	-0.0299	0.0065	-0.0427	-0.0172	21.18	<.0001
Water Area (ha)	1.27	Negative Binomial	1	-0.0101	0.0465	-0.1013	0.0811	0.05	0.8285
Vegetation Area (ha)	1.23	Negative Binomial	1	0.0028	0.0078	-0.0124	0.0180	0.13	0.7186
Avg. SAV biomass/Point	1.23	Negative Binomial	1	-0.0081	0.0219	-0.0509	0.0348	0.14	0.7120
Avg. Inverts./ Activity Trap	1.25	Negative Binomial	1	-0.0558	0.0674	-0.1878	0.0763	0.69	0.4076
Avg. Inverts./ Sticky Trap	1.12	Negative Binomial	1	0.0074	0.0176	-0.0271	0.0418	0.18	0.6757
No. Fish/Trap Hour	1.08	Negative Binomial	1	-0.0379	0.0223	-0.0816	0.0058	2.89	0.0893

Table 3.34- Wetland-scale habitat model results for the relative density of fall-staging diver species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Divers						Wald 95% Confidence Limits			
Model	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.19	Negative Binomial	1	0.0928	0.0381	0.0181	0.1676	5.92	0.0149
Water Area (ha)	1.24	Negative Binomial	1	-0.1197	0.0645	-0.2462	0.0067	3.44	0.0635
Vegetation Area (ha)	0.86	Negative Binomial	1	-0.0168	0.0121	-0.0405	0.0070	1.92	0.1660
Avg. SAV biomass/Point	0.63	Negative Binomial	1	-0.1614	0.0739	-0.3063	-0.0165	4.77	0.0290
Avg. Inverts./ Activity Trap	1.41	Negative Binomial	1	-0.0721	0.2019	-0.4678	0.3235	0.13	0.7209
Avg. Inverts./ Sticky Trap	1.01	Negative Binomial	1	-0.0865	0.0384	-0.1618	-0.0113	5.08	0.0242
No. Fish/Trap Hour	1.27	Negative Binomial	1	-0.0041	0.0721	-0.1454	0.1372	0	0.9547

Table 3.35- Wetland-scale habitat model results for the relative density of fall-staging Blue-winged Teal in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Blue-winged Teal						Wald 95% Confidence Limits			
Model	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower	Wald Chi-Square	<i>p</i>
Average Open Water Depth (cm)	1.93	Negative Binomial	1	0.0301	0.0112	0.0082	0.0520	7.24	0.0071
Water Area (ha)	1.17	Negative Binomial	1	0.0006	0.0364	-0.0708	0.0721	0.00	0.9863
Vegetation Area (ha)	1.16	Negative Binomial	1	0.0009	0.0070	-0.0128	0.0146	0.02	0.8956
Avg. SAV biomass/Point	1.29	Negative Binomial	1	0.0095	0.0293	-0.0479	0.0669	0.10	0.7461
Avg. Inverts./ Activity Trap	1.00	Negative Binomial	1	-0.0457	0.0513	-0.1462	0.0549	0.79	0.3733
Avg. Inverts./ Sticky Trap	1.80	Negative Binomial	1	-0.0411	0.0169	-0.0742	-0.0081	5.95	0.0147
No. Fish/Trap Hour	1.29	Negative Binomial	1	0.0314	0.0227	-0.0132	0.076	1.91	0.1672

Table 3.36- Wetland-scale habitat model results for the relative density of fall-staging Mallard in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Mallard						Wald 95% Confidence Limits		Wald Chi-Square	<i>p</i>
Model	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower		
Average Open Water Depth (cm)	1.32	Negative Binomial	1	-0.0395	0.0061	-0.0514	-0.0275	41.90	<.0001
Water Area (ha)	1.25	Negative Binomial	1	-0.0071	0.0594	-0.1235	0.1092	0.01	0.9045
Vegetation Area (ha)	1.22	Negative Binomial	1	0.0032	0.0098	-0.0160	0.0224	0.10	0.7464
Avg. SAV biomass/Point	1.20	Negative Binomial	1	-0.0082	0.0257	-0.0585	0.0422	0.10	0.7501
Avg. Inverts./ Activity Trap	1.19	Negative Binomial	1	-0.0524	0.0869	-0.2227	0.1179	0.36	0.5468
Avg. Inverts./ Sticky Trap	1.08	Negative Binomial	1	0.0134	0.0209	-0.0277	0.0544	0.41	0.5230
No. Fish/Trap Hour	1.13	Negative Binomial	1	-0.0556	0.0254	-0.1054	-0.0057	4.77	0.0289

Table 3.37- Wetland-scale habitat model results for the relative density of fall-staging Ring-necked Duck in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Ring-necked Duck						Wald 95% Confidence Limits		Wald Chi-Square	<i>p</i>
Model	Pearson Chi-square/df	Distribution	DF	Estimate	Standard Error	Upper	Lower		
Average Open Water Depth (cm)	0.81	Negative Binomial	1	0.1100	0.0478	0.0163	0.2036	5.29	0.0214
Water Area (ha)	0.91	Negative Binomial	1	-0.1169	0.0828	-0.2792	0.0454	1.99	0.1581
Vegetation Area (ha)	0.59	Negative Binomial	1	-0.0160	0.0132	-0.0418	0.0098	1.47	0.2248
Avg. SAV biomass/Point	0.41	Negative Binomial	1	-0.2882	0.1415	-0.5654	-0.0109	4.15	0.0417
Avg. Inverts./ Activity Trap	1.21	Negative Binomial	1	-0.2163	0.4042	-1.0084	0.5759	0.29	0.5926
Avg. Inverts./ Sticky Trap	1.44	Negative Binomial	1	-0.2853	0.1170	-0.5146	-0.0559	5.94	0.0148
No. Fish/Trap Hour	0.87	Negative Binomial	1	0.0012	0.087	-0.1694	0.1718	0	0.9891

Appendix 3.6- Plot-scale Habitat Model Results for Spring-breeding Waterfowl in 2008 and 2009

Table 3.38- Plot-scale habitat model results for the relative density of spring-breeding dabbling species in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	-0.0181	0.0130	18	-1.39	0.1815
Cover	Water Area (ha)	0.1982	0.1008	17	1.97	0.0657
	Water Edge Density (km/ha)	-1.2155	1.6580	17	-0.73	0.4735
Veg. Composition	Sparganium Area (ha)	8.7303	3.5636	13	2.45	0.0292
	Carex Area (ha)	-1.2846	0.6012	13	-2.14	0.0522
	Equisetum Area (ha)	11.7885	4.8443	13	2.43	0.0301
	Phragmites Area (ha)	-0.3364	0.2100	13	-1.60	0.1331
	Scirpus Area (ha)	-1.0785	0.4706	13	-2.29	0.0393
	Typha Area (ha)	-3.3240	1.0757	13	-3.09	0.0086

Table 3.39- Plot-scale habitat model results for the relative density of spring-breeding diver species in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	0.0026	0.0071	18	0.36	0.7213
Cover	Water Area (ha)	0.1489	0.0715	17	2.08	0.0527
	Water Edge Density (km/ha)	0.3595	0.8150	17	0.44	0.6647
Veg. Composition	Sparganium Area (ha)	0.8577	0.7344	13	1.17	0.2639
	Carex Area (ha)	0.0847	0.0896	13	0.95	0.3616
	Equisetum Area (ha)	-0.6639	0.8418	13	-0.79	0.4445
	Phragmites Area (ha)	-0.0406	0.0738	13	-0.55	0.5915
	Scirpus Area (ha)	-0.0883	0.0663	13	-1.33	0.2059
	Typha Area (ha)	-0.4597	0.1791	13	-2.57	0.0234

Table 3.40- Plot-scale habitat model results for the relative density of spring-breeding Blue-winged Teal in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	-0.0103	0.0074	18	-1.39	0.1822
Cover	Water Area (ha)	0.0256	0.0693	17	0.37	0.7160
	Water Edge Density (km/ha)	-1.5471	1.0944	17	-1.41	0.1755
Veg. Composition	Sparganium Area (ha)	-0.0655	0.9771	13	-0.07	0.9476
	Carex Area (ha)	-0.1131	0.0879	13	-1.29	0.2207
	Equisetum Area (ha)	0.2769	0.6014	13	0.46	0.6528
	Phragmites Area (ha)	-0.1385	0.0848	13	-1.63	0.1264
	Scirpus Area (ha)	-0.1351	0.0840	13	-1.61	0.1319
	Typha Area (ha)	-0.1000	0.1262	13	-0.79	0.4423

Table 3.41- Plot-scale habitat model results for the relative density of spring-breeding Mallard in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	-0.0199	0.0163	18	-1.22	0.2397
Cover	Water Area (ha)	0.2754	0.1299	17	2.12	0.0490
	Water Edge Density (km/ha)	-0.7234	1.9460	17	-0.37	0.7147
Veg. Composition	Sparganium Area (ha)	3.5474	1.1042	13	3.21	0.0068
	Carex Area (ha)	-0.5413	0.2169	13	-2.50	0.0268
	Equisetum Area (ha)	2.3373	1.3651	13	1.71	0.1106
	Phragmites Area (ha)	-0.2370	0.0965	13	-2.46	0.0289
	Scirpus Area (ha)	-0.3490	0.1110	13	-3.14	0.0078
	Typha Area (ha)	-0.9509	0.3515	13	-2.71	0.0180

Table 3.42- Plot-scale habitat model results for the relative density of spring-breeding Ring-necked Duck in the Summerberry Marsh Complex, 2008. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	0.0101	0.0079	18	1.28	0.2158
Cover	Water Area (ha)	0.2256	0.0876	17	2.58	0.0196
	Water Edge Density (km/ha)	1.3755	0.8640	17	1.59	0.1298
Veg. Composition	Sparganium Area (ha)	1.4965	0.7169	13	2.09	0.0571
	Carex Area (ha)	0.1542	0.0995	13	1.55	0.1451
	Equisetum Area (ha)	-0.5533	0.7803	13	-0.71	0.4908
	Phragmites Area (ha)	0.0225	0.0735	13	0.31	0.7646
	Scirpus Area (ha)	-0.0368	0.0647	13	-0.57	0.5786
	Typha Area (ha)	-0.4360	0.1717	13	-2.54	0.0247

Table 3.43- Plot-scale habitat model results for the relative density of spring-breeding dabbling species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	-0.0173	0.0082	33	-2.12	0.0412
Cover	Water Area (ha)	0.1851	0.0887	32	2.09	0.0449
	Water Edge Density (km/ha)	-1.2557	0.7874	32	-1.59	0.1206
Veg. Composition	Sparganium Area (ha)	1.5853	0.8397	28	1.89	0.0695
	Carex Area (ha)	0.0138	0.1223	28	0.11	0.9107
	Equisetum Area (ha)	-0.6389	0.4811	28	-1.33	0.1949
	Phragmites Area (ha)	-0.2510	0.1308	28	-1.92	0.0652
	Scirpus Area (ha)	-0.1064	0.0811	28	-1.31	0.2002
	Typha Area (ha)	-0.0888	0.0955	28	-0.93	0.3601
Food	Avg. SAV Biomass/Pt (g)	-0.0336	0.0215	31	-1.56	0.1278
	Total Inverts. Activity Trap	0.0001	0.0011	31	0.06	0.9531
	Total Inverts. Sticky Trap	0.0017	0.0005	31	3.50	0.0014

Table 3.44- Plot-scale habitat model results for the relative density of spring-breeding diver species in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	0.0038	0.0092	33	0.41	0.6844
Cover	Water Area (ha)	0.3776	0.1147	32	3.29	0.0024
	Water Edge Density (km/ha)	-1.9065	0.8806	32	-2.16	0.0380
Veg. Composition	Sparganium Area (ha)	0.6755	0.9144	28	0.74	0.4662
	Carex Area (ha)	0.3085	0.0695	28	4.44	0.0001
	Equisetum Area (ha)	-1.1967	0.4856	28	-2.46	0.0201
	Phragmites Area (ha)	-0.1937	0.1271	28	-1.52	0.1388
	Scirpus Area (ha)	-0.0036	0.0776	28	-0.05	0.9630
	Typha Area (ha)	-0.6171	0.2401	28	-2.57	0.0158
Food	Avg. SAV Biomass/Pt (g)	0.0334	0.0200	31	1.67	0.1058
	Total Inverts. Activity Trap	0.0023	0.0008	31	3.11	0.0040
	Total Inverts. Sticky Trap	0.0015	0.0008	31	1.93	0.0632

Table 3.45- Plot-scale habitat model results for the relative density of spring-breeding Blue-winged Teal in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	-0.0257	0.0122	33	-2.11	0.0426
Cover	Water Area (ha)	0.1568	0.1160	32	1.35	0.1857
	Water Edge Density (km/ha)	-1.8712	1.2403	32	-1.51	0.1412
Veg. Composition	Sparganium Area (ha)	1.3785	1.2473	28	1.11	0.2785
	Carex Area (ha)	0.0540	0.1738	28	0.31	0.7583
	Equisetum Area (ha)	-1.4443	0.6750	28	-2.14	0.0412
	Phragmites Area (ha)	-0.6717	0.2383	28	-2.82	0.0088
	Scirpus Area (ha)	-0.2779	0.1438	28	-1.93	0.0635
	Typha Area (ha)	0.0058	0.0971	28	0.06	0.9529
Food	Avg. SAV Biomass/Pt (g)	-0.1215	0.0323	31	-3.76	0.0007
	Total Inverts. Activity Trap	-0.0062	0.0044	31	-1.42	0.1661
	Total Inverts. Sticky Trap	0.0034	0.0004	31	7.68	<.0001

Table 3.46- Plot-scale habitat model results for the relative density of spring-breeding Mallard in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	-0.0101	0.0087	33	-1.16	0.2553
Cover	Water Area (ha)	0.1706	0.1024	32	1.67	0.1055
	Water Edge Density (km/ha)	-0.9675	0.8630	32	-1.12	0.2706
Veg. Composition	Sparganium Area (ha)	1.1951	0.8480	28	1.41	0.1698
	Carex Area (ha)	0.0176	0.1455	28	0.12	0.9048
	Equisetum Area (ha)	-0.6295	0.5836	28	-1.08	0.2900
	Phragmites Area (ha)	-0.1952	0.1469	28	-1.33	0.1947
	Scirpus Area (ha)	-0.0307	0.0822	28	-0.37	0.7120
	Typha Area (ha)	-0.2374	0.1651	28	-1.44	0.1615
Food	Avg. SAV Biomass/Pt (g)	-0.0202	0.0235	31	-0.86	0.3968
	Total Inverts. Activity Trap	0.0009	0.0009	31	1.03	0.3126
	Total Inverts. Sticky Trap	0.0008	0.0007	31	1.20	0.2390

Table 3.47- Plot-scale habitat model results for the relative density of spring-breeding Ring-necked Duck in the Summerberry Marsh Complex, 2009. Highlighted rows indicate significant variables ($\alpha = 0.1$).

Model	Variables	Estimate	Std Error	DF	t-value	p
Treatment	Average Water Depth (cm)	0.0034	0.0102	33	0.34	0.7384
Cover	Water Area (ha)	0.4214	0.1290	32	3.27	0.0026
	Water Edge Density (km/ha)	-2.3625	1.0230	32	-2.31	0.0275
Veg. Composition	Sparganium Area (ha)	0.6385	1.1535	28	0.55	0.5843
	Carex Area (ha)	0.3537	0.0820	28	4.31	0.0002
	Equisetum Area (ha)	-0.9097	0.5092	28	-1.79	0.0849
	Phragmites Area (ha)	-0.2010	0.1585	28	-1.27	0.2151
	Scirpus Area (ha)	-0.0286	0.1111	28	-0.26	0.7986
	Typha Area (ha)	-0.6742	0.2931	28	-2.30	0.0291
Food	Avg. SAV Biomass/Pt (g)	0.0450	0.0217	31	2.07	0.0464
	Total Inverts. Activity Trap	0.0026	0.0008	31	3.19	0.0033
	Total Inverts. Sticky Trap	0.0017	0.0009	31	2.00	0.0545

Appendix 3.7- Results of Forage Fish, Invertebrate, and Submerged Aquatic Vegetation Sampling

Table 3.48- Number of fish caught per trap hour in 2008 and 2009. Data are absent for wetlands 34HI and 35HI in 2008. Note: in wetland 32C in 2009 the numbers of fish caught were too numerous to identify and were therefore labeled “Mixed Species”, these were a combination of Brook Stickleback and Fathead Minnow.

Wetland	Brook Stickleback		Central Mudminnow		Fathead Minnow		Northern Pike		Mixed Species		Total	
	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009
14R	2.06	1.45	0	0	0	0	0	0	--	--	2.06	1.45
35HI	--	0	--	0	--	0.00	--	0.04	--	--	--	0.04
37C	0.21	0.01	0	0.10	0.02	0.00	0.02	0.01	--	--	0.25	0.12
21C	0.02	0.66	0.25	0.82	1.40	0.13	0	0	--	--	1.67	1.61
32C	14.56	0	0.15	0.11	10.75	0	0	0	--	27.24	25.46	27.35
34HI	--	0.30	--	0.23	--	0	--	0	--	--	--	0.53

Table 3.49- Average number and standard deviation of nektonic invertebrates caught per activity trap during each survey round in 2009 in the Summerberry Marsh Complex, Manitoba. Shaded columns indicate drawdown wetlands.

	14R		35HI		37C		21C		32C		34HI	
	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev
May 15 - May 23 2009	2	0.22	4	0.38	2	0.15	1	0.07	1	0.16	4	0.31
May 15 - May 23 2009	7	0.36	10	0.77	5	0.41	33	3.28	2	0.24	5	0.52
May 15 - May 23 2009	22	1.63	10	0.38	11	0.71	23	1.14	3	0.25	13	1.02
May 15 - May 23 2009	9	0.53	25	4.04	9	0.57	21	1.62	6	0.61	19	2.09

Table 3.50- Average number and standard deviation of flying invertebrates caught per sticky trap during each survey round in 2009 in the Summerberry Marsh Complex, Manitoba. Shaded columns indicate drawdown wetlands.

	14R		35HI		37C		21C		32C		34HI	
	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev	Avg./Pt	Std Dev
May 15 - May 23 2009	3	0.17	25	0.69	20	1.73	11	0.50	44	1.63	32	2.00
May 15 - May 23 2009	109	7.34	74	3.03	78	4.19	122	4.65	62	2.59	62	3.62
May 15 - May 23 2009	177	6.52	66	2.05	116	3.22	112	5.13	63	2.10	58	2.35
May 15 - May 23 2009	80	3.13	83	3.16	50	1.81	119	4.62	67	3.45	67	5.55

Table 3.51- Average weight (g) and standard deviation of submerged aquatic vegetation genera in 2009 from the Summerberry Marsh Complex, Manitoba. Shaded columns indicate drawdown wetlands.

Wetland	14R		35HI		37C		21C		32C		34HI	
	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.	Avg. Wt. (g)	Std. Dev.
<i>Ceratophyllum</i>	3.90	0.40	1.64	0.18	0.92	0.10	6.76	0.33	0.33	0.06	3.53	0.52
<i>Chara</i>	0.00	0.00	2.29	0.31	0.38	0.07	4.24	0.65	0.77	0.11	0.11	0.03
<i>Elodea</i>	0.33	0.05	2.59	0.52	1.38	0.18	0.00	0.00	0.01	0.00	4.00	0.53
<i>Fontinalis</i>	0.00	0.00	1.41	0.19	0.05	0.01	0.00	0.00	0.03	0.01	0.00	0.00
<i>Myriophyllum</i>	1.60	0.21	1.51	0.14	0.59	0.11	1.71	0.25	4.13	0.38	1.55	0.14
<i>Potamogeton</i>	3.89	0.28	5.54	0.56	4.95	0.41	0.93	0.07	1.44	0.13	3.32	0.27
<i>Sagittaria</i>	0.17	0.04	0.01	0.00	0.73	0.07	0.00	0.00	0.15	0.03	0.00	0.00
<i>Sparganium</i>	0.20	0.04	4.89	0.45	1.27	0.15	5.33	0.66	5.29	0.62	0.02	0.01
<i>Stuckenia</i>	0.30	0.05	3.05	0.14	2.49	0.18	0.30	0.05	3.44	0.40	0.27	0.06
<i>Utricularia</i>	0.00	0.00	0.00	0.00	0.09	0.01	0.00	0.00	0.36	0.06	0.00	0.00

Chapter 4 - Management Implications and Future Work

4.1 Management Implications

The partial water-level drawdowns had both positive and negative effects on waterbirds in the SMC. The partial drawdowns circumvented the apparent problems associated with complete water-level drawdowns, such as the creation of monotypic vegetation communities and floating mats of vegetation (see Chapter 1). The partial drawdowns also appeared to promote the growth of submerged aquatic vegetation genera, including *Potamogeton* and *Stuckenia*, which are an important food source for waterfowl (Dirschl 1969; Anderson and Low 1976). However, the partial drawdowns also negatively impacted diverse species of waterfowl and several species of marsh birds, including American Bitterns, American Coots, and Pied-billed Grebes. These findings highlight the need to create a diverse range of wetland habitats to meet the varied requirements of avian species in the SMC.

Due to the numerous, upstream anthropogenic developments on the Saskatchewan River, the frequency and extent of flooding events in the SRD has been greatly diminished (Chapter 1). While the explicit effects of reduced flooding on wetlands in the delta are unknown, the moderation of this disturbance has likely reduced the diversity of wetland habitats in the delta. In an attempt to mitigate this, active water-level management should be performed in the SRD to create a diversity of wetland habitats. Wetland diversity will provide a wide range of available habitats to waterbirds and help and promote avian biodiversity.

Water-level management should take place on a wetland-complex scale rather than basin scale; where possible, water levels should be staggered in the wetland complex to provide a mosaic of deep and shallow wetlands, creating a diverse range of habitats available in relatively close proximity. Partial water-level drawdowns could be used as shallow wetland habitat to promote the use by dabbler species of waterfowl, whereas the Full Supply Level could be used as deep water habitat to promote the use by diver species of waterfowl and marsh birds.

One of the major reasons for creating a mosaic of deep and shallow wetlands is to control forage fish populations. Forage fish abundance appeared to play an important role in attracting piscivorous avian species, including American Bitterns and Pied-billed Grebes. However, forage fish also compete for invertebrate prey and negatively impacted the use of wetlands by waterfowl and insectivorous marsh birds, including Soras and Virginia Rails. Water depth can play an important role in fish abundance by regulating the population through overwinter survival. Partial drawdowns may reduce forage fish abundance by promoting ice formation to the wetland substrate and anoxic conditions during winter. This would benefit insectivorous marsh birds and waterfowl, but would be detrimental to piscivorous species. Therefore, water levels with the marsh complex should be staggered, providing habitat for piscivorous and insectivorous avian species. This could be accomplished by keeping some wetlands at Full Supply Level during the winter to provide deep water areas to promote overwinter survival of forage fish, creating habitat for piscivorous species. Other wetlands could be partially drawn down to reduce

forage fish populations and simultaneously create habitat for dabbling species of waterfowl and insectivorous species of marsh birds.

The timing of partial drawdowns or refloods should coincide with low avian use to avoid disturbance. Ideally, this would occur in late fall, following the peak of waterfowl migration. Partial drawdowns or refloods should not be performed in the spring or summer, to avoid disturbance of nesting waterfowl and marsh birds.

The depth of the partial drawdowns should promote the growth of submerged aquatic vegetation, but not expose mudflats, unless the goal is to establish more emergent vegetation. In this study, the average open water depth of the partial drawdowns was 54 cm, which may promote the growth of *Potamogeton* and *Stuckenia* (Anderson 1978). The high water wetlands should be kept at the Full Supply Level, to provide deep, open water for diving species of waterfowl, and over-water nesting and foraging sites for marsh birds. In this study, the average open water depth of the non-drawdown wetlands was 89 cm, which appeared to meet these requirements.

The water levels within the partial drawdown wetlands should be maintained for a minimum of two years, to allow for a response by submerged aquatic vegetation.

Increasing submerged aquatic vegetation density would provide a carbohydrate-rich food source for waterfowl, which is important for staging and migrating waterfowl in the fall (Dirschl 1969; Anderson and Low 1976; Sugden and Driver 1980; Murkin and Kadlec 1986; Baldassare and Bolen 2006). Greater densities of submerged aquatic vegetation

would also increase the amount of available habitat for invertebrates, potentially increasing invertebrate abundances (Voigts 1976; van den Berg et al. 1997; Hornung and Foote 2006). In this study, invertebrates were important for all species of waterfowl, particularly during the spring-breeding and brooding periods when protein demands are high. Invertebrates were also important to Soras and Virginia Rails during the spring, also likely as a result of high protein demands.

The ideal duration of the partial drawdowns that best promotes submerged aquatic vegetation growth, and subsequently waterfowl use, requires further study. However, the duration of partial drawdowns should not be long enough to promote the establishment of woody vegetation, such as *Salix* spp., in upland portions of the basins. Within the SMC, the duration of partial drawdowns will be directed by the availability of water, either from the Saskatchewan River or surrounding wetlands. Therefore, reflooding partially drawn down basins may need to occur on relatively short notice if water suddenly becomes available i.e., high water levels in the Saskatchewan River due to an ice jam. However, due to the reduced flood frequency of the river, water will likely have to be cycled through the wetland complex in order to refill the partially drawn down basins. In the SMC, this could be accomplished by storing water in Ravensnest Lake, routing this water through the Ravensnest Zone, refilling any partially drawn down wetlands in this area, and into the Back Channel where it could be routed into the Central Zone of the SMC to reflood basins in that area (Figure 1.6). The number of partially drawn down wetlands within the SMC should be determined by the amount of available water stored in Ravensnest Lake. Water from other, upstream high-water wetlands could also be

routed to help achieve Full Supply Level during the reflood stage. By cycling the available water in the SMC, a range of shallow and deep water wetlands can be created, providing a wide range of wetland habitats necessary for the variety of avian species using the area.

4.2 Future Work

The duration and optimum depth of partial water-level drawdowns requires further study. It appears that a minimum of two years may be required for a response from the biota in the partially drawn down wetlands. However, the length of time partial drawdowns may benefit dabbler species of waterfowl is unknown. The depth of the partial drawdowns that best promotes waterfowl use is also unknown. In this study, I found that dabblers preferred shallower water areas within the partial drawdown wetlands. Therefore, a more extensive drawdown, creating even shallower water than the wetlands in this study, may be warranted. Further, it may be beneficial to perform a complete water-level drawdown in order to obtain quantitative data on the effects of this management technique in the area. This would allow an effective comparison between wetlands in the SRD to wetlands in the prairies where complete drawdowns were developed and are commonly used.

To reduce costs and disturbance by future studies, I suggest that work be conducted in a more “remote” manner. Waterfowl surveys should be done with a helicopter and double observer, as this was technique was most efficient for surveying waterfowl within individual wetland basins. Vegetation can be monitored remotely using satellite imagery and the classification technique described in Appendix 4.0. This technique would allow

large-scale monitoring of changes in vegetation species, structure, and potentially vegetation health. However, quantitative data and further refinement of the classification technique are needed to accurately assess plant health.

Undoubtedly, avian research in the delta will involve “on-the-ground” field work. This research could be more efficient by reducing the number of rounds in the call-response surveys for marsh birds from four to three. These surveys should begin immediately after the ice has disappeared and wetlands can be accessed. If continued, these surveys do not need to include the Least Bittern, or Yellow Rail as these species were nearly absent during the surveys in 2008 and 2009. However, researchers may want to perform night surveys for Yellow Rails to confirm low abundances.

One aspect of wetland management, often overlooked, is the presence of fish in wetlands. Future avian studies should include forage fish sampling to identify the presence and abundance of forage fish within wetlands. This will allow for a more accurate examination of the factors affecting avian use of wetlands in the delta.

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Appendix 4.0 – Mapping of wetland emergent vegetation in the Summerberry Marsh Complex in the Saskatchewan River Delta, Manitoba: a comparison among unsupervised, supervised, and object-based classification techniques.

4.1 Introduction

Wetlands are complex ecosystems composed of a diverse community of heterogeneously distributed emergent and submergent vegetation species (Vivian-Smith 1997; Paracuellos 2006). The vegetation structure, species composition, and density of this community will determine the amounts of cover and food resources available to waterfowl and marsh birds and strongly impact their use of wetlands (Kaminski and Prince 1984; Eddleman et al. 1988; Murkin et al. 1997; van der Valk et al. 2000; Paracuellos 2006). Therefore, wetland management often attempts to provide suitable habitat for selected species through vegetation management (Baldassarre and Bolen 2006). The drawdown/refill technique is often used to manage wetland vegetation communities by mimicking the wet-dry cycle of succession (Murkin et al. 2000). This technique is often successfully applied in the prairie pothole region of North America, but has been unsuccessful when applied in the Saskatchewan River Delta (SRD) (Chapter 1; Murkin et al. 1997; Smith 1986; Smith 1987). For this reason partial water-level drawdowns are commonly used in the SRD in attempt to keep wetlands productive. However, quantitative information of the effects on the vegetation community caused by the partial water-level drawdowns is lacking.

4.1.1 Importance of Wetland Vegetation for Waterfowl and Marsh Birds

Vegetation structure, defined as the physical characteristics of the vegetation (i.e., patch size and interspersion), strongly influences wetland use by waterfowl and marsh birds.

For example, interspersions of emergent vegetation play an important role in providing visual isolation between breeding pairs of waterfowl during the spring breeding season. Kaminski and Prince (1981) illustrated that wetlands with an equal interspersions ratio of vegetation to open water supported the greatest numbers of breeding pairs of waterfowl. This interspersions ratio is also preferred by over-water nesting birds, including several species of diving ducks, Pied-billed Grebes (*Podilymbus podiceps*), and American Coots (*Fulica americana*) as it provides the greatest number of potential over-water nesting sites (Muller and Storer 1999; Murkin et al. 2000; Brisbin et al. 2002). Interspersions also provides the American Bittern (*Botaurus lentiginosus*) with a greater number of potential foraging sites by increasing the amount of vegetation/water interface (Lor 2007; Rehm and Baldassarre 2007; Lowther et al. 2009). Alternatively, wetlands with large areas of open water surrounded by emergent vegetation are selected by brooding dabbling species of waterfowl to provide cover for ducklings (Murkin et al. 2000; Raven et al. 2007). Some species of diving ducks and marsh birds, such as the American Coot and Pied-billed Grebe, also use large expanses of open water for foraging and escape from predators (Murkin et al. 1997; Murkin et al. 2000).

Vegetation composition also influences waterfowl and marsh bird use of wetlands. Dry sedge meadows within wetlands may increase the amount of upland nesting sites by increasing the amount of cover available for upland nesting species of waterfowl and marsh birds (Clay 1978; Swanson and Duebbert 1989). Vegetation species such as bulrush (*Schoenoplectus tabernaemontani*) provides deep, over-water nesting habitat for waterfowl species such as Ring-necked Duck (*Aythya collaris*) and Canvasback (*Aythya*

valisineria), as well as marsh birds such as Pied-billed Grebe and American Coot (Welling et al. 1988; Murkin et al 1997; van der Valk et al. 2000). Other species of vegetation such *Typha* spp. and *Phragmites australis* are preferred by walking species of marsh birds such as Sora (*Porzana carolina*) and Virginia Rail (*Rallus limicola*) as they offer dense cover and residual vegetation that enables the birds to move along the water surface (Johnson et al. 1986; Bookhout 1995; Melvin and Gibbs 1996). Sora and Yellow Rail (*Coturnicops noveboracensis*) select areas of *Carex* spp. as the seeds of this species are important food (Bookhout 1995; Melvin and Gibbs 1996).

The importance of wetland vegetation structure and composition to waterfowl and marsh birds highlights the need to develop an efficient way to monitor and quantify changes in these resources. Monitoring the response of vegetation is also important for determining the impacts of management actions, such as partial water-level drawdowns, which will ultimately affect bird use. These requirements can be accomplished efficiently using remote sensing technology to classify and quantify wetland vegetation. These classifications could then be used to determine what habitats are available for waterfowl and marsh birds across broad regions, and could potentially be used to predict the use of other areas by these birds.

4.1.2 Techniques for Mapping Wetland Vegetation

Vegetation can be monitored using traditional ground surveys. However, these are often time consuming and inefficient, particularly in large, inaccessible wetlands such as those found in the SRD. Remote sensing techniques using satellite imagery can be used to

solve these issues (Carter 1982). A number of remote sensing techniques exist that can accomplish this, each with its own advantages and disadvantages.

The first, more traditional approach, involves hand mapping. In this method, aerial or satellite imagery is obtained, visually examined, and the major vegetation zones are delineated by hand. The accuracy of the map is then determined by on-site field verification (ground-truthing), and the boundaries of the vegetation zones are adjusted if required (Cowardin and Myers 1974; Tiner 1997). The advantage of this method includes the relative simplicity of the technique; it does not require an in-depth knowledge of remote sensing statistics or software, and can be performed by an individual with minimal training. This technique may be sufficient if relatively small areas are being examined that provide easy access for on-site verification. However, hand-mapping a large area is labourious and prone to bias due to subjective decision making of cover/vegetation types. Some of this bias may be overcome with ground-truthing. However, if large areas are involved, adequate site verification also becomes labour intensive.

Pixel-based classification of aerial or satellite imagery can be used to overcome some of the problems associated with hand-mapping. This technique uses the spectral signatures of various wavelength bands in aerial imagery to classify individual pixels into different groups. There are two basic types of pixel-based classifications: unsupervised and supervised classification (Lillesand et al. 2008).

Unsupervised classification uses algorithms, such as the K-means or Iterative Self Organizing Data Analysis (ISODATA) algorithm, to classify individual pixels based solely on the pixel's spectral characteristics (Xie et al. 2008). No *a priori* knowledge is required; the user only needs to define the number of classes and threshold class values of the algorithm. In theory, pixels representing similar classes should have similar spectral characteristics and be distinct from other classes, allowing separation by the algorithm (Lillesand et al. 2008; Xie et al. 2008). Unsupervised classification is a good exploratory tool to examine how many "natural" classes the imagery contains. It also has the advantage of being relatively easy to perform, with relatively little effort required by the user. This technique is commonly used to classify wetland vegetation (Everitt et al. 2005; Olmanson et al. 2002; Ghioca-Robrecht et al. 2008), and can be successful when applied appropriately. However, one disadvantage of this technique is that features with similar spectral signatures are difficult to distinguish (Everitt et al. 2004). If the spectral signatures are not distinct enough for the classification algorithm, they will be grouped together as a single class and can result in an underestimation of the number of existing classes.

To increase discrimination among classes, supervised classification may be used. This technique uses "training areas", which are groups of pixels defined by the user, to represent the spectral characteristics of each class. Each pixel is compared to the spectral characteristics of the trained areas, and classified according to which area it best resembles (Lillesand et al. 2008). However, as with unsupervised classification, features that have similar spectral signatures are difficult to distinguish. This method is also more

involved and more time consuming. Selection of training areas needs to be accurate, and often statistical comparisons are used to determine the most accurate training areas in order to yield accurate classification maps (Lillesand et al. 2008).

One disadvantage common in both pixel-based classification methods is the “salt and pepper effect” or speckle produced by misclassification and pixel heterogeneity. Both techniques base classification solely on the spectral signature of individual pixels with no consideration of spatial attributes or heterogeneity within a class. Therefore, the techniques may produce heterogeneous areas with numerous classes giving a salt and pepper, or speckled appearance (Kelly et al. 2004; Kamagata et al. 2007). High resolution imagery such as IKONOS and QuickBird are prone to speckle as more heterogeneity is introduced with the smaller pixels of the very high resolution imagery (Thomas et al. 2003).

Object-based classification can be used to overcome some of the problems associated with pixel-based classifications. This technique uses an algorithm to segment groups of pixels within the image into objects (polygons); these objects are then classified rather than individual pixels (Ryherd and Woodcock 1996; Baatz et al. 2004). Segmentation is a “region-growing” process that begins with single-pixel objects that are subsequently merged into larger objects. The merger of objects is an optimization procedure based on object heterogeneity, which is a measure of the colour, shape, and smoothness of objects. The optimization procedure merges objects while minimizing heterogeneity. The region-growing process stops when the threshold levels of heterogeneity are reached. These

levels are user defined through the scale parameter. The scale parameter thus determines the relative size and shape of the segments in the image (Benz et al. 2004; Carleer et al. 2005). An advantage of this technique is that pixels can be grouped together to delineate different areas of vegetation species, density, etc., allowing the image to be segmented into ecologically meaningful objects, similar to human interpretation (Lillesand et al. 2008). This eliminates the problem of speckle, which is common in the pixel-based techniques. Classification of objects from the segmented image is then performed using a knowledge based approach. Ground-truthed data are compared to the segmented map and objects are classified based upon their spectral and spatial attributes (Lillesand et al. 2008).

4.1.3 Assessment of Classification Accuracy

Determining classification accuracy is important as it enables the comparison of mapping methods, and determines the overall worth of maps produced. Both the supervised classification and object-based classification techniques produce a confusion matrix that can be used as a measure of classification accuracy. The confusion matrix is calculated by comparing the number of pixels or seed objects used training to the number of pixels or seed objects that are classified into the appropriate class they represent (Lillesand et al. 2008). The major diagonal through the matrix from the upper-left to lower-right represents the number of pixels or objects that are correctly classified. The remainder of the matrix represents errors of omission and errors of commission. Errors of omission occur when a pixel or object belonging to a certain class is omitted from that class. Errors of commission occur when a pixel or object are classified into a category which they do

not belong. From these errors the producer's accuracy and user's accuracy can be calculated. The producer's accuracy is the measure of how well the training pixels/objects are classified. It is calculated by dividing the number of correctly classified pixels/objects in a category by the number of training pixels/objects for that category. The user's accuracy is a measure of error of commission, indicating the probability of classification actually representing the category in reality. It is calculated by dividing the number of correctly classified pixels/objects in each category by the total number of pixels/objects in the category. The overall class accuracy can be determined by dividing the total number of correctly classified pixels/objects by the total number of pixels/objects.

4.2 Objectives

My objective was to compare unsupervised and supervised classification techniques to an object-based technique to identify the most efficient method able to delineate different emergent vegetation species within our study wetlands. This technique would then be used to quantify habitat of waterfowl and marsh birds to determine the effects of vegetation composition and structure in our study wetlands in the Summerberry Marsh Complex, Manitoba.

4.3 Study Area and Experimental Design

My study took place during 2008 and 2009 in the managed wetlands of Summerberry Marsh Complex (SMC). The SMC is located within the SRD, approximately 25 km southeast of The Pas, Manitoba. A partial water-level drawdown was initiated in three wetlands in 2007 and continued through 2010. Three additional wetlands were concurrently managed with high water levels to act as control in the experiment. See Chapter 1 for a more detailed description.

4.4 Methods

4.4.1 Pixel-based Classification

QuickBird satellite imagery of the SMC was obtained from DigitalGlobe Inc. on 4 July, 2008, at the approximate the peak of above-ground emergent vegetation biomass. The imagery obtained was 0.6-m resolution colour infrared with three spectral bands: green (520-600 nm), red (630-690 nm), and near-infrared (760-900 nm). All imagery was radiometrically corrected and georeferenced prior to delivery. Images came in four separate image files and were merged to form a mosaic using ArcGIS 9.1 (ESRI, Redlands, CA) as many of the wetlands were on a boundary between two or more of the images. Regions containing each study wetland were clipped from these image mosaics. It should be noted that both pixel-based classifications were performed in 2008. Therefore, no ground-truthed data were available, and only the 2008 imagery was on hand at the time of analysis. Pixel-based classifications were not performed on the 2009 imagery due to the relatively poor performance of the 2008 classifications.

An unsupervised classification was performed using MultiSpec 3.1 (Purdue Research Foundation) on each of the colour infrared images of the study wetlands from 2008 to determine the number of “naturally” distinct classes in the colour infrared imagery. I used the Iterative Self Organizing Data Analysis (ISODATA) algorithm, set to define 10 clusters/classes. After the algorithm was run, the thematic image was visually examined, compared to the aerial imagery, and the original 10 clusters were then merged to combine clusters representing similar areas and reduce speckle.

A supervised classification was performed using MultiSpec 3.1 (Purdue Research Foundation) on each of the colour infrared images of the study wetlands from 2008 using the maximum likelihood algorithm. Approximately 12-20 training areas were defined for each class, in each of the study wetlands, depending on the amount of homogenous areas available for the class. The results from the unsupervised classification were used as a guide to determine the number and location of training areas for the supervised classification.

Nine classes were chosen to be most representative of the cover types available for classification. The class names and respective cover types were: [carex (*Carex* spp.), equis (*Equisetum* spp.), scirpus (*Schoenoplectus tabernaemontani*), typha (*Typha* spp.), phrag (*Phragmites australis*), trees (dominated by *Salix* spp. but also included *Populus* spp., *Acer negundo*, *Fraxinus pennsylvannica*), dead veg (dead/stressed vegetation of various species), water (open water in the wetland), and river (open water in the river)]. Not all classes were found or discernable in each wetland. Therefore, the total number of classes within each wetland was determined by the availability of training areas for the class. The maps from the unsupervised classification were used to assist in identifying homogenous stands of vegetation, large enough to use as a training area. Delineation of training areas was done subjectively, as no ground-truthed data were available at the time of classification. Some of this information was supplemented with information from a simultaneous muskrat project in the SMC, which provided the dominant vegetation species at approximately 100 locations in each wetland. However, most of this information was collected around the periphery of the wetlands and was of limited use.

4.4.2 Ground-truthing for Object-based Classification

Ground-truthing was performed in late July 2009, at the approximate peak of above-ground emergent vegetation biomass. This was done from an airboat that traveled along transects spaced 50 m apart, stopping every 50 m, to create an approximate 50-m by 50-m grid across the entire wetland basin. The number of points per wetland varied with basin size. Several measurements were recorded at each point, including: the Universal Transverse Mercator (UTM) coordinates using a Garmin GPSMAP 76CSx hand-held Global Positioning System (GPS), dominant vegetation species within a 10-m radius, including [*Sparganium* (*Sparganium* spp.), *Carex* (*Carex* spp.), *Equisetum* (*Equisetum* spp.), *Scirpus* (*Schoenoplectus tabernaemontani*), *Typha* (*Typha* spp.), *Phragmites* (*Phragmites australis*), and Water (open water)], whether the vegetation was rooted or floating, alive or dead, and the water depth. Water depth was taken by attaching a metal metre stick to a piece of 2" x 2" lumber. This measuring stick was used to push through detritus and floating vegetation to the substrate to obtain an accurate water depth.

Orthorectification control points were collected using a Garmin GPSMAP 76CSx hand-held GPS. We used the "Tracking" function of the GPS to delineate the locations of prominent landscape features in the SMC, such as water control structures and channel edges. These control points were then used to orthorectify the aerial imagery.

4.4.3 Image Adjustment for Object-based Classification

QuickBird satellite imagery of the SMC was obtained from DigitalGlobe Inc. on 4 July, 2008 and 7 July, 2009, at the approximate the peak of above-ground emergent vegetation

biomass. Two types of imagery were obtained: 0.6-m resolution natural colour and 0.6-m resolution colour infrared. The natural colour imagery contained three spectral bands: blue (450-520 nm), green (520-600 nm), and red (630-690 nm). The colour infrared also contained three spectral bands: green (520-600 nm), red (630-690 nm), and near-infrared (760-900 nm). All methods described hereafter were performed on both the 2008 and 2009 imagery. All imagery was radiometrically corrected and georeferenced prior to delivery. The blue, green, and red spectral bands from the natural colour image were combined with the near-infrared spectral band of the colour infrared image to create a new, four-band image. The new images were then orthorectified using ArcMap 9.1. The orthorectification points were overlaid onto the imagery and the imagery was adjusted to line up with the control points. No stretching or warping of the imagery was performed.

4.4.4 Object-based Image Classification

Segmentation of the raster imagery into polygons was done using Definiens eCognition® software. The scale parameter used on the SMC wetlands was determined by comparing gradually finer segmentation scales to isolate as many unique ground-truthed points within single polygons as possible while still retaining homogenous species stands.

The segmented imagery was clipped around each wetland's Full Supply Level (FSL) to reduce the number of polygons. The FSL is the highest elevation that Ducks Unlimited Canada manages the water levels, and consequently is where the tree line begins.

Using ArcMap 9.2 a spatial join was performed between the segmented polygons and ground-truthed points. This produced polygons with the attributes of the ground-truthed point it contained. Each polygon containing greater than one ground-truthed point was examined to identify if the ground-truthed points were conflicting vegetation types. If the polygon contained conflicting ground-truthed points, it was visually examined, and if necessary, changed to the most likely vegetation type. The most likely vegetation type was determined by visually examining the satellite imagery, while simultaneously taking into account the relative position of the ground-truthed points within the polygon.

Ground-truthed points along the edges of polygons were more often inaccurate due to error in segmentation and GPS accuracy (personal observation). Despite being subjective, only a small portion of the polygons contained conflicting ground-truthed points. For the 2009 segmentation 255 polygons contained greater than one ground-truthed point. Of these, 36 polygons had conflicting points (14.1%), and only 16 polygons required the vegetation type to be changed manually (6.3%).

Trees were not recorded as a vegetation type during ground-truthing. Instead, a “Trees” class was developed *a posteriori*. This was done visually using the segmented satellite imagery. I selected 220 random polygons around the six study wetlands that appeared to be trees and manually assigned the “Trees” attribute. These polygons were then treated as ground-truthed data. The “Trees” class was dominated by *Salix* spp., but also included *Populus* spp., *Acer negundo*, *Fraxinus pennsylvannica*, and numerous other species of trees and shrubs along the riparian edge.

Depth contours of each wetland basin were also created in ArcMap 9.2 by Kriging the ground-truthed depth data. The average water depth within each polygon was calculated by converting the depth contours into a Triangular Irregular Network (TIN) and averaging the water depth values from the TIN that were inside the polygon.

To determine the vegetation species of polygons lacking a ground-truthed point, data were first separated by the drawdown and non-drawdown treatments to account for senescence and stressed vegetation caused by the lowered water levels. Generalized logistic models (GLMs) were performed on the spectral characteristics of the polygons from the drawdown and non-drawdown wetlands separately to assign each polygon to a vegetation class. The vegetation classes used in the model were: *Carex*, *Equisetum*, *Scirpus*, *Typha*, *Phragmites*, Trees, and Water. The *Sparganium* class was not included at this time due to a low number of ground-truthed points of this vegetation type.

Thirteen variables were available for modeling the wetland vegetation species, including: the mean values of the blue, red, green, and near-infrared (NIR) values in each polygon, the standard deviations of the blue, red, green, and NIR values in each polygon, maximum difference between pixel values in each polygon, brightness (sum of mean values of blue, red, green, and NIR divided by the quantities), the normalized difference vegetation index ($NDVI = \frac{\text{mean NIR} - \text{mean red}}{\text{mean NIR} + \text{mean red}}$), the green normalized difference vegetation index ($GNDVI = \frac{\text{mean NIR} - \text{mean green}}{\text{mean NIR} + \text{mean green}}$), and average water depth.

To find the most parsimonious model, several parameter selection methods were used, including: full (all parameters included), reduced (sub-set of the best non-correlated parameters), forward stepwise, and backward stepwise. Because the dataset was large, approximately 20,000 objects, all model selection methods used a retention $\alpha = 0.001$ to reduce the chances of Type I error. The model with the highest overall classification accuracy was used to create the final vegetation map. The overall classification accuracy was calculated by dividing the total number of correctly classified objects by the total number of objects. Class accuracies were estimated using the confusion matrix, which compares the observed vegetation types of polygons containing a ground-truthed point to the expected vegetation type as predicted by the GLM. It was assumed that a polygon containing a ground-truthed point consisted of a homogenous stand of that vegetation species and was correct. The number of polygons in which the observed vegetation type from ground-truthing the GLM were changed by the model gives an estimation of how accurate the predicted probabilities are for each given class and are shown by the respective confusion matrices.

For the final version of the vegetation maps, polygons containing a ground-truthed point that had the vegetation type changed by the GLM were changed back to the observed vegetation type from the ground-truthing. This relied on the assumption that ground-truthing was completely accurate. This added the *Sparganium* class to the maps.

Polygons containing a ground-truthed point observed as *Sparganium* spp. acquired this classification. Finally, the maps were visually inspected and compared to the satellite

imagery to check for misclassified polygons. If a polygon appeared to be misclassified, it was changed to the vegetation type interpreted from the satellite imagery.

4.5 Results

4.5.1 Pixel-based Unsupervised Classification

Overall, the unsupervised classifications performed poorly. In the non-drawdown study wetlands, the technique was unable to distinguish between trees in the riparian areas from *Carex* spp. This resulted in trees being classified throughout the basin, instead of being limited to their actual location around the periphery. Similarly, trees were indistinguishable from *Phragmites australis* in wetland 34HI (Figure 4.1). Other species of emergent vegetation, including *Phragmites*, *Typha* spp., *Schoenoplectus* spp., and *Carex* spp., were indistinguishable from one another and contained large amounts of speckle (Figures, 4.1, 4.2, 4.3). The exceptions to this were *Equisetum* spp. and *Lemna* spp., both of which were given their own separate class by the algorithm. Open water within the wetlands and in the river were also given separate classes by the algorithm. However, areas with robust submergent vegetation growth were misclassified as emergent vegetation rather than open water. This inaccuracy was extensive in the open water areas of all three non-drawdown wetlands. These areas contain considerable amounts of speckle, produced by the submergent vegetation. Alternatively, in areas with relatively sparse emergent vegetation, patches were misclassified as water rather than a vegetation class. This was prevalent in wetland 32C, which supports a large amount of relatively sparse *Schoenoplectus* spp. In this wetland the water below the canopy of the *Schoenoplectus* spp. is visible in the imagery causing water to be over-classified (Figure 4.2).

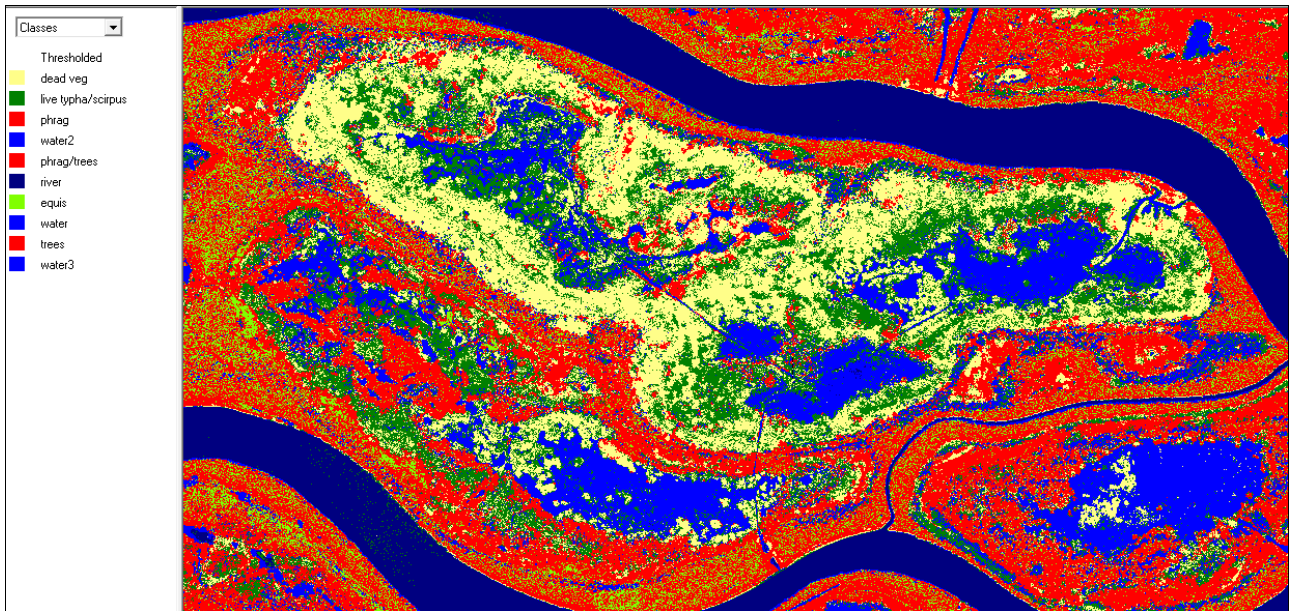


Figure 4.1- Unsupervised classification of wetlands 35HI (top) and 34HI (bottom) in the Summerberry Marsh Complex, Manitoba, 2008, using the maximum likelihood technique and 10 clusters.

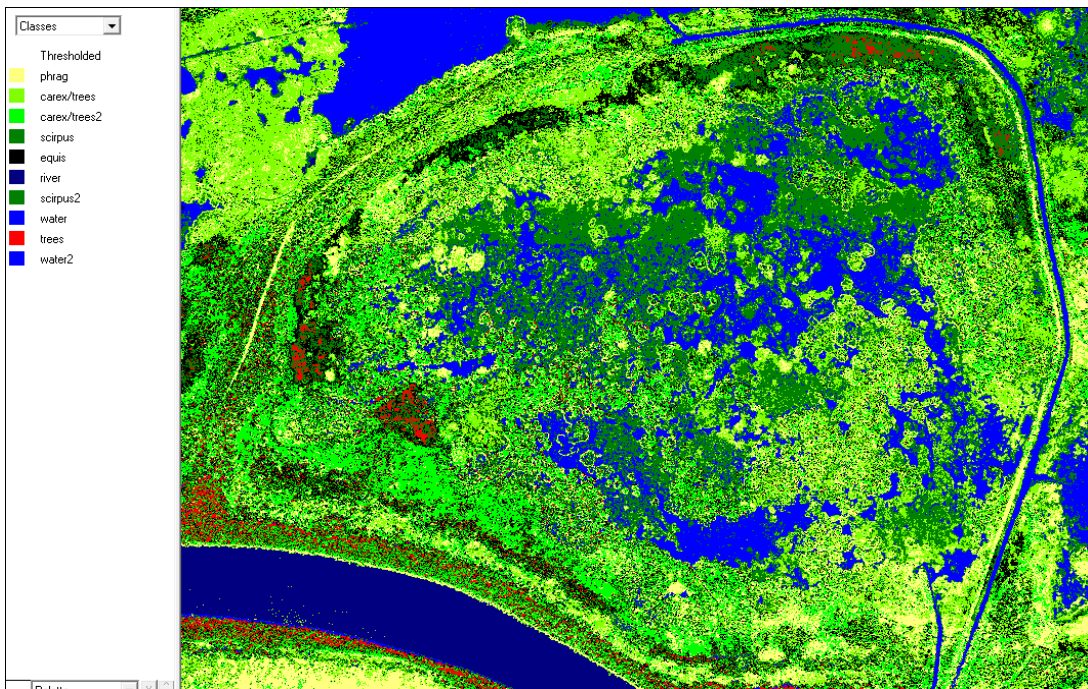


Figure 4.2- Unsupervised classification of wetland 32C in the Summerberry Marsh Complex, Manitoba, 2008, using the maximum likelihood technique and 10 clusters.

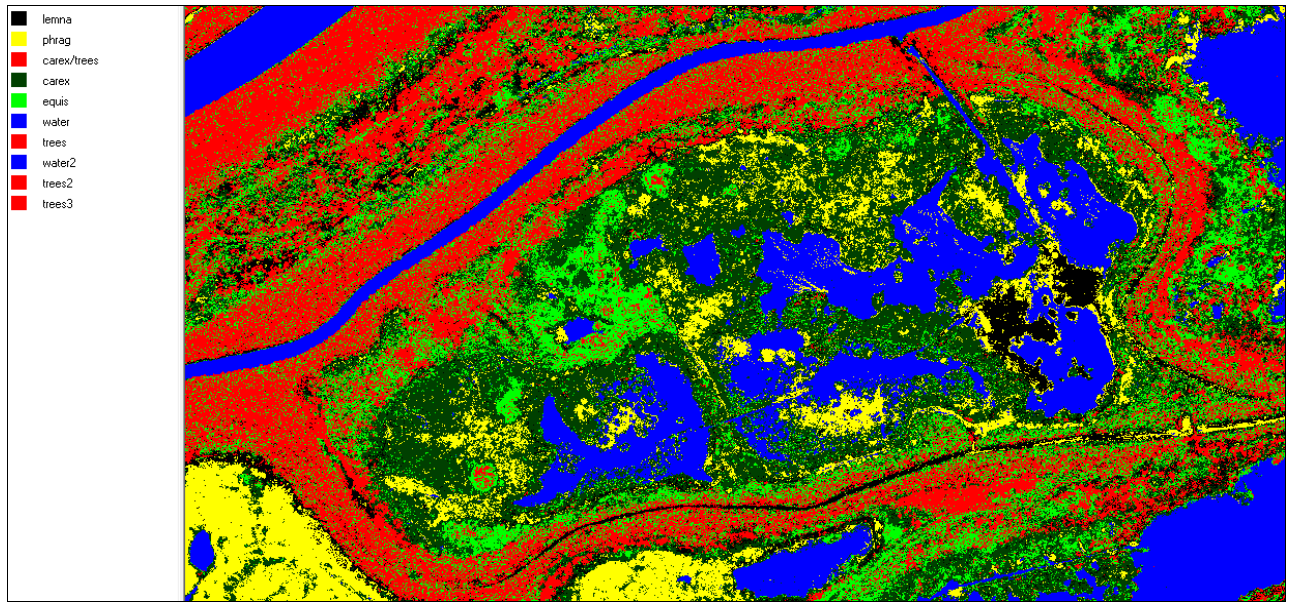


Figure 4.3- Unsupervised classification of wetland 21C in the Summerberry Marsh Complex, Manitoba, 2008, using the maximum likelihood technique and 10 clusters.

In the drawdown study wetlands, the vegetation around the periphery of the basins was stressed and highly reflective, making it difficult for this technique to delineate the different species of vegetation. In all drawdown wetlands the only classes possible to delineate were trees, living vegetation, dead vegetation, and open water in the wetlands and river. Water was delineated accurately due to the near absence of vegetation/open water interspersions and flooded emergent vegetation (Figures 4.1, 4.4, 4.5).

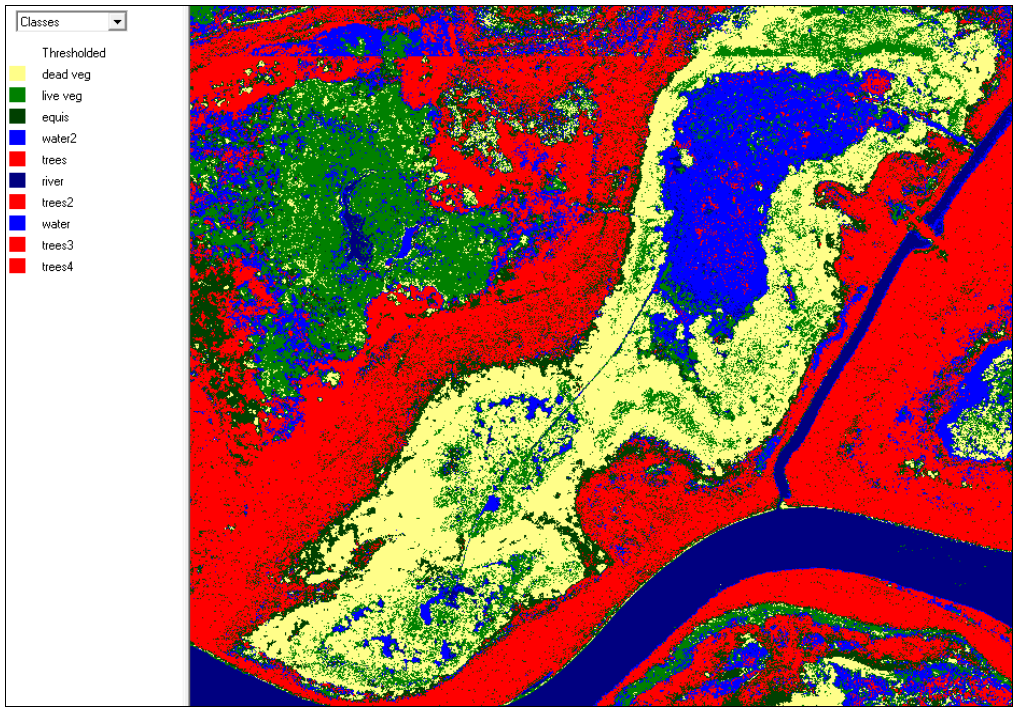


Figure 4.4- Unsupervised classification of wetland 14R in the Summerberry Marsh Complex, Manitoba, 2008, using the maximum likelihood technique and 10 clusters.

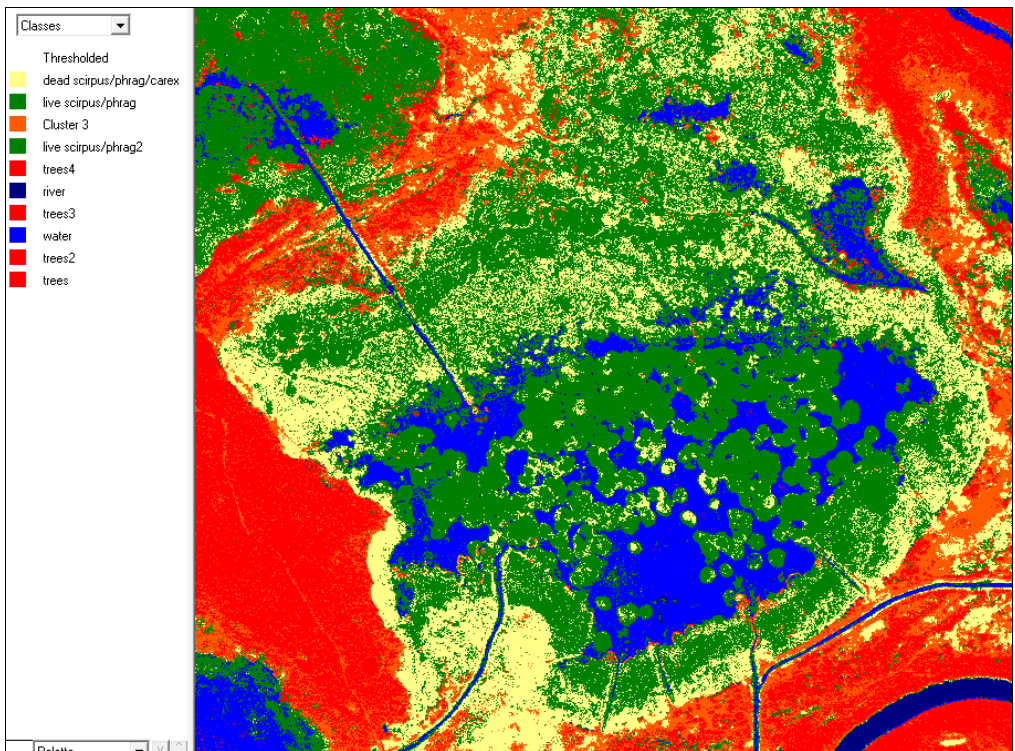


Figure 4.5- Unsupervised classification of wetland 37C in the Summerberry Marsh Complex, Manitoba, 2008, using the maximum likelihood technique and 10 clusters.

4.5.2 Supervised Classification of Non-drawdown Wetlands

I was unable to delineate training areas for at least one class in all wetlands due to the absences of homogenous areas large enough to support training areas. Therefore, the numbers and types of classes varied by wetland.

In the non-drawdown wetlands *Carex* spp. were not delineated accurately, as shown by the low user's and producer's accuracies (Tables 4.1, 4.2, 4.3). *Carex* spp. were often confused with trees or *Phragmites* spp. Other species of emergent vegetation, including *Phragmites*, *Typha* spp., and *Schoenoplectus* spp. were also delineated poorly as shown by large amounts of speckle within the classes, and extensive confusion with trees (Table 4.2).

In wetlands 32C and 34HI, trees were under-classified due to confusion with emergent vegetation classes. These wetlands contain a riparian strip of trees around the periphery of the basins. However, in both wetlands, most of the riparian areas are classified as a mixture of *Phragmites* and *Equisetum* spp. (Figures 4.6 & 4.7). However, trees were classified accurately in wetland 21C. In this wetland the entire treed riparian area appears around the periphery of the basin and small groups of interspersed trees appear farther in the wetland (Figure 4.8).

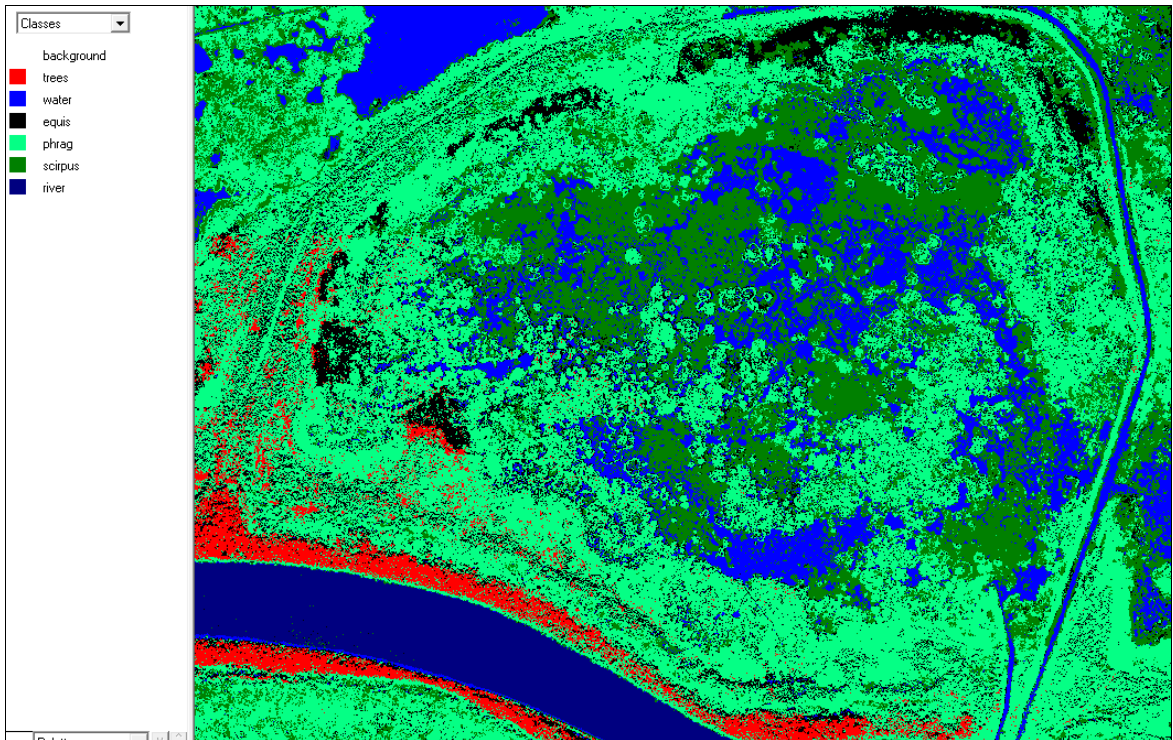


Figure 4.6- Supervised classification of wetland 32C in the Summerberry Marsh Complex, Manitoba, 2008, using six classes.

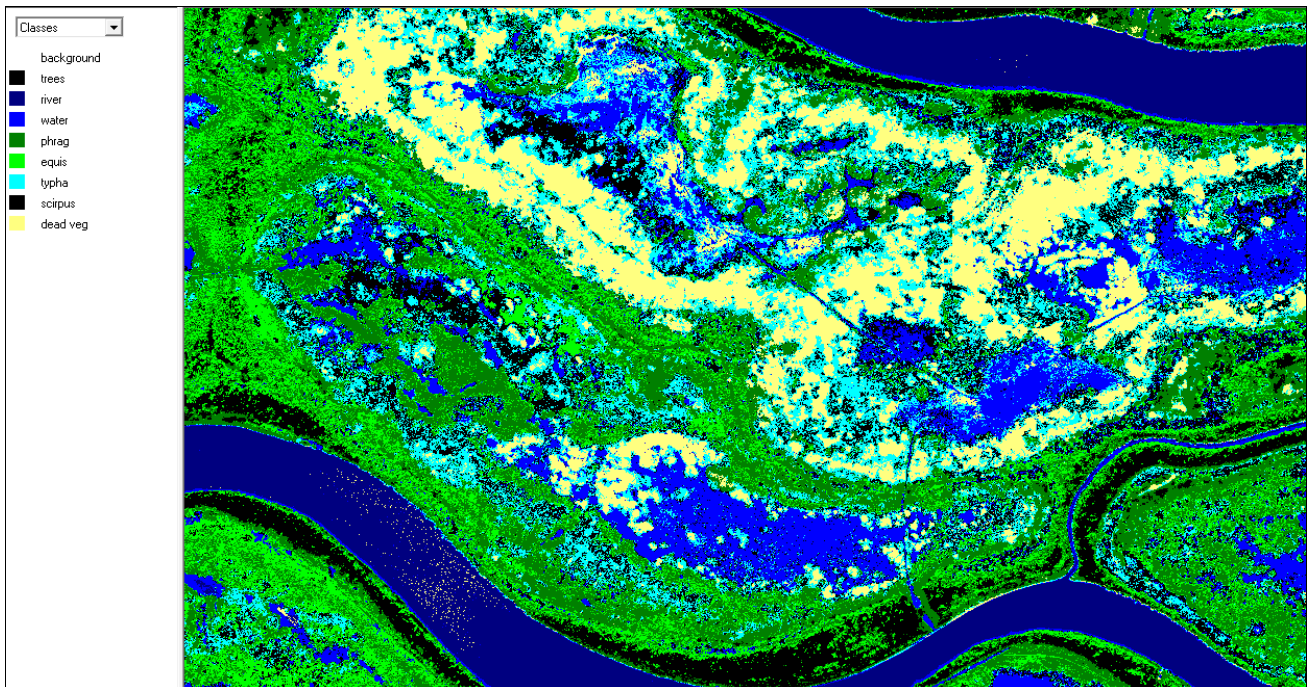


Figure 4.7- Supervised classification of wetlands 35HI (top) and 34HI (bottom) in the Summerberry Marsh Complex, Manitoba, 2008, using eight classes.

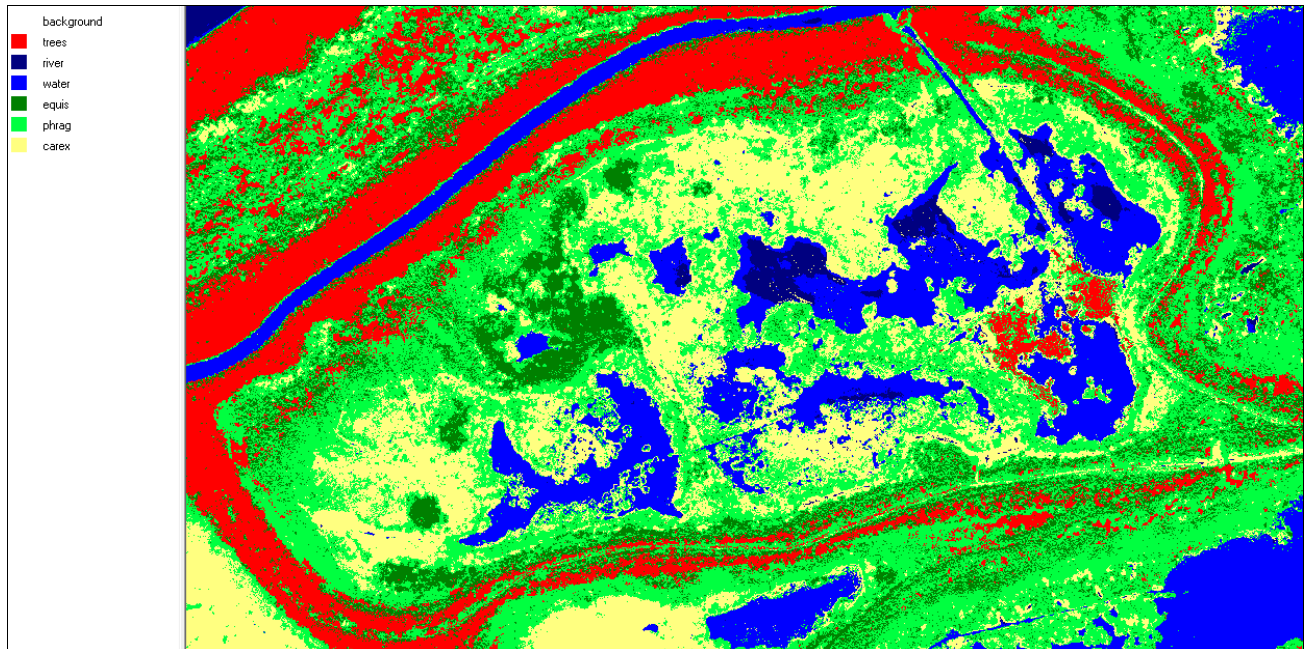


Figure 4.8- Supervised classification of wetland 21C in the Summerberry Marsh Complex, Manitoba, 2008, using six classes.

Equisetum spp., and open water in the wetlands and river were classified accurately.

Large *Equisetum* spp. patches are visible in the maps of wetland 32C and 21C, which contain little speckle (Figures 4.6 & 4.8). Open water in the wetlands was accurate for the most part. Similar to the unsupervised classification, the presence of sparse vegetation mixed with water was a problem for the supervised classification. In wetland 32C (Figure 4.6), areas with sparse *Schoenoplectus* spp. were often misclassified as water. In the other wetlands water appeared to be classified accurately, likely due to the absence of sparse vegetation patches in these wetlands.

4.5.3 Supervised Classification of Drawdown Wetlands

The drawdown wetlands contained large amounts of stressed/dead vegetation, making it difficult to define training areas for different species of emergent vegetation. This caused many classes of emergent vegetation to be grouped into one large class called “dead veg”

(Tables 4.3, 4.4, 4.5). Some classes of emergent vegetation were distinguishable in the wetlands; some *Schoenoplectus* spp. and *Phragmites* were visible in wetlands 35HI and 37C. However, these areas were small, inaccurate, and contained large amounts of speckle (Tables 4.3 & 4.5, Figures 4.7 & 4.10). In wetland 14R, no species-specific vegetation classes were detectable and classes were limited to identifying areas of living and dead vegetation (Table 4.4). These classes appeared to be fairly accurate, with minimal amounts of speckle, and a low amount of confusion with the “trees” class (Table 4.4 & Figure 4.9).

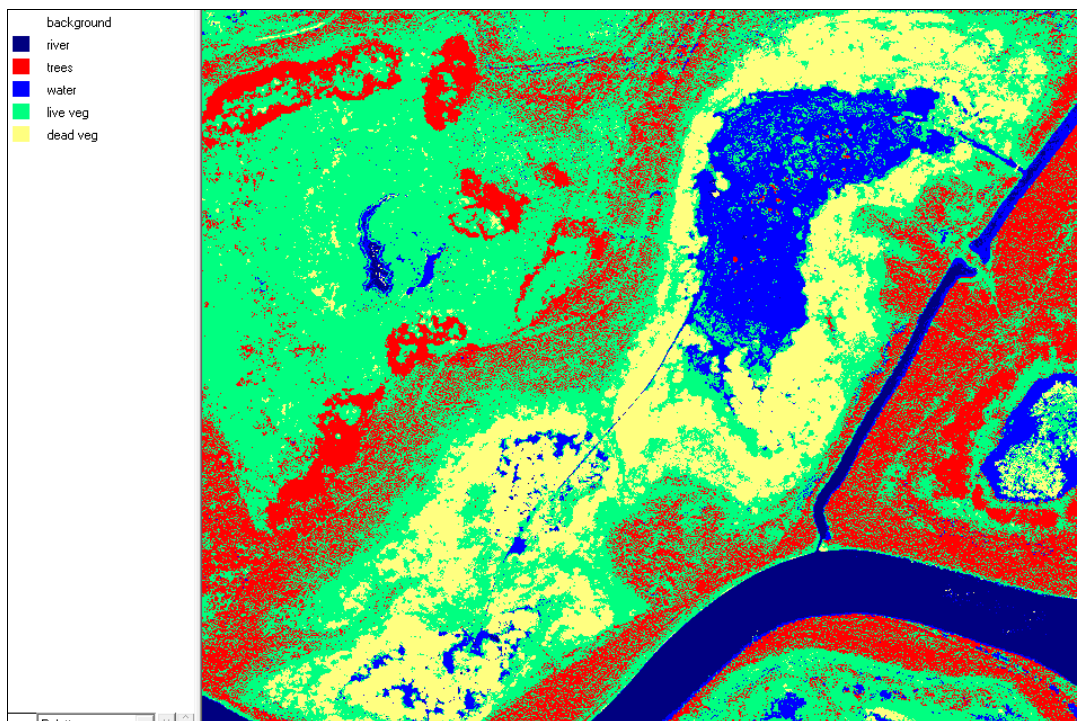


Figure 4.9- Supervised classification of wetland 14R in the Summerberry Marsh Complex, Manitoba, 2008, using five classes.

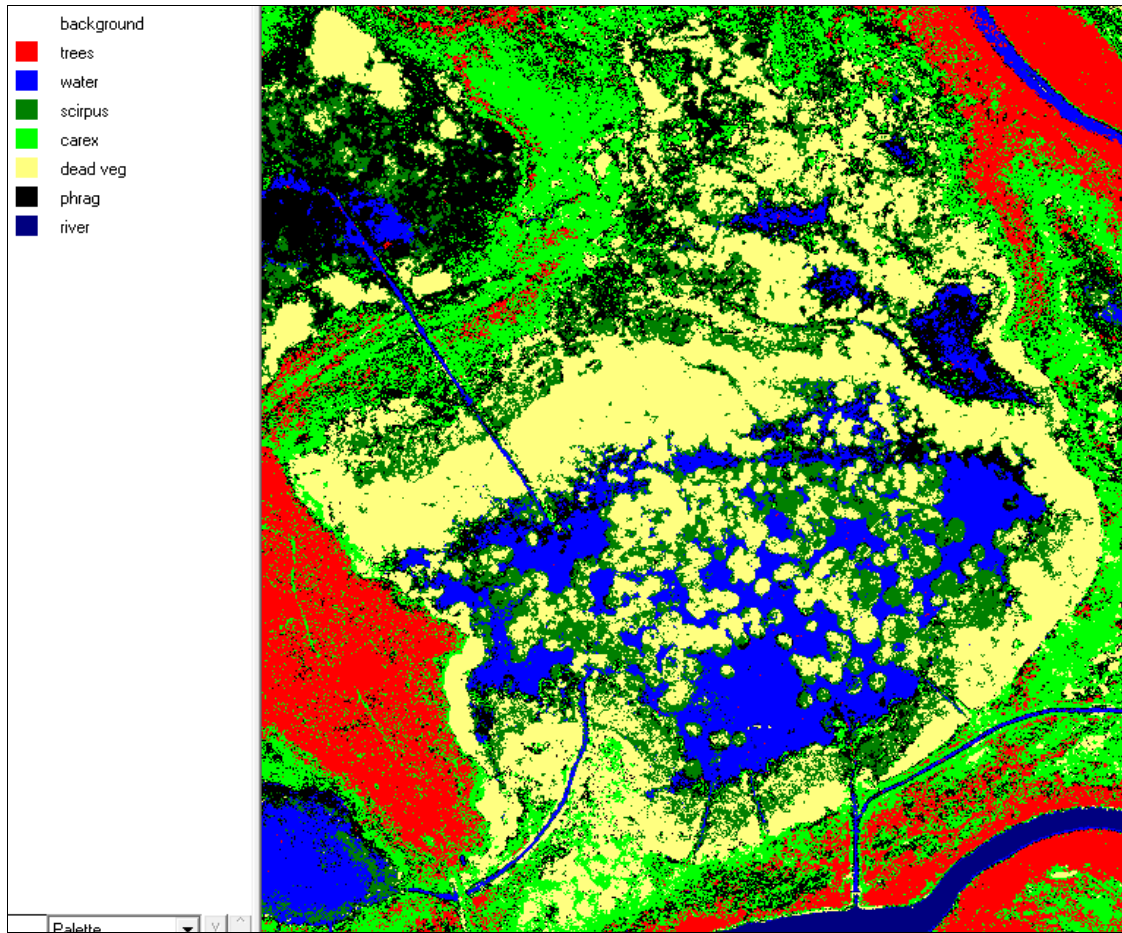


Figure 4.10- Supervised classification of wetland 37C using seven classes.

Table 4.1- Confusion matrix and class accuracies of the six classes from the supervised classification of wetland 21C in the Summerberry Marsh Complex, Manitoba, 2008.

	trees	river	water	equis	phrag	carex	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
trees	141288	0	0	2009	212	0	143509	98.5	99.9	99.2	98.3
river	0	87475	32	0	0	3	87510	100.0	100.0	100.0	
water	0	10	81303	0	63	15	81391	99.9	100.0	100.0	
equis	103	0	0	25424	1001	25	26553	95.7	91.0	93.4	
phrag	0	0	0	517	7281	771	8569	85.0	73.5	79.3	
carex	0	0	0	0	1354	18670	20024	93.2	95.8	94.5	
Total	141391	87485	81335	27950	9911	19484	367556				

Table 4.2- Confusion matrix and class accuracies of the six classes from the supervised classification of wetland 32C in the Summerberry Marsh Complex, Manitoba, 2008.

	trees	water	equis	phrag	scirpus	river	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
trees	78463	21	12124	16565	1	0	107174	73.2	97.9	85.6	93.1
water	0	41676	4	0	1481	0	43161	96.6	95.4	96.0	
equis	877	49	52797	3915	112	0	57750	91.4	78.7	85.1	
phrag	774	7	1979	45859	191	0	48810	94.0	68.4	81.2	
scirpus	0	1942	152	667	72393	0	75154	96.3	97.3	96.8	
river	0	0	36	0	193	261325	261554	99.9	100.0	100.0	
Total	80114	43695	67092	67006	74371	261325	593603				

Table 4.3- Confusion matrix and class accuracies of the eight classes from the supervised classification of wetlands 34HI and 35HI in the Summerberry Marsh Complex, Manitoba, 2008.

	trees	river	water	phrag	equis	typha	scirpus	dead veg	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
trees	46797	0	0	965	1217	0	0	0	48979	95.5	99.5	97.5	93.8
river	0	335926	51	0	0	0	0	42	336019	100.0	100.0	100.0	
water	0	0	44861	0	0	1216	1321	0	47398	94.6	98.8	96.7	
phrag	10	0	3	24917	2104	59	5	6	27104	91.9	88.3	90.1	
equis	246	0	13	2322	11328	0	14	0	13923	81.4	77.3	79.4	
typha	0	9	232	1	1	10216	12319	10410	33188	30.8	60.0	45.4	
scirpus	0	0	178	0	6	4456	21913	74	26627	82.3	61.6	72.0	
dead veg	0	21	46	0	0	1085	4	81504	82660	98.6	88.6	93.6	
Total	47053	335956	45384	28205	14656	17032	35576	92036	615898				

Table 4.4- Confusion matrix and class accuracies of the five classes from the supervised classification of wetland 14R in the Summerberry Marsh Complex, Manitoba, 2008.

	river	trees	water	live veg	dead veg	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
river	304973	0	1664	2	280	306919	99.4	99.9	99.7	88.5
trees	0	179436	3470	86130	13	269049	66.7	99.9	83.3	
water	324	6	71391	279	0	72000	99.2	93.0	96.1	
live veg	0	190	206	35391	136	35923	98.5	28.1	63.3	
dead veg	4	0	64	3928	152340	156336	97.4	99.7	98.6	
Total	305301	179632	76795	125730	152769	840227				

Table 4.5- Confusion matrix and class accuracies of the seven classes from the supervised classification of wetland 37C in the Summerberry Marsh Complex, Manitoba, 2008.

	trees	water	scirpus	carex	dead veg	phrag	river	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
trees	267942	0	0	17033	0	2272	0	287247	93.3	99.8	96.6	94.6
water	301	110490	98	0	0	9201	0	120090	92.0	97.9	95.0	
scirpus	0	72	24647	0	2733	567	0	28019	88.0	90.3	89.2	
carex	147	0	55	44289	520	818	0	45829	96.6	69.1	82.9	
dead veg	0	0	169	24	107430	22	0	107645	99.8	96.8	98.3	
phrag	22	2336	2326	2771	337	16875	0	24667	68.4	56.7	62.6	
river	0	18	0	0	0	0	156065	156083	100.0	100.0	100.0	
Total	268412	112916	27295	64117	111020	29755	156065	769580				

4.5.4 Ground-truthing for Object-based Classification

A total of 3342 points were ground truthed in the six study wetlands. The number in each wetland varied according to size (Table 4.6). See Figure 4.11 for general distributions of points in each wetland. Due to the relatively high number of ground-truthed points taken, I was able to create 10 cm depth contours for each study wetland (Figures x-x).

Table 4.6- Number of ground-truthed points taken in each study wetland in the Summerberry Marsh Complex, Manitoba, 2009.

Wetland	No. Ground-truthed Points
14R	609
21C	608
32C	675
34HI	407
35HI	521
37C	522

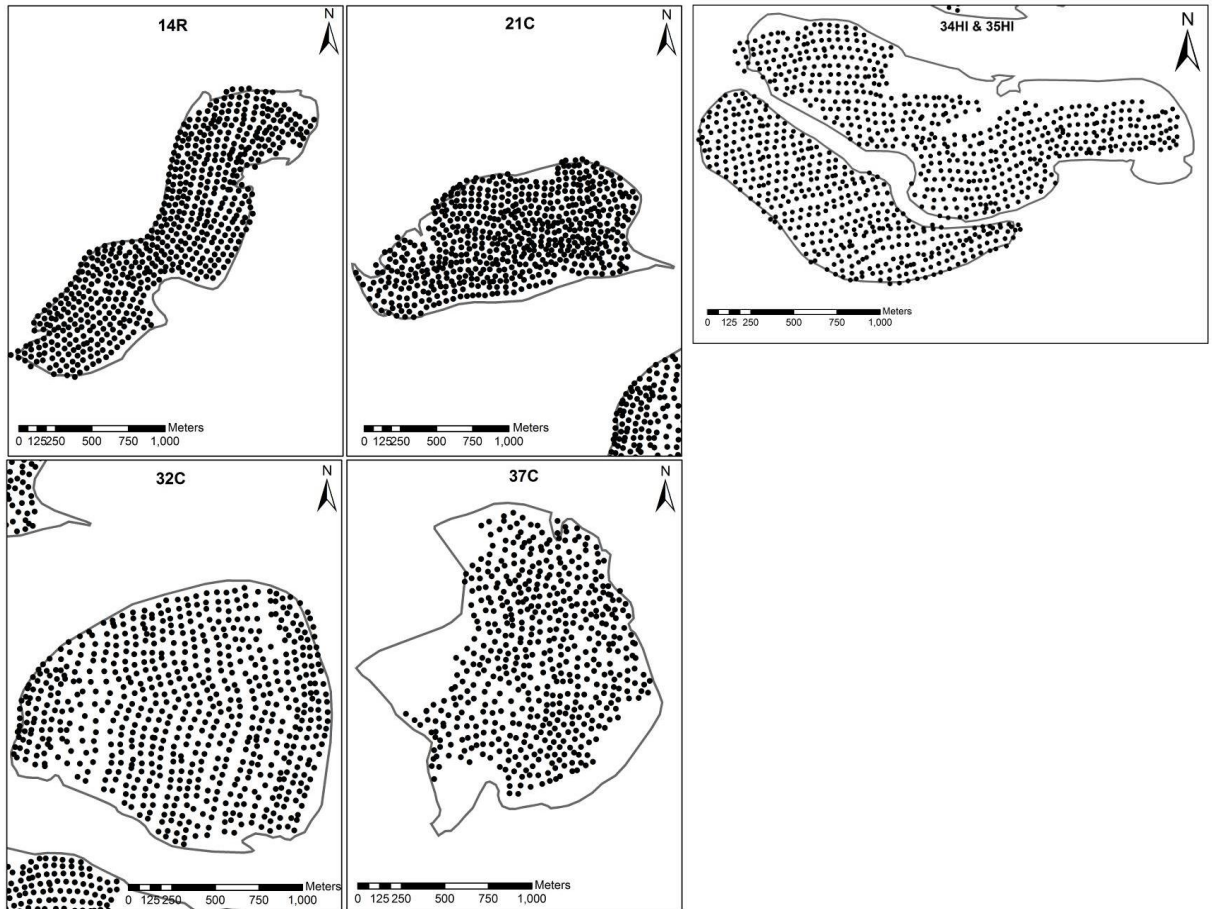


Figure 4.11- Ground-truthed points taken in each wetland's Full Supply Level (FSL) in the Summerberry Marsh Complex, Manitoba, 2009.

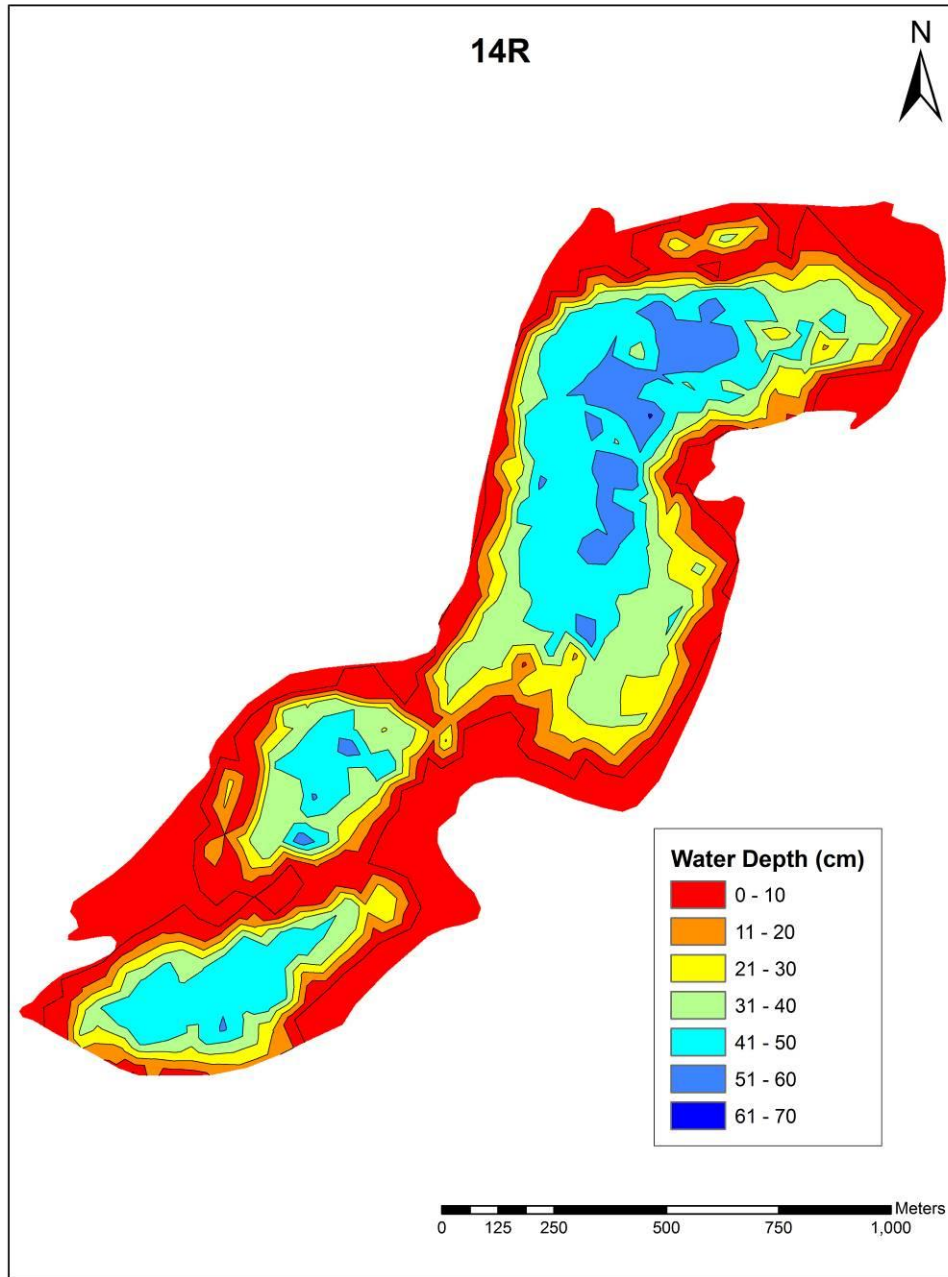


Figure 4.12- Water depth contours of wetland 14R in the Summerberry Marsh Complex, Manitoba taken during ground-truthing in 2009.

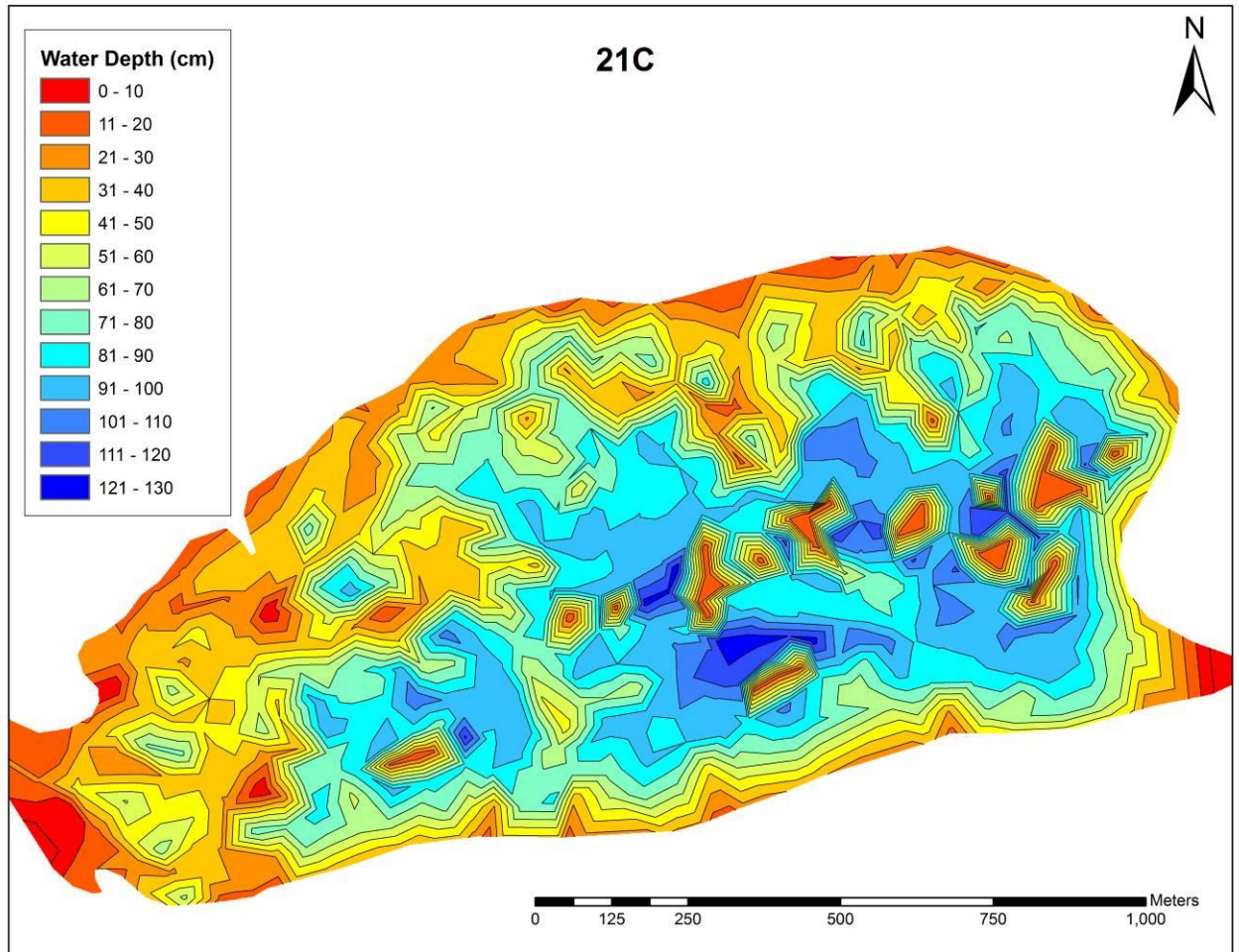


Figure 4.13- Water depth contours of wetland 21C in the Summerberry Marsh Complex, Manitoba taken during ground-truthing in 2009.

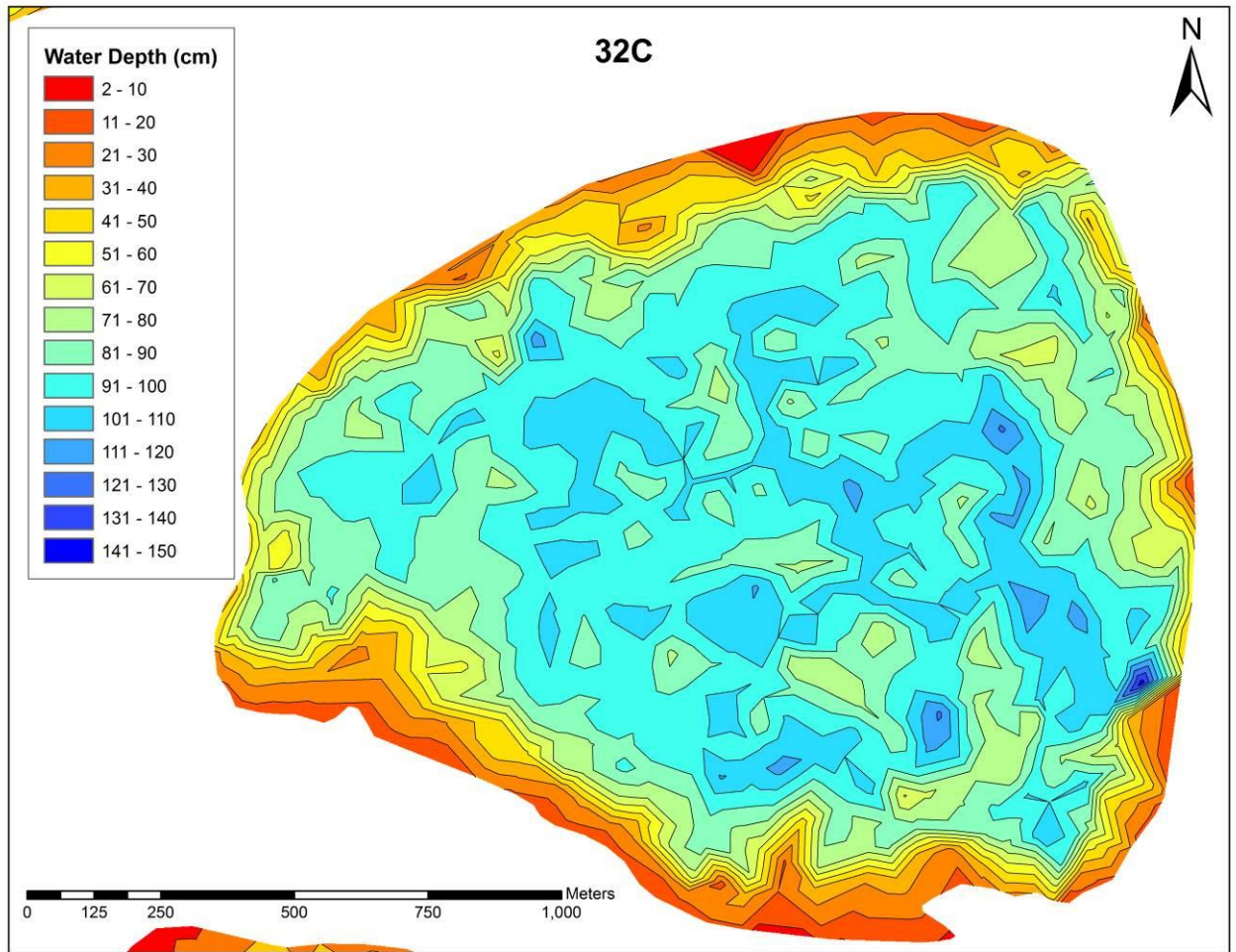


Figure 4.14- Water depth contours of wetland 32C in the Summerberry Marsh Complex, Manitoba taken during ground-truthing in 2009.

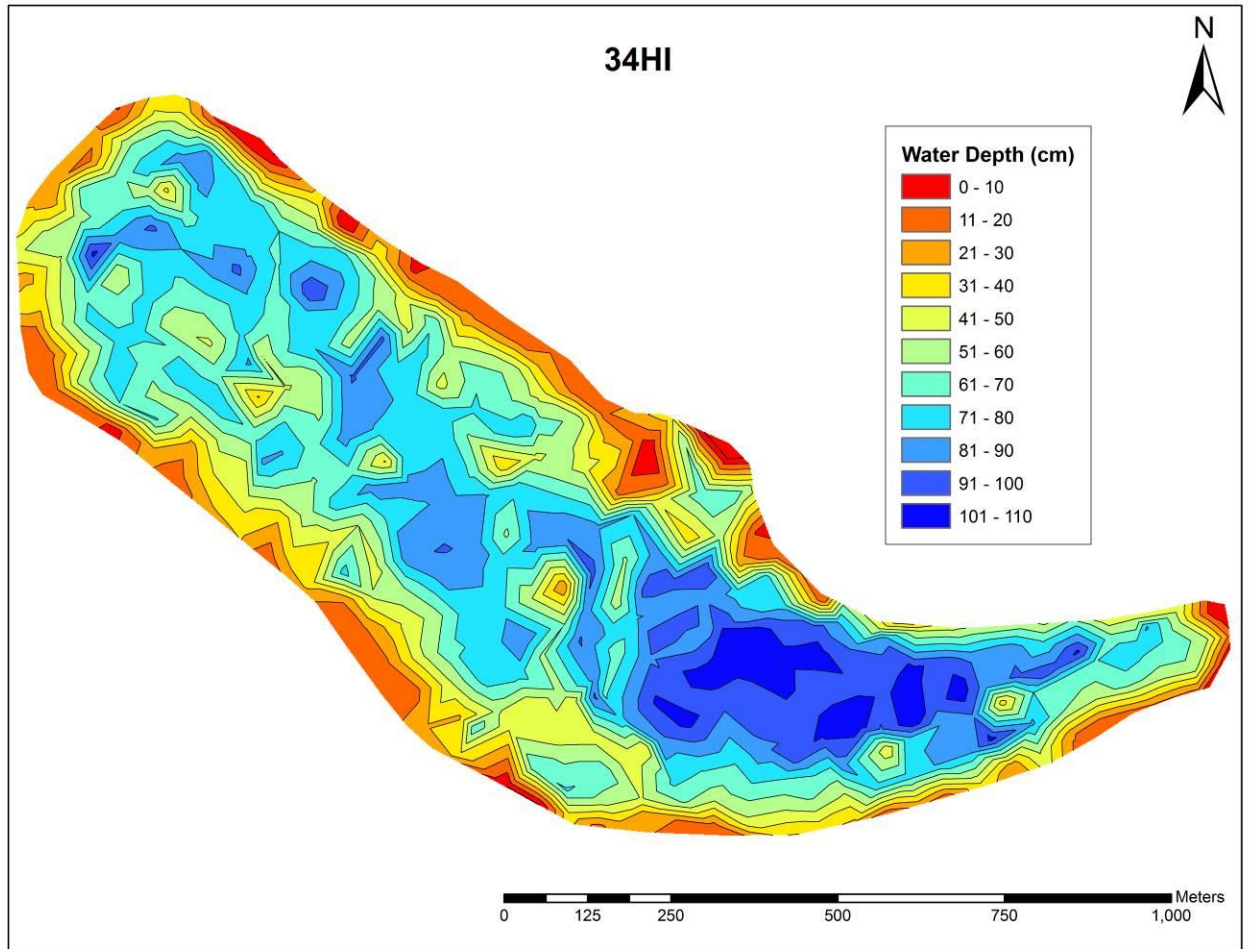


Figure 4.15- Water depth contours of wetland 34HI in the Summerberry Marsh Complex, Manitoba taken during ground-truthing in 2009.

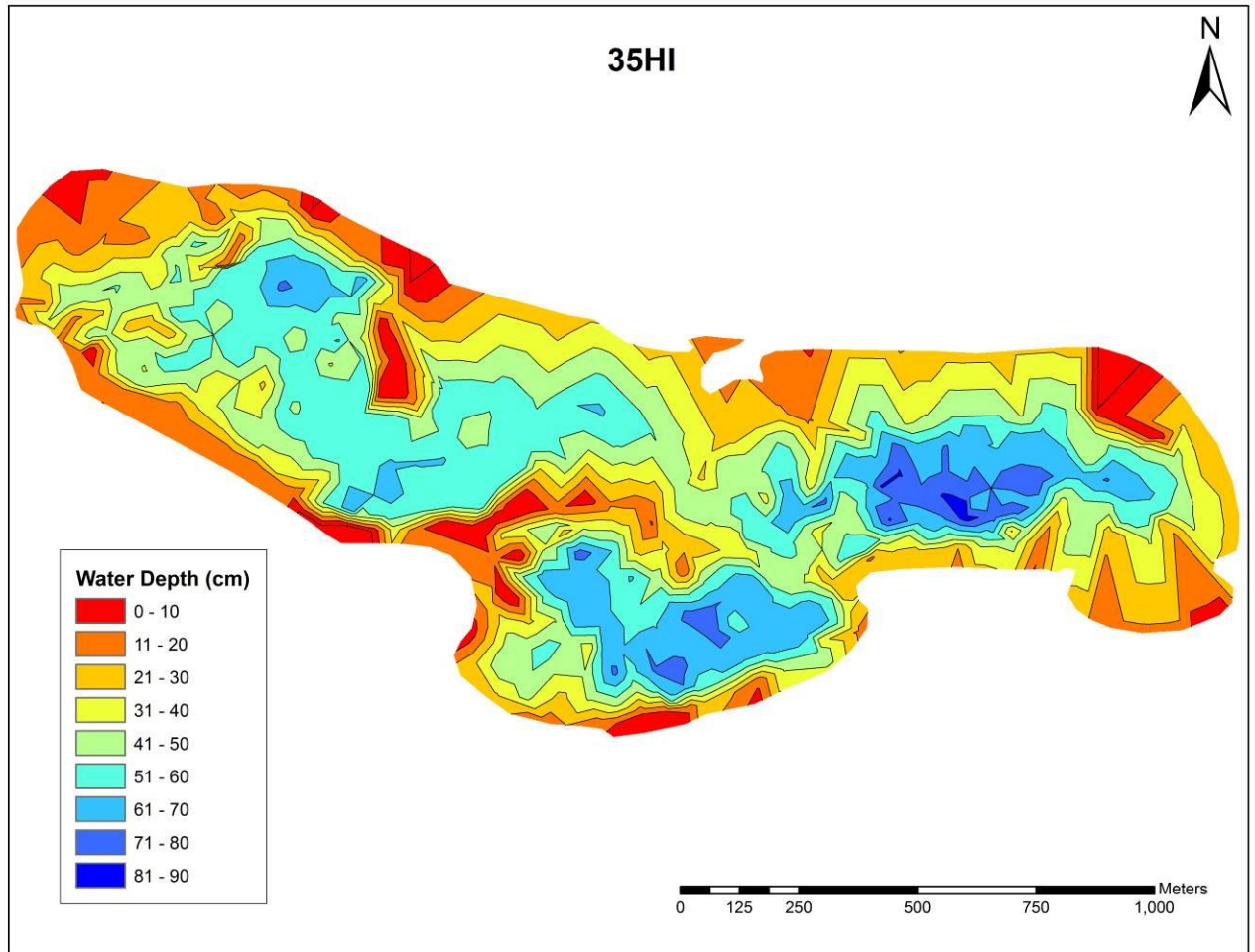


Figure 4.16- Water depth contours of wetland 35HI in the Summerberry Marsh Complex, Manitoba taken during ground-truthing in 2009.

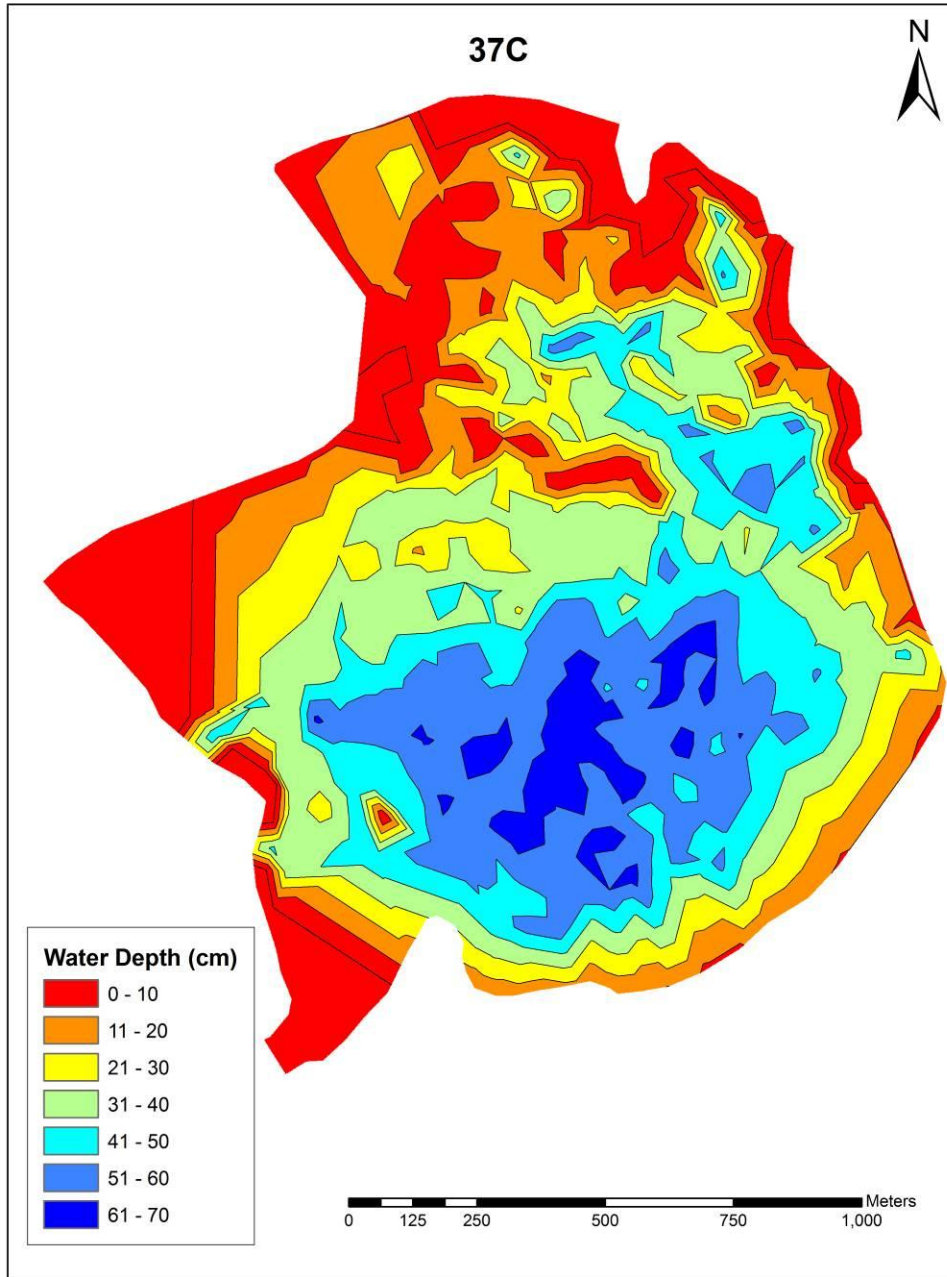


Figure 4.17- Water depth contours of wetland 37C in the Summerberry Marsh Complex, Manitoba taken during ground-truthing in 2009.

4.5.5 Object-based Image Classification

The most accurate model for 2008 used the full parameter estimation method for both the drawdown and non-drawdown wetlands. The most accurate model for 2009 used the full parameter estimation method for the drawdown wetlands and the stepwise selection method for the non-drawdown wetlands (Table 4.7). The stepwise selection procedure for the 2009 non-drawdown wetlands removed three variables, including the maximum difference between pixel values in each polygon, the standard deviation of the blue band, and the normalized difference vegetation index. Since the stepwise procedure produced similar accuracy results with fewer variables than the full procedure, it was used for classification.

Table 4.7- Fit statistics and overall class accuracies of GLMs using various parameter selection methods to classify vegetation in the three drawdown and three non-drawdown study wetlands in the Summerberry Marsh Complex, Manitoba, 2008-09. Shaded rows indicate models used to produce final vegetation maps.

Year	Treatment	Parameter Selection Method	Overall Class Accuracy (%)
2008	Non-drawdown	Full	76.48
	Drawdown	Full	71.74
	Non-drawdown	Stepwise	75.59
	Drawdown	Stepwise	71.15
	Non-drawdown	Backward	76.35
	Drawdown	Backward	71.15
	Non-drawdown	Reduced	64.28
	Drawdown	Reduced	64.29
2009	Non-drawdown	Full	79.90
	Drawdown	Full	80.74
	Non-drawdown	Stepwise	79.90
	Drawdown	Stepwise	79.74
	Non-drawdown	Backward	79.90
	Drawdown	Backward	79.74
	Non-drawdown	Reduced	72.51
	Drawdown	Reduced	76.60

In 2008 the drawdown wetlands had a lower overall class accuracy compared to the non-drawdown wetlands. *Carex*, *Equisetum*, and *Phragmites* had lower user's and producer's accuracies in the drawdown wetlands compared to the non-drawdown wetlands (Tables 4.8 & 4.9). *Equisetum* could not be accurately mapped in the drawdown wetlands. This was caused by a small sample size, and confusion with the *Typha* class (Table 4.8). The *Typha* class had a relatively low accuracy in both types of wetlands, while the Trees and Water classes had the highest accuracies of all classes in both drawdown and non-drawdown wetlands. However, the Trees and Water classes had lower accuracies in the non-drawdown wetlands compared to the drawdown wetlands (Table 4.9).

Table 4.8- Confusion matrix and class accuracies produced by the GLM using the full parameter selection method for the three drawdown wetlands (14R, 35HI, 37C) in the Summerberry Marsh Complex, Manitoba, 2008.

	Carex	Equisetum	Phragmites	Scirpus	Trees	Typha	Water	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
Carex	207	1	43	4	1	81	14	351	59.0	71.1	65.1	71.8
Equisetum	7	0	12	3	0	31	1	54	0.0	0.0	0.0	
Phragmites	33	0	170	9	0	39	5	256	66.4	60.9	63.7	
Scirpus	3	0	11	120	0	8	8	150	80.0	77.9	79.0	
Trees	0	0	0	0	115	0	0	115	100.0	99.1	99.6	
Typha	37	0	39	9	0	253	10	348	72.7	61.0	66.8	
Water	4	0	4	9	0	3	224	244	91.8	85.5	88.7	
Total	291	1	279	154	116	415	262	1518				

Table 4.9- Confusion matrix and class accuracies produced by the GLM using the full parameter selection method for the three non-drawdown wetlands (21C, 32C, 34HI) in the Summerberry Marsh Complex, Manitoba, 2008.

	Carex	Equisetum	Phragmites	Scirpus	Trees	Typha	Water	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
Carex	225	10	22	0	6	9	3	275	81.8	74.0	77.9	76.5
Equisetum	11	86	21	2	0	8	3	131	65.7	81.1	73.4	
Phragmites	18	7	416	4	9	29	11	494	84.2	77.8	81.0	
Scirpus	2	0	6	113	2	7	12	142	79.6	74.3	77.0	
Trees	1	0	17	0	104	0	0	122	85.3	85.3	85.3	
Typha	45	3	28	21	0	104	5	206	50.5	61.5	56.0	
Water	2	0	25	12	1	12	168	220	76.4	83.2	79.8	
Total	304	106	535	152	122	169	202	1590				

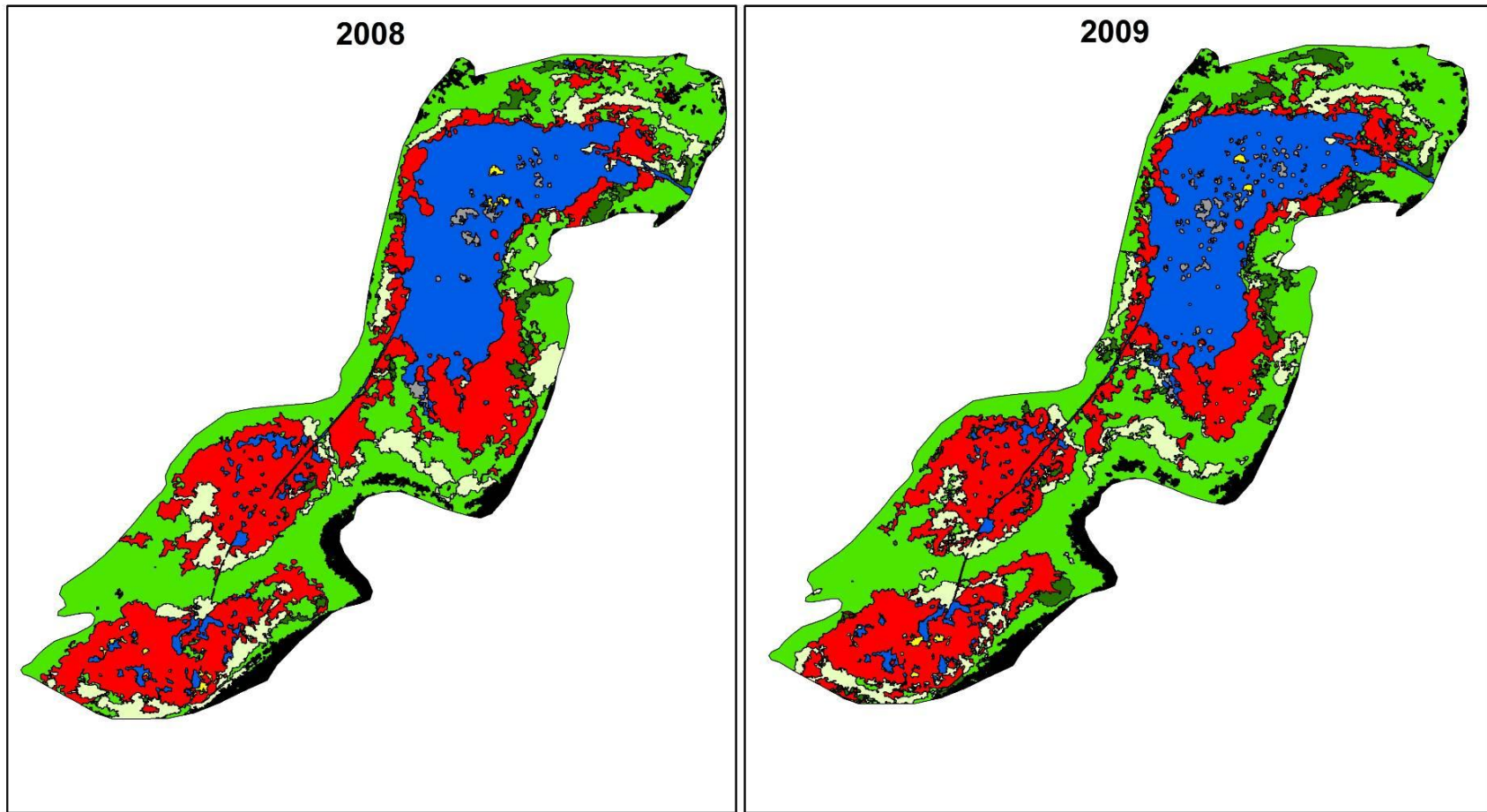
In 2009 the drawdown wetlands had slightly higher overall class accuracy than the non-drawdown wetlands. The overall class accuracy was higher for both the drawdown and non-drawdown wetlands in 2009 compared to 2008 (Tables 4.8-4.11). *Carex*, *Phragmites*, and *Scirpus* had comparable user's and producer's accuracies in the drawdown and non-drawdown wetlands in 2009 (Tables 4.10 & 4.11). *Equisetum* had a much lower accuracy in the drawdown wetlands due to a small sample size (Table 4.10). The Water and Trees classes had the highest accuracies of all classes in both types of wetlands. However, the Water class had lower accuracies in the non-drawdown wetlands compared to the drawdown wetlands. *Typha* accuracy was lower in the non-drawdown wetlands, but overall higher in 2009 compared to 2008 in both types of wetlands (Tables 4.10 & 4.11).

Table 4.10- Confusion matrix and class accuracies produced by the GLM using the full parameter selection method for the three drawdown wetlands (14R, 35HI, 37C) in the Summerberry Marsh Complex, Manitoba, 2009.









	Carex	Equisetum	Phragmites	Scirpus	Trees	Typha	Water	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
Carex	307	3	26	1	0	34	2	373	82.3	77.9	80.1	80.7
Equisetum	32	8	7	2	0	11	2	62	12.9	38.1	25.5	
Phragmites	20	4	204	10	0	33	1	272	75.0	71.3	73.2	
Scirpus	3	1	15	126	0	6	5	156	80.8	83.4	82.1	
Trees	0	0	0	0	116	0	0	116	100.0	100.0	100.0	
Typha	32	5	33	9	0	304	2	385	79.0	77.6	78.3	
Water	0	0	1	3	0	4	222	230	96.5	94.9	95.7	
Total	394	21	286	151	116	392	234	1594				

Table 4.11- Confusion matrix and class accuracies produced by the GLM using the step-wise parameter selection method for the three non-drawdown wetlands (21C, 32C, 34HI) in the Summerberry Marsh Complex, Manitoba, 2009.

	Carex	Equisetum	Phragmites	Scirpus	Trees	Typha	Water	Total	User's Accuracy (%)	Producer's Accuracy (%)	Average Accuracy (%)	Overall Class Accuracy (%)
Carex	241	12	34	0	5	7	1	300	80.3	76.5	78.4	79.9
Equisetum	20	99	13	0	1	4	0	137	72.3	77.3	74.8	
Phragmites	35	11	426	3	0	27	15	517	82.4	76.9	79.7	
Scirpus	0	1	9	128	0	7	5	150	85.3	85.9	85.6	
Trees	2	0	0	0	102	0	0	104	98.1	94.4	96.3	
Typha	15	5	52	10	0	133	5	220	60.5	74.7	67.6	
Water	2	0	20	8	0	0	179	209	85.7	87.3	86.5	
Total	315	128	554	149	108	178	205	1637				



Vegetation Class

 Sparganium	 Equisetum	 Scirpus	 Typha
 Carex	 Phragmites	 Trees	 Water

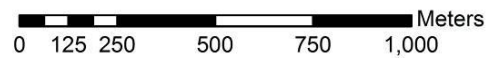


Figure 4.18- Object-based classification of wetland 14R in the Summerberry Marsh Complex, Manitoba, 2008 and 2009.

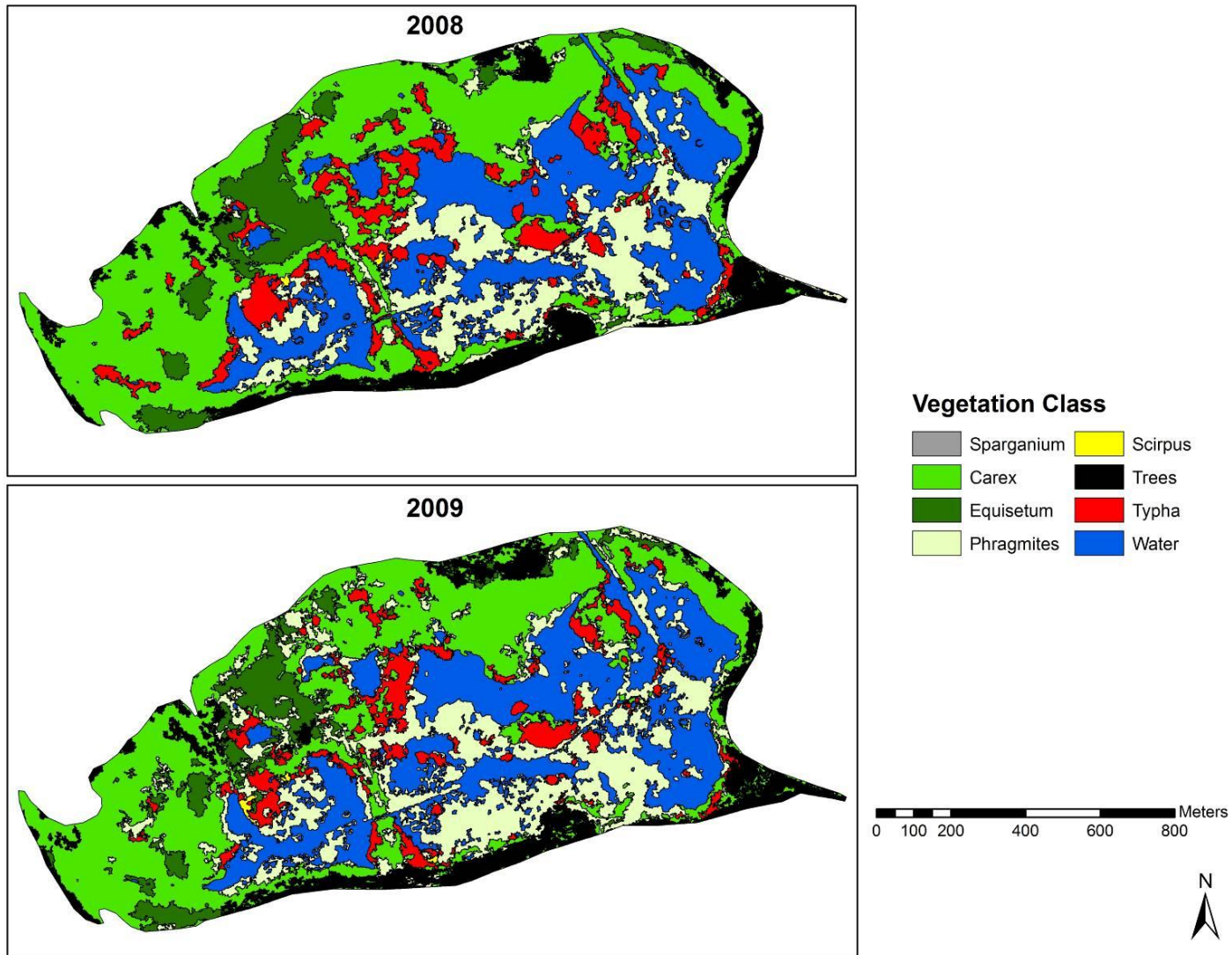


Figure 4.19- Object-based classification of wetland 21C in the Summerberry Marsh Complex, Manitoba, 2008 and 2009.

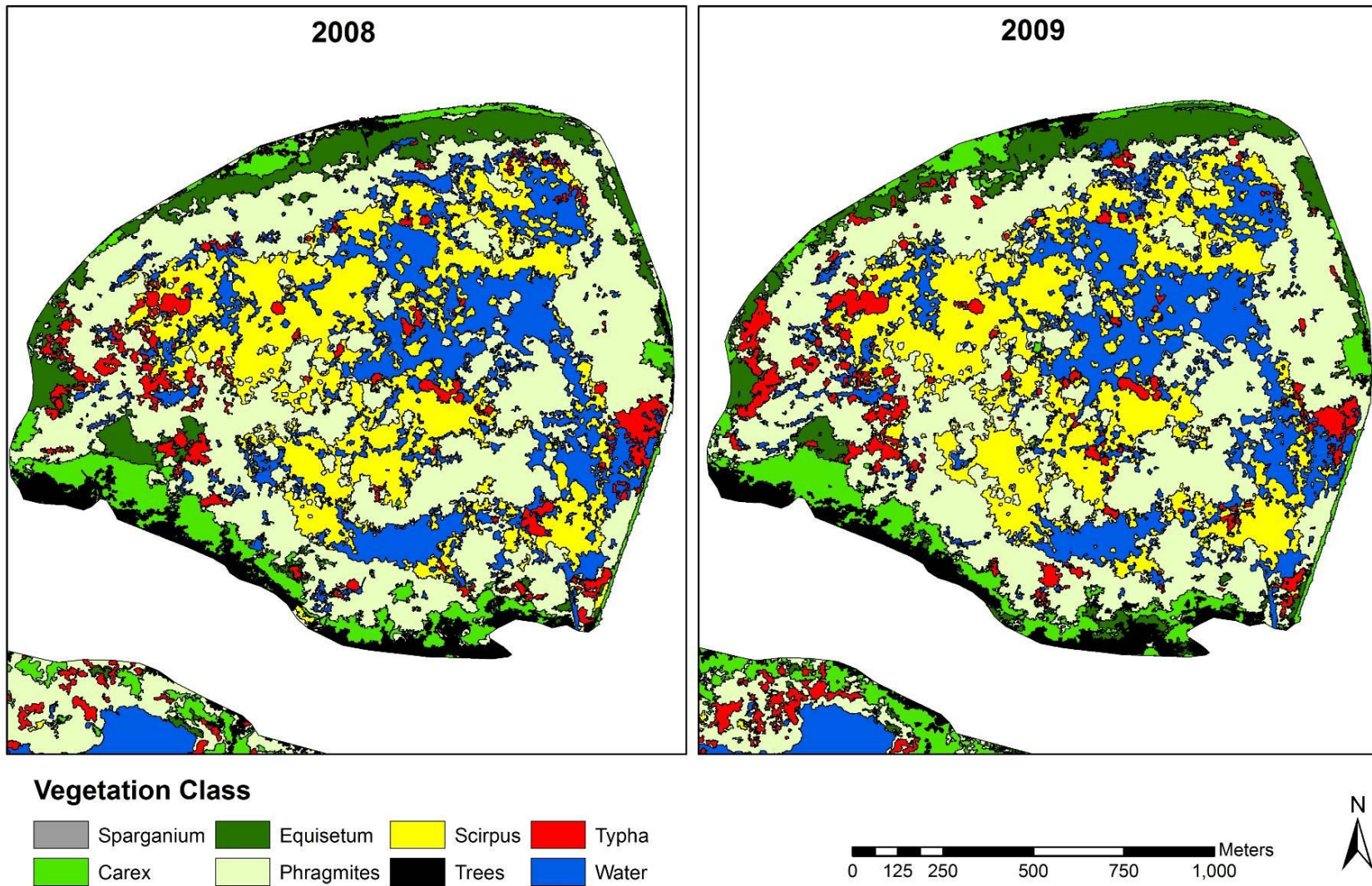


Figure 4.20- Object-based classification of wetland 32C in the Summerberry Marsh Complex, Manitoba, 2008 and 2009.

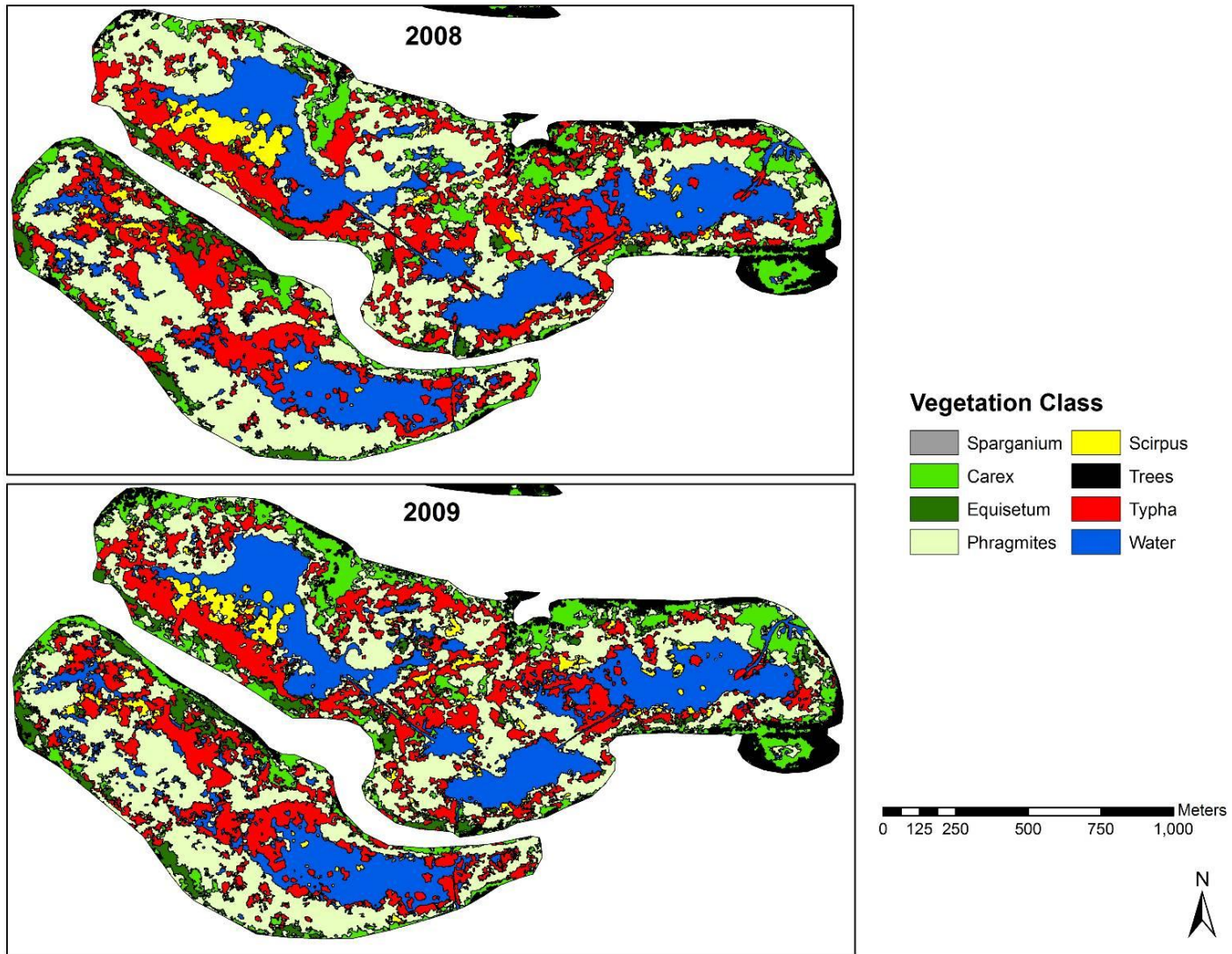


Figure 4.21- Object-based classification of wetlands 34HI and 35HI in the Summerberry Marsh Complex, Manitoba, 2008 and 2009.

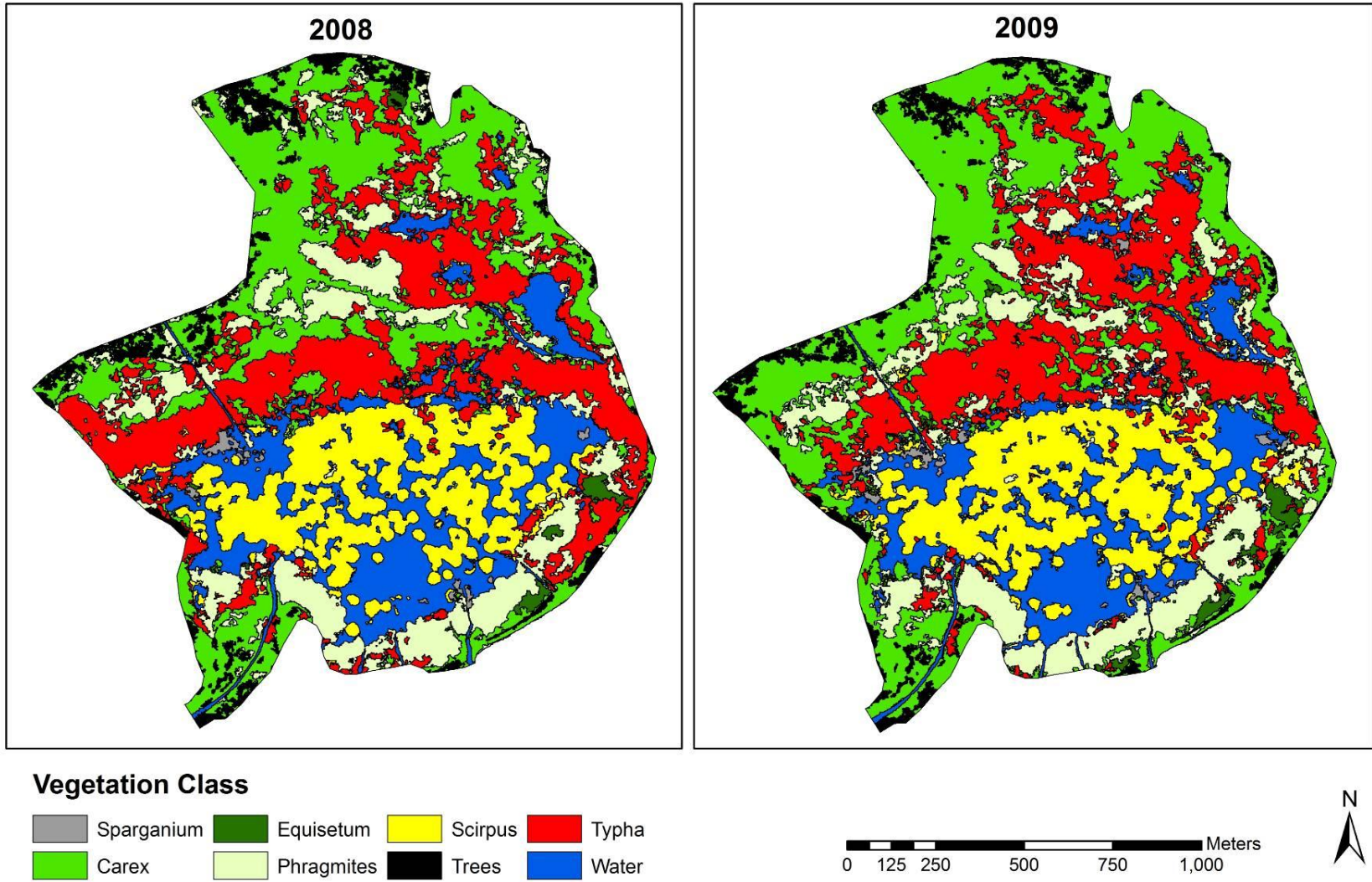


Figure 4.22- Object-based classification of wetland 37C in the Summerberry Marsh Complex, Manitoba, 2008 and 2009.

4.6 Discussion

4.6.1 Pixel-based Classifications

The unsupervised classification technique was not effective for delineating species-specific emergent vegetation types within the study wetlands. Some researchers have successfully used unsupervised classification to delineate different groups of wetland vegetation (Olmanson et al. 2002; Sawaya et al. 2003; Everitt et al. 2004; Everitt et al. 2005; Everitt et al. 2008). However, these researchers used more general vegetation classes and did not attempt to distinguish between vegetation species. The inability to detect different vegetation species limits the use of unsupervised classification for delineating waterfowl or marsh bird habitat. However, unsupervised classification was useful for quickly identifying the amounts of water, living vegetation, and dead vegetation within wetlands. Consequently, it could be used for determining amounts of vegetation, water, and vegetation/water interspersion.

Supervised classifications were slightly better than the unsupervised classifications. This technique produced less confusion among classes and identified more vegetation types. The benefit of the supervised classification was the ability to determine errors of omission and commission using the confusion matrices. From these matrices, more accurate training areas could be delineated, resulting in more accurate maps.

There were some disadvantages with the supervised classification. Several species of emergent vegetation were not accurately detected by the supervised classifications, and most classes had considerable amounts of speckle making the outputs very hard to

decipher. The classification accuracy may have been improved if ground-truthed data were available for selection of training areas. However, the supervised classification was done in 2008 and the ground-truthing in the SMC was not conducted until 2009. The lack of ground-truthed data may have reduced the accuracy of training areas and consequently, the accuracy of the classification. Laba et al. (2008) used supervised classification in combination with ground-truthed data to delineate 20 cover classes, including eight species-specific classes of wetland vegetation in the tidal wetlands of the Hudson River, New York State. Similar to my study, Laba et al. (2008) found that the accuracy of more general vegetation classes were higher than the species-specific vegetation classes; several species-specific classes had a user's accuracy of less than 50%, including *Typha* spp. and *Schoenoplectus* spp. Cumulatively, our research suggests that supervised classification of high resolution imagery is only adequate for determining general wetland vegetation characteristics, such as the amount of living and dead vegetation, and amount of open water. However, it is not sufficient to determine species-specific areas within the study wetlands.

The biggest challenge for the unsupervised and supervised classifications was the high resolution, 0.6 m, of the QuickBird satellite imagery. High resolution imagery contains many different pixel values within small areas. Combined with the pixel-based classification techniques only using spectral signatures of each individual pixel for classification, this resulted in large amounts of speckle and confusion between classes (Thomas et al. 2003). This was most evident from the confusion among the Trees class and other vegetation classes. The tree canopies were substantially heterogeneous in

vertical structure and texture, creating various degrees of shadowing and a vast array of spectral signatures among pixels. The other vegetation classes may have had unique spectral signatures, but due to the wide spectral range of the trees, these signatures overlapped, causing different pixels to be indistinguishable and creating large amounts of speckle in the outputs. This is a common problem when very high resolution imagery is used for pixel-based classifications (Nagendra 2001; Thomas et al. 2003; Nagendra and Rocchini 2008). This problem can be overcome using object-based classification (Thomas et al. 2003) or hyperspectral imagery that measures reflectance in many narrow spectral bands (Nagendra and Rochinni 2008).

The ability to separate different vegetation species by the unsupervised and supervised classifications was limited by the low number of spectral bands present in the colour infrared imagery. The imagery used in the analysis only contained three bands, reducing the likelihood of vegetation species having unique spectral signatures that could be separated (Nagendra 2001). The pixel-based classifications could be improved if hyperspectral imagery, such as AVIRIS, were used. This imagery can provide up to 224 spectral bands, drastically increasing the chances of finding unique spectral signatures of vegetation species (Green et al. 1998). Thenkabail et al. (2004) and Clark et al. (2005) both found that hyperspectral imagery, despite the lower spatial resolution, produced more accurate results when identifying tree species compared to higher spatial resolution, multispectral imagery. Therefore, even though the spatial resolution of imagery is important for identifying small areas/objects, spectral resolution is more useful for identifying subtle spectral differences among vegetation species..

In this study, the use of colour infrared imagery and pixel-based techniques essentially provided a classification of vegetation health rather than a classification of vegetation species. Healthy plants contain large amounts of chlorophyll, which strongly absorbs blue and red energy with wavelengths from 450-670 nm (Carter 1993). The internal structure of the plant leaves strongly influences the absorption of energy in the 700-1300 nm range, or near-infrared range. Since internal structure of leaves varies among species, reflectance measurements at these wavelengths are useful for distinguishing among different species. However, when vegetation is stressed, as caused from a water-level drawdown, the amount of chlorophyll it contains decreases, and the internal structure of the leaves may also be altered. This increases the amount of reflectance in the red, blue, and near-infrared wavelengths (Lillesand et al. 2008). Therefore, in the drawdown wetlands of this study, vegetation species were not as distinct because the spectral signatures no longer represent the differences between vegetation species. Instead, the spectral signatures represent the stress the vegetation is undergoing.

4.6.2 Ground-truthing for Object-based Classification

Ground-truthing was time consuming but provided thorough coverage of each wetland basin. Lillesand et al. (2008) warns the confusion matrix should not be used to determine the overall accuracy of a map, as it represents the accuracy within training areas, not the entire study area. However, I believe with our extensive ground-truthing, which covered the entire study area, the confusion matrices are a good indication of overall map accuracy.

4.6.3 Object-based Image Classification

The use of the full parameter selection method produced the highest, or equally highest, overall class accuracy results compared to the other parameter selection methods. Due to the relatively high number of vegetation classes in my study, using all the variables in the selection model increased the chances of accurately discriminating vegetation classes from one another. Future studies should also compare parameter selection methods in order to avoid over-fitting models and obtain the most accurate classification possible.

Overall, the accuracies of the object-based classification maps were high (>70%). The object-based classification was able to distinguish all vegetation species with much less confusion between classes compared to the pixel-based classifications. Classifying areas of contiguous groups of pixels rather than individual pixels reduced the amount of spectral variation caused by vegetation texture, eliminating speckle, and reducing confusion between classes. Benoit et al. (2008) also noted these advantages of object-based classification for delineating habitat of Grasshopper Sparrow (*Ammodramus savannarum*), and recommended its use by biologists and ecologists. Due to the high accuracy of the maps produced and the absence of speckle, this technique was also the best choice for delineating emergent vegetation in my study.

The accuracy of most vegetation species classes from the object-based classification were high (>70%). This was comparable to the findings of Gilmore et al. (2008) who obtained an average accuracy of 78% when mapping emergent vegetation species. My accuracy

rate may be lower because, (1) Gilmore et al. (2008) used LiDAR data and multi-season imagery to increase classification accuracy, and (2) I had twice the number of classes in my study. Similar to Gilmore et al. (2008), I also found that *Typha* was the most difficult vegetation species to distinguish in our classification. This is due to *Typha* having a similar spectral signature to *Phragmites* (Ghioaca-Robrecht et al. 2008; Lopez et al. 2004). However, the use of LiDAR data to distinguish the differences in height between the species could be used to increase classification accuracy as shown in Gilmore et al. (2008).

Despite the relatively high accuracy of object-based classifications, there were some classes that caused some difficulty. *Equisetum* was uncommon in the drawdown wetlands (Figures 4.17, 4.21, 4.22). This resulted in very few ground-truthed points in these areas, which increased the chances of omission and commission errors, and caused the low accuracy of the *Equisetum* class in the drawdown wetlands. Despite the low accuracy of this class, areas of *Equisetum* were consistently found in the same areas of wetlands in 2008 and 2009, suggesting the low accuracy was due to small sample sizes and not spectral signature. This is supported by relatively high accuracies found in the non-drawdown wetlands, where sufficient samples sizes were present.

The Water class in the non-drawdown wetlands was less accurate compared to the drawdown wetlands. This was due to a greater amount of vegetation/water interspersion present in the non-drawdown wetlands (i.e., *Phragmites* in 21C) as well as more areas of sparse vegetation growing out of water (i.e., *Scirpus* in 32C). The small clumps of

vegetation and sparse vegetation over water caused the background reflectance to dominate the reflectance of the vegetation causing the confusion. This resulted in more errors of omission and commission, reducing the overall accuracies of the Water class. Fuller (2005) had this problem when attempting to detect small groups of invasive trees in the wetlands of the Everglades National Park, Florida, USA. He found that background reflectance dominated the spectral signature of pixels containing small groups of trees, causing them to be indistinguishable from the surrounding vegetation. However, Fuller (2005) used IKONOS multispectral imagery with 4-m resolution. Since we used higher resolution imagery, 0.6 m, we did not encounter the same extent of background interference as Fuller (2005).

There were some problems associated with the object-based classification. Some segments contained ground-truthed points with conflicting vegetation types. This was likely due to two sources of error. The first was due to error in the segmentation. Some areas do not consist of totally homogenous vegetation species, causing segments to contain a mixture of vegetation. We took precautions to prevent this, such as choosing a scale parameter that was fine enough to detect some of this vegetation heterogeneity (shown by a stand of a single species of vegetation being broken up by multiple segments). However, there were undoubtedly segments that were under-segmented, containing a mixture of vegetation species in a single segment, which would have decreased the accuracy of our GLM models.

Some of the error in segmentation accuracy could potentially be decreased by lowering the scale parameter to account for vegetation heterogeneity. This would increase the sensitivity of the segmentation process, creating smaller polygons that could delineate different vegetation species on a finer scale. However, decreasing the scale parameter also increases the chances of over-segmentation (Schiewe 2002), which occurs when a homogenous area is segmented needlessly. This is less of a problem than under-segmentation, as similar, neighbouring segments can later be merged together (Schiewe 2002). Overall, the scale parameter chosen is subjective and should be based on the requirements of the user (Schiewe 2002). We chose a scale parameter that appeared to slightly over-segment our image so the similar, neighbouring polygons could later be merged.

The second source of error was caused by inaccuracy in the GPS units used during ground-truthing. The GPS unit used has an accuracy of 3-5 m 95% of the time (Garmin Ltd. 2007) causing the ground-truthed points to have some associated spatial error. This spatial error may have resulted in some ground-truthed points being located in the wrong segment. This could have caused a misrepresentation of the vegetation type within the segment, which would have decreased the accuracy of our GLM models. We attempted to reduce the possibility of this by placing ground-truthed points in the centres of the vegetation patch, rather than at the edges. However, this error still occurred. The error associated with the GPS unit could be reduced if a more accurate GPS unit was used. Hand-held units are available that can achieve an accuracy of <30 cm, which would eliminate this problem.

4.7 Conclusions

The use of very high resolution satellite imagery and pixel-based classification techniques did not work well together as spectral values were highly variable in small areas. This caused a limited number of species-specific vegetation classes to be identified and large amounts of confusion between classes. These problems were more prevalent in the drawdown wetlands due to the stressed conditions of the vegetation. These techniques may be suitable for delineating basic wetland features, such as the amounts of living and dead vegetation, and the amount of open water. However, they are not suitable for delineating species-specific areas within wetlands.

The object-based classification was much better suited for the high resolution imagery. It was able to segment areas of pixels, and to classify these with higher accuracy. All classes of vegetation were able to be delineated using this technique, allowing detailed vegetation maps of each study wetland to be created. Similar to the pixel-based classifications, the drawdown wetlands had lower overall classification accuracies due to the stressed conditions of the vegetation. To increase classification accuracies, another class, identifying stressed/senescing vegetation could be included in future analyses. Future analyses should also consider the use of different parameter selection methods. In this study, the full models provided the highest overall classification accuracies, likely due to the large number of vegetation classes used. However, if future studies use a different number or type of vegetation classes, parameter selection methods should be used to find the most accurate and parsimonious models.

One drawback associated with the technique was the amount of ground-truthing performed. The number of ground-truthed points could be reduced if detailed bathymetric maps of the wetland basins are available or if depth is not included as a variable in the classification model. The number of ground-truthed points could also be reduced by delineating homogenous stands of vegetation species on the ground to develop “training areas” to use for modeling in the GLM, rather than using data from individual points.

In the future, this technique could be applied over large areas of the SRD to allow for large-scale monitoring of wetlands. This would allow managers to monitor, in detail, the response of wetlands to management activities, such as water-level manipulations.

Changes in vegetation areas, structure, and species composition could be monitored with relatively high accuracy. This technique is currently being applied in an attempt to delineate and quantify the changes of vegetation species and health in the SMC study area during the partial water-level drawdowns from 2007-2009 (M. Ervin, 2010, Iowa State University, Thesis).

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Appendix 4.1 - Vegetation Class Areas Produced by Object-based Classification

Table 4.12- Area (ha) of each cover type mapped in the study wetlands in 2008 and 2009 in the Summerberry Marsh Complex, Manitoba.

	Partial Drawdown						Non-drawdown					
	14R		35HI		37C		21C		32C		34HI	
	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009
Carex	49.83	52.85	16.89	21.34	50.59	55.88	50.51	46.42	14.34	14.77	6.74	6.10
Equisetum	2.90	4.83	3.38	4.77	1.45	2.89	9.79	8.65	14.63	14.51	4.64	5.07
Phragmites	14.52	14.04	55.23	50.62	30.43	31.20	22.80	29.23	82.54	81.64	34.91	35.80
Scirpus	0.64	0.39	6.70	6.73	29.39	30.15	0.19	0.32	43.20	44.71	2.27	1.64
Sparganium	0.95	1.44	0.05	0.07	0.94	1.40	0.00	0.00	0.02	0.16	0.00	0.00
Trees	5.80	5.57	7.29	7.64	7.70	6.66	10.41	11.06	7.64	7.34	0.92	0.95
Typha	39.33	35.89	32.97	31.24	45.10	41.36	12.19	9.74	9.85	9.95	20.36	19.71
Water	29.01	27.96	33.28	33.38	29.21	25.30	30.77	31.25	35.01	34.40	14.13	14.71

Table 4.13- Percent of total area of each cover type mapped in the study wetlands in 2008 and 2009 in the Summerberry Marsh Complex, Manitoba.

	Partial Drawdown						Non-drawdown					
	14R		35HI		37C		21C		32C		34HI	
	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009	2008	2009
Carex	35	37	11	14	26	29	37	34	7	7	8	7
Equisetum	2	3	2	3	1	1	7	6	7	7	6	6
Phragmites	10	10	35	32	16	16	17	21	40	39	42	43
Scirpus	0	0	4	4	15	15	0	0	21	22	3	2
Sparganium	1	1	0	0	0	1	0	0	0	0	0	0
Trees	4	4	5	5	4	3	8	8	4	4	1	1
Typha	28	25	21	20	23	21	9	7	5	5	24	23
Water	20	20	21	21	15	13	23	23	17	17	17	18