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The Effects of Invasion by Reed Canarygrass (Phalaris arundinacea) on Avian Communities and Nesting Success in Minnesota Wetlands

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THE EFFECTS OF INVASION BY REED CANARYGRASS (*PHALARIS ARUNDINACEA*) ON AVIAN COMMUNITIES AND NESTING SUCCESS IN MINNESOTA WETLANDS

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ABSTRACT

Invasive plants are a primary contributor to loss of biodiversity worldwide. In southern Minnesota, many wetlands have been invaded by reed canarygrass (*Phalaris arundinacea*). The current perception among ecologists and resource managers is that these wetlands are of little value to wildlife, yet little is known about the effects on birds of the widespread conversion of diverse wetlands to apparent monocultures of *P. arundinacea*. The purpose of this study was to determine the effects of *P. arundinacea*-mediated changes in the wetland plant community on avian communities and nesting success. During 2006 and 2007, I studied four diverse sedge wetlands paired with four wetlands dominated by *P. arundinacea* in the farmland region of southern Minnesota. I measured vegetative structure and composition, surveyed birds year-round via the fixed-radius point count technique, and conducted nest searching and monitoring to assess nesting success of Red-winged Blackbirds (*Agelaius phoeniceus*). Vegetation in wetlands invaded by *P. arundinacea* was taller and had greater visual obstruction readings than vegetation in sedge wetlands, but sedge wetlands had greater plant species richness and number of woody stems/100 m^2 that were < two meters tall. Plant species diversity, litter depth, horizontal heterogeneity, and number of woody stems/100 m^2 that were > two meters were not different between habitat types. Bird species richness was greater in wetlands invaded by *P. arundinacea* during the breeding season but did not differ between habitat types during the non-breeding season. Bird species diversity was not different between habitat types during either season. The abundance of individual species, including rare and listed species, also was not different between habitat types for

either season, with one exception. The Ring-necked Pheasant (*Phasianus colchicus*) was more abundant in wetlands dominated by *P. arundinacea* during the non-breeding season. Rare species collectively contributed similar percent composition to the bird communities of each habitat type. Furthermore, nesting success and density of nests/10 hectares of Red-winged Blackbirds was not different between habitat types. Results of this study did not indicate that invasion by *P. arundinacea* has a negative effect on bird communities or nesting success of Red-winged Blackbirds in wetlands of southern Minnesota. The invasion by *P. arundinacea* does not appear to have altered the structure of wetland vegetation in a way that negatively affects birds and may provide better avian habitat than is currently perceived. Although invasion by *P. arundinacea* had mixed effects on the plant community in this study, it has had marked negative effects on other native plant communities and is likely to be a continual problem in the restoration and management of wetlands in Minnesota.

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CHAPTER 1

INTRODUCTION

Composition of avian communities is influenced by structure of vegetation (MacArthur and MacArthur 1961, Cody 1968, Wiens 1969, Wiens 1974). Foliage height diversity (MacArthur and MacArthur 1961) is a measure of the variation in vertical structure and layering of vegetation, and is one of the most important characteristics of vegetation that affects bird species diversity (MacArthur and MacArthur 1961, Cody 1968, Wiens 1969, Karr and Roth 1971, Willson 1974). More vertical layers of vegetation yield a more structurally complex plant community, which leads to greater bird species diversity (MacArthur and MacArthur 1961, Cody 1968, Wiens 1969, Karr and Roth 1971, Willson 1974). Additionally, horizontal patchiness, or heterogeneity, (MacArthur et al. 1962, Cody 1968, Wiens 1974, Roth 1976) is a measure of the variation in horizontal form and structure of a plant community. Bird species diversity increases with increasing horizontal heterogeneity (MacArthur et al. 1962, Cody 1968, Karr and Roth 1971, Wiens 1974, Roth 1976). Plant communities with a variety of plants in distinct patches support greater bird species diversity than plant communities that are monotypic or low diversity (MacArthur et al. 1962, Cody 1968, Karr and Roth 1971, Wiens 1974, Roth 1976). Lastly, most birds appear to respond to the structure of vegetation more than plant species composition (MacArthur 1961, Cody 1968, Wiens 1974, Willson 1974). However, plant species composition influences the structure of vegetation, and therefore indirectly affects bird species diversity (MacArthur 1957, MacArthur and MacArthur 1961, MacArthur et al. 1962).

The choice of nesting sites and nesting success of birds also are influenced by structure of vegetation (Crabtree et al. 1989, Johnson and Temple 1990, Mankin and Warner 1992, Martin 1993, Camp and Best 1994, McCoy et al. 2001, Davis 2005). Tall, dense vegetation may restrict the activity of predators and conceal nests, resulting in greater nesting success than in sparser vegetation (Dwernychuk and Boag 1972, Martin and Roper 1988, Johnson and Temple 1990, Mankin and Warner 1992, Martin 1993, Davis 2005). Many species of birds avoid homogenous vegetation, selecting nest sites with more heterogeneous and diverse cover (Mankin and Warner 1992, McCoy et al. 2001). In fact, vegetation around successful nests often has greater heterogeneity (Bowman and Harris 1980, Crabtree et al. 1989, Mankin and Warner 1992, McCoy et al. 2001) and plant diversity than around depredated nests (Crabtree et al. 1989, McCoy et al. 2001). Although plant species composition influences vegetative structure, it is less important for birds in the choice of nesting sites (Kantrud and Higgins 1992, McCoy et al. 2001). Birds may select vegetative features at the nest-site scale and at larger spatial scales such as the habitat patch surrounding the nest (Davis 2005).

Whereas the structure of vegetation influences birds, anthropogenic factors can influence the structure of vegetation, including the introduction and invasion of exotic species (Wilcox 1995). Invasive plants are a growing concern for conservation of native plant communities and are a primary contributor to loss of biodiversity (Vitousek et al. 1996). Invasive plants can displace native species, thereby affecting the composition and structure of native plant communities (Wilson and Belcher 1989, Vitousek et al. 1996). Wetlands are particularly susceptible to invasive plants because even small changes to a wetland's physio-chemical environment, such as addition of nutrients and sediment or altered hydrology, can result in major changes to the biological community (Wilcox 1995, Zedler and Rea 1998, Zedler and Kercher 2004). If these changes are beyond the natural range of variation and sources of invasive plants are available, natural vegetation may be displaced by invasive plants, especially if they have rapid growth and high reproductive rates and wide tolerance to the physical environment (Zedler and Rea 1998). Changes to a biological community can lead to alteration of nutrient cycling and disturbance regimes that may result in a nonreversible change in ecosystem function (Vitousek 1990, Vitousek et al. 1996). Furthermore, such changes in the plant community may lead to changes in the structure and function of higher trophic levels, such as avian communities (Rawinski and Malecki 1984, Wilson and Belcher 1989, Benoit and Askins 1999, Whitt et al. 1999, Scheiman et al. 2003, Maddox and Wiedenmann 2005) and nesting success (Schmidt and Whelan 1999, Lloyd and Martin 2005).

Plant invasions can have variable effects on native bird communities and nesting success, and effects often vary between bird species within the same community. For example, coastal wetlands dominated by phragmites (*Phragmites australis* Cav. Trin. ex Steud.) had lower bird species richness than diverse coastal wetlands, and state listed birds were less abundant in phragmites than diverse wetlands. However, Marsh Wrens (*Cistothorus palustris*) and Swamp Sparrows (*Melospiza georgiana*) had higher densities in wetlands dominated by phragmites (Benoit and Askins 1999). Similarly, wetlands dominated by purple loosestrife (*Lythrum salicaria* L.) had lower avian diversity but higher avian densities than wetlands with native vegetation (Whitt et al. 1999). Invasive plants can adversely affect nesting birds by causing increased predation (Schmidt and Whelan 1999, Lloyd and Martin 2005), avoidance of invaded habitats by some nesting species (Rawinski and Malecki 1984, Maddox and Wiedenmann 2005), slower weight gain and longer nestling periods, which increases vulnerability, and decreased final mass in nestlings that may reduce future survival (Lloyd and Martin 2005). Conversely, invasive plants can positively affect nesting birds. Although grasslands invaded with leafy spurge (*Euphorbia esula* L.) had lower nest densities and fewer nesting species, nesting success of Western Meadowlarks (*Sturnella neglecta*) was positively correlated with percent cover of spurge (Scheiman et al. 2003). Lastly, plant invasions may cause changes in nesting phenology by affecting when the substrate is suitable for nesting (Maddox and Wiedenmann 2005), and later nesting dates can lead to reduced nesting success (Mayfield 1975, Hochachka 1990). In all studies, invasive plants altered vegetative structure and affected the availability of resources for birds, such as food (insects) and suitable nesting substrates. The shift in available resources resulted in shifts in composition of the bird community and abundance of individual species (Rawinski and Malecki 1984, Wilson and Belcher 1989, Benoit and Askins 1999, Schmidt and Whelan 1999, Whitt et al. 1999, Scheiman et al. 2003, Lloyd and Martin 2005, Maddox and Wiedenmann 2005).

Reed canarygrass (*Phalaris arundinacea* L.) is an invasive, perennial grass that has altered plant communities in wetlands of North America (Apfelbaum and Sams 1987, Galatowitsch et al. 1999, Lavergne and Molofsky 2004, Schooler et al. 2006). Though native to North America (Anderson 1961, Apfelbaum and Sams 1987, Lavergne and Molofsky 2004), *P. arundinacea* has become increasingly invasive with repeated introductions of Eurasian strains since 1850 (Lavergne and Molofsky 2004). Hybridization with introduced strains and changes in nutrient loading and hydrology of wetlands may be contributing to the increased invasiveness of this species (Green and Galatowitsch 2002, Maurer et al. 2003). Once established, *P. arundinacea* is able to rapidly out-compete diverse wetland vegetation. Dominance of *P. arundinacea* alters the structure of plant communities by decreasing plant diversity and spatial heterogeneity of vegetation (Apfelbaum and Sams 1987, Galatowitsch et al. 1999, Kercher et al. 2004, Lavergne and Molofsky 2004, Schooler et al. 2006). Furthermore, under high nutrient conditions that often facilitate invasion, *P. arundinacea* grows taller and produces more aboveground biomass than other wetland plants (Green and Galatowitsch 2001, Green and Galatowitsch 2002, Lindig-Cisneros and Zedler 2002, Maurer and Zedler 2002, Maurer et al. 2003).

The popular consensus among ecologists and resource managers in the Midwestern United States is that *P. arundinacea*-dominated wetlands are of little value to wildlife, especially birds (Steinauer 1999, Groshek 2000). In reality, the consequences for birds of the widespread conversion of diverse wetland plant communities to apparent monocultures of *P. arundinacea* are largely unknown (but see Kirsch et al. 2007). The purpose of this study was to determine the effects of *P. arundinacea*-mediated changes in the wetland plant community on the avian community and on nesting success. More specifically, I investigated 1) the differences in structure of the plant community in wetlands dominated by *P. arundinacea* compared to diverse sedge wetlands and determined if the difference in structure had an effect on 2) species richness and diversity of birds, 3) abundance of individual species, and 4) nesting success. Because *P. arundinacea* out-competes many native plants for nutrients and light, I predicted that invaded wetlands would be dominated by *P. arundinacea* and have lower plant species richness and diversity than sedge wetlands. Consistent with previous research, I expected that vegetation in wetlands invaded by *P. arundinacea* would be taller, have greater visual obstruction readings (VOR), and have lower horizontal heterogeneity than vegetation in sedge wetlands. Because litter depth is inversely proportional to horizontal heterogeneity (Weins 1974), I expected that wetlands invaded by *P. arundinacea* would produce greater litter depths than sedge wetlands. Lastly, I hypothesized that altered vegetative structure associated with invasion by *P. arundinacea* would cause changes in resources available to birds and impact bird species richness and diversity, especially rare and listed species, and nesting success.

CHAPTER 2

METHODS

Study Area

I conducted the study from spring 2006 to fall 2007 in the farmland region of southern Minnesota, USA. The study area spanned five counties located in the Prairie Pothole Region (PPR) of North America (Table 1). Prior to European settlement, the PPR was characterized by myriad shallow wetlands interspersed in a matrix of tallgrass prairie and aspen parkland that provided habitat for a vast abundance and diversity of wildlife (Dinsmore 1994). Intensive row-crop farming now dominates the southern Minnesota landscape, where less than one percent of native prairies and wetlands remain (Dahl 1990, Noss et al. 1995). Many remaining native habitats are degraded due to fragmentation, loss of diversity, invasive species, altered hydrology, changes in nutrient availability, and altered disturbance regimes (Dahl 1990, Vitousek 1990, Noss et al. 1995).

The study design consisted of four diverse sedge wetlands paired with four wetlands dominated by *P. arundinacea* (Table 1; Figure 1). The paired sites were close in proximity to each other and similar in landscape context with regard to surrounding habitat, land use, and size. Paired sites also appeared to contain similar amounts and size classes of woody vegetation. Diverse sedge wetlands were rare, limiting the available sites to four. Once I located diverse sedge wetlands using data from the Minnesota County Biological Survey (2006), I selected sites dominated by *P. arundinacea* that were proximal to the diverse sites. Because geographic location and surrounding landscape

Table 1. Location and size of four sedge wetlands paired with four wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) used to evaluate the influence of invasion by *P. arundinacea* on birds and plant communities in southern Minnesota during 2006 and 2007.

Site	Block	Habitat type	County	Size (ha)	UTM N	UTM E
Cannon River[a]	1	Sedge	Rice	5.81	466580 N	4898546 E
Cannon River[a]	1	RCG	Rice	3.79	466405 N	4898570 E
Ottawa[a]	2	Sedge	Le Sueur	10.55	426698 N	4910629 E
Rasmussen Woods[b]	2	RCG	Blue Earth	12.35	419151 N	4888823 E
Judson[c]	3	Sedge	Blue Earth	8.79	407790 N	4894057 E
Swan Lake[a]	3	RCG	Nicollet	6.49	403049 N	4896197 E
Pogones[a]	4	Sedge	Steele	1.25	487784 N	4860628 E
Oak Glen[a]	4	RCG	Steele	1.5	491719 N	4864682 E

[a] State Wildlife Management Area
[b] City park
[c] Private

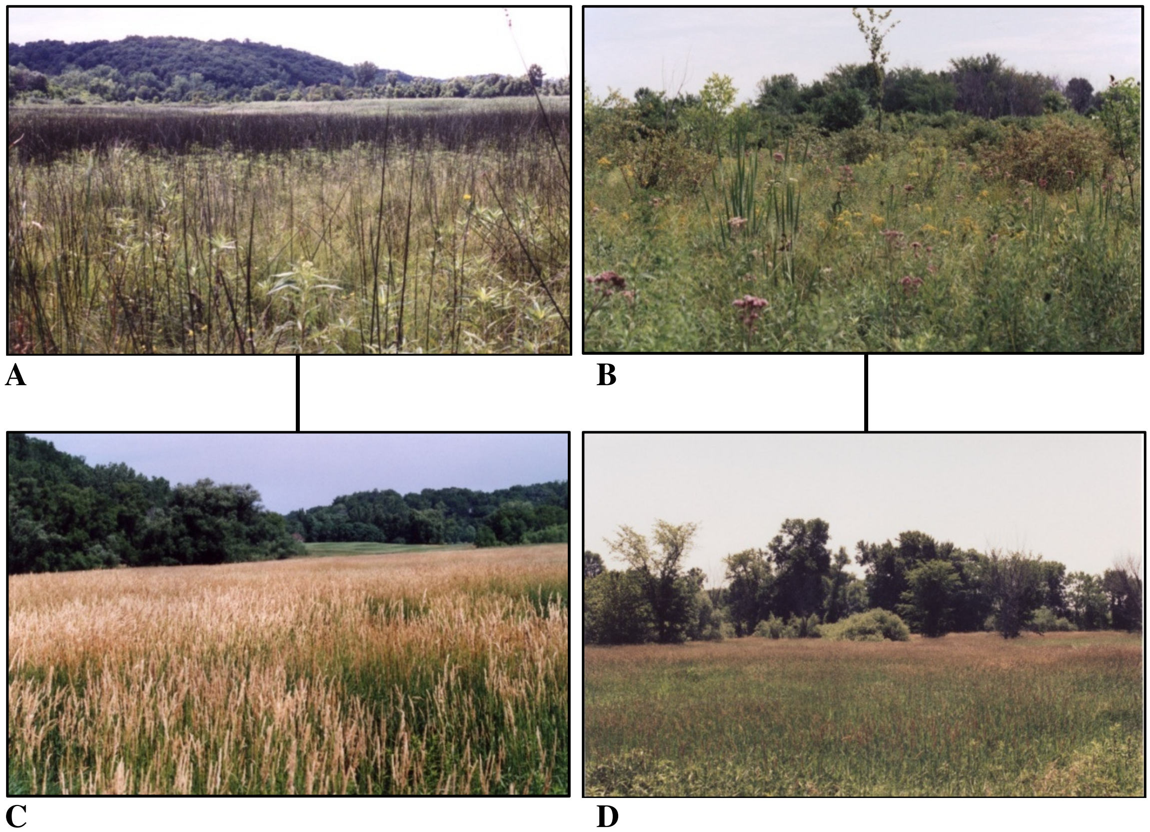

Figure 1. Sedge wetlands A) Ottawa State Wildlife Management Area and B) Cannon River State Wildlife Management Area paired with wetlands dominated by *P. arundinacea* C) Rasmussen Woods City Park and D) Cannon River State Wildlife Management Area, respectively.

can have dramatic influences on the bird community of a given area (Mossman and Sample 1990, Pearson 1993, Herkert 1994, Naugle et al. 2000), each pair of sites was designated to a block based on these features. Blocking potentially helps to remove the effect of geographic location and surrounding landscape on birds. The paired sites of Block 1 on the east side of the study area (Figure 2) were within the riparian corridor of the Cannon River and had woody vegetation on and adjacent to them. The riparian corridor was surrounded by a largely agricultural landscape. The paired sites of Block 2 on the west side of the study area (Figure 2) were each embedded in a larger wetland that was adjacent to wooded bluffs in the Minnesota River Valley. The sedge wetland of Block 2 was located 2.5 kilometers from the city of St. Peter, and the invaded wetland was located on the south edge of the city of Mankato. The paired sites of Block 3 were within the riparian corridor of the Minnesota River (Figure 2) and differed from Block 2 in that they were adjacent to cropland as well as a mix of upland habitats and riparian forest. Both sites were within 1.6 kilometers of the village of Judson. The paired sites of Block 4 on the east side of the study area (Figure 2) had woody vegetation on and adjacent to them and were surrounded by an agricultural landscape. None of the study sites had experienced active management, such as prescribed burning, haying, or seeding, at least three years prior to the study. Thus, differences in bird communities and reproductive success between habitat types were assumed to be attributed to differences in the dominant plant communities.

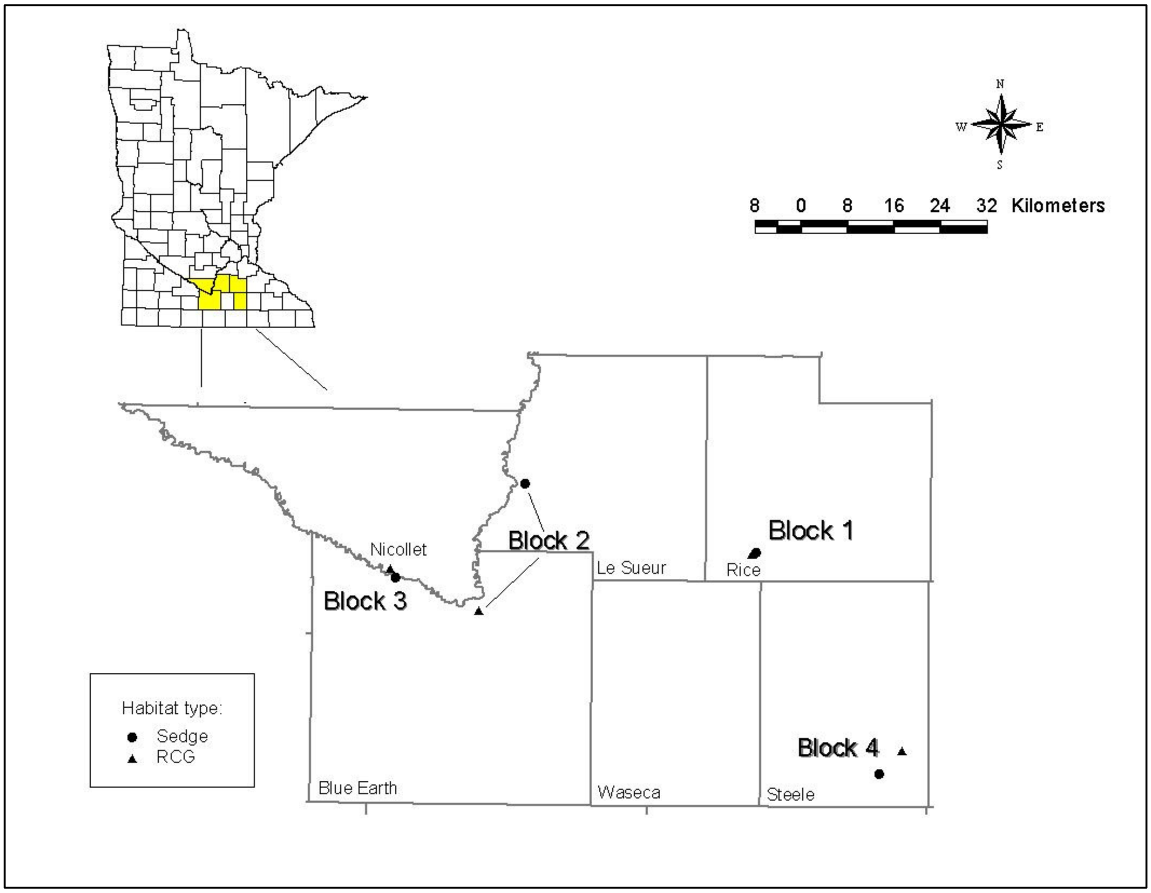

Figure 2. Location of blocks in five southern Minnesota counties used to study the differences in vegetative structure, bird communities, and nesting success in four sedge wetlands paired with four wetlands invaded by reed canarygrass (*Phalaris arundinacea*) during 2006 and 2007.

Structure and Composition of Vegetation

I assessed characteristics of the plant community to determine if the vegetative structure of my study sites was similar to other wetlands invaded by *P. arundinacea* (Green and Galatowitsch 2002, Lindig-Cisneros and Zedler 2002, Maurer and Zedler 2002, Maurer et al. 2003, Kercher et al. 2004, Schooler et al. 2006) and to determine if differences existed in vegetation between habitat types, so that I could in turn determine if the structural differences affected the bird community. To facilitate sampling of vegetation, I used ArcMap 9.1 Geographic Information System (ESRI 2006) to establish a grid system of reference stakes located at 100-meter intervals across each study site. Vegetation plots were located 10 meters from each reference stake along a random compass bearing that was within the study area. The number of vegetation plots varied from site to site and was determined by the area of each wetland (n = 4-11 plots). I measured the physical structure of vegetation once in June and once in July 2007 on each study site in order to determine if vegetative structure varied within and between habitat types during the growing season. I measured vegetation on paired sites in the same week or in consecutive weeks to minimize temporal bias and used the same sampling points at each site in June and July.

I recorded visual obstruction readings (VOR) at each point as well as maximum height of live vegetation and litter depth. I used a 17-decimeter Robel pole graduated in one-decimeter intervals. At each sampling point, I viewed the Robel pole from the four cardinal directions at a height of one meter and a distance of four meters and recorded the first visible interval for each quadrant (Robel et al. 1970). I recorded maximum height of live vegetation within one meter of the Robel pole in each cardinal direction and measured litter depth by lowering a ruler through the litter layer on the north side of the base of the Robel pole at each point. Lastly, I recorded the number of woody stems within a 10-meter radius circle around the point in two size classes—$<$ two meters and $>$ two meters. I recorded the number of woody stems only once, in June 2007. I averaged the VOR and height measurements at each sampling point and obtained an overall mean for each measurement at each site. I also averaged the litter depths to arrive at an overall mean and calculated an overall mean number of woody stems/100 m^2 in each size class for each site.

I calculated horizontal heterogeneity of vegetation from the VOR using a formula developed by Wiens (1974). For horizontal heterogeneity, the index for a sample unit (one sampling point) is Max-Min, which is defined as the maximum minus the minimum visual obstruction reading at that point. For the overall study site, the index is calculated as $\sum$ (Max-Min)/$\sum \bar{x}$.

To determine plant species richness and diversity, I conducted plant inventories at each site once during the 2007 growing season. I used a stratified-systematic design to establish a series of randomly-located transects at each site. The number of transects varied from site to site and was determined by the area of each wetland (n = 14-40 plots). Each transect was 100 meters long, and I sampled vegetation in plots located at 20-meter intervals along each transect. I used hybrid Daubenmire-Releve methodology to record plant species composition and estimate absolute cover of each species within a 1-meter^2 rectangular quadrat (Mueller-Dombois and Ellenberg 1974). I used the absolute coverage per species at each wetland to calculate relative abundance for each species by site. I used this data to calculate plant species diversity via the Shannon-Wiener diversity index. I also calculated plant species richness as the number of species recorded on a site.

To assess composition of the plant community, I compared mean percent cover of individual species and frequency and relative abundance of *P. arundinacea* between habitat types with a two-tailed, paired t-test. In addition, I calculated the percent cover rare or listed species contributed to the community of each habitat type (Minnesota Department of Natural Resources 2007) and the percent composition of graminoids and forbs by habitat type.

I calculated beta-diversity as percent similarity of plant communities between habitat types for summer 2007. I first calculated relative abundance of each species for each habitat type as a percentage. I then added the lowest percentage for each species the habitat types had in common to arrive at the percent similarity.

I used the General Linear Model (GLM) procedure in SPSS to determine if differences existed in vegetative structure among blocks, months, and habitat types (SPSS Inc. 2009). I included block in the model to account for differences that existed among paired sites in addition to the month the measurements were taken (June or July). My third independent variable was habitat type, referring to sedge and *P. arundinacea* sites. Because I was mainly interested in the differences in vegetative structure between habitat types, my model for the dependent variables VOR, maximum height, litter depth, and horizontal heterogeneity was Y = block + month + habitat type + month × habitat type. For the remaining dependent variables (number of woody stems/100 m^2, species richness, diversity) my model was Y = block + habitat type because these measurements were taken only once in 2007. I conducted all statistical tests using a significance level of $P \leq 0.05$. I used Tukey's post-hoc test to compare differences among blocks and one-tailed, paired t-tests to compare significant interactions.

Bird Community

I established survey points at each site to sample the bird community. I randomly selected survey points from the same grid system used to measure vegetation that were located at least 200 meters apart to minimize the likelihood of counting birds twice (Reynolds et al. 1980). All blocks had two survey points per site, except Block 4 had one survey point per site because the sites were small and could not accommodate two points. The edge of each plot was located ≥ 25 meters from the nearest habitat transition when possible to reduce potential bias associated with edges (Arnold and Higgins 1986).

I surveyed birds using the fixed-radius point count technique (Ralph et al. 1995). For this method, I commenced surveys upon arriving at the center of the 50-meter radius plot (Ralph et al. 1995) and conducted surveys for five minutes. During the five minutes at each station, I recorded all birds seen and heard actively utilizing the site (Reynolds et al. 1980), including birds that foraged over the survey plot, such as swallows and raptors (Bryan and Best 1991). Additionally, I counted birds that flew over the survey plot during a survey if they originated or landed within the study site. I also recorded birds that flushed from within a plot as I approached a survey point (Fowler and McGinnes 1973, Reynolds et al. 1980) and birds that flushed upon leaving the survey point that I was certain were within the plot during the survey but were undetected.

I conducted surveys during standard climatic and temporal conditions across multiple seasons. I conducted surveys from sunrise to four hours after sunrise (Fowler and McGinnes 1973, Robbins 1981) on days with little or no precipitation or fog and winds less than 12 mph (North American Breeding Bird Survey 2001). During the 2006 and 2007 breeding seasons, I conducted weekly surveys on all sites from May through mid July. During the non-breeding season, however, I conducted monthly surveys on all sites from August 2006-April 2007 and August-October 2007. Paired sites were surveyed on the same day and the order of points within sites was reversed each survey period to minimize temporal bias. Three observers assisted with surveys during the 2006 and 2007 breeding seasons. We alternated weekly surveys on paired sites between observers to minimize observer bias (Bibby et al 2000).

I calculated species richness, diversity, and relative abundance of birds for each habitat type across seasons and years. Because detectability and density of birds varies by season due to changes in behavior and habitat (Dawson 1981), my methods were slightly different for the breeding seasons and the non-breeding season. During the breeding season, I summed the greatest number of individuals of each species recorded at each survey point within a site on any one day. During the non-breeding season, I used the total number of individuals of each species recorded at each site (Dawson 1981). I used these numbers to calculate relative abundance of each species and bird species diversity via Simpson’s Reciprocal Index. I calculated bird species richness as the number of species recorded on a site each season.

To assess composition of the bird community, I compared relative abundance of species between habitat types with a two-tailed, paired t-test. Because composition of the bird community can be an indication of habitat quality (ie. composition and structure of vegetation) and anthropogenic disturbance (Benoit and Askins 1999, Browder et al. 2002), I calculated the percent composition that species of greatest conservation need (SGCN; Minnesota Department of Natural Resources 2006) collectively contributed to the community of each habitat type. Species of greatest conservation need are species that are rare, declining, or vulnerable in Minnesota. They include federal and/or state listed species (endangered, threatened, or of special concern) or have been identified as experiencing significant population declines largely due to habitat loss and degradation both within and outside of Minnesota (Minnesota Department of Natural Resources 2006).

I calculated beta-diversity as percent similarity of bird communities between habitat types for the 2006 and 2007 breeding seasons and for the non-breeding season. I first calculated relative abundance of each species for each habitat type as a percentage. I then added the lowest percentage for each species the habitat types had in common to arrive at the percent similarity.

I used the Repeated Measures GLM procedure in SPSS to compare species richness and diversity of breeding birds between habitat types and the GLM to compare species richness and diversity of non-breeding birds (SPSS Inc. 2009). Year was the repeated measure in the Repeated Measures GLM, and block and habitat type were the independent variables. Because I was mainly interested in the differences in avian communities between habitat types, my model for each dependent variable (species richness and diversity) in the GLM was Y = year + block + habitat type. Due to small sample sizes, I combined data across years for the non-breeding seasons and my model was Y = block + habitat type. I included block in the models to account for differences that existed among paired sites, and habitat type, referring to sedge and *P. arundinacea* sites. I conducted all statistical tests using a significance level of $P \leq 0.05$ and used Tukey's post-hoc test to compare differences among blocks.

Nesting Success

In order to assess nesting success, I searched for and monitored nests within the same grid system used to measure vegetation and survey birds. In 2006, I searched for and monitored nests of all species on all sites. Red-winged Blackbird (*Agelaius phoeniceus*) nests were the most numerous found in 2006, and in 2007 I focused my search efforts solely on Red-winged Blackbirds. However, because I did not find any Red-winged nests in the wetland invaded by *P. arundinacea* in Block 4 in 2006, I omitted this wetland and its paired sedge wetland from nesting analysis. My assessment of nesting success is based only on Red-winged Blackbirds.

I conducted nest searches from mid May through late July in 2006 and 2007 by using a sweeping stick to flush adult birds off nests and by observing adults building nests or feeding young (Martin and Geupel 1993, Ralph et al. 1993). I held my search effort constant across habitat types to minimize bias in comparisons of nesting success and nest density arising from differential sampling of nests. I marked nests with pin flags in alternating distances of five or eight meters north of each nest to minimize the risk of attracting predators (Picozzi 1975) and to aid in relocating nests during monitoring. I also placed a small piece of flagging tape 20 centimeters-one meter south of each nest. I referenced the location of each nest to the nearest stake in the grid system. I monitored nests every three-five days until the nestlings fledged or the nest failed (Martin and Geupel 1993, Ralph et al. 1993).

Data collected on each nest included species, nest ID (year-observer's initials-nest number), site, date and time found, distance from pin flag, direction and distance from nearest reference stake, observer, nest stage (nest building, incubation, or nestling), nest substrate, nest height, number of eggs/nestlings, and parent location relative to the nest (on, close, or absent) (Martin and Geupel 1993, Ralph et al. 1993). I also recorded the incidence of Brown-headed Cowbird (*Molothrus ater*) parasitism and the number of cowbird eggs and nestlings (Martin and Geupel 1993, Ralph et al. 1993) to help determine fate of the nest. Data recorded during each revisit included date and time, observer, nest stage, number of eggs or nestlings, parent location, condition of the nest when it was found empty, and nest fate on the last visit (Martin and Geupel 1993, Ralph et al. 1993).

I used standard criteria to help me determine nest fate. I determined a nest was successful if $\geq$ one host nestling fledged. Evidence of a successful nest included a flattened nest rim, feces in or around the nest, feather sheaths in the nest, continuous chipping from the parents, parents carrying food, or a fledgling seen or heard near the nest (Martin and Geupel 1993, Ralph et al. 1993). I considered nestlings to have fledged successfully if I observed them in the nest at seven-eight days of age but were absent at the subsequent visit, and I found no evidence of mortality (Camp and Best 1994). I determined that a nest failed if I found the eggs missing or the nest empty before young reached fledging age, the nest was damaged or disturbed, eggshell fragments remained in the nest, some or all eggs were present but cold and the parents were absent on two consecutive visits (abandoned), or all nestlings were dead in the nest (starved/abandoned) (Martin and Geupel 1993, Ralph et al. 1993, Camp and Best 1994, Greenwood and Sargeant 1995). I used the midpoint from the time a nest was last checked to when it was found empty to calculate the termination date (Mayfield 1961).

I assessed nesting success for Red-winged Blackbirds using the Mayfield method (Mayfield 1961, Mayfield 1975). I first calculated daily survival rates (DSR) by site for the egg-laying, incubation, and nestling stages. Daily survival rates among nesting stages within sites were not different as determined by a two-tailed, paired t-test, so I combined these probabilities in calculating nest success for each site. I computed the percentage of successful nests from the DSR by raising the DSR to the power of the number of days of the nesting cycle. I used three days for egg-laying, 11 days for incubation, and 10 days for the nestling stage (Ehrlich et al. 1988, Yasukawa and Searcy 1995). I multiplied the product of the three DSR by the probability eggs would hatch to arrive at the overall percent nesting success by site (Mayfield 1961, Mayfield 1975).

I compared density of nests/10 hectares and percent nesting success between habitat types with the GLM procedure in SPSS (SPSS Inc. 2009). Because I was mainly interested in the differences in nesting success between habitat types, my model for the dependent variables was Y = block + habitat type. I included block in the model to account for differences that existed among paired sites, and habitat type, referring to sedge and *P. arundinacea* sites. I combined data between years due to small sample sizes and, therefore, did not include year in the model. I used a significance level of $P \leq 0.05$ and Tukey's post-hoc test to compare differences among blocks.

I tested factors that may affect nesting success using a logistic regression model in SPSS (SPSS Inc. 2009) that included block, habitat type, stage found (eggs or nestlings), and Julian date found. My model for the dependent variable (nest fate) was Y = block + habitat type + stage found + Julian date found. I included stage found and Julian date found because they are important nest-survival covariates that may influence success rates of nests. For instance, nests found during the nestling stage and nests found earlier in the season may have higher success rates (Mayfield 1975). I conducted the statistical test using a significance level of $P \leq 0.05$.

CHAPTER 3

RESULTS

Structure and Composition of Vegetation

Visual obstruction readings for the 2007 growing season exhibited mixed results for the main effects and the interaction. Visual obstruction readings were different among blocks (F = 19.312, d.f. = 3, P < 0.001). Block 3 had the lowest VOR among blocks, and Block 2 had lower VOR than Block 4 (Table 2). Additionally, VOR differed between habitat types (F = 11.243, d.f. = 1, P = 0.001). Wetlands invaded by *P. arundinacea* had greater VOR (10.39 ± 0.39) than sedge wetlands (7.99 ± 0.61; Figure 3). However, VOR were not different between the June and July measurements (F = 0.228, d.f. = 1, P = 0.634; Table 3) or in the habitat type × month interaction (F = 1.576, d.f. = 1, P = 0.212; Table 4, Figure 4).

Maximum height of live vegetation for the 2007 growing season also exhibited mixed results for the main effects and the interaction. Maximum height was different among blocks (F = 12.998, d.f. = 3, P < 0.001), as Block 3 exhibited the shortest vegetation (Table 2). Height also differed between habitat types (F = 15.118, d.f. = 1, P < 0.001). Wetlands invaded by *P. arundinacea* had taller vegetation (13.96 ± 0.39) than sedge wetlands (11.28 ± 0.56; Figure 5). Maximum height was not different between the June and July measurements (F = 0.001, d.f. = 1, P = 0.980; Table 3). However, the habitat type × month interaction was significant (F = 7.239, d.f. = 1, P = 0.008). Therefore, I conducted one-tailed, paired t-tests between habitat types for each month and

Table 2. Parameters of vegetative structure (mean ± SE) in four sedge wetlands paired with four wetlands invaded by reed canarygrass (*Phalaris arundinacea*) (blocks) in southern Minnesota during the 2007 growing season.

	Block			
Parameter	**1**	**2**	**3**	**4**
VOR (dm)[b]	10.90 ± 0.50 a[a]	9.09 ± 0.47 ab	5.93 ± 0.74 c	12.74 ± 0.82 ad
Vegetation height (dm)[c]	13.54 ± 0.50 a	13.52 ± 0.55 a	9.63 ± 0.73 b	14.99 ± 0.73 a
Litter depth (cm)[d]	8.64 ± 1.08 a	10.47 ± 1.60 a	6.63 ± 0.66 a	10.63 ± 1.81 a
No. woody stems < 2 m[e]	14.24 ± 4.29 a	1.10 ± 0.77 ab	21.10 ± 8.55 ac	11.82 ± 4.00 a
No. woody stems > 2 m[f]	3.24 ± 0.93 a	0.02 ± 0.02 ab	0.87 ± 0.72 a	4.46 ± 2.78 ac
Horizontal heterogeneity[g]	0.33 ± 0.03 a	0.36 ± 0.06 a	0.43 ± 0.07 a	0.31 ± 0.05 a
Plant species richness	40.00 ± 3.00 a	38.00 ± 4.00 a	36.50 ± 7.50 a	27.50 ± 18.50 a
Plant species diversity[h]	2.17 ± 0.06 a	2.48 ± 0.47 a	2.43 ± 0.24 a	1.67 ± 1.10 a

[a] According to Tukey's post-hoc test, means sharing the same letter are not different ($P \leq 0.05$).
[b] Visual obstruction reading of vegetation in decimeters
[c] Maximum height of live vegetation in decimeters
[d] Litter depth in centimeters
[e] Number of woody stems/100 m^2 that are < 2 meters tall
[f] Number of woody stems/100 m^2 that are > 2 meters tall
[g] Horizontal heterogeneity of vegetation calculated via Wiens Index (Wiens 1974)
[h] Plant species diversity calculated via Shannon-Wiener diversity index

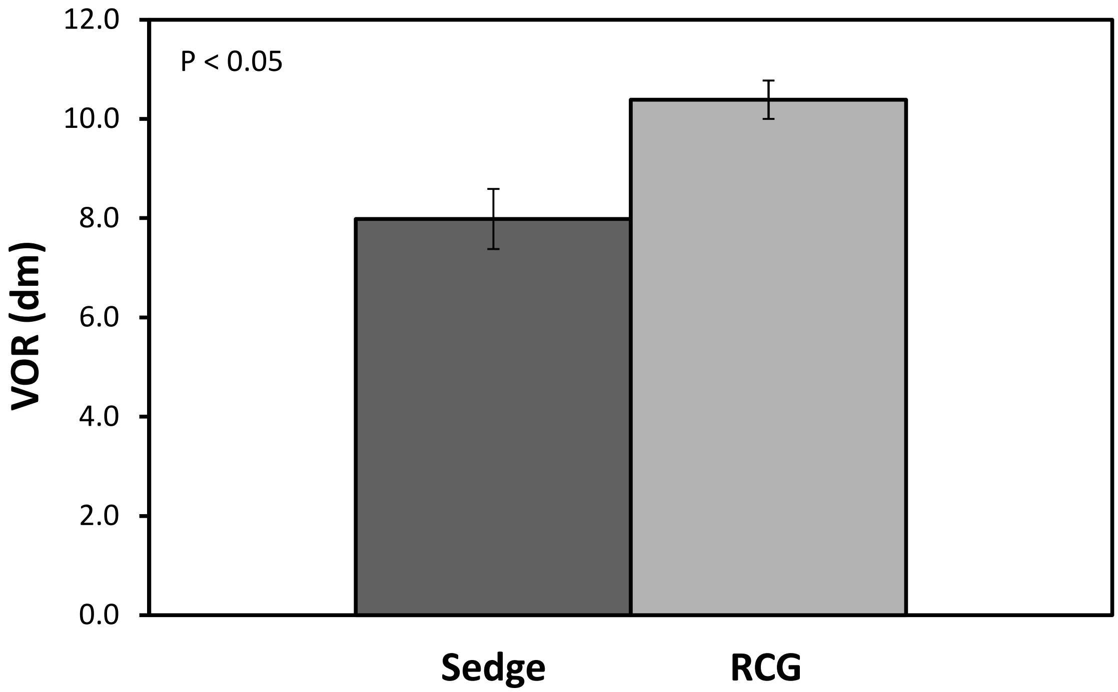

Figure 3. Mean visual obstruction reading (VOR) of vegetation in decimeters (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

Table 3. Parameters of vegetative structure (mean ± SE) in southern Minnesota wetlands during June and July 2007.

	Month	
Parameter	**June**	**July**
VOR (dm)[b]	8.94 ± 0.52 a[a]	9.33 ± 0.56 a
Vegetation height (dm)[c]	12.50 ± 0.53 a	12.62 ± 0.50 a
Litter depth (cm)[d]	9.15 ± 0.79 a	8.53 ± 1.03 a
Horizontal heterogeneity[e]	0.33 ± 0.02 a	0.39 ± 0.05 a

[a] According to Tukey's post-hoc test, means sharing the same letter are not different ($P \leq 0.05$).
[b] Visual obstruction reading of vegetation in decimeters
[c] Maximum height of live vegetation in decimeters
[d] Litter depth in centimeters
[e] Horizontal heterogeneity of vegetation calculated via Wiens Index (Wiens 1974)

Table 4. Parameters of vegetative structure (mean + SE) for sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during June and July 2007.

	Habitat type	
Parameter and Month	**Sedge**	**RCG**
VOR (dm)[b]		
June	7.43 ± 0.77 a[a]	10.62 ± 0.55 a
July	8.57 ± 0.95 a	10.15 ± 0.55 a
Vegetation height (dm)[c]		
June	10.46 ± 0.72 a	14.77 ± 0.55 b
July	12.12 ± 0.84 ab	13.16 ± 0.51 ab
Litter depth (cm)[d]		
June	8.39 ± 1.28 ab	10.00 ± 0.86 a
July	10.40 ± 1.87 ab	6.52 ± 0.61 b
Horizontal heterogeneity[e]		
June	0.30 ± 0.04 a	0.35 ± 0.02 a
July	0.40 ± 0.09 a	0.39 ± 0.06 a

[a] According to one-tailed, paired t-tests, means for each parameter sharing the same letter across habitat types and months are not different ($P \leq 0.05$).
[b] Visual obstruction reading of vegetation in decimeters
[c] Maximum height of live vegetation in decimeters
[d] Litter depth in centimeters
[e] Horizontal heterogeneity of vegetation calculated via Wiens Index (Wiens 1974)

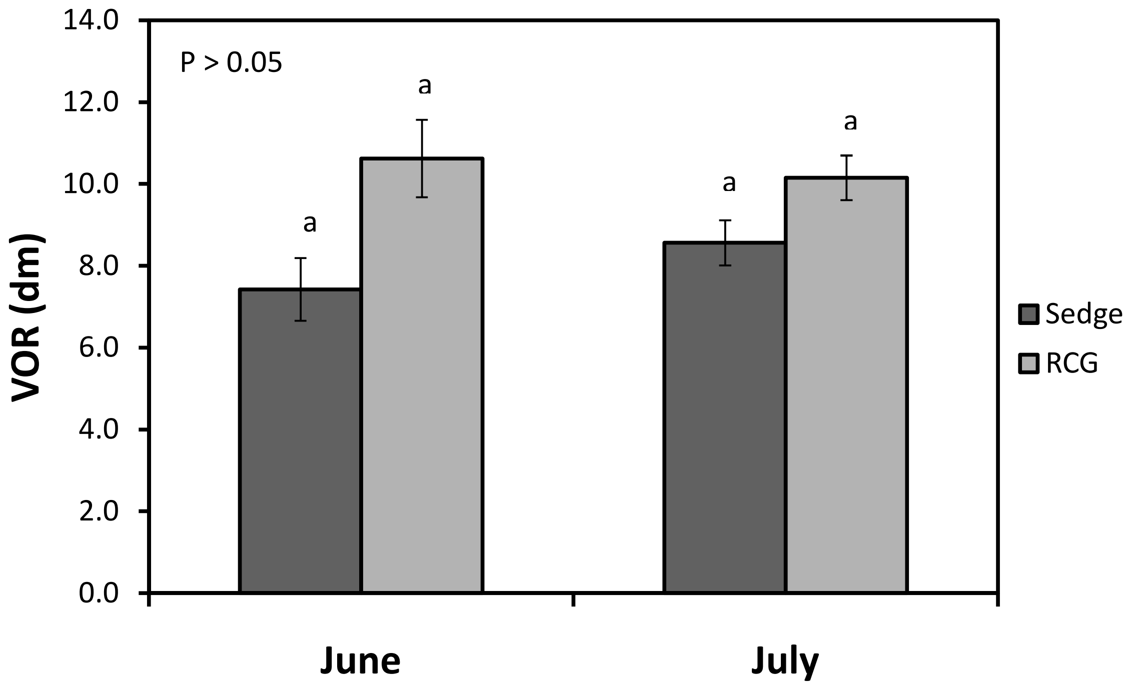

Figure 4. Mean visual obstruction reading (VOR) of vegetation in decimeters (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during June and July 2007. Bars sharing the same letter are not different.

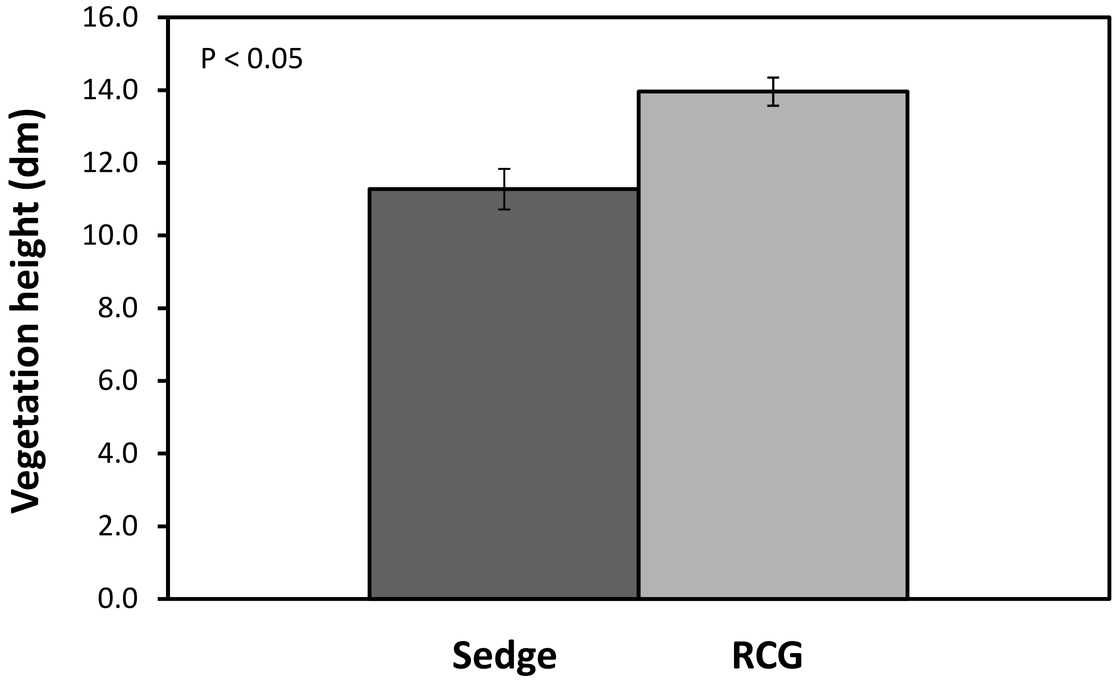

Figure 5. Mean maximum height of live vegetation in decimeters (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

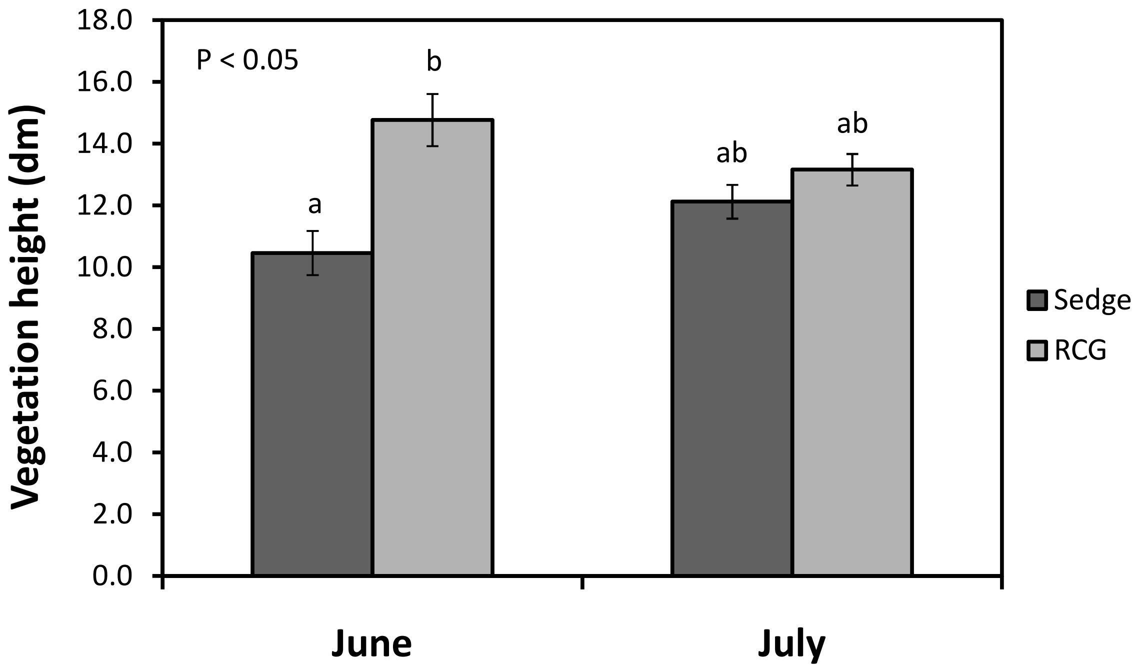

Figure 6. Mean maximum height of live vegetation in decimeters (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during June and July 2007. Bars sharing the same letter are not different.

between months for each habitat type. Wetlands invaded by *P. arundinacea* had taller vegetation than sedge wetlands in June (P = 0.039; Table 4, Figure 6).

Litter depth for the 2007 growing season was similar among the main effects but exhibited a difference in the interaction. Litter depth was not different among blocks (F = 2.424, d.f. = 3, P = 0.07; Table 2) or between habitat types (F = 1.355, d.f. = 1, P = 0.247), with a mean litter depth of 9.38 ± 1.12 centimeters for sedge wetlands and 8.26 ± 0.57 centimeters for wetlands invaded by *P. arundinacea* (Figure 7). Litter depth also was not different between the June and July measurements (F = 0.376, d.f. = 1, P = 0.541; Table 3). However, the habitat type × month interaction was significant (F = 4.672, d.f. = 1, P = 0.03). Therefore, I conducted one-tailed, paired t-tests between habitat types for each month and between months for each habitat type. Wetlands invaded by *P. arundinacea* had greater litter depths in June than in July (P = 0.002; Table 4, Figure 8).

The number of woody stems/100 m^2 that were < two meters and > two meters had mixed results for the main effects of block and habitat type. The number of woody stems < two meters tall for the 2007 growing season was not different among blocks (F = 2.178, d.f. = 3, P = 0.101; Table 2) but was different between habitat types (F = 11.774, d.f. = 1, P = 0.001). Sedge wetlands had a greater number of woody stems/100 m^2 that were < two meters tall (21.65 ± 5.21) than wetlands invaded by *P. arundinacea* (1.79 ± 0.73; Figure 9). The number of woody stems/100 m^2 that were > two meters tall was different among blocks (F = 3.671, d.f. = 3, P = 0.018). Block 4 had a greater number of woody stems > two meters than Block 2 (Table 2). The number of woody stems > two meters tall did not differ between habitat types (F = 2.0, d.f. = 3, P = 0.163), as sedge wetlands

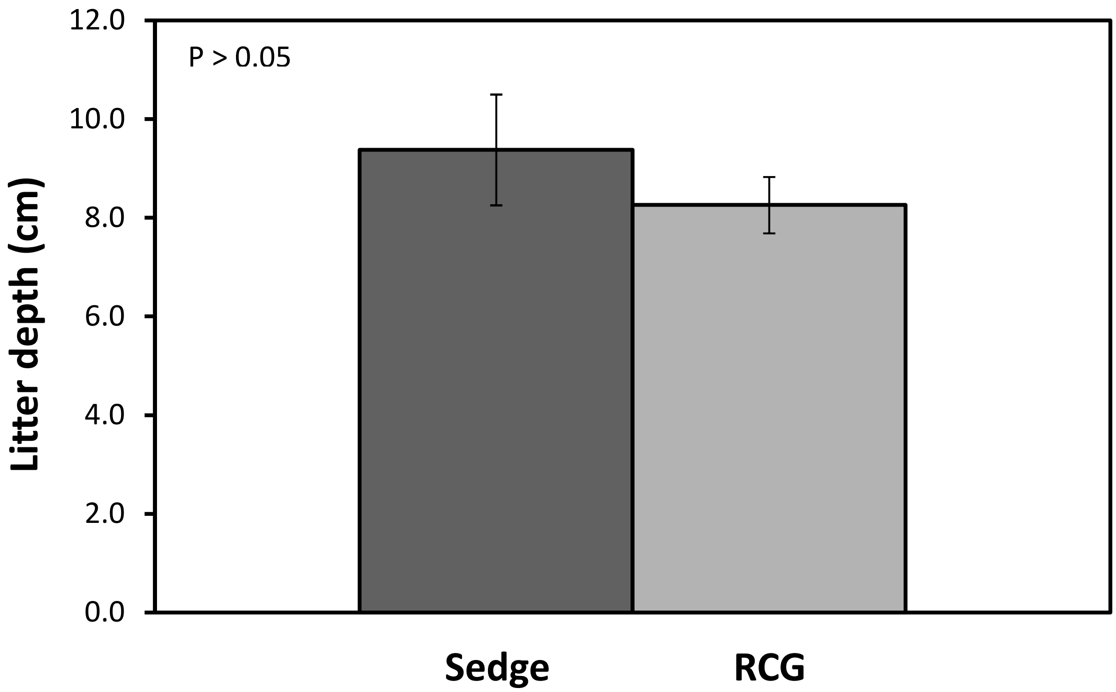

Figure 7. Mean litter depth in centimeters (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

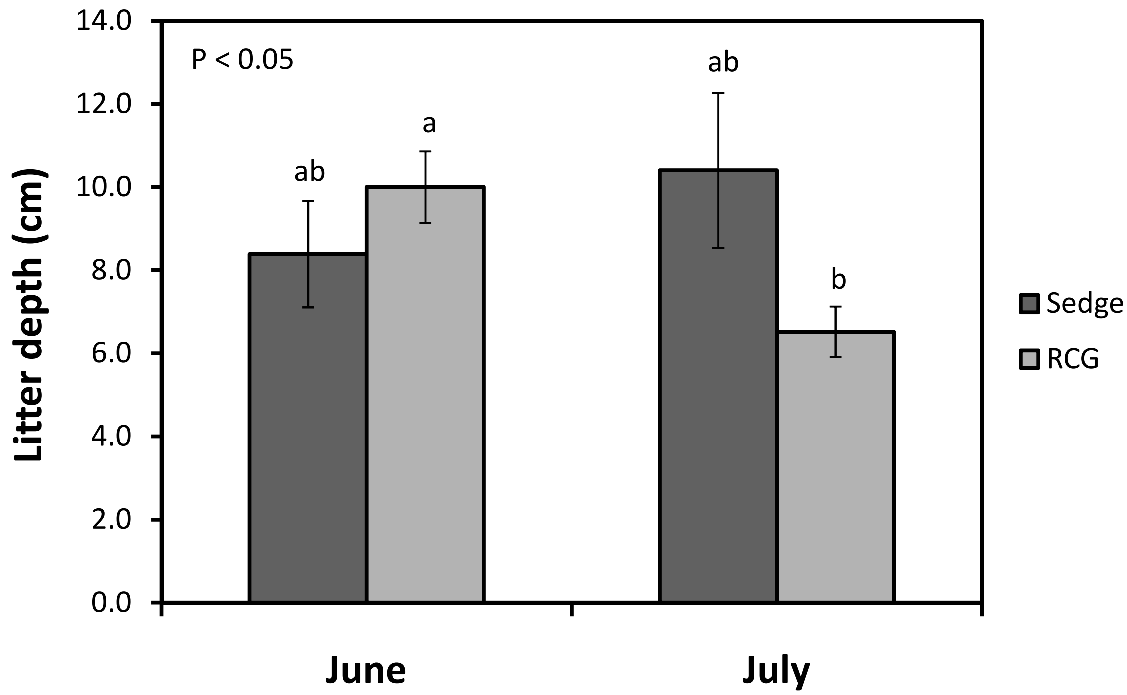

Figure 8. Mean litter depth in centimeters (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during June and July 2007. Bars sharing the same letter are not different.

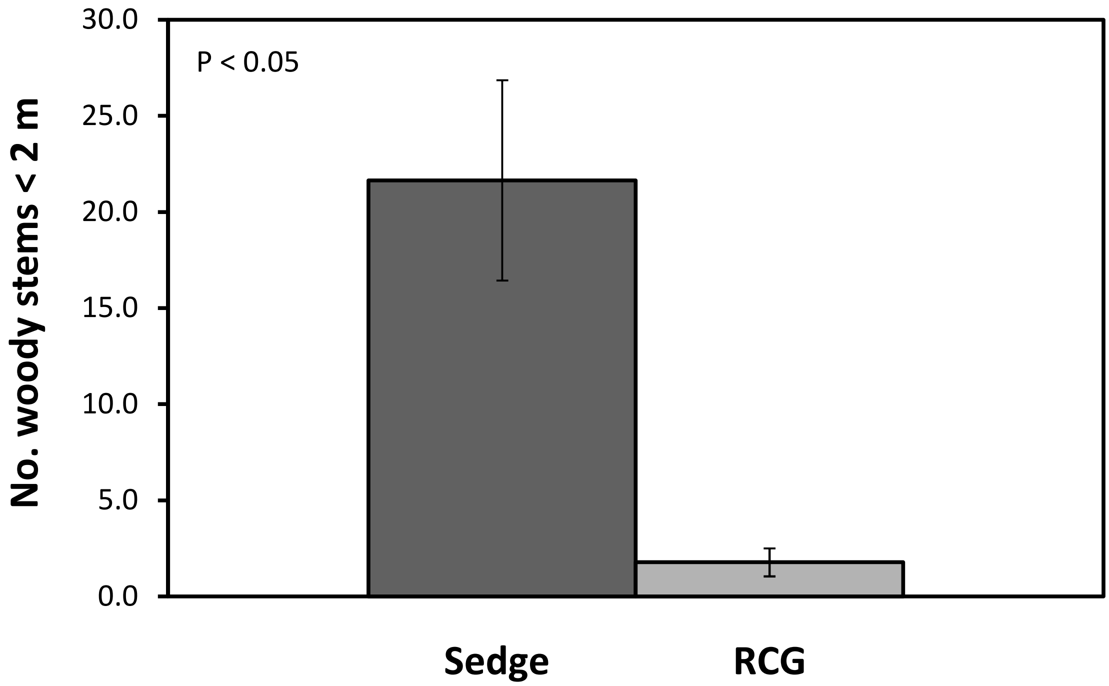

Figure 9. Mean number of woody stems/100 meters^2 that are < two meters tall (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

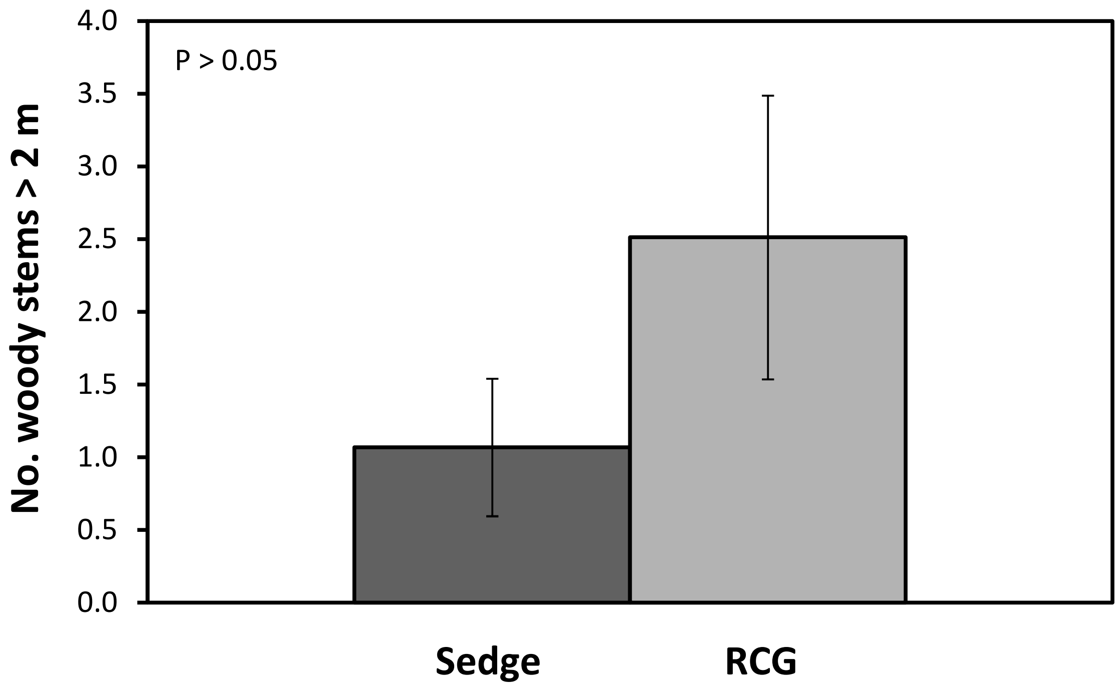

Figure 10. Mean number of woody stems/100 meters^2 that are > two meters tall (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

had a mean of 1.07 ± 0.47 woody stems/100 m^2 and invaded wetlands 2.51 ± 0.98 woody stems/100 m^2 (Figure 10).

Horizontal heterogeneity for the 2007 growing season was similar for all main effects and the interaction. Horizontal heterogeneity was not different among blocks (F = 0.743, d.f. = 3, P = 0.553; Table 2) or between habitat types (F = 0.143, d.f. = 1, P = 0.714), with a mean heterogeneity index of 0.35 ± 0.05 for sedge wetlands and 0.37 ± 0.03 for wetlands invaded by *P. arundinacea* (Figure 11). Horizontal heterogeneity also was not different between the June and July measurements (F = 1.285, d.f. = 1, P = 0.286; Table 3) or in the habitat type × month interaction (F = 1.576, d.f. = 1, P = 0.212; Table 4, Figure 12).

During summer 2007, I recorded 85 species of plants across habitat types. Eighty species occurred in sedge wetlands and 57 species occurred in wetlands invaded by *P. arundinacea* (Table 5). Sedge wetlands had 27 species with ≥ 1% mean cover. Of these, six species had ≥ 5% mean cover, including *Carex stricta* (27.57%), *P. arundinacea* (13.32%), *C. vulpinoidea* (10.39%), *C. lacustris* (6.97%), *Typha angustifolia* (5.62%), and *Scirpus atrovirens* (5.43%; Table 5). Wetlands invaded by *P. arundinacea* had 19 species with ≥ 1% mean cover. Of these, four species had ≥ 5% mean cover, including *P. arundinacea* (65.08%), *Carex stricta* (12.27%), *C. vulpinoidea* (7.21%), and *Scirpus fluviatilis* (4.96%; Table 5). *Phalaris arundinacea* dominated invaded wetlands, and mean percent cover of *P. arundinacea* was greater in invaded wetlands (65.08%) than in sedge wetlands (13.32%; P = 0.003; Table 5). *Phalaris arundinacea* occurred in 96.07% of the plots in invaded wetlands compared to 54.43% in sedge wetlands (P = 0.04).

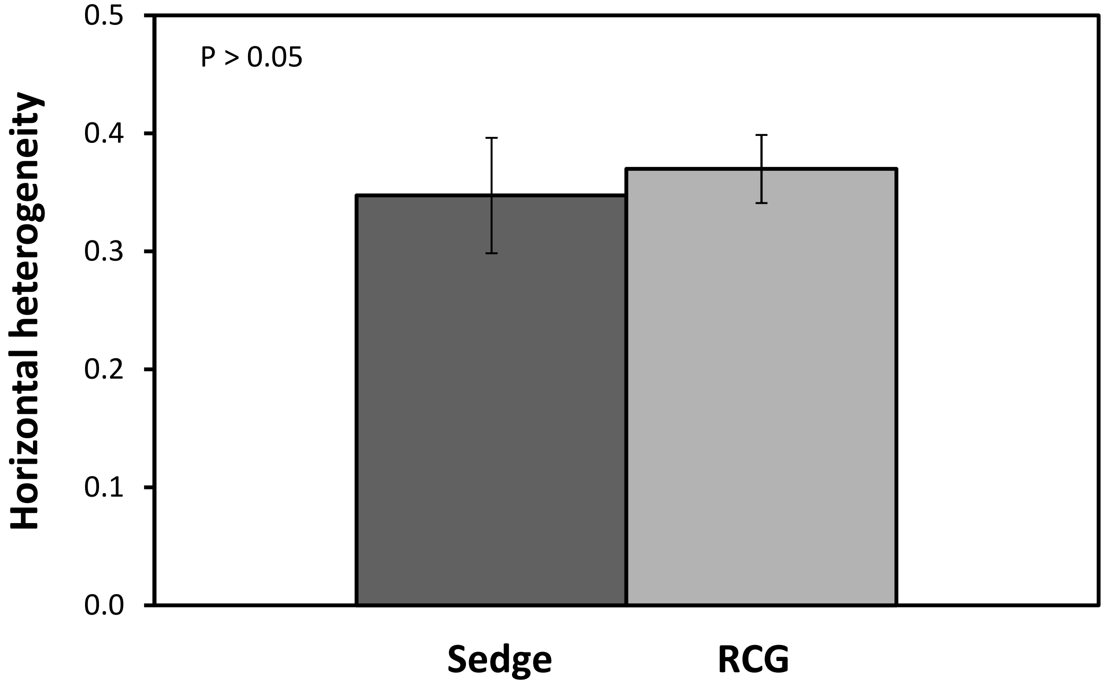

Figure 11. Mean horizontal heterogeneity (± SE) of vegetation calculated via Wiens Index (Wiens 1974) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

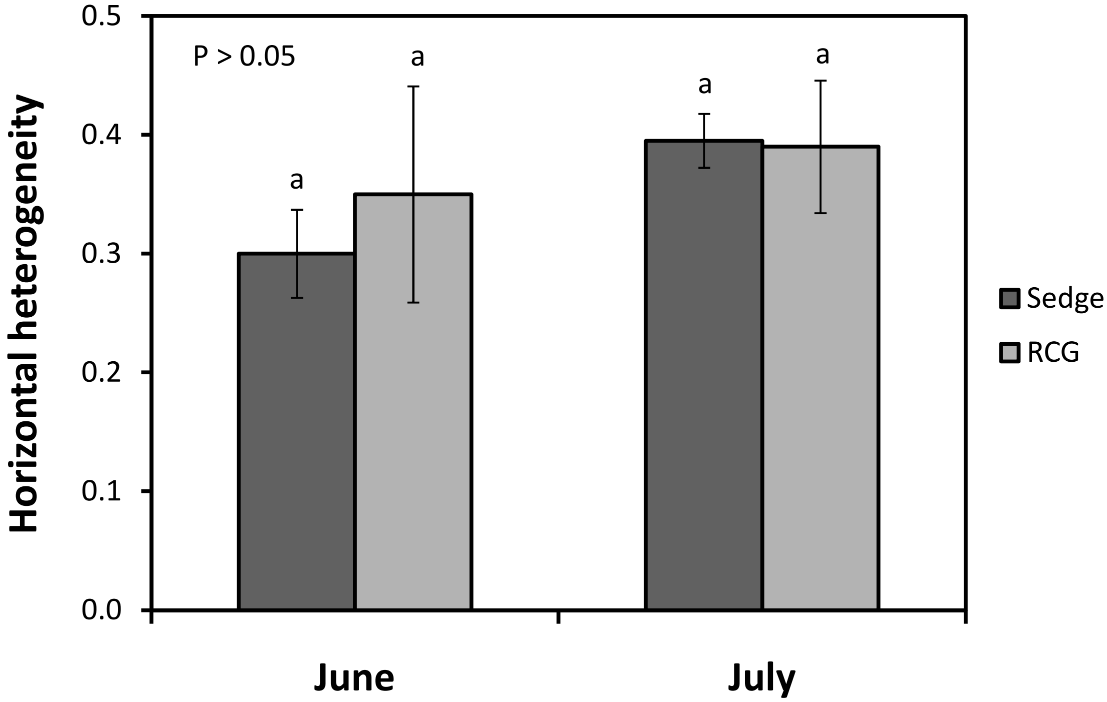

Figure 12. Mean horizontal heterogeneity (± SE) of vegetation calculated via Wiens Index (Wiens 1974) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during June and July 2007. Bars sharing the same letter are not different.

Table 5. Mean percent (%) cover of plants (± SE) in sedge wetlands paired with wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during summer 2007.

Species		Sedge	RCG
Scientific name	**Common name**	**%**	**%**
Ambrosia artemisifolia	Common Ragweed	0.02 ± 0.02*	0.00 ± 0.00
Apocynum sibiricum	Prairie Dogbane	0.49 ± 0.49*	0.00 ± 0.00
Asclepias sullivantii	**Sullivant's Milkweed**	0.24 ± 0.10	0.02 ± 0.02*
Asclepias syriaca	Common Milkweed	0.52 ± 0.32	0.16 ± 0.09
Asclepias verticillata	Narrow-leaved Milkweed	0.40 ± 0.26	0.56 ± 0.49
Aster lucidulus	Swamp Aster	0.44 ± 0.24	0.07 ± 0.07*
Aster puniceus	Purple Stemmed Aster	0.40 ± 0.13	0.37 ± 0.35
Aster simplex	Marsh Aster	1.37 ± 1.02	0.02 ± 0.02*
Calamagrostis canadensis	Canada Bluejoint	0.24 ± 0.24*	0.16 ± 0.16*
Caltha palustris	Marsh Marigold	2.54 ± 1.53	0.75 ± 0.54
Cardamine rhomboidea	Spring Cress	0.12 ± 0.12*	0.00 ± 0.00
Carex aquatilis	Water Sedge	0.38 ± 0.38*	0.00 ± 0.00
Carex hysternica	Porcupine Sedge	2.95 ± 1.92	1.35 ± 0.64
Carex lacustris	Lake Sedge	6.97 ± 6.97*	0.00 ± 0.00
Carex rostrata	Beaked Sedge	1.06 ± 0.97	0.95 ± 0.88
Carex sterilis	**Sterile Sedge**	4.44 ± 2.10	1.12 ± 0.88
Carex stricta	Tussock Sedge	27.57 ± 8.47	12.27 ± 8.12
Carex vulpinoidea	Fox Sedge	10.39 ± 4.56	7.21 ± 4.89
Chenopodium album	Goosefoot	0.05 ± 0.05*	0.03 ± 0.03*
Cirsium discolor	Field Thistle	0.35 ± 0.29	0.07 ± 0.07*
Cirsium muticum	Swamp Thistle	0.38 ± 0.19	1.13 ± 0.76
Cirsium vulgare	Bull Thistle	0.26 ± 0.14	0.11 ± 0.06
Conzya canadensis	Horseweed	0.51 ± 0.51*	0.00 ± 0.00
Cryptotaenia canadensis	Honewort	0.18 ± 0.18*	0.00 ± 0.00
Daucus carota	Queen Anne's Lace	1.14 ± 0.96	0.00 ± 0.00
Eleocharis rostella	Beaked Spike-rush	2.12 ± 2.12*	0.71 ± 0.71*
Equisetum arvense	Common Horsetail	0.67 ± 0.17	0.84 ± 0.45
Equisetum palustre	Marsh Horsetail	0.18 ± 0.18*	0.00 ± 0.00
Erigeron annuus	Daisy Fleabane	0.03 ± 0.03*	0.16 ± 0.16*
Eupatorium maculatum	Joe-Pye Weed	1.29 ± 0.43	0.55 ± 0.34
Eupatorium perfoliatum	Boneset	0.40 ± 0.16	0.19 ± 0.14
Galium boreale	Northern Bedstraw	0.15 ± 0.15*	0.03 ± 0.02

Table 5. *continued*

Galium triflorum	Fragrant Bedstraw	0.15 ± 0.15*	0.00 ± 0.00
Glyceria grandis	Manna Grass	1.46 ± 1.26	0.16 ± 0.16*
Helenium autumnale	Sneezeweed	1.14 ± 1.04	0.00 ± 0.00
Helianthus grosseserratus	Sawtooth Sunflower	1.31 ± 0.86	0.51 ± 0.50
Helianthus maximilliani	Maximillian's Sunflower	0.28 ± 0.22	0.00 ± 0.00
Herlacleum maximum	Cow Parsnip	0.00 ± 0.00	0.03 ± 0.03*
Hesperis matronalis	Dame's Rocket	0.00 ± 0.00	0.01 ± 0.01*
Hierochloe odorata	Sweetgrass	0.11 ± 0.11*	0.20 ± 0.20*
Hydrophyllum virginianum	Virginia Waterleaf	0.06 ± 0.06*	0.00 ± 0.00
Impatiens capensis	Jewel Weed	3.10 ± 1.29	4.13 ± 1.28
Iris versicolor	Blueflag Iris	0.03 ± 0.03*	0.00 ± 0.00
Juncus effusus	Common Rush	1.09 ± 0.70	0.04 ± 0.04*
Juncus tenuis	Poverty Rush	0.34 ± 0.31	0.00 ± 0.00
Lemna sp.	Duckweed	0.15 ± 0.15*	0.00 ± 0.00
Lepidium virginicum	Poor Man's Pepper	0.05 ± 0.05*	0.00 ± 0.00
Liatris pycnostachya	Meadow Blazing Star	0.15 ± 0.09	0.00 ± 0.00
Lysimachia punctata	Yellow Loosestrife	0.03 ± 0.03*	0.11 ± 0.11*
Melilotus officinales	Yellow Sweetclover	0.19 ± 0.13	0.10 ± 0.09
Onoclea sensibilis	Sensitive Fern	0.05 ± 0.05*	0.04 ± 0.04*
Packera pseudaurea	False Groundsel	0.05 ± 0.05*	0.00 ± 0.00
Parthenocissus cinquefolia	Virginia Creeper	0.12 ± 0.12*	0.04 ± 0.04*
Pedicularis canadensis	Canadian Lousewort	0.06 ± 0.06*	0.00 ± 0.00
Phalaris arundinacea	Reed Canarygrass	13.32 ± 5.01	65.08 ± 7.55†
Phlox pilosa	Prairie Phlox	0.14 ± 0.14*	0.02 ± 0.02*
Phragmites australis	Giant Reed	4.33 ± 2.64	1.88 ± 1.88
Poa pratensis	Kentucky Bluegrass	3.17 ± 3.10	1.11 ± 0.67
Polygonatum biflorum	Solomon's Seal	0.00 ± 0.00	0.02 ± 0.02*
Polygonum amphibium	Water Smartweed	0.00 ± 0.00	0.13 ± 0.13*
Pycnanthemum virginianum	Virginia Mountain Mint	0.66 ± 0.66*	0.00 ± 0.00
Ranunculus bulbosus	Bulbous Buttercup	0.06 ± 0.06*	0.00 ± 0.00
Rumex crispus	Curly Dock	0.13 ± 0.13*	0.07 ± 0.07*
Sagittaria latifolia	Broad-leaved Arrowhead	0.34 ± 0.31	1.82 ± 1.68
Saxifraga pensylvanica	Swamp Saxifrage	0.32 ± 0.23	0.00 ± 0.00
Scirpus atrovirens	Green Bulrush	5.43 ± 2.13	1.51 ± 1.48
Scirpus fluviatilis	River Bulrush	1.60 ± 1.42	4.96 ± 3.83
Scirpus validus	Softstem Bulrush	2.88 ± 2.01	1.08 ± 0.65
Senecio pseudaureus	Ragwort	0.35 ± 0.35*	0.00 ± 0.00

Table 5. *continued*

Solanum dulcamora	Bittersweet Nightshade	0.00 ± 0.00	0.11 ± 0.11*
Solidago altissima	Tall Goldenrod	1.87 ± 1.22	0.84 ± 0.54
Solidago gigantea	Giant Goldenrod	3.12 ± 1.82	2.90 ± 2.03
Solidago ohioensis	Ohio Goldenrod	1.91 ± 1.72	0.20 ± 0.18
Sparganium angustifolium	Narrow-leaved Bur-Reed	0.15 ± 0.15*	0.06 ± 0.06*
Sphagnum	Moss	0.71 ± 0.71*	0.59 ± 0.59*
Thalictrum venulosm	Northern Meadow-rue	0.41 ± 0.19	0.00 ± 0.00
Thelypteris palustris	Marsh Fern	0.46 ± 0.46*	0.00 ± 0.00
Toxicendron radicans	Poison Ivy	0.20 ± 0.10	0.69 ± 0.40
Triglochin palustris	**Marsh Arrow-grass**	0.22 ± 0.22*	0.00 ± 0.00
Typha angustifolia	Narrow-leaved Cattail	5.62 ± 3.48	4.52 ± 1.51
Typha latifolia	Broad-leaved Cattail	0.80 ± 0.31	1.06 ± 0.44
Typha x glauca	Hybrid Cattail	1.30 ± 0.80	0.95 ± 0.91
Urtica dioica	Stinging Nettle	0.05 ± 0.05*	1.79 ± 1.25
Verbascum thapsis	Mullein	0.14 ± 0.08	0.00 ± 0.00
Vicia americana	Purple Vetch	0.24 ± 0.11	0.24 ± 0.13

*Species only found on one site.
†Abundance significantly different between habitat types ($P \leq 0.05$).
Bold font denotes rare and listed species (Minnesota Department of Natural Resources 2007).

Relative abundance of *P. arundinacea* was greater in invaded wetlands (53.98%) than in sedge wetlands (10.30%; P = 0.02).

Plant species richness and diversity were similar among the main effects of block and habitat type. Plant species richness did not differ among blocks (F = 0.605, d.f. = 3, P = 0.655; Table 2) or between habitat types (F = 5.40, d.f. = 1, P = 0.103) during summer 2007, with a mean species richness of 43.75 ± 0.85 for sedge wetlands and 27.25 ± 6.30 for wetlands invaded by *P. arundinacea*. Similarly, plant species diversity in summer 2007 was not different among blocks (F = 0.663, d.f. = 3, P = 0.628; Table 2) or between habitat types (F = 4.222, d.f. = 1, P = 0.132), with a mean diversity index of 2.65 ± 0.15 for sedge wetlands and 1.72 ± 0.39 for wetlands invaded by *P. arundinacea*. Because block had little effect on plant species richness (F = 0.605, d.f. = 3, P = 0.655) and diversity (F = 0.663, d.f. = 3, P = 0.628), I re-analyzed the data with only habitat type as a main effect. Plant species richness was greater in sedge wetlands (F = 6.729, d.f. = 1, P = 0.041; Figure 13), but plant species diversity was not different between habitat types (F = 5.078, d.f. = 1, P = 0.065; Figure 14).

Percent composition of individual species, rare and listed species collectively, and plant functional groups were similar between habitat types. I recorded only three rare and listed species in vegetation plots. All three species occurred in sedge wetlands and two occurred in wetlands invaded by *P. arundinacea* (Table 5). The percent composition of rare and listed species collectively was similar between plant communities of sedge wetlands (4.13%) and wetlands invaded by *P. arundinacea* (0.81%; P = 0.116). Two other listed species, small white lady's slipper (*Cypripedium albidum* Muhl. ex Willd.)

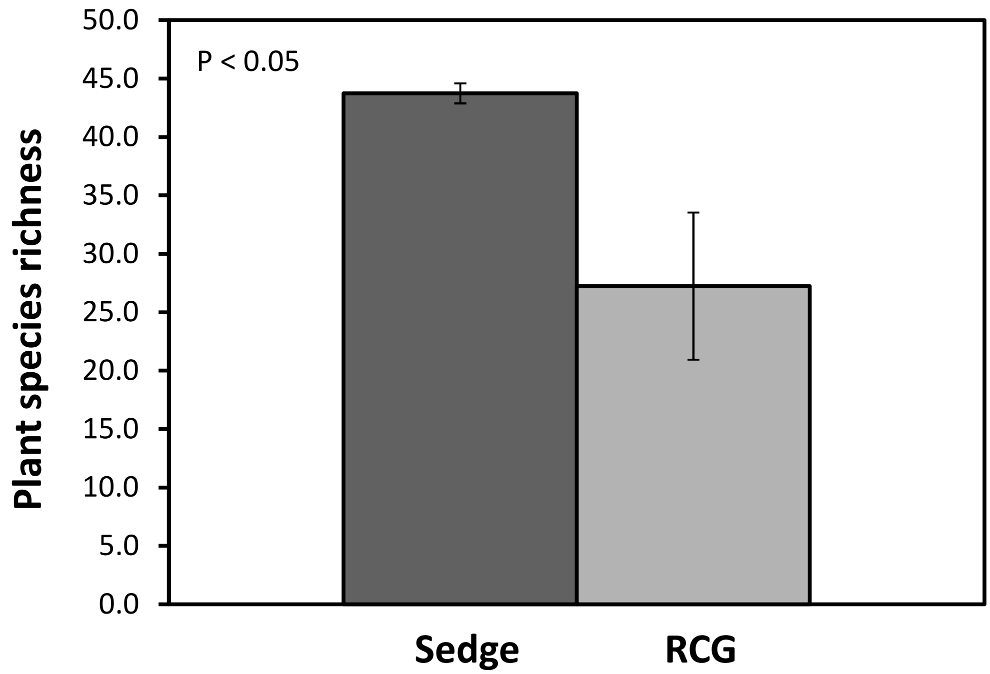

Figure 13. Mean plant species richness (± SE) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

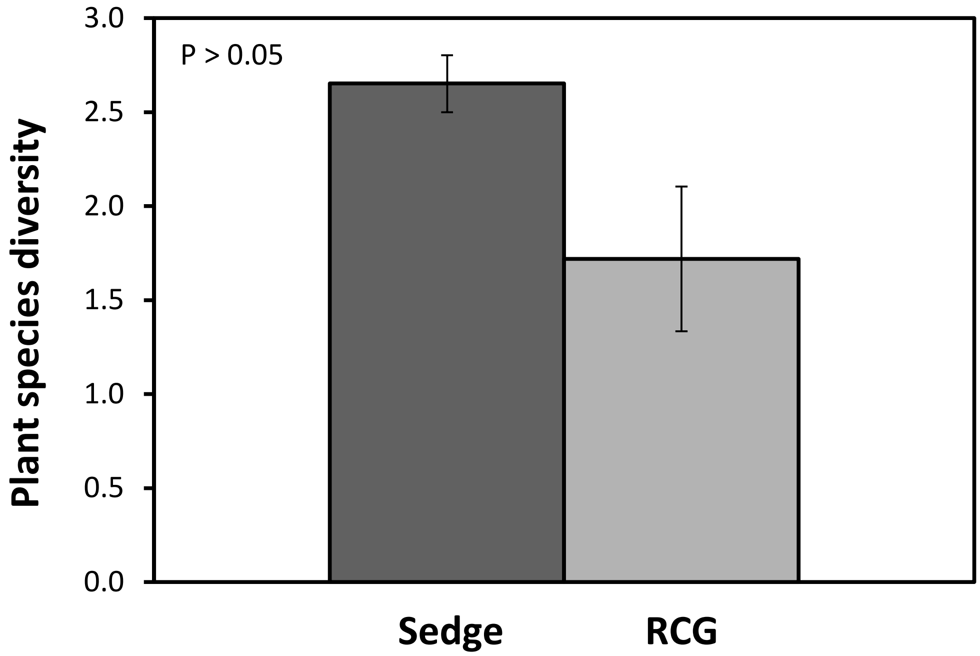

Figure 14. Mean Shannon-Wiener diversity (± SE) of plants in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2007 growing season.

and tuberous Indian-plantain (*Arnoglossum plantagineum* Raf.), were observed outside of vegetation plots in the sedge wetland of Block 3. Other than *P. arundinacea*, mean percent cover of individual plant species between habitat types, including rare and listed species, were not different (P > 0.05; Table 5). Both habitat types were dominated by graminoids. Composition of the plant community for sedge wetlands was 71.18% graminoids and 28.82% forbs. For wetlands dominated by *P. arundinacea*, graminoids comprised 80.33% composition and forbs 19.67%. Percent similarity was 47.8% between habitat types.

Bird Community

During summer 2006, I recorded 41 species of birds across habitat types. Twenty-seven species occurred in sedge wetlands and 37 species occurred in wetlands invaded by *P. arundinacea* (Table 6). The most abundant species in sedge wetlands included Red-winged Blackbird (20.96%), Common Yellowthroat (*Geothlypsis trichas*; 12.65%), Barn Swallow (*Hirundo rustica*; 8.21%), American Goldfinch (*Carduelis tristas*; 6.63%), and Swamp Sparrow (*Melospiza georgiana*; 6.21%; Table 6). Similarly, the most abundant species in wetlands invaded by *P. arundinacea* included Red-winged Blackbird (16.26%), Common Yellowthroat (12.86%), Barn Swallow (6.46%), American Goldfinch (5.68%), and Sedge Wren (*Cistothorus platensis*; 5.61%; Table 6). I recorded ten SGCN during 2006 surveys, seven in sedge wetlands and eight in wetlands invaded by *P. arundinacea* (Table 6). The percent composition of SGCN collectively was similar between bird communities of sedge wetlands (24.12%) and invaded wetlands (14.21%; P = 0.170). Additionally, abundance of individual species between habitat types was not different (P > 0.05; Table 6). Percent similarity of bird communities between habitat types was 72.85%.

For summer 2007, I recorded 52 species of birds, of which 37 occurred in sedge wetlands and 38 occurred in wetlands invaded by *P. arundinacea* (Table 7). The most abundant species in sedge wetlands included Red-winged Blackbird (18.32%), European Starling (*Sturnus vulgaris*; 8.87%), American Goldfinch (8.26%), Common Yellowthroat (8.20%), and American Robin (*Turdus migratorius*; 6.89%; Table 7). The most abundant species in wetlands invaded by *P. arundinacea* included Common Grackle (*Quiscalus quiscula*; 12.25%), Red-winged Blackbird (10.99%), American Goldfinch (9.40%), Common Yellowthroat (9.06%), and Sedge Wren (7.97%; Table 7). I recorded 16 SGCN during 2007 surveys, twelve in sedge wetlands and ten in wetlands invaded by *P. arundinacea* (Table 7). The percent composition of SGCN collectively was similar between bird communities of sedge wetlands (15.32%) and invaded wetlands (17.88%; P = 0.334). Additionally, abundance of individual species between habitat types for the 2007 breeding season was not different (P > 0.05; Table 7). Percent similarity of bird communities between habitat types was 62.45%.

Differences in bird species richness occurred among main effects as opposed to bird species diversity for the 2006 and 2007 breeding seasons. Bird species richness was different among blocks (F = 163.743, d.f. = 3, P = 0.001). Block 1 had the greatest species richness, and Block 3 had greater species richness than Block 2 (Table 8). Species richness also was different between habitat types (F = 37.8, d.f. = 1, P = 0.009). Wetlands invaded by *P. arundinacea* had greater bird species richness (17.50 ± 1.94)

Table 6. Mean percent (%) composition (± SE) of breeding birds in sedge wetlands paired with wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota from May-July 2006.

Species		Sedge			RCG		
Scientific name	**Common name**	**%**			**%**		
Carduelis tristis	American Goldfinch	6.63	±	1.13	5.68	±	0.91
Turdus migratorius	American Robin	3.14	±	1.84	4.53	±	2.15
Icterus galbula	Baltimore Oriole	0.00	±	0.00	0.52	±	0.52*
Hirundo rustica	Barn Swallow	8.21	±	3.50	6.46	±	3.70
Parus atricapillus	Black-capped Chickadee	0.00	±	0.00	2.38	±	2.38*
Megaceryle alcyon	Belted Kingfisher	0.00	±	0.00	0.52	±	0.52*
Molothrus ater	Brown-headed Cowbird	1.13	±	0.69	4.55	±	2.64
Dolichonyx oryzivorus	**Bobolink**	4.17	±	4.17*	0.00	±	0.00
Chaetura pelagica	Chimney Swift	0.00	±	0.00	0.44	±	0.44*
Quiscalus quiscula	Common Grackle	1.04	±	1.04*	0.54	±	0.54*
Geothlypis trichas	Common Yellowthroat	12.65	±	5.73	12.86	±	1.61
Spiza americana	**Dickcissel**	3.13	±	3.13*	1.56	±	1.56*
Picoides pubescens	Downy Woodpecker	1.48	±	0.99	0.54	±	0.54*
Tyrannus tyrannus	Eastern Kingbird	0.00	±	0.00	1.56	±	1.56*
Sturnella magna	**Eastern Meadowlark**	3.13	±	3.13*	0.00	±	0.00
Saynornis phoebe	Eastern Phoebe	0.00	±	0.00	0.54	±	0.54*
Contopus virens	**Eastern Wood-pewee**	0.00	±	0.00	0.54	±	0.54*
Dumetella carolinensis	Gray Catbird	2.36	±	1.38	3.99	±	2.05
Picoides villosus	Hairy Woodpecker	0.88	±	0.88*	1.19	±	1.19*
Troglodytes aedon	House Wren	1.75	±	1.24	3.47	±	2.26
Passerina cyanea	Indigo Bunting	0.00	±	0.00	2.23	±	1.29
Empidonax minimus	**Least Flycatcher**	0.00	±	0.00	0.54	±	0.54*
Anas platyrhynchos	Mallard	0.00	±	0.00	0.44	±	0.44*
Cistothorus palustris	**Marsh Wren**	0.69	±	0.69*	1.32	±	1.32*
Zenaida macroura	Mourning Dove	1.32	±	1.32*	1.19	±	1.19*
Cardinalis cardinalis	Northern Cardinal	0.44	±	0.44*	0.54	±	0.54*
Colaptes auratus	N. Flicker (Yellow-shafted)	1.32	±	1.32*	0.00	±	0.00
Icterus spurius	Orchard Oriole	1.04	±	1.04*	1.04	±	1.04*
Pheucticus ludovicianus	**Rose-breasted Grosbeak**	0.00	±	0.00	1.73	±	1.13
Phasianus colchicus	Ring-necked Pheasant	1.32	±	1.32*	0.44	±	0.44*
Columba livia	Rock Pigeon	1.04	±	1.04*	0.00	±	0.00
Agelaius phoeniceus	Red-winged Blackbird	20.96	±	4.06	16.26	±	7.36

Table 6. *continued*

Cistothorus platensis	**Sedge Wren**	4.17	±	4.17*	5.61	±	3.09
Melospiza melodia	Song Sparrow	2.63	±	1.52	3.80	±	1.90
Melospiza georgiana	**Swamp Sparrow**	6.21	±	2.31	2.38	±	1.08
Vermivora peregrina	Tennessee Warbler	0.00	±	0.00	0.54	±	0.54*
Iridoprocne bicolor	Tree Swallow	4.35	±	1.75	4.20	±	2.38
Vireo gilvus	Warbling Vireo	0.00	±	0.00	0.54	±	0.54*
Empidonax trailii	**Willow Flycatcher**	2.63	±	1.52	0.52	±	0.52*
Aix sponsa	Wood Duck	0.00	±	0.00	1.40	±	0.86
Dendroica petechia	Yellow Warbler	2.19	±	2.19*	3.36	±	2.10

*Species only found on one site.
†Abundance significantly different between habitat types ($P \leq 0.05$).
Bold font denotes species of greatest conservation need (SGCN; Minnesota Department of Natural Resources 2006).

Table 7. Mean percent (%) composition (± SE) of breeding birds in sedge wetlands paired with wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota from May-July 2007.

Species		Sedge			RCG		
Scientific name	**Common name**	%			%		
Corvus brachyrhynchos	American Crow	0.00	±	0.00	0.29	±	0.29*
Carduelis tristis	American Goldfinch	8.26	±	1.72	9.40	±	2.66
Setophaga ruticilla	American Redstart	0.00	±	0.00	0.57	±	0.57*
Turdus migratorius	American Robin	6.89	±	4.25	2.83	±	1.00
Philohela minor	**American Woodcock**	0.25	±	0.25*	0.00	±	0.00
Halieetus leucocephalus	**Bald Eagle**	0.44	±	0.44*	0.00	±	0.00
Icterus galbula	Baltimore Oriole	1.56	±	1.26	2.92	±	1.19
Hirundo rustica	Barn Swallow	5.16	±	2.72	5.51	±	3.22
Coccyzus erythropthalmus	**Black-billed Cuckoo**	0.25	±	0.25*	0.00	±	0.00
Parus atricapillus	Black-capped Chickadee	0.49	±	0.49*	0.00	±	0.00
Polioptila caerulea	Blue-gray Gnatcatcher	0.00	±	0.00	0.29	±	0.29*
Molothrus ater	Brown-headed Cowbird	2.05	±	1.28	5.83	±	2.96
Cyanocitta cristata	Blue Jay	0.00	±	0.00	0.29	±	0.29*
Dolichonyx oryzivorus	**Bobolink**	0.88	±	0.88*	0.47	±	0.47*
Spizella pallida	Clay-colored Sparrow	0.49	±	0.49*	0.00	±	0.00
Bombycilla cedrorum	Cedar Waxwing	0.49	±	0.49*	0.00	±	0.00
Petrochelidon pyrrhonata	Cliff Swallow	5.74	±	2.96	0.00	±	0.00
Quiscalus quiscula	Common Grackle	2.53	±	1.01	12.25	±	5.82
Geothlypsis trichas	Common Yellowthroat	8.20	±	3.81	9.06	±	1.36
Spiza americana	**Dickcissel**	1.32	±	1.32*	0.94	±	0.94*
Picoides pubescens	Downy Woodpecker	0.44	±	0.44*	1.02	±	0.69
Tyrannus tyrannus	Eastern Kingbird	0.25	±	0.25*	1.05	±	0.61
Sturnella magna	**Eastern Meadowlark**	1.32	±	1.32*	0.00	±	0.00
Sturnus vulgaris	European Starling	8.87	±	7.92	0.00	±	0.00
Contopus virens	**Eastern Wood-pewee**	0.00	±	0.00	0.57	±	0.57*
Spizella pusilla	**Field Sparrow**	0.88	±	0.88*	0.00	±	0.00
Dumetella carolinensis	Gray Catbird	1.07	±	0.62	2.05	±	1.39
Ammodramus savannarum	**Grasshopper Sparrow**	0.44	±	0.44*	0.00	±	0.00
Troglodytes aedon	House Wren	0.00	±	0.00	1.76	±	1.40
Passerina cyanea	Indigo Bunting	0.00	±	0.00	0.74	±	0.74*
Empidonax minimus	**Least Flycatcher**	0.25	±	0.25*	1.31	±	0.77
Cistothorus palustris	**Marsh Wren**	0.00	±	0.00	1.39	±	1.39*

Table 7. *continued*

Zenaida macroura	Mourning Dove	0.88	±	0.88*	2.41	±	1.21
Cardinalis cardinalis	Northern Cardinal	0.25	±	0.25*	1.02	±	0.69
Colaptes auratus	N. Flicker (Yellow-shafted)	0.25	±	0.25*	0.00	±	0.00
Dryocopus pileatus	Pileated Woodpecker	0.25	±	0.25*	0.00	±	0.00
Pheucticus ludovicianus	**Rose-breasted Grosbeak**	0.00	±	0.00	1.02	±	0.69
Melanerpes carolinus	Red-bellied Woodpecker	0.00	±	0.00	0.29	±	0.29*
Phasianus colchicus	Ring-necked Pheasant	0.25	±	0.25*	0.00	±	0.00
Agelaius phoeniceus	Red-winged Blackbird	18.32	±	2.45	10.99	±	3.34
Passerculus sandwichensis	Savannah Sparrow	0.00	±	0.00	0.47	±	0.47*
Cistothorus platensis	**Sedge Wren**	3.11	±	2.02	7.97	±	3.25
Melospiza melodia	Song Sparrow	2.96	±	1.05	2.05	±	1.39
Melospiza georgiana	**Swamp Sparrow**	4.57	±	1.58	3.44	±	0.90
Vermivora peregrina	Tennessee Warbler	0.00	±	0.00	0.29	±	0.29*
Iridoprocne bicolor	Tree Swallow	6.22	±	1.76	5.64	±	2.01
Vireo gilvus	Warbling Vireo	0.49	±	0.49*	0.29	±	0.29*
Sitta carolinensis	White-breasted Nuthatch	0.00	±	0.00	0.47	±	0.47*
Empidonax trailii	**Willow Flycatcher**	1.65	±	1.10	0.47	±	0.47*
Aix sponsa	Wood Duck	0.00	±	0.00	0.93	±	0.93*
Sphyrapicus varius	**Yellow-bellied Sapsucker**	0.00	±	0.00	0.29	±	0.29*
Dendroica petechia	Yellow Warbler	2.34	±	1.00	1.44	±	1.44*

*Species only found on one site.
†Abundance significantly different between habitat types ($P \leq 0.05$).
Bold font denotes species of greatest conservation need (SGCN; Minnesota Department of Natural Resources 2006).

Table 8. Avian community and nesting parameters (mean ± SE) of four sedge wetlands paired with four wetlands invaded by reed canarygrass (*Phalaris arundinacea*) (blocks) in southern Minnesota during 2006 and 2007.

	Block			
Parameter	**1**	**2**	**3**	**4**
Bird species richness, BS[b]	24.00 ± 2.80 a[a]	11.75 ± 0.75 b	15.75 ± 1.31 c	13.25 ± 1.25 bc
Bird species diversity, BS[c]	16.98 ± 2.22 a	8.14 ± 1.27 a	13.19 ± 1.08 a	12.84 ± 4.53 a
Bird species richness, NB[d]	21.5 ± 0.50 a	13.00 ± 3.00 b	14.00 ± 1.00 b	17.00 ± 1.00 ab
Bird species diversity, NB[e]	13.43 ± 4.16 a	7.09 ± 0.14 a	7.90 ± 0.89 a	7.94 ± 5.42 a
Nest density/10 ha[f]	35.06 ± 16.61 a	22.05 ± 8.28 a	12.45 ± 4.49 a	-
Nesting success[g]	27.60 ± 1.02 a	9.05 ± 7.59 ab	38.91 ± 3.48 ac	-

[a] According to Tukey's post-hoc test, means sharing the same letter are not different ($P \leq 0.05$).
[b] Bird species richness for the 2006-2007 breeding seasons
[c] Bird species diversity for the 2006-2007 breeding seasons calculated via Simpson's Reciprocal Index
[d] Bird species richness for the non-breeding season
[e] Bird species diversity for the non-breeding season calculated via Simpson's Reciprocal Index
[f] Nest density/10 hectares for Red-winged Blackbirds (*Agelaius phoeniceus*) in 2006 and 2007
[g] Nesting success for Red-winged Blackbirds in 2006 and 2007 calculated with the Mayfield method (Mayfield 1961, Mayfield 1975)

than sedge wetlands (14.88 ± 2.16; Figure 15). Species richness also differed between years ($F = 25.485$, d.f. = 1, $P = 0.015$) and was greater in 2007 (18.00 ± 2.42) than 2006 (14.38 ± 1.45). Bird species diversity for the 2006 and 2007 breeding seasons was not different among blocks ($F = 1.283$, d.f. = 3, $P = 0.421$; Table 8) or between habitat types ($F = 1.536$, d.f. = 1, $P = 0.303$), with a mean diversity of 10.81 ± 1.30 in sedge wetlands and 14.77 ± 2.46 in invaded wetlands (Figure 16). Species diversity also was not different between years ($F = 0.499$, d.f. = 1, $P = 0.531$), with a mean diversity of 13.5 ± 2.49 in 2006 and 12.07 ± 1.58 in 2007.

During the non-breeding season, I recorded 54 species of birds across habitat types. Thirty-eight species occurred in sedge wetlands and 42 species occurred in wetlands invaded by *P. arundinacea* (Table 9). The most abundant species in sedge wetlands included Red-winged Blackbird (26.17%), American Goldfinch (13.13%), Swamp Sparrow (10.35%), Yellow-rumped Warbler (*Dendroica coronata*; 7.35%), and Black-capped Chickadee (*Parus atricapillus*; 7.02%; Table 9). The most abundant species in wetlands invaded by *P. arundinacea* included American Goldfinch (9.88%), Swamp Sparrow (9.84%), Canada Goose (*Branta canadensis*; 7.41%), Ruby-throated Hummingbird (*Archilochus colubris*; 6.26%), and Mallard (*Anas platyrhynchos*; 5.58%; Table 9). I recorded ten SGCN during surveys, eight in sedge wetlands and six in wetlands invaded by *P. arundinacea* (Table 9). The percent composition of SGCN collectively was similar between bird communities of sedge wetlands (19.39%) and invaded wetlands (17.42%; $P = 0.853$). For the non-breeding season, the abundance of only one species was different between habitat types. The Ring-necked Pheasant

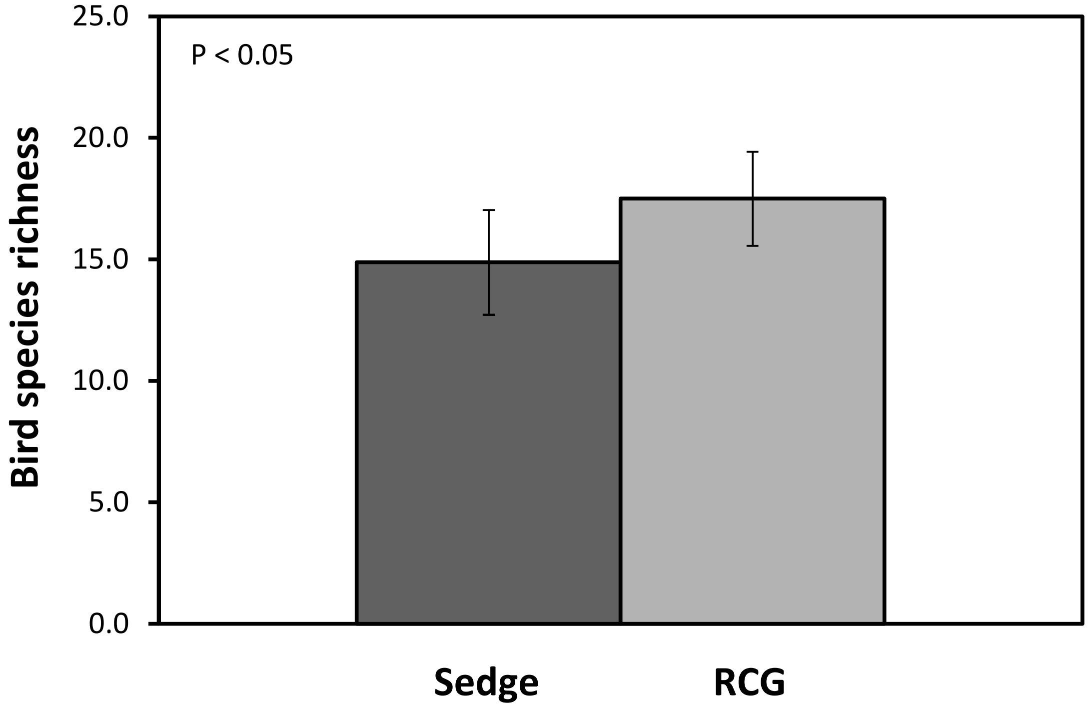

Figure 15. Mean species richness (± SE) of breeding birds in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2006 and 2007 breeding seasons.

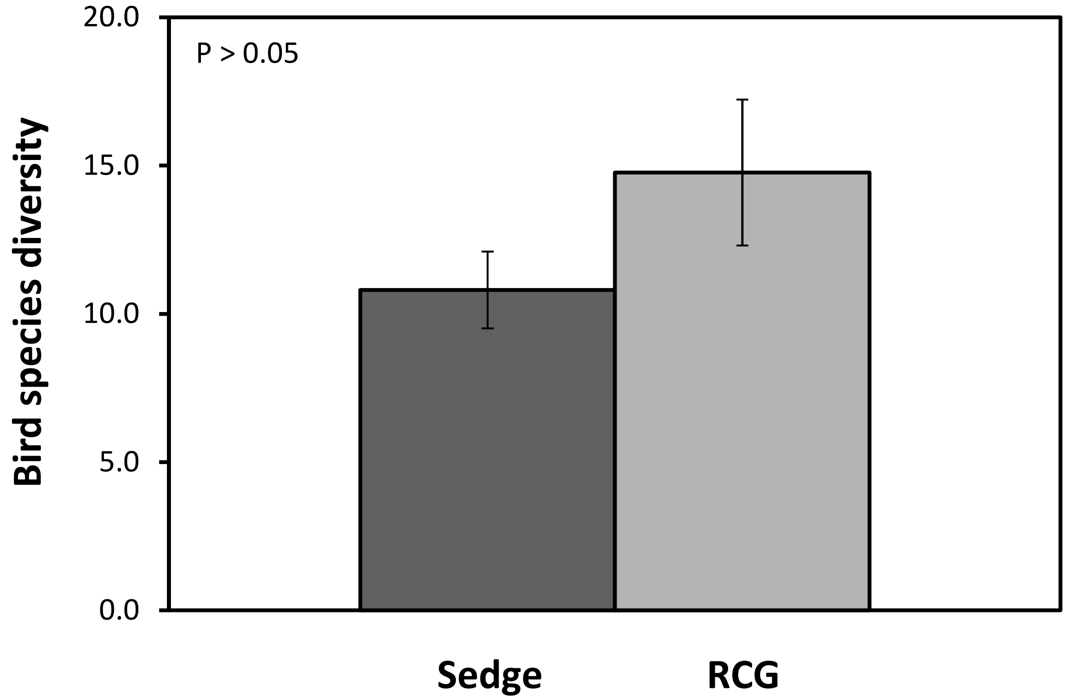

Figure 16. Mean species diversity (± SE) of breeding birds calculated via Simpson's Reciprocal Index in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2006 and 2007 breeding seasons.

Table 9. Mean percent (%) composition (± SE) of non-breeding birds in sedge wetlands paired with wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota from August 2006-April 2007 and August-October 2007.

Species		Sedge	RCG
Scientific name	**Common name**	**%**	**%**
Carduelis tristis	American Goldfinch	13.13 ± 2.68	9.88 ± 3.52
Falco sparverius	American Kestrel	0.00 ± 0.00	0.46 ± 0.46*
Turdus migratorius	American Robin	1.54 ± 0.94	3.18 ± 1.08
Spizella arborea	American Tree Sparrow	0.58 ± 0.58*	1.88 ± 1.33
Hirundo rustica	Barn Swallow	4.90 ± 4.90*	0.86 ± 0.50
Parus atricapillus	Black-capped Chickadee	7.02 ± 4.30	3.84 ± 2.04
Molothrus ater	Brown-headed Cowbird	0.00 ± 0.00	0.53 ± 0.53*
Cyanocitta cristata	Blue Jay	2.44 ± 2.19	3.51 ± 1.71
Branta canadensis	Canada Goose	0.00 ± 0.00	7.41 ± 7.41*
Bombycilla cedrorum	Cedar Waxwing	0.32 ± 0.32*	2.52 ± 1.88
Chaetura pelagica	Chimney Swift	0.64 ± 0.64*	0.00 ± 0.00
Quiscalus quiscula	Common Grackle	0.49 ± 0.49*	1.54 ± 1.01
Accipiter cooperii	Cooper's Hawk	0.19 ± 0.19*	0.00 ± 0.00
Chordeiles minor	**Common Nighthawk**	0.00 ± 0.00	0.53 ± 0.53*
Capella gallinago	Common Snipe	0.49 ± 0.49*	0.94 ± 0.94*
Geothlypsis trichas	Common Yellowthroat	1.48 ± 1.10	2.52 ± 2.03
Junco hyemalis	Dark-eyed Junco	0.32 ± 0.32*	4.37 ± 2.70
Spiza americana	**Dickcissel**	0.98 ± 0.98*	0.00 ± 0.00
Picoides pubescens	Downy Woodpecker	1.38 ± 0.60	1.27 ± 0.78
Sialia sialis	Eastern Bluebird	0.54 ± 0.54*	0.47 ± 0.47*
Tyrannus tyrannus	Eastern Kingbird	1.62 ± 0.98	0.86 ± 0.50
Sturnella magna	**Eastern Meadowlark**	0.98 ± 0.98*	0.00 ± 0.00
Sturnus vulgaris	European Starling	1.60 ± 1.60*	0.00 ± 0.00
Contopus virens	**Eastern Wood-pewee**	0.00 ± 0.00	0.40 ± 0.40*
Spizella pusilla	**Field Sparrow**	1.16 ± 1.16*	0.00 ± 0.00
Regulus satrapa	Golden-crowned Kinglet	0.00 ± 0.00	0.40 ± 0.40*
Dumetella carolinensis	Gray Catbird	0.71 ± 0.41	2.66 ± 2.66*
Troglodytes aedon	House Wren	0.00 ± 0.00	2.92 ± 2.01
Melospiza lincolnii	Lincoln's Sparrow	0.00 ± 0.00	1.06 ± 1.06*
Anas platyrhynchos	Mallard	0.00 ± 0.00	5.58 ± 3.93
Vermivora ruficapilla	Nashville Warbler	0.00 ± 0.00	0.47 ± 0.47*
Cardinalis cardinalis	Northern Cardinal	0.64 ± 0.64*	0.40 ± 0.40*

Table 9. *continued*

Colaptes auratus	N. Flicker (Yellow-shafted)	0.32	±	0.32*	0.00	±	0.00
Pheucticus ludovicianus	**Rose-breasted Grosbeak**	0.19	±	0.19*	2.52	±	2.03
Regulus calendula	Ruby-crowned Kinglet	1.47	±	1.47*	0.00	±	0.00
Phasianus colchicus	Ring-necked Pheasant	0.00	±	0.00	1.47	±	0.49†
Buteo jamaicensis	Red-tailed Hawk	0.00	±	0.00	0.46	±	0.46*
Archilochus colubris	Ruby-throated Hummingbird	1.18	±	0.69	6.26	±	5.08
Agelaius phoeniceus	Red-winged Blackbird	26.17	±	12.52	5.14	±	2.99
Passerculus sandwichensis	Savannah Sparrow	0.98	±	0.98*	0.00	±	0.00
Cistothorus platensis	**Sedge Wren**	3.58	±	3.17	3.74	±	2.16
Porzana carolina	Sora	0.00	±	0.00	0.46	±	0.46*
Melospiza melodia	Song Sparrow	1.60	±	0.95	5.08	±	1.59
Accipiter striatus	Sharp-shinned Hawk	0.00	±	0.00	0.47	±	0.47*
Melospiza georgiana	**Swamp Sparrow**	10.35	±	5.24	9.84	±	6.01
Iridoprocne bicolor	Tree Swallow	0.49	±	0.49*	1.19	±	1.19*
Vireo gilvus	Warbling Vireo	0.32	±	0.32*	0.40	±	0.40*
Sitta carolinensis	White-breasted Nuthatch	0.00	±	0.00	0.79	±	0.79*
Empidonax trailii	**Willow Flycatcher**	0.19	±	0.19*	0.40	±	0.40*
Wilsonia pusilla	Wilson's Warbler	0.49	±	0.49*	0.00	±	0.00
NA	Unknown Woodpecker	0.00	±	0.00	0.53	±	0.53*
Zonotrichia albicollis	**White-throated Sparrow**	1.95	±	1.55	0.00	±	0.00
Dendroica petechia	Yellow Warbler	0.19	±	0.19*	0.00	±	0.00
Dendroica coronata	Yellow-rumped Warbler	7.35	±	7.35*	0.79	±	0.79*

*Species only found on one site.
†Abundance significantly different between habitat types ($P \leq 0.05$).
Bold font denotes species of greatest conservation need (SGCN; Minnesota Department of Natural Resources 2006).

(*Phasianus colchicus*) was more abundant and only occurred in sites invaded by *P. arundinacea* (P = 0.058; Table 9). Percent similarity of bird communities between habitat types was 51.23%.

Bird species richness for the non-breeding season had mixed results for the main effects as opposed to bird species diversity. Species richness of non-breeding birds was different among blocks (F = 11.847, d.f. = 3, P = 0.036), as Block 1 had greater species richness than Blocks 2 and 3 (Table 8). Bird species richness was not different, however, between habitat types (F = 6.153, d.f. = 1, P = 0.089), with a mean richness of 15.00 ± 2.35 for sedge wetlands and 17.75 ± 1.55 for invaded wetlands (Figure 17). Species diversity of non-breeding birds was not different among blocks (F = 1.324, d.f. = 3, P = 0.412; Table 8) or between habitat types (F = 4.357, d.f. = 1, P = 0.128), with a mean diversity of 6.44 ± 1.41 for sedge wetlands and 11.74 ± 2.34 for invaded wetlands (Figure 18).

Nesting Success

In 2006, I found and monitored nests of 12 species, 11 of which occurred in sedge wetlands and eight in wetlands invaded by *P. arundinacea* (Table 10). I found more Red-winged Blackbird nests than all other species in both habitat types. Yellow Warbler nests were the second most abundant in sedge wetlands, but I found few nests in wetlands invaded by *P. arundinacea* (Table 10). Two SGCN, Dickcissel (*Spiza americana*) and Sedge Wren, nested in both habitat types in 2006. Three SGCN, Eastern Meadowlark (*Sturnella magna*), Swamp Sparrow, and Willow Flycatcher (*Empidonax trailii*), nested only in sedge wetlands. No SGCN nested exclusively in wetlands invaded by *P. arundinacea* in 2006 (Table 10).

I found 118 Red-winged Blackbird nests in 2006 (n = 47) and 2007 (n = 71). However, three were inactive when found, one contained only a cowbird nestling, nine were abandoned during nest building presumably due to observer disturbance, and one was only checked once. Therefore, I analyzed 104 usable nests, 69 in sedge wetlands and 35 in wetlands invaded by *P. arundinacea* (Table 11). I found 17 nests during the egg-laying stage, 65 during incubation, and 22 during the nestling stage (Table 11). The mean Julian date of nests found in sedge wetlands and wetlands invaded by *P. arundinacea* did not differ (P = 0.110) and was 149.9 and 156.8, respectively. Mean density of nests/10 hectares was not different among blocks (F = 1.141, d.f. = 2, P = 0.467; Table 8) or between habitat types (F = 1.229, d.f. = 1, P = 0.383), with a density of 29.99 ± 12.62 nests/10 hectares for sedge wetlands and 16.38 ± 1.38 nests/10 hectares for wetlands invaded by *P. arundinacea* (Figure 19). Nesting success was different among blocks (F = 20.616, d.f. = 2, P = 0.046). Block 3 had a higher success rate than Block 2 (Table 8). However, nesting success was not different between habitat types (F = 4.417, d.f. = 1, P = 0.170), with a mean success rate of 29.21% ± 7.44 for sedge wetlands and 21.16% ± 10.17 for wetlands invaded by *P. arundinacea* (Figure 20). Habitat type was not a significant predictor of nest fate for Red-winged Blackbirds (d.f. = 1, P = 0.605). Furthermore, nest survival did not vary by Julian date found (d. f. = 1, P = 0.909), by stage found (d.f. = 1, P = 0.068), or by block (d.f. = 2, P = 0.174).

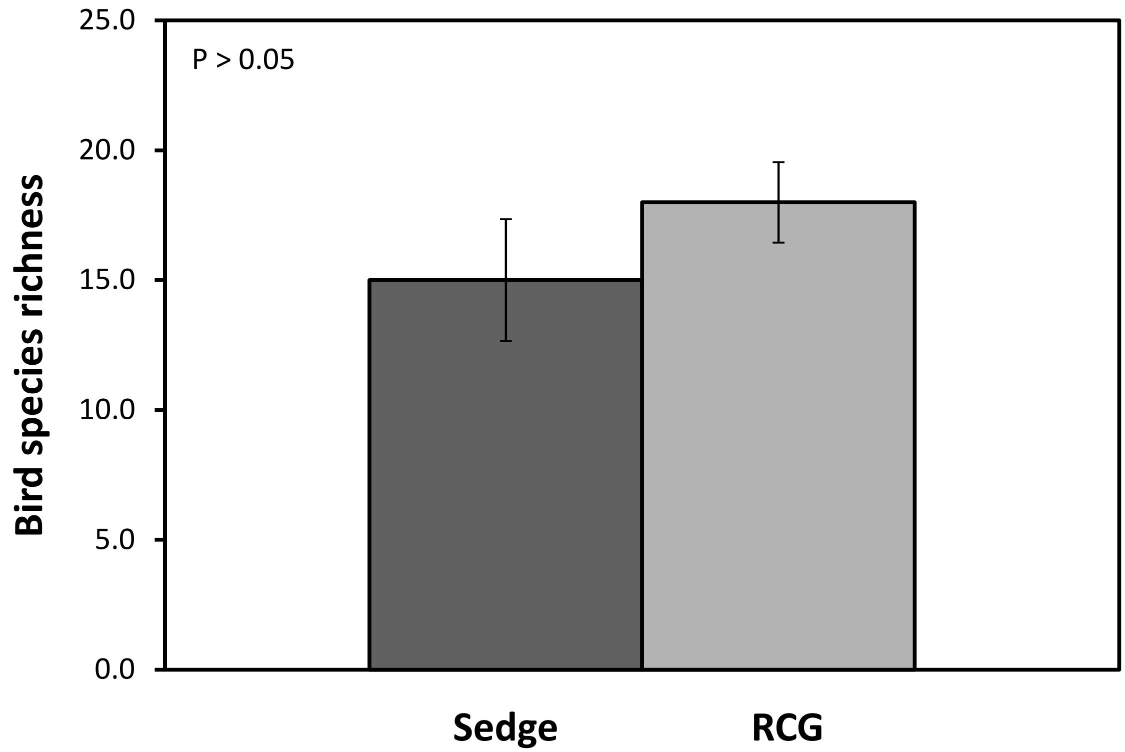

Figure 17. Mean species richness (± SE) of non-breeding birds in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota from August 2006-April 2007 and August-October 2007.

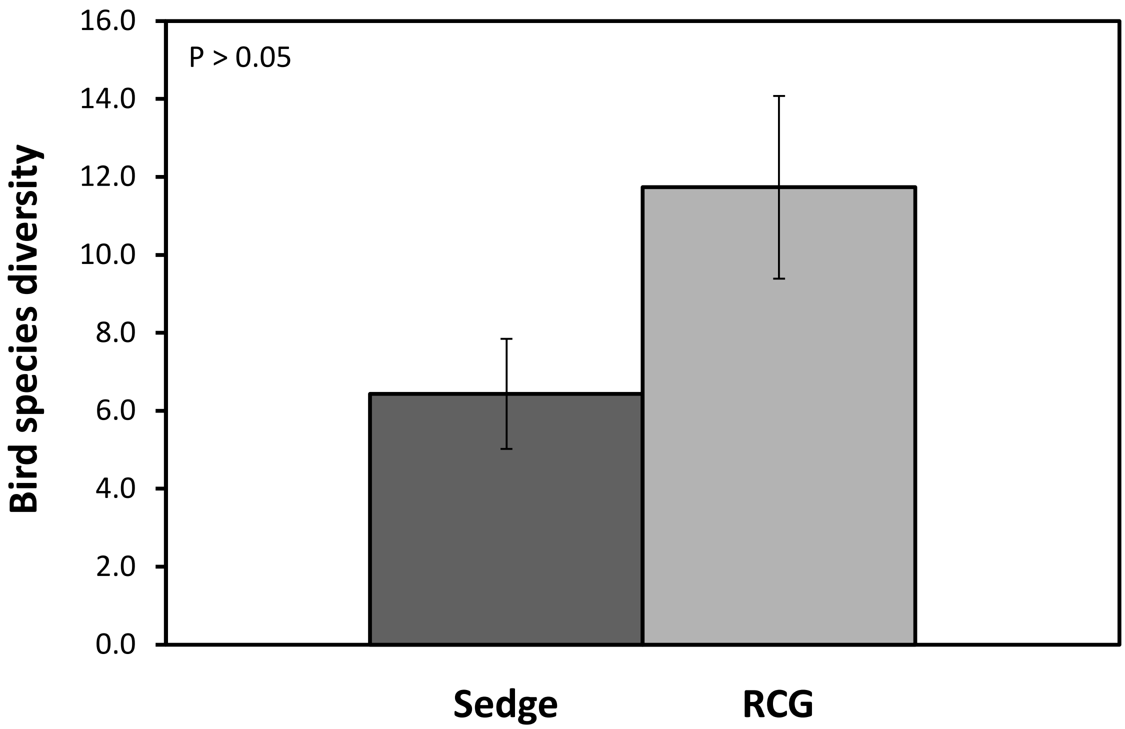

Figure 18. Mean species diversity (± SE) of non-breeding birds in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota from August 2006-April 2007 and August-October 2007.

Table 10. Number (No.) of nests of all species found in four sedge wetlands paired with four wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota from May-July 2006.

Species		Sedge	RCG
Scientific name	**Common name**	**No.**	**No.**
Carduelis tristis	American Goldfinch	3	2
Turdus migratorius	American Robin	4	0
Geothlypsis trichas	Common Yellowthroat	2	2
Spiza americana	**Dickcissel**	1	1
Sturnella magna	**Eastern Meadowlark**	1	0
Dumetella carolinensis	Gray Catbird	1	1
Troglodytes aedon	House Wren	0	1
Agelaius phoeniceus	Red-winged Blackbird	36	16
Cistothorus platensis	**Sedge Wren**	1	4
Melospiza georgiana	**Swamp Sparrow**	3	0
Empidonax trailii	**Willow Fycatcher**	1	0
Dendroica petechia	Yellow Warbler	17	2
Total		70	29

Bold font denotes species of greatest conservation need (SGCN; Minnesota Department of Natural Resources 2006).

Table 11. Mean daily survival rates during stages of the nest cycle for Red-winged Blackbirds (*Agelaius phoeniceus*) in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during 2006 and 2007.

Habitat type	Nest cycle stage	Total nests (n)	Failed nests (n)	Stage length[b]	Total exposure days: Nest-days	Egg-days	Nestling-days	Mean daily survival rate[c] ± SE	Mean nesting success[c] (%) ± SE
Sedge	Egg-laying	10	4	3	25.5	[d]	-	0.89 ± 0.06	
	Incubation	42	9	11	298	1075	-	0.97 ± 0.01	
	Nestling	17	16	10	433.5	-	1285.5	0.96 ± 0.01	
	Total	69	29	24	-	-	-	-	29.21 ± 7.44
RCG	Egg-laying	7	2	3	14	[d]	-	0.73 ± 0.20	
	Incubation	23	9	11	194.5	676.5	-	0.95 ± 0.02	
	Nestling	5	6	10	210	-	635	0.98 ± 0.01	
	Total	35	17	24	-	-	-	-	21.16 ± 10.17

[a] Data was analyzed by site but is summarized by habitat type.
[b] Stage lengths for the nesting cycle of Red-winged Blackbirds (*Agelaius phoeniceus*) from Ehrlich et al. (1988) and Yasukawa and Searcy (1995).
[c] Daily survival rates and nesting success were calculated with the Mayfield method (Mayfield 1961, Mayfield (1975).
[d] No individual eggs were lost without the loss of the entire nest during the egg-laying stage, and therefore no egg-days were calculated.
† Indicates a significant difference ($P \leq 0.05$).

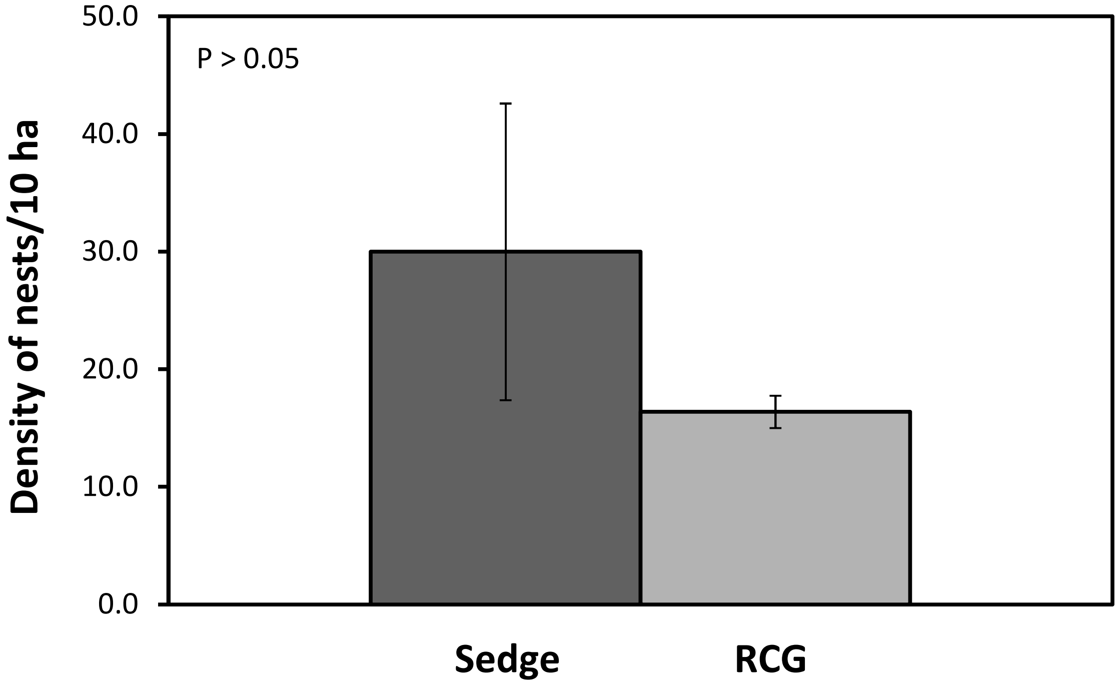

Figure 19. Mean density of Red-winged Blackbird (*Agelaius phoeniceus*) nests (± SE) per 10 hectares in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2006 and 2007 breeding seasons.

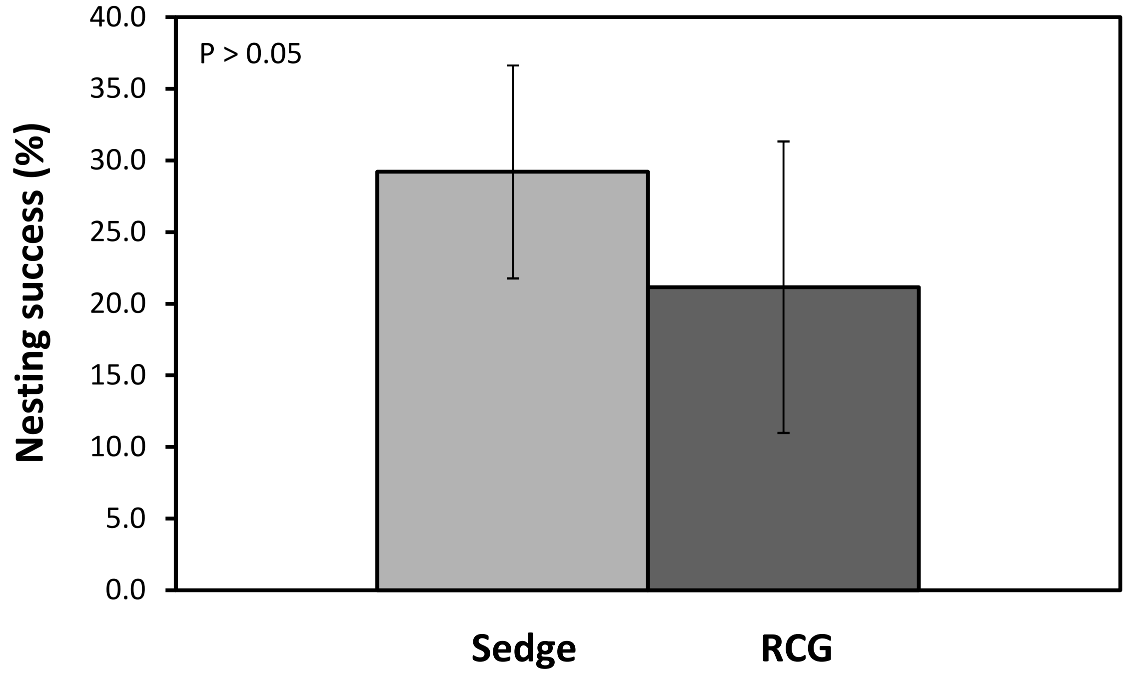

Figure 20. Mean percent nesting success from egg-laying to fledging (± SE) calculated with the Mayfield method (Mayfield 1961, Mayfield 1975) for Red-winged Blackbird (*Agelaius phoeniceus*) nests found in sedge wetlands and wetlands invaded by reed canarygrass (*Phalaris arundinacea*; RCG) in southern Minnesota during the 2006 and 2007 breeding seasons.

CHAPTER 4

DISCUSSION AND CONCLUSIONS

Invasive plants can decrease biodiversity in some communities (Vitousek et al. 1996). *Phalaris arundinacea* has contributed to decreases in plant species diversity and heterogeneity in wetlands (Apfelbaum and Sams 1987, Galatowitsch et al. 1999, Kercher et al. 2004, Lavergne and Molofsky 2004, Schooler et al. 2006) and grows taller and produces more aboveground biomass than other wetland plants (Green and Galatowitsch 2001, Green and Galatowitsch 2002, Lindig-Cisneros and Zedler 2002, Maurer and Zedler 2002, Maurer et al. 2003). For these reasons, I expected that plant species richness, diversity, and heterogeneity would be lower in wetlands invaded by *P. arundinacea* and that vegetation would be taller with greater VOR than sedge wetlands. I also expected that wetlands invaded by *P. arundinacea* would produce greater litter depths than sedge wetlands. Lastly, I hypothesized that the alteration of vegetative structure caused by invasion of *P. arundinacea* would impact bird species richness, diversity, and nesting success of Red-winged Blackbirds.

Results of this study share some similarities with previous research on the effects of invasion by *P. arundinacea* on native plant communities. Invasion by *P. arundinacea* appeared to decrease plant species richness during summer 2007, as invaded wetlands had lower richness than sedge wetlands. However, plant species diversity was not different between habitat types. Although dominated by *P. arundinacea*, the invaded wetlands were not monotypes. The lack of detecting a difference in plant species diversity may indicate these wetlands were still in a state of invasion, such that *P.*

arundinacea was in the process of invading a more diverse wetland. Furthermore, invasion by *P. arundinacea* did not adversely affect percent cover of individual plant species, percent composition of listed species collectively, or percent composition of plant functional groups. Wet and sedge meadows, like the wetlands in this study, in the upper Midwestern United States are typically dominated by tall, dense graminoids (Reuter 1986, Mossman and Sample 1990). The dominance of *P. arundinacea*, a species native to North America (Anderson 1961, Apfelbaum and Sams 1987), does not appear to have altered the physical structure of vegetation in these wetlands, at least to the extent of the vegetative characteristics measured during this period of time.

As expected, vegetation in wetlands invaded by *P. arundinacea* was taller and had greater VOR than sedge wetlands, but litter depth and horizontal heterogeneity were not different between habitat types. The horizontal heterogeneity of all my sites was rather low (< 1) compared to sites studied by Wiens (1974), who reported heterogeneity indices of 1-3 for grasslands. Truncated readings may have contributed to lower-than-expected measurements of VOR, vegetative height, and heterogeneity as the Robel pole was 17 decimeters tall. The VOR exceeded this limit in 8.61% and 4.02% of measurements in sedge and invaded wetlands, respectively, and the height exceeded this limit in 15.16% and 19.20% of measurements, respectively. Furthermore, Wiens (1974) reported litter depths of generally ≤ two centimeters in a range of grassland communities. Litter depths of wetlands in my study were comparatively greater at > eight centimeters for both habitat types and are similar to litter depths reported for other wetlands invaded by *P. arundinacea*, ranging from 0.9-9.6 centimeters (Kirsch et al. 2007). Even though sedge wetlands had shorter vegetation with lower VOR and greater plant species richness than invaded wetlands, horizontal heterogeneity was not different between habitat types. Wiens (1974) concluded that vegetation in the tallgrass prairie region is tall and dense with a high percent cover of grass, generally low horizontal heterogeneity, and relatively deep litter. My findings of a high percent cover of graminoids (>70%), low heterogeneity, and deep litter for both habitat types parallel this research.

Sedge wetlands had more woody stems/100 m^2 that were < two meters tall than wetlands invaded by *P. arundinacea*, but the number of woody stems that were > two meters tall was not different between habitat types. Invasion by *P. arundinacea* may prevent the establishment and growth of shrubs in wetlands. Furthermore, despite the fact sedge wetlands had a greater number of woody stems < two meters tall, horizontal heterogeneity was not different between habitat types. This finding contradicts previous research that demonstrates increased heterogeneity in plant communities with woody vegetation (ie. MacArthur et al. 1962, Karr and Roth 1971, Wiens 1974, Roth 1976). The tall, dense vegetation of these wetlands may have masked any heterogeneity provided by shrubs < two meters tall.

Physical structure of the plant community varied little within and between habitat types over the course of the growing season. Invaded wetlands had taller vegetation than sedge wetlands in June, but VOR and litter depth did not differ between habitat types during the months of June and July. Additionally, litter depth was greater in invaded wetlands in June than July. Structure of vegetation may differ more between habitat types during the fall and winter months, as I observed that *P. arundinacea* exhibited a characteristic structural collapse in late summer and early fall (Klopatek and Stearns 1978, Conchou and Fustec 1988). Finding no difference in bird species richness, diversity, or abundance of individual species between habitat types during the non-breeding season (except that the Ring-necked Pheasant was more abundant in invaded wetlands) indicates that invaded wetlands still provide cover for birds. Because invaded wetlands were not monotypes, plants with more rigid structures may have continued to provide upright cover during fall and winter. Further research on the physical structure of vegetation and bird communities of these habitat types during the fall and winter months is needed.

Differences in vegetative structure and the avian community occurred among blocks. Vegetative structure differed among blocks for three out of eight parameters, including VOR, maximum height, and number of woody stems/100 m^2 that were > two meters tall. Additionally, differences in the avian community and nesting occurred among blocks for three out of six parameters, including bird species richness during the 2006 and 2007 breeding seasons as well as the non-breeding season and nesting success. Surprisingly, Block 3 had greater bird species richness during the breeding season and higher nesting success than Block 2, even though Block 3 had the shortest vegetation with the lowest VOR among blocks. Furthermore, Block 2 had similar bird species richness during the breeding season compared to Block 4, even though Block 4 had greater VOR and a greater number of woody stems > two meters tall. These results contradict previous research that demonstrates an increase in bird species richness in plant communities with taller grass and greater vertical structure (MacArthur and

MacArthur 1961, Cody 1968) and research that demonstrates nests with greater concealment are more successful (Dwernychuk and Boag 1972, Martin and Roper 1988, Johnson and Temple 1990, Mankin and Warner 1992, Martin 1993, Davis 2005).

Results of this study contradict the current perception that invasion by *P. arundinacea* negatively affects birds. Bird species diversity was not different between habitat types during the breeding season, and wetlands invaded by *P. arundinacea* actually had greater species richness of breeding birds than sedge wetlands. This phenomenon may be explained, in part, by the fact that invaded wetlands had greater height and VOR than sedge wetlands. Cody (1968) concluded that in structurally simple habitats like grasslands, the species richness of birds could be predicted by the mean height of the grass and its standard deviation. More species can coexist in very tall vegetation by feeding at different heights (Cody 1968). Other factors not measured may certainly affect bird species richness, such as the variation in wetland vegetation between years. Furthermore, invasion by *P. arundinacea* did not adversely affect abundance of individual bird species or percent composition of listed species collectively. In Wisconsin, sedge wetlands typically do not have highly diverse plant and bird communities (Mossman and Sample 1990), and the invasion by *P. arundinacea* does not appear to have changed the structure of vegetation in a way that negatively affects composition of the bird community in southern Minnesota wetlands. In fact, Mossman and Sample (1990) found that the bird communities of Wisconsin sedge wetlands are similar to bird communities of wetlands invaded by *P. arundinacea* and upland areas planted to monotypic stands of switchgrass (*Panicum virgatum* L.).

Invasion by *P. arundinacea* did not affect nesting success or density of nests of Red-winged Blackbirds, as both variables were similar between habitat types. Red-winged Blackbirds often prefer to nest in tall, dense vegetation (Albers 1978, Bryan and Best 1994, Camp and Best 1994). In linear habitats like roadsides and grassed waterways in agricultural fields, nest densities and nesting success of Red-winged Blackbirds were greater at nest sites with tall, dense vegetation with a high percent cover of grass (Bryan and Best 1994, Camp and Best 1994). In fact, densities of Red-winged Blackbird nests in roadsides were highly correlated with percent cover of *P. arundinacea* as well as height and density of vegetation (Camp and Best 1994). In my study, both habitat types were comprised of a high percent composition of graminoids (> 70%), and although vegetation height and VOR were greater in invaded wetlands, nest density and nesting success did not differ between habitat types. In some cases, no clear relationship exists between vegetative structure and nesting success of birds (Best and Stauffer 1980, Patterson and Best 1996). However, a possible explanation may be that although wetlands invaded by *P. arundinacea* had taller vegetation than sedge wetlands (14.0 dm ± 0.4 vs. 11.3 dm ± 0.6) with greater VOR (10.4 dm ± 0.4 vs. 8.0 dm ± 0.6), sedge wetlands in southern Minnesota were still relatively tall and dense compared to other nesting habitats for Red-winged Blackbirds. Camp and Best (1994) reported a mean maximum height of live vegetation of 8.1 dm ± 0.71 and mean VOR of 2.7 dm ± 0.21 in the vicinity of Red-winged Blackbird nests in roadsides, and Bryan and Best (1994) reported mean height and VOR measurements in waterways of 8.6 dm ± 2.6 and 3.9 dm ± 1.0, respectively. Additionally, Red-winged Blackbirds nest in both heterogeneous (Weller and Spatcher

1965) and homogenous habitats (McCoy et al. 2001), but the scale at which heterogeneity occurs may vary. For instance, Red-winged Blackbirds may select a homogenous nest site within a more heterogeneous habitat patch (Burger 1985). I took vegetative measurements at the scale of the habitat patch and found that horizontal heterogeneity was low and did not differ between habitat types. If heterogeneity had differed at the patch scale, nesting success may have been different between habitat types. Furthermore, measurements taken at the nest may have differed in heterogeneity compared to the habitat patch overall.

Although nests of species other than the Red-winged Blackbird were not abundant enough to warrant analysis, the presence-absence of some nesting species may be important. The Yellow Warbler is a shrub-nesting species that frequently nests in shrubby wetlands (Ehrlich et al. 1988). This species nested more frequently in sedge wetlands, where shrub cover was more abundant. Invasion of *P. arundinacea* may prevent the establishment and growth of shrubs, restricting the Yellow Warbler's opportunity to nest in this habitat type. Sedge Wrens, a SGCN, nested in both habitat types in this study, though I found only several nests. In southern Wisconsin sedge wetlands, Sedge Wrens are negatively affected by brush invasion (Mossman and Sample 1990). Furthermore, placement of Sedge Wren territories was positively correlated with cover of *P. arundinacea* in Minnesota and Wisconsin wetlands (Kirsch et al. 2007). Conversely, Swamp Sparrows—also a SGCN—placed their territories to avoid areas with high cover of *P. arundinacea*, but this phenomenon was probably related to less standing water in areas dominated by *P. arundinacea* (Kirsch et al. 2007). In this study, I found only three Swamp Sparrow nests, all in sedge wetlands. Focused search efforts on Sedge Wrens and Swamp Sparrows may have yielded results similar to previous research (Mossman and Sample 1990, Kirsch et al. 2007).

In conclusion, the structure of vegetation in sedge wetlands and wetlands invaded by *P. arundinacea* exhibited several differences. The main differences were that wetlands invaded by *P. arundinacea* had greater vegetative height and VOR than sedge wetlands whereas sedge wetlands had greater plant species richness and more woody stems/100 m^2 that were < two meters tall. Plant species diversity, litter depth, horizontal heterogeneity, and number of woody stems/100 m^2 that were > two meters were not different between habitat types. Although invasion by *P. arundinacea* had mixed effects on the plant community in this study, it has had marked negative effects on other native plant communities (ie. Apfelbaum and Sams 1987, Galatowitsch et al. 1999). Therefore, *P. arundinacea* is likely to be a continual problem in the restoration and management of diverse wetlands in Minnesota and other Midwestern states. Results of this study did not indicate that invasion by *P. arundinacea* has a negative effect on bird communities in Minnesota wetlands with regard to species richness, diversity, abundance of individual species, or nesting success of Red-winged Blackbirds. Ultimately, the invasion by *P. arundinacea* does not appear to have altered the structure of wetland vegetation in a way that negatively affects birds and may provide better avian habitat than is currently perceived.

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