THE EFFECTS OF WETLAND SEDIMENTATION AND SEDIMENT REMOVAL ON RAINWATER BASIN VEGETATION, SEED BANK, AND WATERBIRD COMMUNITIES

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CHAPTER I

EFFECTS OF SEDIMENT REMOVAL ON VEGETATION COMMUNITIES IN RAINWATER BASIN PLAYA WETLANDS

ABSTRACT

Alterations of natural hydrologic regimes through sedimentation from cultivated agricultural land use have affected most depressional wetlands in the Great Plains. These alterations can negatively affect native wetland plant communities. Our objective was to test the efficient-community hypothesis which suggests that restored wetlands will develop plant communities similar to reference conditions following hydrologic restoration. For this study, hydrology was restored via sediment removal. Thirty-four playa wetlands in reference, restored, and agricultural condition within the Rainwater Basin Region of Nebraska were sampled in 2008 and 2009. In 2008, reference and restored wetlands had higher species richness and more native, annual, and perennial species than agricultural wetlands. Restored and reference wetlands had similar exotic species richness, however restored wetlands contained more than agricultural wetlands. In 2009, reference and restored wetlands had higher species richness, and more perennial, and native species than agricultural wetlands. Restored wetlands contained a greater number and proportion of annuals than reference and agricultural wetlands. Restored

wetlands proportion of exotics was 3.5 times less than agricultural wetlands. Canonical Correspondence Analysis showed that reference, restored and agricultural wetlands are dominated by different plant species and guilds, and restored wetland plant communities do not appear to be acting as intermediates between reference and agricultural conditions or on a trajectory to reach reference condition. This may be attributed to differing seed bank communities between reference and restored wetlands, dispersal limitations of perennial plant guilds associated with reference wetland conditions, and/or management activities at restored wetlands may be preventing restored wetlands from reaching reference status.

INTRODUCTION

In the U.S. Great Plains, agricultural practices have altered terrestrial and wetland habitats to make way for crop and livestock pastures (Samson and Knopf 1996). This has ultimately led to changes in ecosystem services provided in these landscapes (Smith et al. 2011a). This is especially true in the Rainwater Basin (RWB) of Nebraska where up to 90% of playa wetlands have been drained or modified for agricultural purposes (Stutheit et al. 2004). A stopover site to over 12 million migrating waterfowl, geese, and shorebirds every year, this environmentally sensitive area has been deemed as one of nine areas in the contiguous United States with the highest wetland loss (Tiner 1984) and contains one of the most threatened and least studied wetland complexes in North America (Smith 1998).

Throughout the Great Plains, sedimentation from upland erosion from surrounding agricultural fields is the largest threat to the continued existence of properly functioning depressional wetlands (Luo et al. 1997, 1999, Tsai et al. 2007). Playa wetlands, the dominant hydrogeomorphic feature of the RWB, are the lowest point within a watershed and are thus highly susceptible to sedimentation (LaGrange 2005). Excessive sediment loads within wetlands can bury hydric soils, reduce wetland volume, increase surface area, and shorten hydroperiods (Luo et al. 1997, Tsai et al. 2007). These changes can alter plant community structure through burial of seed banks (Jurik et al. 1994, Gleason et al. 2003), allow non-native species to colonize and dominate an area (Smith and Haukos 2002), and select for monotypic stands of invasive native or exotic species (Galatowitsch et al. 1999).

The efficient-community hypothesis states that wetland vegetation should reestablish naturally following wetland hydrologic restoration, and all plant species that can become established and survive under the environmental conditions found at the site will eventually be found growing there or occurring within the seed bank (Galatowitsch and van der Valk 1996). The plant communities found at wetland sites with restored hydrology are determined by pre-sedimentation environmental conditions (Galatowitsch and van der Valk 1996) and the underlying seed bank. A successful hydrologic restoration by the removal of sediment should develop plant communities similar to historic conditions or be on a restoration trajectory to reach reference conditions.

Removal of sediment from agriculturally impacted wetlands aids in restoring the natural hydrology of a wetland by removing non-hydric soils that may absorb water rather than ponding it. Restoring hydrology is critical in establishing native wetland plant communities (Keddy 2000). Sediment removal has also been shown to lower nutrient availability (Klimkowska et al. 2007), remove persistent pesticides (Kiehl and Wagner

2006), and remove persistent weedy and invasive species from the seed bank (Constance et al. 2007). In addition, sediment removal also removes established vegetation (Kiehl et al. 2006) that prevents the seed bank from contributing to the development of standing vegetation and restores ecosystem function (Odum and Barrett 2005). RWB wetland restoration typically involves removal of up to 30 cm of sediment, filling irrigation reuse pits, and the reestablishment of upland buffers to protect wetlands from future sedimentation. These practices should allow establishment of pre-impact vegetation, however, only seeds persistent enough in the seed bank prior to impact will initially become established. Other species found will arrive via dispersal.

Because most restored wetlands within the RWB are allowed to revegetate naturally following sediment removal and are assumed to resemble historic conditions or be on a trajectory to reach reference conditions, the objective of this study was to test this goal via the efficient-community hypothesis. We predicted that restored RWB wetlands will develop plant community characteristics (species richness, number of annuals, perennials, native and invasive species as well as composition of each) similar to reference wetlands once sediment has been removed.

METHODS

Study Area

The RWB Region encompasses 15,907 km^2 and includes all or parts of 21 counties on the Central Loess Plains of south-central Nebraska (LaGrange 2005). The area was named for its abundant natural wetlands that formed where clay-bottom depressions catch and hold precipitation from rain and run-off (Stutheit et al. 2004).

Annual precipitation averages 460 mm in the western portion of the region and 710 mm in the east; evapotranspiration generally exceeds precipitation (Stutheit et al. 2004). Within this region, playa wetlands are the most notable hydrogeomorphic feature on the landscape. Playas range from 0.1 ha to 1,000 ha in size and are defined by the presence of Massie, Scott and Fillmore soil series (Stutheit et al. 2004). The area was originally mixed grass prairie in the western region and tall grass prairie in the eastern region (Kaul 1975), but presently the region is intensively cultivated with corn and soybeans. Domestic livestock graze most uncultivated areas.

Study Sites

Thirty-four wetlands were sampled in 2008 and 2009 among three land use treatments: reference standard (from here forward known as reference), restored, and cropland (defined below). In 2008, 12 reference, 11 restored, and 11 cropland wetlands were sampled and in 2009, 11 reference, 11 restored, and 12 cropland wetlands were sampled. Most wetlands were sampled both years (one reference and one restored wetland was removed in 2009, one agricultural wetland was restored in late 2008, and two agricultural wetlands were added in 2009).

Reference wetlands were selected using the hydrogeomorphic approach (Brinson 1993) by the Nebraska Game and Parks Commission (NGPC) and represented the most highly functioning wetlands within the region (Stutheit 2004). Reference wetlands have had no prior physical manipulation to the basin or water levels, vegetation with little to no invasive species problems, an unmanipulated watershed, and hydric soils present match wetland type (e.g., semi-permanent, seasonal, temporary). The 12 best reference wetlands from the HGM study (Stutheit et al. 2004) were selected for this study. In 2008, 6 of the sampled reference wetlands were seasonal and 6 were semi-permanent. In 2009, 5 were seasonal and 6 were semi-permanent.

Restoration of wetlands impacted by sediment was performed by NGPC, U.S. Fish and Wildlife Service (USFWS), and Ducks Unlimited (DU). Each of these sites was at one time impacted by cropping. Restored wetlands had an average of 30.4 cm of sediment removed from the center and then were graded out to a depth of 10.6 - 15.2 cm around the perimeter. Restored wetlands sampled in 2008 ranged in age from 2 to 11 years since sediment removal and in 2009 from 1 to 12 years since sediment removal. Due to the limited number of wetlands with the entire basin restored via sediment removal, all wetlands with this restoration technique were used in this study. In 2008, 1 of the restored wetlands was temporary, 6 were seasonal, and 4 were semi-permanent. In 2009, 1 was temporary, 7 were seasonal, and 3 were semi-permanent. Within the RWB, temporary and seasonal wetlands function similarly and are often grouped together as one class.

Agricultural wetlands were surrounded by crop production on at least two sides of the wetland. All sites had upland sediments covering hydric soils (Smith et al. 2011b) and were similar in size to reference wetlands (Appendix A). In 2008, 2 of the cropland wetlands were temporary, 6 were seasonal, and 3 were semi-permanent. In 2009, 3 were temporary, 5 were seasonal, and 4 were semi-permanent.

Field Studies

We surveyed the vegetation at each wetland once a month from June-August to account for cool- and warm-season species occurrence, high species turnover, and hydrologic variability (Smith and Haukos 2002). Vegetation was sampled using step-point sampling (Bonham 1989) along two parallel transects to determine plant-species occurrence. Transects ran the length of the longest basin axis, usually northwest to southeast, starting and ending at the basin edge and passing through the center of the wetland. Smith and Huakos (2002) showed that species richness is not correlated with playa size. However, to account for playa size, we generated species accumulation curves. Species accumulation curves indicated that 400 steps were sufficient in encountering 90% of the species present at each wetland site. All wetland sites contained a minimum of 400 steps. Smith and Huakos (2002) showed that species richness is not correlated with playa size. Water depth was measured at 10 random locations along each vegetation transect where water was encountered. Water depth was measured to the nearest centimeter and averaged for each wetland. In 2008, all sampled wetlands contained water during the growing season. In 2009, 4 reference, 5 restored, and 1 cropland wetland contained water during the growing season, the rest were dry.

Nomenclature followed The Flora of Nebraska (Kaul et al. 2006) and plants were classified as perennial or annual and as exotic or native based on the Flora of the Great Plains (1991) and USDA PLANTS data base (USDA & NRCS 2010). Species descriptions from the Flora of the Great Plains (1991) were used to place plants with biennial life history modes into either an annual or perennial category. Each plant was assigned a region 5 (Central Plains) wetland indicator status according to the USDA PLANTS database (USDA 2010.) We classified “species of management concern” as exotic species plus the native species, *Phalaris arundinacea* (reed canarygrass) and *Scirpus fluviatilis* (three-stem river bulrush). *Phalaris arundinacea* and *S. fluviatilis* can form dense monotypic stands and are actively removed in the RWB through grazing and disking. “Species of management concern” was primarily composed of *Phalaris arundinacea, Typha angustifolia,* and *Scirpus fluviatilis.*

Monthly precipitation records for the 2008 field season were recorded from September 1, 2007 – August 30, 2008 and from September 1, 2008 – August 30, 2008 for the 2009 field season from the Nebraska Rainfall Assessment and Information Network (NeRain 2010). Precipitation totaled 103.17 cm in 2008 and 57.53 in 2009; the 20 year average for the area is 68.68 cm of precipitation per year.

Statistical Analysis

The 2008 and 2009 data were analyzed independently due to differences in precipitation. Plant community proportions (annuals, perennials, natives, exotics, and species of management concern) was determined by dividing the number of steps encountered for a particular characteristic by the total number of steps encountered for each wetland. For each plant species, the maximum frequency from the 3 sampling periods (June, July, August) within each year was retained for analysis (Hickman et al. 2004).

Kolmogorov-Smirnov tests were used to determine if species richness, number of annuals, perennials, natives, exotics, and the proportion of annuals, perennials, natives, exotics, and species of concern among wetland land use treatments were normally distributed. If the data were normally distributed, analysis of variances (ANOVA) were used to compare factors (e.g., species richness) among wetland land use treatments (Smith and Haukos 2002). If an ANOVA factor (e.g., species richness) was significant ($P<0.05$), a Scheffe test was performed to determine differences between groups. If the data was not normally distributed, a Kruskal-Wallis one-way analysis of variance was used to compare factors (i.e., proportion of annuals) among wetland land use treatments. If significant, a post-hoc test was used to determine differences between groups. χ^2s were performed to determine differences in wetland type (e.g., semi-permanent, seasonal, temporary) among land use treatments sampled in 2008 and 2009.

Canonical Correspondence Analysis (CCA) (Palmer 1993) was used to examine relationships between plant species and wetland treatments. Results of the CCA were plotted using biplot scaling, rare species were downweighted, and a Monte Carlo permutation, using 999 permutations, was used to identify axis with significant eigenvalues and species-environment correlations.

A regression was performed on the 2008 and 2009 field season to determine if there was a relationship between age since restoration and species richness. Restored sites were also categorized as newly restored (2-5 years in 2008 and 1-6 years in 2009) or old restored (6-11 years in 2008 and 7-12 years in 2009) and an ANOVA was performed to determine differences in species richness between restored age groups.

RESULTS

Plant Community and Composition Characteristics

2008

Species richness differed among land use treatments ($F_{2,33} = 30.03$, $P < 0.001$) (Table 1.1). Species richness in reference and restored wetlands were similar, but species richness in both land use types was higher than in agricultural wetlands. The number of annuals ($F_{2,33} = 5.28$, $P = 0.01$), perennials ($F_{2,33} = 25.41$, $P < 0.001$), and native ($F_{2,33} = 30.21$, $P < 0.001$) species differed among land use treatments. The numbers of annual, perennial, and native species were similar in reference and restored wetlands, but the number of annual, perennial, and native species in both land use types was higher than in agricultural wetlands. There was no difference in the number of invasive species among land use treatments ($F_{2,33}$ – 4.68, P = 0.10). There was no significant difference in the composition of annuals (K-W = 0.5953, P = 0.74), perennials (K-W = 2.79, P = .25), natives (F2,33 = 1.31, P = 0.28), and invasives ($F_{2,33}$ = 4.681, P = 0.10) among land use treatments. Species of management concern differed among land use treatments (K-W = 12.52, P = (0.002). Species of management concern in reference and restored wetlands were similar, but species of management concern in agricultural wetlands was over 2 times higher than reference wetlands and 3 times higher than restored wetlands. There were no differences in average water depth ($F_{2,33}$ = 1.14, P = 0.333) and maximum water depth ($F_{2,33}$ = 0.80, P = 0.458) among land use treatments (Table 1.2). There was no difference in wetland type sampled among land use treatments (χ^2 = 2.97, df = 4, P = 0.563).

2009

Species richness differed among land use treatments ($F_{2,33}$ = 12.37, P < 0.001) (Table 1.3). Species richness in reference and restored wetlands were similar, but species richness in both land use types was higher than in agricultural wetlands. The number of annuals ($F_{2,33}$ = 9.04, P < 0.001), perennials ($F_{2,33}$ = 9.59, P < 0.001), and native ($F_{2,33}$ = 15.10, P < 0.001) species differed among land use treatments. The numbers of annual, perennial, and native species were similar in reference and restored wetlands, but the number of annual, perennial, and native species in both land use types was higher than in agricultural wetlands. There was no difference in the number of invasive species among land use treatments ($F_{2,33}$ = 2.58, P = 0.0922). The composition of annuals ($F_{2,33}$ = 9.84, P < 0.001) and perennials ($F_{2,33}$ = 4.96, P = 0.01) differed among land use treatments. The composition of annuals and perennials in reference and agricultural wetlands were similar, however, restored wetlands has a greater composition of annuals and a decreased composition of perennials than reference and agricultural wetlands. There was no difference in the composition of native (K-W = 6.19, P = 0.05) and invasive (K-W = 5.99, P = 0.05) among land use treatments. Species of management concern differed among land use treatments ($F_{2,33}$ = 7.5, P = 0.002). Species of management concern in reference and restored wetlands were similar, but species of management concern in agricultural wetlands was over 2.5 times higher than reference wetlands and 2 times higher than restored wetlands. There were no differences in average water depth ($F_{2,33}$ = 01.06, P = 0.357) and maximum water depth ($F_{2,33}$ = 1.78, P = 0.185) among land use treatments (Table 1.2). There was no difference in wetland type sampled among land use treatments (χ^2 = 4.95, df = 4, P = 0.292).

Associated Communities: results from CCA

2008

Axis one, accounted for 7.5% of the variation between vegetation and land use treatments (*F* = 2.49 ; *P* = 0.002) (Fig. 1.1). Axis two, accounted for 3.9% of the variation. Land use treatments explained 35% of the variation in species composition. Assuming that restoration of wetlands progresses in a linear path, restored wetlands do not appear to be on a trajectory to reach reference wetland status/ condition. Reference, restored, and agricultural land use wetlands are associated with differing plant species as well as differing plant guilds. Reference wetlands are highly associated with wet prairie perennials such as *Leersia oryzoides* (ricecut grass), *Vernonia fasciculate* (prairie ironeweed), *Poa pratensis* (Kentucky bluegrass) and deep emergent perennials such as *Schoenoplectus acutus* (hardstem bulrush) and *Schoenoplectus heterochaetus* (softstem bulrush). Restored wetlands are associated with mudflat annuals such as *Coreopsis tinctoria* (golden tickseed), *Ambrosia grayi* (woollyleaf bur ragweed), *Hordeum jubatum* (foxtail barley), submergents such as *Ceratophyllum demersum* (coontail), *Potamogeton nodosus* (longleaf pondweed), and shallow emergent perennials *Eleocharis palustris* (common spikerush), *E. erythropoda* (bald spikerush) and *Bacopa Americana* (disk waterhyssop). Agricultural wetlands are associated with 3 species of management concern *Typha angustifolia* (narrowleaf cattail) a deep emergent perennial, and *Scirpus fluviatilis* (river bulrush), and *P. arundinacea* (reed canarygrass) shallow emergent perennials.

2009

Axis one, accounted for 6.8% of the variation between vegetation and land use ($F = 2.26$, $P = 0.004$) (Fig. 1.2). Axis two, accounted for 2.6% of the variation. Land use treatments explained 31.3% of the variation in species composition. As with the 2008 CCA, assuming that restoration of wetlands progresses in a linear path, restored wetlands do not appear to be on a trajectory to reach reference wetland status/ condition. Reference, restored, and agricultural land use wetlands are associated with differing plant species as well as differing plant guilds. Reference wetlands are highly associated with wet prairie perennials such as *Leersia oryzoides* (ricecut grass), *Vernonia fasciculate* (prairie ironeweed), and deep emergent perennials such as *Schoenoplectus acutus* (hardstem bulrush) and *Schoenoplectus heterochaetus* (softstem bulrush). Restored wetlands are associated with mudflat annuals such as *Echinochloa crus-galli* (barnyard grass), *Coreopsis tinctoria* (golden tickseed), *E. acicularis* (needle spikerush), *Hordeum jubatum* (foxtail barley), *Amaranthus* rudis (redroot amaranthus) and a shallow emergent perennials *E. compressa* (common spikerush). Agricultural wetlands are associated with 3 species of management concern *Typha angustifolia* (narrowleaf cattail) a deep emergent perennial, and *Scirpus fluviatilis* (river bulrush), and *P. arundinacea* (reed canarygrass) shallow emergent perennials.

Age since restoration

There was no association between restoration time and species richness for 2008 ($F = 0.18$, $P = 0.68$) and 2009 ($F = 1.43$, $P = 0.26$). In 2008, newly restored wetlands averaged 42 species and older restored wetlands averaged 45 ($F = 0.43$, $P = 0.53$). In 2009, newly restored wetlands averaged 43 species (reference for the same year averaged 40 species) and older restored wetlands averaged 57 species. There was no difference in species richness among newly restored and older restored wetlands ($F = 2.56$, $P = 0.14$).

DISCUSSION

Sediment removal along with passive revegetation in the RWB does not support the efficient-community hypothesis that restored wetlands will resemble reference conditions following restoration. Restored wetlands within the RWB had similar plant community characteristics (e.g., species richness) compared to reference wetlands (except for the number of annuals, and composition of annuals and perennials in 2009), however the plant guilds and species associated with restored wetlands differ from those found at reference wetlands. These results indicate that examining restoration success based solely on plant community characteristics (e.g. species richness, proportion of natives) may not be the best way to examine relative differences or similarities between restored and reference conditions because, restored wetlands seldom reach reference wetland status outside general plant community characteristics (Seabloom and van der Valk 2003; Gutrich et al. 2009).

Within the Great Plains, annual precipitation is highly variable and has profound effects on wetland hydroperiod and therefore wetland plant communities (Smith and Haukos 2002). However, in the RWB, regardless of differences in rainfall among years (2008 had nearly twice the precipitation compared to 2009), the plant guilds and species associated with each land use treatment did not vary. Restored wetlands are highly associated with mudflat annuals and are missing wet prairie and deep emergent perennials that are associated with reference wetland conditions, similar to results found by Galatowitsch et al 2006 in prairie potholes. This may suggest that precipitation has little association with the differing plant guilds and species found between reference and restored wetlands within the RWB (or water depth since there was no difference among land use treatments) and other factors such as perennial plant species dispersal limitations (O'Connell et al. 2011), limited seed bank availability following sediment removal (see chapter 2), age since restoration, and/or management activities may be driving the differences in plant guild and species difference between restored and reference wetlands.

In many recently restored wetlands, species richness and diversity is often higher than in reference wetlands (Gutrich et al. 2009) consistent with our findings for the RWB in 2009. This may be attributed to wetland habitats present. A greater number of habitats present should correspond to an increase in species richness (MacArthur and Wilson 1967; Rosenzweig 1995). Up to five different plant zones can be found within playas of the RWB (Gilbert 1989), however, playas in cropland typically had only one or two zones present (based species associated with RWB wetland habitats (Gilbert 1989)). Loss of plant zonation in wetlands situated within a cropland landscape can be attributed to wetland sedimentation (Gleason and Euliss 1998). For example, increased sediment loads can decrease playa volume, spreading water over the landscape and decrease the hydroperiod (Luo et al. 1997). This results in a loss of wetland zonation and a decrease in the number of possible wetland plant species present. In addition, nutrients carried in by sedimentation can increase species such as *P. arundinacea*, that can exclude other species from becoming established.

An area of concern regarding wetland restoration is that these sites may be more susceptible to reinvasion by exotic species. Native plant diversity may have little influence on initial exotic species establishment in recently restored wetlands, because habitats most suitable for native species establishment may also provide conditions most suitable for invasion (Mathews et al. 2009b). For example, the removal of sediment can create hydroperiods similar to reference wetlands, however the soil disturbance created through sediment removal causes a disturbance regime suitable for exotics. In the RWB, restored wetlands contained the least coverage of exotic species as well as the greatest ratio of native to exotic species between the three land use treatments in 2008. However, in 2009, restored wetlands contained more coverage of exotics than reference wetlands and 4 times the coverage of exotics from the previous year, but still contained the greatest native to exotic species ratio among all land use treatments. Discrepancies between years for restored wetlands may partially be due to the amount of *E. crus-galli*. There was an 11% increase in the coverage of *E. crus-galli* from 2008 to 2009 in restored wetlands. In addition, *E.crus-galli* accounted for over 81% of exotic cover in restored wetlands in 2009. *Echinochloa crus-galli* is federally listed as an exotic species for this region (USDA 2010); however, wetland managers in many areas promote the growth of this species through moist soil management for waterfowl. When we examined species of management concern, (exotic species coverage along with coverage of *S. fluviatilis and P. arundinacea*) there was no difference between reference and restored wetlands and agricultural wetlands contained 3 times the coverage in 2008 and 2 times the coverage in 2009 compared to restored wetlands; agricultural wetlands contained 4 times the amount compared to restored wetlands, in 2009, if *E. crus-galli* is not considered exotic. This may indicate that sediment removal is removing species of management concern (Constance et al. 2007), exotic species are not persisting in the seed bank (see chapter 2), or these species were not present in the seed bank prior to sedimentation (see chapter 2).

Unlike plant community characteristics that can be highly variable between years, CCA results depict restored wetlands having plant guilds and species independent of reference and agricultural land use for both years. Constrained ordination possibly provides a more consistent way in determining if restored wetlands resemble reference wetland conditions. In addition, if we assume that restoration continues on a linear path towards reference condition, restored wetlands do not appear to be on a trajectory to reach reference wetland status. The differing plant guilds and species among land use treatments may be attributed to several causes:

Reference wetlands may not be indicative of historic conditions. The Rainwater Basin is heavily cultivated (USDA 2002) and there likely have been anthropogenic impacts that have occurred in reference wetlands. The HGM protocol used to select reference wetlands takes into account the best functioning wetlands which removes the difficulty in defining wetlands that resemble presumed pre-settlement conditions (White and Walker 1997). In this case, restored wetlands may be more representative of historic conditions rather than reference wetlands that were chosen based on functioning ability rather than solely on plant communities, but further investigation is needed. However, with reference wetlands having an established perennial plant community and a low composition of exotics, reference wetlands not being indicative of historic norms is most likely not the case (see below).

Restored wetlands have not had the time to develop perennial plant communities associated with reference conditions. With wetland mitigation projects, 3 – 5 year monitoring periods are usually used to monitor vegetation success (Mitsch and Wilson 1996, Breaux and Serefiddin 1999), especially in terms of species richness. However, this time frame may not be adequate for restored wetlands to develop the perennial guilds and species associated with the reference conditions (Mitsch and Wilson 1996). If we use Pianka (1970) loosely associated scheme that annuals are r-selected and perennials are k-selected, restored wetlands also contain a greater composition of r-selected species. r-selected species produce greater seed densities and have better dispersal mechanisms than most k-selected species and are thus more easily dispersed. With a lack of k-selected species associated with restored wetlands, this may indicated that restored wetlands have not had time to develop k-selected plant species that are associated with reference wetlands, k-selected species are not reaching restored wetlands, or that r-selected species are preventing k-selected species from becoming established in numbers to reduce r-selected species (Pollock et al. 1998). In addition, lack of anthropogenic disturbance (Hobbs and Huenneke 1992) within reference wetlands may make establishment of exotic (r-selected) species difficult; since K-selected species are superior competitors in crowded niches making it difficult for exotics and r-selected species to become established.

Species richness in restored sites has been shown to peak within the first few years after restoration (Campbell et al. 2002) and often exceeds reference wetlands (Mathews et al. 2009a). However, our results indicate that restored wetlands are most similar to reference condition, in terms of species richness, within the first 6 years following restoration. Older restorations (7-12 years in age) contained, on average, 16 more species than reference wetlands. These results indicate that short-term, rapid monitoring can provide results that are not indicative of the long term response of the wetland (Mitsch and Wilson 1996).

Restored wetlands are not on a reference wetland trajectory. Following sediment removal, the buried seed bank prior to impact can aid restoration (Weinstein et al 2001); however restored wetlands may be developing differing plant communities than those found at reference wetlands, similar to results found by Campbell et al. (2002). CCA results depict that restored wetlands of the RWB are not acting as intermediates (if restoration follows a linear path) between reference and agricultural land use conditions, indicating that restored wetlands may never reach reference land use conditions in terms of plant species composition. This may be the result of seed availability within the seed bank following restoration. Within the RWB, very few plants (97 in 450, 7.62 cm diameter soil cores) germinated after a foot of sediment had been removed (see chapter 2) indicating that most plant arrive at restored wetlands via dispersal mechanism

(Galatowitsch and van der Valk 1996). However, dispersal limited perennial plant species may not be reaching restored wetlands within the RWB (O’Connell et al. 2011). This is likely the result of restored wetlands being situated within a heavily cultivated matrix, playa wetlands being hydrologically isolated from other wetlands, and few reference (standard) wetlands remaining (O’Connell et al. 2011). With all of these factors, obtaining species associated with reference conditions may prove difficult (Seabloom and van der Valk 2003) and reseeding may be needed to recover missing guilds.

Management activities are preventing restored wetlands from reaching reference status.

Many restored and reference wetlands of the RWB are often periodically managed through grazing (Davis and Bidwell 2008), artificial flooding, prescribed burning (Brennan et al. 2005), and/or disking (Davis and Bidwell 2008). These management activities play a role in determining the plant communities present by eliminating non-fire tolerant vegetation, reducing species coverage of plants most palatable through grazing, providing niches where less competitive species can establish, and promoting increased species richness by eliminating monotypic stands of vegetation. However, reference and restored wetlands may respond differently to these management activities. For example, reference wetlands that have established perennial plant communities, many management activities help to reduce monotypic species cover. In restored wetlands where perennial plant species (or guilds) may not have become established or are in the beginning stages of becoming established, management activities may prevent perennial plant species establishment by providing open niches for annual plant species. In addition, activities such as disking followed by flooding can eliminate the germinability of perennial plant seeds from the seed bank. Conversely, management activities can also increase species richness and reduce exotic cover (Strykstra et al 1996) and long term restoration success likely depends on these management practices (Klimkowska et al. 2007).

CONCLUSION

The efficient-community hypothesis is not supported in restored playas of the RWB. Restored and reference wetlands within the RWB are associated with differing plant species and guilds. Restored wetlands are dominated by mudflat annuals and are missing the wet prairie and deep emergent perennials that are associated with reference wetlands. However, restored wetlands may never establish the plant communities associated with reference wetland conditions due to a heavily fragmented landscape, hydrologic isolation, and poor dispersal ability of some perennial plant species. Reseeding may be needed to establish missing guilds in restored wetlands if perennial seeds are not being dispersed from reference to restored wetlands or are not available within the seed bank following initial restoration. In addition, when comparing wetland plant community characteristics (e.g., species richness), results can vary from year to year based on climatic conditions. However, the plant species and guilds associated with wetland land use types remained constant between years.

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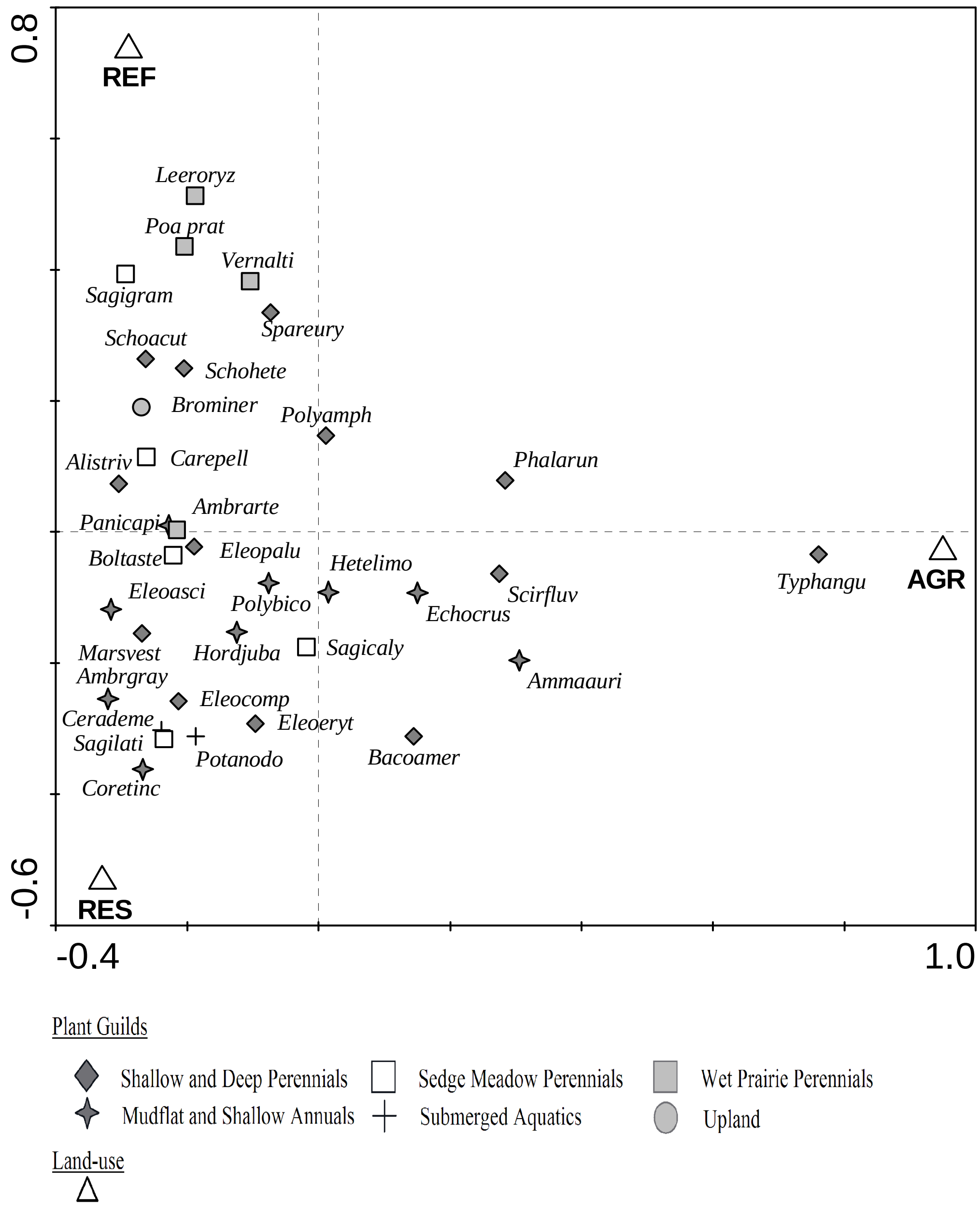

Figure 1.1 Canonical Correspondence Analysis (CCA) biplot of 2008 plant species and wetland land use treatments for Rainwater Basin playas. Inclusion of only species that occurred in at least three percent abundance are shown. Abbreviations: REF, reference wetlands; RES, restored wetlands; AGR, wetlands situated in an agricultural landscape. Species were indicated by the first four letters of the genus and species names respectively. Species symbols indicate guild classification.

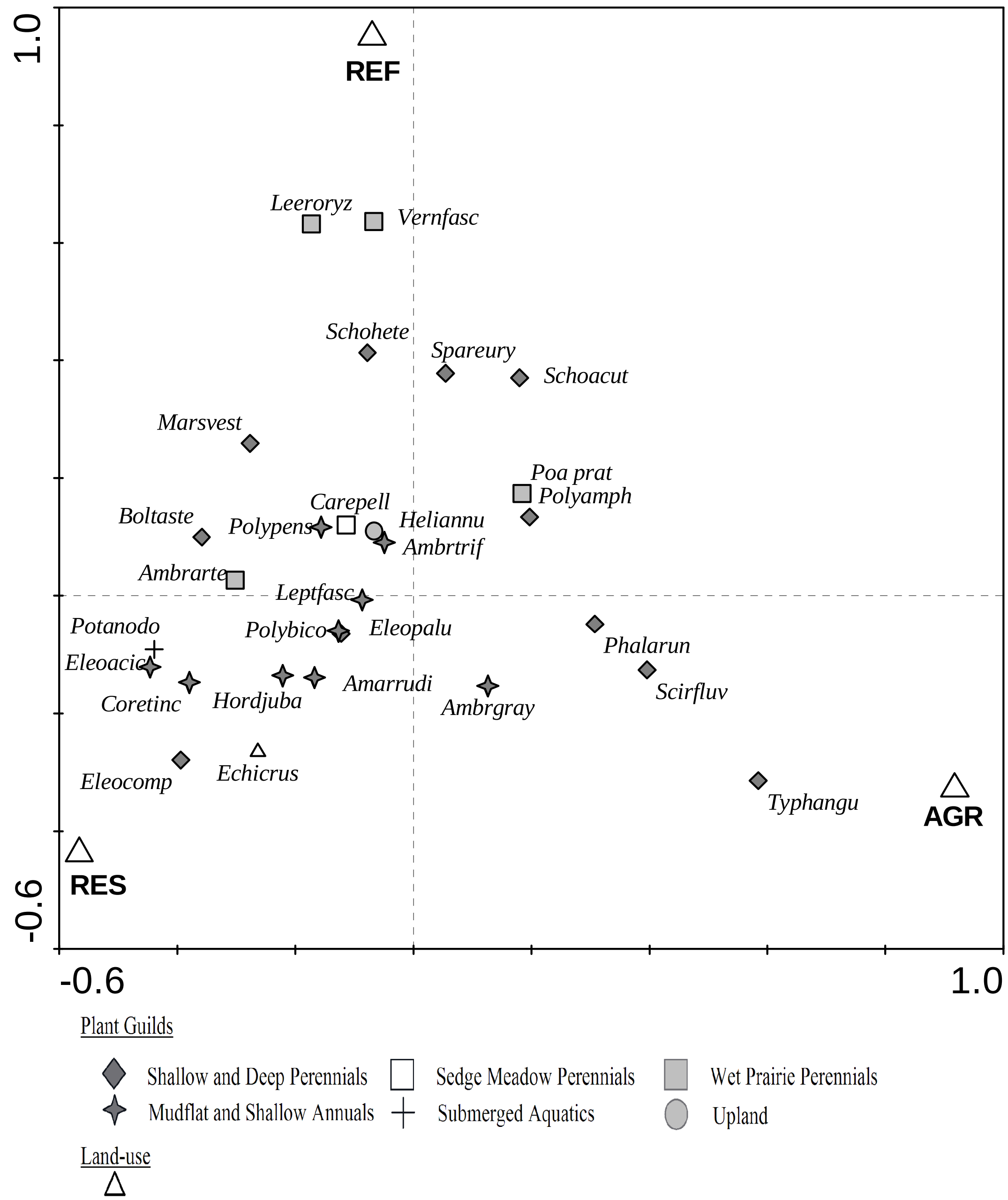

Figure 1.2 Canonical Correspondence Analysis (CCA) biplot of 2009 plant species and wetland land use treatments for Rainwater Basin playas. Inclusion of only species that occurred in at least three percent abundance are shown. Abbreviations: REF, reference wetlands; RES, restored wetlands; AGR, wetlands situated in an agricultural landscape. Species were indicated by the first four letters of the genus and species names respectively. Species symbols indicate guild classification.

Table 1.1: Plant community characteristics by land use treatments from wetlands sampled in the Rainwater Basin during the 2008 field season.

	Reference		Restored		Agriculture				
	Mean	SE	Mean	SE	Mean	SE	F-Value[1]	K-S Value[2]	P-Value
Species Richness	38.42[A]	1.53	42.82[A]	2.37	23.91[B]	1.64	30.03		<0.001
Annuals Species	10.83[A]	0.9	11.36[A]	0.86	7.64[B]	0.96	5.28		0.011
Perennial Species	27.58[A]	1.48	31.45[A]	1.72	16.09[B]	1.73	25.41		<0.001
Native Species	31.08[A]	1.27	35.73[A]	2.24	18.82[B]	1.29	30.21		<0.001
Exotic Species	7.33[AB]	0.61	7.09[B]	0.52	5.00[A]	0.8	4.22		0.02
Proportion of Annuals	0.071	0.013	0.093	0.022	0.115	0.049		0.6	0.743
Proportion of Perennials	0.776	0.044	0.689	0.045	0.699	0.069		2.79	0.248
Proportion of Natives	0.772	0.042	0.742	0.055	0.669	0.049	1.31		0.283
Proportion of Exotics	0.075	0.022	0.04	0.007	0.146	0.041		4.68	0.1
Proportion of Sp. Mgmt Concern	0.227[A]	0.051	0.163[A]	0.028	0.481[B]	0.061		12.53	0.002

[1] If data was normally distributed, an ANOVA was used.
[2] If data was not normally distributed, a Kruskal-Wallis test was used.

Table 1.2: Average water depth and max water depth by land use treatments from wetlands sampled in the Rainwater Basin during the 2008 and 2009 field seasons

	Reference		Restored		Agriculture			
2008 Sampling Season	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
Average water depth (cm)	15.39	3.41	23.78	4.3	2.68	5.39	1.14	0.3329
Max water depth (cm)	24.85	5.17	35.57	6.05	7.21	6.79	0.8	0.458
2009 Sampling Season								
Average water depth (cm)	4.73	2.57	5.72	2.55	1.53	1.59	1.06	0.3572
Max water depth (cm)	7.4	4.33	12.13	5.31	1.95	2.03	1.78	0.1854

Table 1.3: Plant community characteristics by land use treatments from wetlands sampled in the Rainwater Basin during the 2009 field season.

	Reference		Restored		Agriculture				
	Mean	SE	Mean	SE	Mean	SE	F-Value[1]	K-S Value[2]	P-Value
Species Richness	40.09^{A}	2.61	49.36^{A}	4.91	25.50^{B}	3.01	12.37		<0.001
Annuals Species	15.64^{A}	1.61	22.00^{B}	2.34	11.92^{B}	1.31	9.04		<0.001
Perennial Species	24.45^{A}	1.62	26.45^{A}	3.36	13.58^{B}	1.97	9.59		<0.001
Native Species	30.91^{A}	2.02	38.82^{A}	3.81	18.25^{B}	2.40	15.10		<0.001
Exotic Species	9.18	1.14	10.55	1.24	7.25	0.89	2.58		0.092
Proportion of Annuals	0.281^{A}	0.050	0.498^{B}	0.055	0.200^{A}	0.049	9.84		<0.001
Proportion of Perennials	0.675^{A}	0.053	0.417^{B}	0.072	0.695^{A}	0.086	4.96		
Proportion of Natives	0.899	0.018	0.745	0.077	0.734	0.075		6.00	0.050
Proportion of Exotics	0.057	0.013	0.170	0.048	0.161	0.038		6.00	0.045
Proportion of Sp. Mgmt Concern	0.207^{A}	0.046	0.244^{A}	0.046	0.453^{B}	0.059	7.500		

[1] If data was normally distributed, an ANOVA was used.
[2] If data was not normally distributed, a Kruskal-Wallis test was used.

CHAPTER II

SEED BANK REPONSES TO WETLAND RESTORATION: DO RESTORED WETLANDS RESEMBLE REFERENCE WETLAND CONDITIONS FOLLOWING SEDIMENT REMOVAL?

ABSTRACT

Sedimentation and alterations of natural hydroperiods from watershed cultivation have affected most depressional wetlands in the Great Plains. This can result in altered plant community structure through changes in water availability and depth as well as burial of seed banks. The vegetation and seed banks of 15 wetlands were sampled within the Rainwater Basin Region of Nebraska. Our objectives were to: (1) compare wetland seed bank communities among wetlands with different watershed land uses (reference, restored and impacted by watershed cultivation); (2) determine the available seed bank following sediment removal and establish if wetland zonation occurs in the deeper sediment layer of pre-scraped cropland wetlands; and (3) determine the similarity between extant vegetation and the seed banks for each wetland land use treatment. There were no significant differences in seed bank species richness and the number and composition of annual, perennial, native, or exotic species among reference, restored, and cropland playas. Restored wetlands had a greater number of upland species germinate

from the surface soil seed bank compared to reference and crop land use playas. Availability of seeds after 30 cm of soil was removed (to simulate available seed bank if the wetland was to be restored) in crop land use wetlands was low (2 - 52 seeds/ wetland) making determination of wetland zonation difficult. Reference wetlands had the highest similarity between seed bank species and extant vegetation. Sediment removal appears to be successful in removing weedy and exotic species from the seed bank; however, the seed bank is not the primary source for playa wetland revegetation. Restored and reference wetlands have similar seed bank community characteristics (i.e., richness) however, each wetland land use treatment was associated with differing plant species and plant guilds.

INTRODUCTION

Sedimentation from conversion of native grassland watershed to cropland is the largest immediate threat to the continued existence of properly functioning depressional wetlands within the Great Plains (Luo et al. 1997, 1999, Gleason et al. 2003, Tsai et al. 2007). However, recently there has been increased effort to restore depressional wetlands that provide key ecosystem services such as nutrient retention, binding of pesticides, groundwater recharge, and sites of biodiversity provisioning (Smith et al. 2011a). Restoration of these wetlands often involves sediment removal to restore the wetland’s natural hydrology (LaGrange 2010). However, the importance of existing seed banks in wetland revegetation after sediment removal has received little study.

Seed banks provide information on past vegetation (Adams and Steigerwalt 2008), distribution and relative abundance of species (Smith and Kadlec 1983, Haukos and Smith 1993), and regeneration potential (Hopfensperger 2007). The seed bank composition of wetlands, along with hydrologic conditions, natural disturbance, and management activities aid in determining the vegetation that develops each year. However, excessive sediment loads resulting from watershed cultivation can result in altered plant community structure through burial of seed banks (Jurik et al. 1994, Luo et al. 1997, Gleason et al. 2003) and changes in water availability and depth of flooding (Gleason and Euliss 1998).

Playa wetlands are the lowest point within their individual watersheds and are therefore highly susceptible to sedimentation in cultivated landscapes (Luo et al. 1997). Increased sediment loads within these wetlands bury hydric soils, reduce volume (resulting in a loss of wetland zonation), and shorten hydroperiods (Tsai et al. 2007). The decreased ponding time can result in reductions of hydric vegetation germinating from the wetland seed bank (Battaglia and Collins 2006). In addition, increased nutrients carried in with cropland sediment can promote invasive species (Zedler and Kercher 2004), many which form dense monotypic stands (i.e., *Phalaris arundinacea, Typha angustifolia*) that can prevent the penetration and germination of seeds through dense litter layers (Vaccaro 2005). This causes seed bank communities to differ greatly from the extant vegetation (During and Willems 1984).

Removal of sediment from cropland wetlands can restore the natural hydrology (Verhagen et al. 2001), lower nutrient availability (Klimkowska et al. 2007), eliminate persistent pesticides (Kiehl and Wagner 2006), remove established weedy and invasive species from the seed bank (Zedler and Kercher 2004, Bakker et al. 2005, Constance et al. 2007), remove existing established vegetation that prevents the seed bank from contributing to the development of standing vegetation (Bekker et al. 2000, Kiehl et al. 2006), and restore ecosystem function (Odum and Barrett 2005). Following sediment removal to restore wetland hydroperiod, wetlands are allowed to self-design. Wetland self-design relies on recruitment from the seed bank and natural dispersal (following restoration of hydrology and geomorphology) as two mechanisms responsible for the passive reestablishment of depressional wetland vegetation (Galatowitsch and van der Valk 1996). Self-design allows for plants to self-assemble based on the new hydrologic conditions; since hydrologic conditions primarily determine wetland plant species composition (Mitsch et al.1998, Weinstein et al. 2001). Previous studies have examined seed availability following sediment removal, but no study has shown if zonation occurs within the deeper sediment layer of impacted wetlands prior to sediment removal and restoration of hydrology. Depending on time since sediment accumulation, the persistent seed bank of impacted wetlands may exhibit remnants of vegetative zonation that has been removed due to sedimentation. This may allow us to determine the wetland zones and vegetation communities that were present prior to impact from sedimentation.

Therefore, our objectives were to: (1) compare wetland seed bank communities among land use treatments (reference, restored and impacted by cultivation) to determine if restored wetland seed banks resemble reference wetlands more than their previous cropland condition; (2) determine the available seed bank following sediment removal; and establish if wetland zonation occurs within the deeper sediment layer of post-sediment removed cropland wetlands, and (3) determine the similarity between the seed bank community and extant vegetation.

METHODS

Study Site

The Rainwater Basin (RWB) encompasses 15,907 km^2 and includes all or parts of 21 counties in the Central Loess Plains of south-central Nebraska (LaGrange 2005). The area is named for its abundant natural wetlands that formed where clay-bottom depressions catch and hold the only two inputs of water; precipitation and run-off (Stutheit et al. 2004). Annual precipitation averages 460 mm in the western portion of the region and 710 mm in the eastern portion; evapotranspiration generally exceeds precipitation (Stutheit et al. 2004). Within this region, playa wetlands are the most notable hydrogeomorphic feature on the landscape. Playas range from 0.1 ha to 1,000 ha in size (Kuzila 1984) and are defined by the presence of Massie, Scott or Fillmore soil series (Stutheit et al. 2004). The area was originally mixed-grass prairie in the western RWB and tallgrass prairie in the eastern region (Kaul 1975). Presently the region is intensively cultivated with corn and soybeans as the dominant crops and domestic livestock graze most uncultivated areas.

Fifteen wetlands were sampled among three wetland land use treatments: reference, cropland, and restored. Reference wetlands were selected using the hydrogeomorphic approach (Brinson 1993) by the Nebraska Game and Parks Commission (NGPC) and represented the most highly functioning wetlands within the region (Stutheit 2004). Reference wetlands have had no prior physical manipulation to the basin or water levels, vegetation with little to no invasive species problems, an unmanipulated watershed, and hydric soils present match wetland type (e.g., semi-permanent, seasonal, temporary). The 5 best reference wetlands from the HGM study (Stutheit et al. 2004) were selected for this study.

Cropland wetlands were surrounded by crop production on at least two sides of the wetland. All sites had upland sediments covering hydric soils (Smith et al. 2011b).

Restoration of cropland wetlands has been performed by numerous conservation partners. Each of these restoration sites was at one time impacted by cropping. Restored wetlands had an average of 30.4 cm of sediment removed from the center and then were graded out to an average depth of 10.6 - 15.2 cm around the perimeter. Following sediment removal, wetlands are allowed to self-design. Within the RWB, 13 wetlands have had sediment removal across the entire basin. From these 13 wetlands, 5 were randomly chosen for the study.

Soil Seed Bank Sampling

Soil cores were taken from 5 wetlands of each land use treatment in March 2009. At each wetland, 10, 1m^2 plots were randomly placed across the length of the wetland (basin edge to basin edge). Within each 1m^2 plot, 9, 7.62 cm diameter soil cores were taken to a depth of 5 cm for a total of 90 cores per wetland. Soil cores from each plot were homogenized (ter Heerdt et al. 1996). For cropland wetlands, an additional 10 sample plots with 9 cores per plot were taken at each wetland after 30.4 cm of soil had been removed to simulate the seed bank that would be available post sediment removal. Samples were stored at 4^{o}C prior to processing (Boedeltje et al. 2002).

Concentrating Samples

Each soil sample was handled according to the concentrated-emergence method (ter Heerdt et al. 1996). Samples were washed with water first through a coarse sieve to remove coarse debris (rhizomes, roots, and plant matter) and then through a fine 0.2 mm sieve to remove clay and silt.

Germination Experiment

Planting trays (21.6 x 30.4 cm) were filled with an equal mixture of sterilized sand and potting soil (3-4 cm deep). The sand-soil mixture was covered with 1 cm of sterilized sand to prevent algal blooms (Boedeltje et al. 2002). The concentrated seed samples from the fine sieve were divided in half and spread in a thin layer no more than 5mm thick (ter Heerdt et al. 1996) on top of the 1 cm of sand in two different planting trays. One planting tray was then placed in a submerged setting (4 cm of standing water) and the other in a moist soil setting and arranged randomly in the Oklahoma State University greenhouse to account for differing germination requirements of wetland species (Smith and Kadlec 1983).

The germination experiment was conducted from 7 January to 7 May 2010 in a controlled greenhouse with temperatures ranging from 15^{o}C to 25^{o}C, consistent with the temperature averages for the growing season in RWB NE. A 15:9 hr photoperiod was maintained throughout the germination period with 400 W sodium and metal halide overhead lights. The moist soil treatment trays were watered daily and the submerged treatments were refilled as needed to account for evaporative water loss. Seedlings were removed within one week of germination and identified. All seedlings that could not be identified within the first week of germination were transferred to separate pots and were grown until identification was possible.

Nomenclature follows Kaul et al. (2006) and plants were classified as perennial or annual and as exotic or native based on Flora of the Great Plains (1991) and USDA

PLANTS database (USDA & NRCS 2010). Each plant was assigned a region 5 (Central Plains) wetland indicator status (e.g., obligate, facultative, upland) according to the USDA PLANTS database (USDA & NRCS 2010). Plant species were placed into guilds with incorporated life history traits (annual or perennial) and water tolerance (Galatowitsch 2006; O'Connell et al. 2011) (Appendix B). Perennial guilds in order of increasing water tolerance: wet prairie, sedge meadow, shallow emergent, deep emergent, and submerged. Annual guilds in order of increasing water tolerance: mudflat annuals and shallow emergent annuals. For species not listed in Galatowitsch (1996) or O'Connell et al. (2011) we categorized them using field observations, Flora of Nebraska (Kaul et al. 2006), and life history designation.

Vegetation Sampling

Wetlands were surveyed using step-point sampling (the nearest species to the end of each 1 m step recorded; Bonham 1989) along two parallel transects to determine plant-species occurrence. Transects ran the length of the longest basin axis, usually northwest to southeast, starting and ending at the basin edge and passing through the center of the wetland. Basin edge was determined by examining changes in soil color (Luo et al. 1997) and vegetation. We surveyed each wetland once a month from June-August to account for cool- and warm-season species occurrence, high species turnover, and hydrologic variability (Smith and Haukos 2002).

Data Analysis

Objective 1: To compare wetland seed bank communities among land use treatments, we grouped plant species into obligate (OBL) and facultative wetland (FACW) categories (67%-100% probability of occurring in a wetland) as "wetland" species and facultative upland (FACU) and upland (UPL) categories (67%-100% not to occur in wetlands) as “upland” species (de Steven et al. 2006). Facultative (FAC) species were categorized as species equally likely to occur in wetland or upland habitats and were not included in “wetland” or “upland” analyses. Germinating seed density among treatments was expressed as the number of seeds per square meter in a layer of soil 5 cm thick. Separate analyses of variances (ANOVAs) were used to compare seed bank species richness, number of annual, perennial, native, exotic, wetland, upland, and FAC species and the composition of each among wetland land uses. The density of germinating seeds from each treatment (moist soil or submerged) among land use treatments were analyzed with separate ANOVAs. The 7 species (*Alisma triviale, Coreopsis tinctoria, Eleocharis palustris, Polygonum pensylvanicum, Sagittaria* spp., *Schoenoplectus tabernaemontani,* and *Typha angustifolia*) with the greatest germinating seed densities among treatments (the species had to occur in at least one sample in each land use category) were analyzed with an ANOVA. If an ANOVA factor (e.g., richness) was significant ($P<0.05$), a LS Means test was performed to determine significance among land uses.

Canonical Correspondence Analysis (CCA) (Palmer 1993) was used to examine relationships among seed bank species and land use treatments. Results of the CCA were plotted using biplot scaling, rare species were down weighted, and a Monte Carlo permutation, using 999 permutations, was used to identify axes with significant eigenvalues and species-environment correlations.

Objective 2: To determine the available seed bank following sediment removal and establish if wetland zonation occurs within the deeper sediment layer of post-sediment removed cropland wetlands, a χ^2 was used to determine differences in frequency of wetland (OBL and FACW) and upland (FACU and UPL) germinating plants by wetland zone for the deeper sediment layer (30.2 cm) of agricultural wetlands. We divided the wetland into three zones: zone 1 corresponded with the transition and outer marsh zone, zone 2 with the persistent emergent zone, and zone 3 with the inner marsh zone (Gilbert 1989).

Objective 3: Sorenson index was used to calculate the similarity between the seed bank community and extant vegetation among land use treatments and for similarity between agricultural land use pre- and post-agricultural sediment removal (Hopfensperger 2007). An ANOVA was used to compare Sorenson index scores among land use treatments.

RESULTS

Germinating Plant Community Characteristics (Seed Bank)

There were no differences in species richness, the number of annual, perennial, native, and exotic species among wetland treatments (Table 1). There was no difference in the proportion of germinating annuals, perennials, native, and exotic plants among land use treatments. Restored and reference wetlands had similar numbers of upland plant species germinate from the seed bank, however, restored wetland had significantly more germinating upland species than cropland wetlands ($F_{2,14} = 4.19$, $P = 0.04$). Restored and reference wetlands had a similar proportion of facultative plants from the seed bank, however, restored wetland had significantly more facultative plants germinate from the seed bank than cropland wetlands ($F_{2,14} = 4.19$, $P = 0.04$) (Fig. 2.1). There was no difference in the number of wetland species and the proportion of germinating wetland or upland plants among land use treatments.

There was no difference among wetland treatments in the density of seeds from the moist soil or the submerged treatments (Table 2.1). *Eleocharis palustris* had the highest germinating seed densities among all three land use treatments (Table 2.2). There was no difference in the density of *A. triviale* ($F = 3.40$, $P = 0.07$), *C. tinctoria* ($F = 1.48$, $P = 0.27$), *E. palustris* ($F = 0.57$, $P = 0.58$), *P. pensylvanicum* ($F = 2.12$, $P = 0.16$), *Sagittaria* spp ($F = 0.94$, $P = 0.16$), *S. tabernaemontani* ($F = 0.67$, $P = 0.53$) and *T. angustifolia* ($F = 2.01$, $P = 0.18$) among land use treatments. *Polygonum amphibium* was a common species found in the vegetation of reference and restored wetlands but was not represented within their seed banks (Table 2.3). *Scirpus fluviatilis* was not detected in the seed banks of any land use treatment and *Phalaris arundinacea* seed bank densities were low compared to the presence of *P. arundinacea* found in the standing vegetation of cropland wetlands.

CCA Results

Axis one accounted for 17.6% of the variation between seed bank species and land use treatment ($F = 2.558$; $P = 0.001$) (Fig. 2.2). Axis two accounted for 4.8% of the variation between seed bank species and land use treatment. Land use treatments explained 60.9% of the variation in species composition. Restored wetlands do not appear to be intermediates between reference and cropland wetlands as different plant species and guilds were associated with each land use treatment. Restored wetlands were associated with mudflat annuals such as *Ambrosia grayii*, *Chenopodium leptophyllum*, *Lepidium densiflorium*, and *Polygonium ramosissium*; reference wetlands were associated with shallow emergent perennials such as *Sparganium eurycarpum* and *Eleocharis erythropoda*, and wet prairie perennials such as *Leersia oryzoides*; cropland wetlands were associated with a deep emergent invasive perennial, *Typha angustifolia*, a mudflat annual, *Erechtites hieraciifolia*, and wet prairie perennials *Polygonium amphibium* and *Eleocharis compressa*.

Available Seed Bank and Wetland Zonation

Only 97 plants comprising 14 species germinated from the deeper sediment layer of cropland wetlands, 47 times less (seed germinations) than the upper impacted layer. Of the 97 individuals, 40% were *E. palustris* and 20% were *Schoenoplectus tabernaemontani* (Table 2.2). There was no difference in the frequency of germinating wetland plants or upland plants among the three wetland zones (transition/outer marsh zone, emergent zone, inner marsh zone) ($\chi^2 = 1.43$, df = 2, P = 0.4869).

Sorenson similarity index comparisons

There was no difference in similarity between the seed bank and extant vegetation among the three land use treatments (F = 1.28, P = 0.3159) (Table 2.4). All species found in the seed bank were observed in extant vegetation surveys. There was moderate similarity (48%) between the exposed and deeper (30 cm) sediment layer of agricultural wetland seed banks.

DISCUSSION

Previous studies have shown that 3-5 years post-restoration is not long enough to measure the restoration success of a wetland (Mitsch and Wilson 1996, Breaux and Serefiddin 1999, NRC 2001), however, out study indicates this may be attributed to the wetland plants that are associated with reference wetlands are not found within the seed bank of restored wetlands. Furthermore, basing restoration success solely on plant community characteristics (e.g., richness) may not be the best approach in evaluating restoration success since the plant communities associated with restored wetlands seldom resemble reference conditions (Seabloom and van der Valk 2003; Gutrich et al. 2009) without additional input such as reseeding. Even though sediment removal along with passive revegetation in the RWB can establish plant and seed bank communities that have similar overall community metrics (e.g., richness) to reference conditions, the plant species and guilds associated with restored wetlands differ from the plant species and guilds associated with reference conditions (de Steven et al. 2006; O'Connell et al. 2011).

Sediment removal of agriculturally impacted wetlands in the RWB appears to remove most seeds of strong competitors and/or invasive species from the seed bank. However, deeper soil layers often contain little viable seed for plant recolonization (Jensen 1998). Therefore, like prairie pothole wetlands, playas may rely primarily on seed dispersal from local wetlands and transport by waterfowl and shorebirds to re-establish plant populations (Galatowitsch and van der Valk 1996) if reseeding does not follow restoration. However, seed dispersal of perennial plant species between reference and restored wetlands is likely limited with the RWB being a heavily fragmented landscape due to agriculture (Kocer 2004, Webb et al. 2010), playa wetlands being hydrologically isolated (Smith 2003), few intact reference wetlands remaining on the landscape, and limited seed dispersal ability of these plants (O'Connell et al. 2011). With over 12 million migrating waterfowl, geese, and shorebirds using the RWB during spring migration (Bishop and Vrtiska 2008) avian dispersal may aid in dispersal of perennial plant species associated with reference conditions. However, the quantity of seeds dispersed to restored wetlands from reference wetlands via avian dispersal is likely limited especially when cropland playas are the dominant hydromorphic feature on the landscape and few true reference wetlands remain.

Within the RWB, most restored wetlands are periodically managed (e.g. mowing, grazing) to decrease the abundance of invasive species such as *Typha* and *Phalaris arundinacea* and during drier years, some cropland wetland basins are cultivated. Cultivation (Smith et al. 2002), mowing (Reine et al. 2004), and cattle grazing (Sternberg et al. 2003) can diminish soil seed banks. Even though there was no difference in seed density among the three land use treatments in our study, cultivation and management practices can affect species richness and numbers of seeds within the seed bank (Cardina et al. 1991). The one cropland wetland in our study that was cultivated through the basin had the lowest species richness of all wetlands in the study. The two restored wetlands that were grazed heavily by cattle the previous year averaged 3 less species and over 3 times less germinating plants compared to the other three restored wetlands. Intensive grazing regimes in these wetlands may have prevented many plants from reproducing by seed. In contrast, seed density can be positively correlated with wetland management (Thompson 1978, Haukos and Smith 1993). One reference wetland in our study is lightly grazed by horses annually. This wetland contained the greatest density of seeds among all restored and reference wetlands and is considered the most highly functioning wetland within the RWB. Even though management practices (such as cattle grazing) may lower seed production, management activities can increase standing species richness, reduce monotypic vegetation (Tesauro 2001, Kotowski and van Diggelen 2004), and accelerate vegetative succession (Strykstra et al. 1996). Thus, long term wetland restoration success likely depends on these management practices (Klimkowska et al. 2007).

The length and depth of inundation of wetlands determines the type of species that occur at a wetland (Keddy 2000). Although rainfall is greater and evapotranspiration rates are lesser in RWB playa compared to Southern High Plains (SHP) playas, playas from both regions exhibit similar plant community characteristics. Not unlike vegetation of reference playa wetlands of the SHP (Huakos and Smith 1993; O'Connell et al 2011b), RWB reference seed banks had a greater proportion of germinating perennials compared to annuals. Also, RWB restored wetland seed banks appear to resemble extant vegetation characteristics that are similar to Conservation Reserve Program (CRP) enrolled wetlands in the SHP and Wetland Reserve Program (WRP) playa wetlands in the RWB. Restored wetlands of the RWB, CRP playas of the SHP, and WRP playas in the RWB all had similar annual to perennial seedbank/extant plant proportions and more upland plants occurring within the wetlands compared to reference and cropland wetlands from their respected regions (O'Connell et al. 2001). Rainwater Basin playa wetlands also have vegetation characteristics similar to prairie pothole wetlands with the high numbers of perennial species that germinated from the seed bank (especially from reference wetlands) (Galatowitsch and van der Valk 1996).

Restored wetlands in our study had a similar proportion of germinating annuals and perennials from the seed bank, whereas reference wetlands had a 1:3 ratio of annuals to perennials and cropland wetlands had a 1:2 ratio germinate from the seed bank. This may indicate restored wetlands have not had the time to fully recover the perennial species that reference wetlands may contain (Mitsch and Wilson 1996) or that disturbance from restoration or management regimes (such as disking) are creating conditions more suitable for annual species. For example, management activities (such as disking) followed by flooding can possibly eliminate any perennial species that may have arrived at the wetland via dispersal or were present in the seed bank and prevent the establishment of these species. However, similar proportions of germinating annual and perennial plants along with their moderate similarity scores in restored wetlands may also be attributed to a more even mixture of transient (viability <1 year) and persistent (viability >1 year) seeds within the seed bank (Hopfensperger 2007). Cropland wetlands should have had a greater proportion of annual seeds germinating from the seed bank if disturbance is a driver of species contributing transient seeds. However, the presence of monotypic stands of perennial plants may have reduced the numbers of these seeds from entering the soil column (via thick litter layer) or that these monotypic species have persisted in the wetlands long enough for transient seeds to no longer be viable.

Increased inundation leads to anoxic soil conditions selecting for wetland species, whereas decreased ponding can allow upland species to encroach the edges and spread inward during dry periods (Smith and Haukos 2002, de Steven et al. 2006). Removing 30 cm of sediment from cropland wetlands allows the wetland to pond water for a longer duration during the growing season. However, removing sediment from the center of the wetland may cause the perimeter of the wetland to dry faster, earlier, or not to be inundated. This may account for the increased number of upland species and 20% less proportion of germinating wetland plants compared to reference and cropland wetlands. In addition, restored wetlands also contained a greater proportion of plants with no affinity for wetland or upland habitats (FAC species) than reference and cropland wetlands with 50% of these species occurring around the perimeter (transition and outer marsh zone) of the wetland. This may be attributed to the closest seed source around restored wetlands being FAC (mud flat annuals) and not FACW or OBL (shallow emergent, sedge meadow perennial, and deep emergent perrenial) species or that these FACW and OBL plants of reference wetlands not having the dispersal mechanism to reach restored wetlands (O'Connell et al. 2011a). Also, the edge of restored wetlands may be dry long enough each season to support FAC and UPL species and may not have the germination requirements that are needed for FACW species.

Large influxes of nutrients from agricultural uplands help to promote establishment of native invasive and exotic wetland species such as *T. latifolia*, *S. fluviatilus*, and *P. arundinacea*. Once established, these perennial species form thick stands that reduce sunlight penetration to the soil and reduce seedling germination (Vaccaro 2005). This can result in a reduced seed bank contribution to the extant vegetation as well as reduced extant vegetation species richness (Bekker et al. 2000). This may account for cropland wetlands reduced similarity scored compared to reference and restored wetlands. In addition, some cropland wetlands can be tilled through in drier years, further reducing similarity scores by reducing extant vegetation species richness through the application of herbicides and the addition of monotypic crops. Though not statistically significant in our study, cropland seed banks contained 19 times the composition of germinating exotic species compared to reference wetlands and 4 times the germinating composition compared to restored wetlands. In addition, *Typha* germinated 125 times and 36 times more in cropland wetlands than restored and reference wetlands (respectively). This likely has a significant biological effect on the ecosystem such as food resources for migrating waterfowl. Within all land use treatments, the proportion of exotics that germinated from the seed bank was less than the coverage found in the extant vegetation possibly indicating the spread of these species via vegetative processes rather than seed production.

Previous studies have shown that wetland species and dominant perennial grasses and sedges are absent from the seed bank following restoration (Galatowitsch and van der Valk 1996; de Steven et al. 2006). However, in the RWB, only perennial grasses were absent from the deeper sediment layer; wetland species comprised 93% of the germinating species and 40% of germinating plants was the sedge *E. palustris*. During a 2009 survey of a recently restored wetland (less than 6 months after sediment removal) in the RWB, sedges and perennial grasses were absent, however, wetlands plants accounted for over 90% of the standing vegetation (Beas unpub). Discrepancies between the presence of sedges found in the seed bank of the deeper sediment layer and the standing vegetation of a recently restored wetland may be attributed to the environmental conditions suitable for germination of these plant guilds not being met (Haukos and Smith 1993). However, the perennial sedges (sedge meadow perennials/ shallow emergent perennials) that may be present in the deeper sediment are not establishing at the rate or quantity to associate RWB restored wetlands with these species and/or guilds.

With only 97 individual seeds germinating from the deeper sediment layer (an average of 1 seed per every 5 cores), determining whether zonation was present was difficult. The deeper sediment layer had 15 species germinate, similar to species richness of restored and reference land use treatments. This may possibly indicate that prior to agricultural practices, these wetlands had similar diversity. However, removing 30 cm of sediment is often a conservative amount to remove in the RWB. Written accounts from the 1930s indicate that more than 30 cm of sediment was observed piled next to fence posts (McMurtrey et al. 1972). With playa wetlands being the lowest points within a closed watershed, they may have experienced sediment loads exceeding those documented along fence posts. In addition, wetland zones present today at crop land wetlands most likely do not a line with their historic zones. Within the RWB, agricultural wetlands are, on average, 26 times smaller than their hydric footprint (Smith et al. 2011b) leaving only the middle of the wetland remaining. This may account for the reduction in facultative species and facultative upland species largely missing for the deeper sediment layer.

Our study has shown that sediment removal within RWB was successful in removing exotic species (e.g., *Typha angustifolia, Phalaris arundinacea, Scirpus fluviatilis*) from the seed bank and having overall seed bank community characteristics (e.g., richness) that are similar to reference wetlands. However, the seed bank species and plant guilds that are closely associated with restored wetlands vary from reference wetlands. Restored wetlands ranging in age from 3-6 years post sediment removal may not have had enough time to develop the seed bank communities of reference condition, however they may never be on a trajectory to reach reference condition. If this is the case, future restorations may need to be seeded to reestablish wet prairie and shallow emergent perennials that are missing from the seed bank of restored wetlands.

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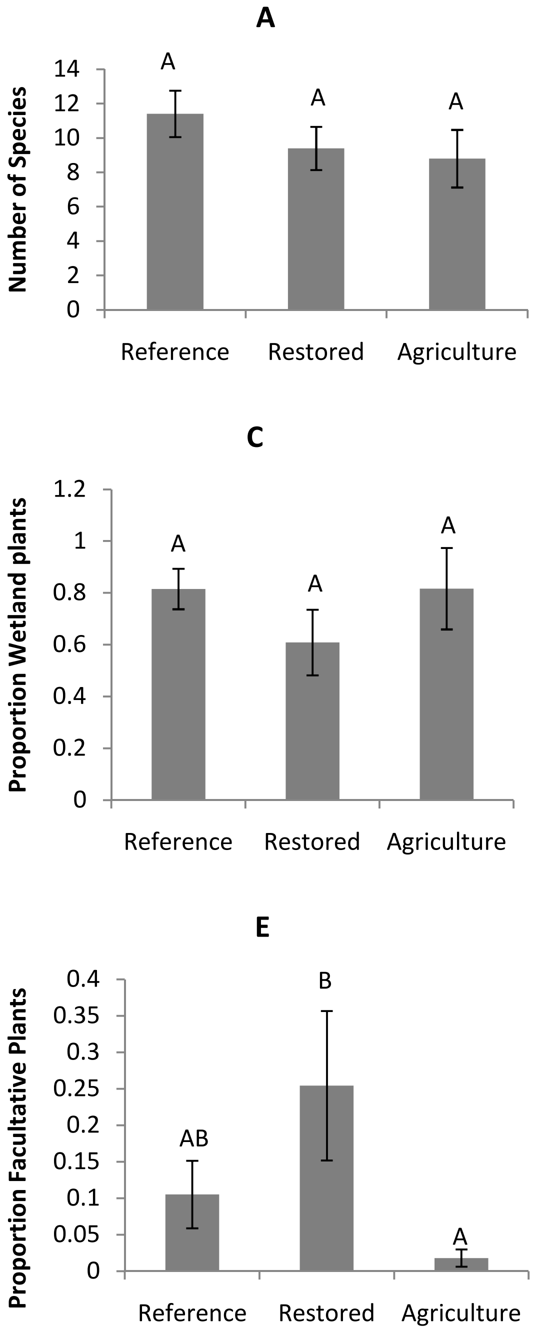

Figure 2.1 Mean wetland and upland plant characteristics among playa land use treatments in the Rainwater Basin Region of Nebraska. Wetland and upland characteristics are based on the region 5 wetland indicator status as defined by the USDA plant data base. Figure A: number of wetland species ($F_{2,14}$ = 1.12, P = 0.3578); figure B: number of upland species ($F_{2,14}$ =4.89, P = 0.0279); figure C: proportion of germinating wetland plants ($F_{2,14}$ = 15.32, P = 0.0005); figure D: proportion of germinating upland plants ($F_{2,14}$ = 4.89, P = 0.0280); figure E: proportion of germinating facultative plants ($F_{2,14}$ =4.19, P = 0.0400.

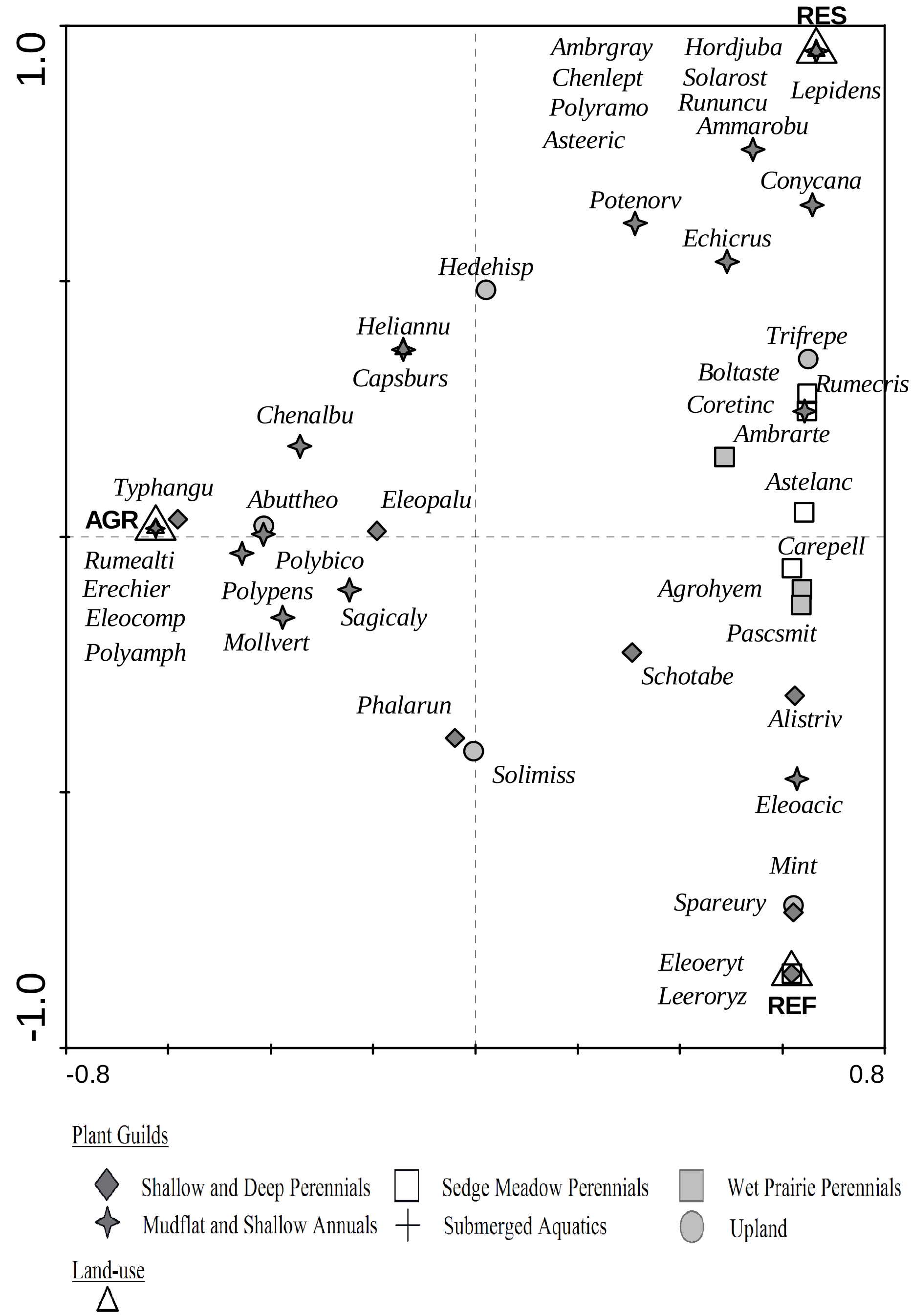

Fig. 2.2 Canonical Correspondence Analysis (CCA) biplot of seed bank species and wetland land use treatments for the Rainwater Basin playas. Abbreviations: REF, reference wetlands; RES, restored wetlands; AGR cropland wetlands. Species names were the first four letter of the genus and species (see Appendix B). Ambrgray, Chenlept, Lepidens, Polyramo, Hordjuba, Runcncu, Solarost, and Asteeric are directly behind RES land use; Leeroryz and Eleoeryt are directly behind REF land use; Polyamph, Rumealti, Erechier, and Eleocomp are directly behind AGR.

Table 2.1. Plant community characteristics from the germinating seed banks of reference, restored, and agricultural land use wetlands in the Rainwater Basin region of Nebraska. Seed density is expressed as the number of seeds per square meter. Restored wetlands range in age from 3 to 8 years post sediment removal and agricultural land use wetlands were surrounded by crop on at least three sides.

	Reference (n=5)		**Restored** (n=5)		**Agriculture** (n=5)			
Characteristics	mean	*SE*	mean	*SE*	mean	*SE*	F value	P value
Species Richness	15.80	*2.04*	17.60	*2.14*	12.60	*2.93*	1.39	0.2869
Number of Annual Species	5.40	*0.84*	9.00	*1.54*	6.60	*1.89*	1.89	0.1927
Number of Perennial Species	10.40	*1.52*	8.60	*1.25*	6.00	*1.22*	3.40	0.0677
Proportion of Germinating Annuals	0.27	*0.13*	0.52	*0.10*	0.36	*0.18*	1.00	0.3963
Proportion of Germinating Perennials	0.73	*0.13*	0.48	*0.10*	0.64	*0.18*	1.00	0.3964
Number of Native Species	13.40	*1.57*	15.00	*1.70*	10.00	*2.42*	2.18	0.1554
Number of Invasive Species	2.40	*0.84*	2.60	*0.67*	2.60	*0.76*	0.03	0.9715
Proportion of Germinating Natives	0.99	*0.00*	0.95	*0.02*	0.80	*0.10*	3.40	0.0678
Proportion of Germinating Invasives	0.01	*0.00*	0.05	*0.02*	0.20	*0.10*	3.39	0.0679
Density of Seeds from Moist-soil trt	501.10	*277.07*	560.46	*213.08*	1933.98	*1538.53*	0.99	0.4000
Density of Seeds from Submerged trt	912.57	*480.40*	657.66	*447.40*	747.87	*522.28*	0.09	0.9100

Table 2.2. Species from seed bank samples from wetlands sampled from the Rainwater Basin region, NE. Seed density was estimated from basins where each sample was detected.

	Restored		Reference		Cropland (surface)		Cropland (30 cm removed)	
Species	# of basin	Mean seed density (# seeds/m2)	# of basin	Mean seed density (# seeds/m2)	# of basin	Mean seed density (# seeds/m2)	# basin	Mean seed density (#seeds/m2)
Abutilon theophrasti	1	2.9	1	2.9	2	14.6	0	0
Agrostis hyemalis	3	29.2	2	61.2	0	0	0	0
Alisma trivial	2	128.3	5	119.6	2	2.9	1	2.9
Amaranthus retroflexus	4	56.9	3	36	4	128.3	3	3.9
Ambrosia artemisifoliia	5	30.3	4	28.4	1	37.9	0	0
Ambrosia grayi	1	5.8	0	0	0	0	0	0
Ammania robusta	4	138.5	2	19	4	15.3	0	0
Aster ericoides	1	2.9	0	0	0	0	0	0
Aster lanceolatus	1	2.9	1	2.9	0	0	0	0
Bacopa rotundifolia	2	11.7	1	14.6	3	16.5	0	0
Boltonia asteroids	2	52.5	1	67.1	0	0	0	0
Capsella bursa-pastoris	1	8.7	0	0	1	14.6	0	0
Carex spp.	4	57.6	3	98.2	2	4.4	2	2.9
Chenopodium album	3	7.8	1	2.9	2	46.7	2	2.9
Chenopodium leptophyllum	1	5.8	0	0	0	0	0	0
Conyza Canadensis	3	4.9	1	2.9	0	0	0	0
Coreopsis tinctoria	4	498.7	3	362.6	2	4.4	0	0
Echinochloa crus-galli	2	8.7	2	7.3	3	3.9	0	0
Eleocharis acicularis	1	303.3	1	1131.5	0	0	0	0
Eleocharis compressa	0	0	0	0	1	29.2	0	0

Table 2.2 (cont.)

	Restored		Reference		Cropland (surface)		Cropland (30 cm removed)	
Species	# of basin	Mean seed density (# seeds/m2)	# of basin	Mean seed density (# seeds/m2)	# of basin	Mean seed density (# seeds/m2)	# basin	Mean seed density (#seeds/m2)
Eleocharis erythropoda	0	0	1	2.9	0	0	1	2.9
Eleocharis palustris	5	326	5	294	5	1077.8	3	37.9
Erechtites hieraciifolia	0	0	0	0	1	2.9	0	0
Hedeoma hispida	1	5.8	0	0	1	5.8	0	0
Helianthus annuus	1	8.7	0	0	1	14.6	1	2.9
Hordeum jubatum	1	2.9	0	0	0	0	0	0
Leersia oryzoides	0	0	4	24.8	0	0	0	0
Lepidium densiflorum	1	2.9	0	0	0	0	0	0
Mollugo verticillata	0	0	1	2.9	1	11.7	0	0
Pascopyrum smithii	4	2.9	2	8.7	0	0	0	0
Phalaris arundinacea	0	0	3	15.6	1	52.5	0	0
Polygonum amphibium	0	0	0	0	1	40.8	0	0
Polygonum bicorne	3	7.8	1	20.4	3	87.5	2	5.8
Polygonum pensylvanicum	3	30.1	4	57.6	3	688.2	1	32.1
Polygonum ramosissimum	3	2.9	0	0	0	0	0	0
Potamogeton nodosus	0	0	2	7.3	1	5.8	0	0
Potentilla norvegica	2	43.7	2	2.9	1	35	0	0
Rorippa palustris	2	45.2	4	40.1	3	45.7	1	2.9
Rumex altissimus	0	0	0	0	1	2.9	0	0
Rumex crispsus	2	32.1	3	12.6	0	0	0	0
Runuculus spp.	1	29.2	0	0	0	0	1	2.9
Sagittaria spp.	5	28.6	5	80.5	5	252.6	3	4.9

Table 2.2 (cont.)

	Restored		Reference		Cropland (surface)		Cropland (30 cm removed)	
Species	# of basin	Mean seed density (# seeds/m2)	# of basin	Mean seed density (# seeds/m2)	# of basin	Mean seed density (# seeds/m2)	# basin	Mean seed density (#seeds/m2)
Teucrium canadense	1	5.8	1	72.9	0	0	0	0
Trifolium repens	1	5.8	1	2.9	0	0	0	0
Typha angustifolia	3	26.2	4	5.8	5	578	1	14.6
Schoenoplectus tabernaemontani	3	112.8	4	196.1	2	196.9	2	29.2
Solanum rostratum	1	2.9	0	0	0	0	0	0
Solidago missouriensis	0	0	1	2.9	1	2.9	0	0
Sparganium eurycarpum	1	2.9	1	40.8	0	0	0	0

Table 2.3. Coverage (%) of most common plant species found in the extant vegetation of each land use treatment.

	Reference		Restored		Agriculture	
Species	mean	*SE*	mean	*SE*	mean	*SE*
Ambrosia artemisiifolia	3.50	*1.08*	5.94	*2.99*	0.55	*0.55*
Echinochloa crus-galli	1.95	*1.65*	10.59	*11.25*	1.05	*0.55*
Eleocharis compressa	0.16	*0.001*	7.47	*6.87*	0.23	*0.25*
Eleocharis palustris	2.99	*1.45*	7.78	*4.24*	1.90	*1.067*
Phalaris arundinacea	6.42	*4.93*	3.02	*1.28*	12.21	*10.96*
Polygonum amphibium	25.88	*4.95*	11.38	*5.93*	22.78	*13.60*
Polygonum bicorn	5.98	*2.58*	9.16	*4.65*	5.75	*3.49*
Scirpus fluviatilis	5.82	*3.52*	3.24	*2.00*	17.58	*15.80*
Sparganium eurycarpum	4.49	*4.04*	0.24	*0.002*	0.001	*0.000*
Typha angustifolia	0.79	*0.63*	0.37	*0.36*	5.73	*4.11*
Zea mays	0.00	*0.00*	0.00	*0.00*	9.08	*9.91*

Table 2.4. Average number of species found in the seed bank, aboveground plant community, and average Sorenson's index for each land use condition. Due to a limited amount of seed germinating from the 30 cm layer of cropland wetlands, similarity between the exposed sediment layer and 30 cm sediment layer of cropland wetlands was not included.

	Average ± SE		
Land use	Seed bank	Vegetation	Sorenson's similarity index (%)
Reference	15.8 ± 2.04	42.2 ± 4.3	40.9
Restored	16.75 ± 2.55	48.0 ± 8.23	38.3
Agriculture	12.6 ± 2.93	23.4 ± 4.14	32.7

CHAPTER III

THE USE OF MODELS TO PREDICT VALUE OF RESTORING PLAYA WETLANDS ON WATERBIRD ABUNDANCE DURING SPRING MIGRATION

ABSTRACT

Spring migratory bird stopover sites are important links between wintering and breeding grounds and should provide birds the resources needed for continuing migration and reproduction. Within the Central Flyway, the Rainwater Basin Region of Nebraska provides critical stopover habitat, but 90% of the wetlands have been destroyed for agricultural practices. Of the remaining wetlands, most are situated within crop fields and have lost much of their function as migratory bird habitat. Our objective was to use models developed by Webb et al. (2010) to determine if restored wetlands, via sediment removal, passive revegetation, and installation of an upland buffer, have the potential to improve migratory waterbird. We compared comparing model predictions among reference, restored, and non-restored (cropland) wetlands. Restored wetlands within the Rainwater Basin Region were twice the size of reference and cropland wetlands and area alone predicted greater abundances of dabbling ducks, diving ducks, and species richness relative to cropland and reference wetlands in years of increased precipitation. However, when taking area into account by analyzing wetlands of similar size, there was no

significant difference in abundance of dabbling and diving ducks, shorebirds, geese, or species richness between reference, restored, and cropland wetlands. However, restored wetlands were predicted to have nearly twice the abundance of dabbling and diving ducks

as reference and cropland playas, twice as many geese, and contain 5 more species of waterbirds compared to reference wetlands. In years of low precipitation, there were no statistical differences in abundance of dabbling ducks, diving ducks, shorebirds, geese, or species richness between the reference, restored, and cropland wetlands. However, restored wetlands were predicted to have the greatest abundance of dabbling ducks, diving ducks, shorebirds, and geese among the three land use treatments. In years of low precipitation, reference and restored wetlands are the primary habitat available for waterbird use during migration because most cropland wetlands are dry. Models predict restored wetlands within the Rainwater Basin will provide improved habitat needed for migratory waterbirds during spring migrations and are most critical in drier years when upwards of 90% of cropland wetlands do not hold water.

INTRODUCTION

Migratory stopover sites provide long-distant migrants a critical link between wintering and breeding grounds by providing vital habitat that is fundamental for continuation of migration and reproduction (Myers et al. 1987; LaGrange and Dinsmore 1998; Farmer and Parent 1997; Davis and Smith 1998). Within the U.S. Great Plains, agricultural practices have altered wetland habitats to permit crop production (Bolen et al. 1989; Samson and Knopf 1996). These practices have resulted in increased wetland sediment loads and pose as the largest immediate threat to the continued existence of properly functioning depressional wetlands (Luo et al. 1997, 1999, Gleason et al. 2003, Tsai et al. 2007). This is especially true in the Rainwater Basin Region (RWB) of Nebraska where nearly of 90% of the depressional wetlands have been lost to agricultural production (Raines et al. 1990; Stutheit et al. 2004). The RWB of Nebraska is situated at the narrowest point along the Central Flyway and provides stopover habitat for over 10 million migrating ducks, over 1 million geese, and 38 species of shorebirds every spring (Gersib et al. 1992; Jorgensen 2004).

Although agricultural practices have resulted in the majority of wetland area lost within this region, recent efforts have been made to reduce additional wetland loss and restore depressional wetlands that provide key ecosystem services such as migratory bird habitat (Smith et al. 2011a). Within the RWB, wetlands are primarily restored through removal of sediment from the wetland basin followed by the installation of a buffer around the perimeter of the wetland (LaGrange 2005). Following restoration, wetlands are allowed to self-design through recruitment of vegetation from the seed bank and dispersal from wind and waterbirds (Galatowitsch and van der Valk 1996; O’Connell et al. 2011).

Because the primary objective of wetland restoration within the RWB is to provide habitat for migrating waterbirds, we used models generated by Webb et al. (2010) to determine if restoration has the potential to improve migratory waterbird use by comparing model predictions for restored wetlands to reference and cropland wetlands. Wetland bird models that were tested included: geese, shorebirds, dabbling ducks, diving ducks, and species richness (Webb et al. 2010).

METHODS

Study Area

The RWB encompasses 15,907 km^2 and includes all or parts of 21 counties in the Central Loess Plains of south-central Nebraska (LaGrange 2005). The area was named for its abundant natural playas that formed where clay-bottom depressions catch and hold water from precipitation and surface water run-off (Stutheit 2004). Playas in this region range from 0.1 ha to 1,000 in size (Stutheit 2004). Playa formation in this region is not entirely known, but likely involved water erosion followed by wind deflation (Kuzila 1984, Smith 2003). Playas are defined by the presences of Massie, Scott, and Fillmore hydric soil series (Stutheit 2004). Annual precipitation averages 460 mm in the western region and 710 mm in the east; evapotranspiration generally exceeds precipitation (Stutheit 2004). Historically, the RWB was mixed-grass prairie in western region and mixed- to tall grass prairie in the eastern region (Kaul 1975), but presently the region is dominated by corn and soybeans. The RWB has been deemed one of the nine areas in the contiguous United States with the highest wetland loss (Tiner 1984) and contains one of the most threatened and least studied wetland complexes in North America (Smith 1998).

Study Sites

Thirty-four wetlands were sampled in 2008 and 2009 among three land use treatments: reference standard (from here forward known as reference), restored, and cropland (defined below). In 2008, 12 reference, 11 restored, and 11 cropland wetlands were sampled and in 2009, 11 reference, 11 restored, and 12 cropland wetlands were sampled. Most wetlands were sampled both years (one reference and one restored wetland was removed in 2009, one agricultural wetland was restored in late 2008, and two agricultural wetlands were added in 2009).

Reference wetlands were selected using the hydrogeomorphic approach (Brinson 1993) by the Nebraska Game and Parks Commission (NGPC) and represented the most highly functioning wetlands within the region (Stutheit 2004). Reference wetlands have had no prior physical manipulation to the basin or water levels, vegetation with little to no invasive species problems, an unmanipulated watershed, and hydric soils present match wetland type (e.g., semi-permanent, seasonal, temporary). The 12 best available reference wetlands from the HGM study (Stutheit et al. 2004) were selected for this study. One reference wetlands was removed in 2009 due to sampling logistics. In 2008, 6 of the sampled reference wetlands were seasonal and 6 were semi-permanent. In 2009, 5 were seasonal and 6 were semi-permanent.

Restoration of cropland wetlands was performed by NGPC, U.S. Fish and Wildlife Service (USFWS), and Ducks Unlimited (DU). Each of these sites was at one time impacted by sedimentation from row crop run-off. Each restored wetland had 20.3 - 30.4 cm of sediment removed from the center of the basin and graded to a depth of 10.6 - 15.2 cm around the perimeter. Following sediment removal, each wetland was allowed to self-design and was surrounded by a native grassland buffer. Many wetlands within in the RWB are restored, however, only 11 wetlands had the criteria of sediment removal across the entire basin followed by natural vegetation. In 2008, 1 of the restored wetlands was temporary, 6 were seasonal, and 4 were semi-permanent. In 2009, 1 was temporary, 7 were seasonal, and 3 were semi-permanent. Within the RWB, temporary and seasonal wetlands function similarly and are often grouped together as one class.

Cropland wetlands were privately owned wetlands that were surrounded by row crop production on at least two sides of the wetland. In some cases crops were planted through the wetland basin. Cropland wetlands had upland sediments covering hydric soils (D. Daniels unpublished data). In 2008, 2 of the cropland wetlands were temporary, 6 were seasonal, and 3 were semi-permanent. In 2009, 3 were temporary, 5 were seasonal, and 4 were semi-permanent.

Models

Local wetland and landscape-scale variables (see below) were input into models developed by Webb et al. (2010) to predict the abundance of geese, shorebirds, dabbling and diving ducks, and overall species richness for each individual wetland sampled among each land use treatment (Appendix C). This allowed us to determine which land use treatment could potentially obtain the highest species richness as well as which land use treatment was most suited for each wetland bird group.

Local wetland characteristics

Vegetation was sampled using step-point sampling (Bonham 1989) along two parallel transects in June 2008 and 2009 to determine plant species occurrence. Transects ran the length of the longest basin axis, usually northwest to southeast, starting and ending at the basin edge and passing through the center of the wetland (O'Connell et al. 2011). These data were used to calculate the percentage of emergent and inner marsh vegetation (defined below) for each wetland. Percent emergent vegetation was calculated by dividing the total number of emergent plants (not including submergents or floating vegetation) by the total number of steps encountered for each wetland. The composition of inner marsh was calculated by totaling the number of inner marsh plants encountered

(*Alisma triviale*, *Bacopa rotundifolia*, *Ceratophyllum demersum*, *Heteranthera limosa*, *H. reniformis*, *Potamogeton nodosus*, *P. pectinatus*, *Sagittaria brevirostra*, *S. calysina*, *S. graminea*, *S. latifolia*, and *Sparganium eurycarpum* (Gilbert 1989)) by the total number of steps encountered at each wetland.

Water depth was measured at 10 random locations along each vegetation transect where water was encountered. Water depth was measured to the nearest centimeter and averaged for each wetland. Hunting was characterized as open to hunting (designated as a 1 in the model) or closed to hunting (designated as a 0 in the model). Closed to hunting would remove the impact of hunting variable from the model. All private lands were considered open to hunting.

Landscape-scale variables

We analyzed 5 landscape variables for each wetland (Webb et al. 2010). Landscape GIS data were provided by the Rainwater Basin Joint Venture (RWBJV). These variables included: area (ha) of the sampled wetland, number of wetlands within 10 km, area (ha) of semi-permanent wetlands within 10 km, area (ha) of riparian within 5 km, and area (ha) of grassland within 5 km.

Analysis of wetland area and type (semi-permanent, temporary, seasonal) were determined using the 2008 USFWS National Wetland Inventory (NWI). NWI wetland types were classified according to Cowardin et al (1979). This classification scheme separated individual wetlands into different flooded zones that were identified by an alphabetic code. Wetlands with deeper water levels, such as semi-permanent wetlands are composed of one (sometimes two) central wetland polygons with semi-concentric seasonal and temporary zones surrounding them. To simplify classification, the RWBJV dissolved the polygons corresponding to individual wetlands into a single outline (footprint) then designated wetlands accordingly (e.g. semi-permanent, seasonal, temporary) (R. Grosse, pers. comm.). PatchAnalyst (ArcGIS 9.0; Environmental Systems Resource Institute, Redlands, CA) was used to calculate area (area of semipermanent wetlands within 10 km, area of riparian within 5 km, and area of grassland within 5 km) for each land use category and the number of wetlands within 10 km from the sampled wetlands from the 2010 Rainwater Basin land cover dataset (Bishop et al. 2011).

Historic wetland hydric footprint data provided by the RWBJV was used to determine differences between current NWI wetland area and historic hydric footprint area. The difference between hydric foot print area and NWI wetland area would give a relative measure of wetland area lost. This analysis was done for all restored and cropland wetlands in the study to determine how much of the historic hydric footprint area has been lost to agricultural practices and how much of the footprint has been gained due to sediment removal.

Statistical Analysis

We used all models for each waterbird group and species richness that had AIC_C weights of 0.01 or greater to predict number of birds at each site (Tables 3.1-3.5). We used multiple models within each group to account for the likelihood that models other than the model with the lowest AICc score had support from the data (Burnham and Anderson 2002). For each model, we entered site-specific local wetland and landscape-level variable data to predict bird abundance and multiplied calculated model outputs (for each individual wetland bird group) by the AICc weight of the given model. All models (for each individual wetland bird group) used were summed and then multiplied by the percentage (since the weights of all models sums to 1.0) of the weighted models used to obtain the final abundance for each given wetland. For example, the AIC_C weights of the top four models that best predict diving duck abundance sum to 0.99 (Table 3.2). The output of model 1 was multiplied by 0.72, output of model 2 by 0.24, output of model 3 by 0.02, and output of model 4 by 0.01. The resulting outputs of each model were then summed and multiplied by 0.99 to obtain the best estimate of avian abundance for each wetland (citation). If a wetland did not contain water within the basin, the wetland was assumed unsuitable and given a value for zero for all wetland bird models. Due to differences in precipitation among years, we analyzed data separately each year.

Analyses of variance (ANOVAs) were used to compare the projected abundance of geese, shorebirds, dabbling and diving ducks, as well as species richness among wetland land use treatments. If an ANOVA factor (e.g., shorebird abundance, species richness) was significant ($P<0.05$), a LS Means test was performed to determine differences between land use treatments. ANOVAs were used to calculate mean water depth, max water depth, composition of emergent vegetation, and composition of inner marsh vegetation among land use treatments. χ^2s were performed to determine differences in the number of wet and dry playas and to determine differences in wetland type (e.g., semi-permanent, seasonal, temporary) among land use treatments sampled in 2008 and 2009.

A subset of the 34 wetlands was re-analyzed to take into account differences (though not statistically significant) in wetland area for 2008 and 2009 (Table 3.6). Reference and restored wetlands averaged half the size compared to restored wetlands.

To account for differences in area among the land use treatments, we eliminated the 2 largest restored (1 semi-permanent, 1 seasonal), and the 3 smallest reference (3 seasonal), and 3 smallest cropland (3 temporary) wetlands in 2008. For the 2009 data, we removed the 2 largest restored (2 semi-permanent), and the 2 smallest reference (2 seasonal), and the 3 smallest cropland (3 temporary) wetlands. ANOVAs were used to compare differences in projected abundances of geese, shorebirds, dabbling ducks, diving ducks, and species richness. If an ANOVA factor (e.g., shorebird abundance, species richness) was significant ($P<0.05$), a LS Means test was performed to determine differences between land use treatments. ANOVAs were used to calculate mean water depth, max water depth, composition of emergent vegetation, and composition of inner marsh vegetation of the subset of wetland among land use treatments to determine differences in these variables after accounting for differences in area. χ^2s were performed to determine differences in the number of wet and dry playas and to determine differences in wetland type (e.g., semi-permanent, seasonal, temporary) among land use treatments sampled in 2008 and 2009.

To eliminate an area effect on predicted species richness and dabbling duck, diving duck, shorebird, and geese abundance, we removed area from all of the models and re-ran each model. In doing this, we looked at the variables other than area that affect predicted abundances and allowed us to relatively measure differences among the three land use treatments. The outputs from these models, with area removed, will not give an accurate estimate of predicted abundances, rather a relative measure of the local and landscape level variables (without area) as they relate to abundances among land use treatments. All wetlands sampled wetlands in 2008 and 2009 were analyzed. Wetlands that were dry were not given a value of zero in this analysis.

RESULTS

2008 All Sampled Wetlands

Even though there were no statistical differences in area of the wetlands sampled in 2008, restored wetlands were nearly twice as large as reference and cropland wetlands (F = 2.39; P = 0.099) (Table 3.6). There were no differences in average water depth (F = 1.14, P = 0.333), max water depth (F = 0.80, P = 0.458), composition of emergent vegetation (F = 1.26, P = 0.297), and composition of inner marsh vegetation (F = 1.33, P = 0.279) among land use treatments. All sampled wetlands in 2008 contained water. There was no difference in wetland type sampled among land use treatments (χ^2 = 2.97, df = 4, P = 0.563). Models for restored wetlands were predicted to have greater numbers of dabbling ducks (F = 6.07; P = 0.006) and higher species richness (F = 5.22; P = 0.011) than reference and cropland wetlands (Table 3.7). Restored wetlands were also predicted to contain more diving ducks than cropland wetlands (F = 3.66; P = 0.037). There were no differences in the predicted number of shorebirds (F = 1.11; P = 0.343). or geese (F = 5.22; P=0.011) among land use treatments.

2008 Subset of sampled wetlands

After removing the 2 largest restored wetlands and the 3 smallest reference and 3 smallest cropland wetlands, land use treatments were more similar in average area (F=0.13, P=0.877). There were no differences in average water depth (F = 0.50, P = 0.611), max water depth (F = 0.52, P = 0.601), composition of emergent vegetation (F =

1.48, P = 0.248), and composition of inner marsh vegetation (F = 0.53, P = 0.595) among land use treatments. All sampled wetlands contained water. There was no difference in wetland type sampled among land use treatments (χ^2 = 2.74, df = 4, P = 0.532). There were no differences in the predicted number of diving ducks (F = 1.55; P = 0.233), dabbling ducks (F = 2.48; P = 0.106), shorebirds (F = 0.06; P = 0.942), geese (F = 0.85; P = 0.442), and species richness (F = 3.19; P = 0.060) among land use treatments (Table 3.8).

2009 All sampled wetlands

Even though there were no differences in area of the wetlands sampled in 2009, restored wetlands were nearly twice as large as reference and cropland wetlands (F = 1.86; P = 0.173). There were no differences in average water depth (F = 01.06, P = 0.357), max water depth (F = 1.78, P = 0.185), composition of emergent vegetation (F = 0.70, P = 0.503), and composition of inner marsh vegetation (F = 1.98, P = 0.155) among land use treatments. There was no difference in the number of wet and dry playas among land use treatments (χ^2 = 4.18, df = 2, P = 0.123) with 36% of reference wetlands, 45% of restored, and 8% of cropland wetlands containing water. There was no difference in wetland type sampled among land use treatments (χ^2 = 4.95, df = 4, P = 0.292). There was no difference in the predicted abundance of diving ducks (1.50; P = 0.238), dabbling ducks (F = 1.89; P = 0.168), shorebirds (F = 2.45; F = 0.103), geese (F = 1.84; P = 0.176), and species richness (F = 2.29; P = 0.119) among land use treatments (Table 3.9).

2009 Subset of sample wetlands

After removing the 2 largest restored wetlands and the 2 smallest reference and 3 smallest cropland wetlands, land use treatments were more similar in average area (F =

0.04; P = 0.963). There were no differences in average water depth (F = 0.65, P = 0.530), max water depth (F = 0.98, P = 0.391), composition of emergent vegetation (F = 2.37, P = 0.116), and composition of inner marsh vegetation (F = 1.17, P = 0.328) among land use treatments. There was no statistical difference in the number of wet and dry playas among land use treatments (χ^2 = 3.00, df = 2, P = 0.223) with 44% of reference, 44% of restored, and 11% of cropland wetlands containing water. There was no difference in wetland type sampled among land use treatments (χ^2 = 3.67, df = 4, P = 0.452). There were no differences in the predicted abundance of diving ducks (F = 0.83; P = 0.449), dabbling ducks (F =1.37; P = 0.273), shorebirds (F = 1.85; P = 0.178), geese (F = 1.56; P = 0.230), and species richness (F = 1.52; P = 0.239) among land use treatments (Table 3.10).

Removal of area from the models

When removing area from the models, there was no significant difference in species richness and dabbling duck, diving duck, shorebird, and geese predicted relative abundance among land use treatments for 2008 or 2009 (Appendix D).

Area lost

There was no difference in wetland area lost among restored and cropland wetlands when comparing the NWI data to the historic footprint data (F = 0.22, P = 0.646). NWI data indicated that restored wetlands were 9% smaller than their hydric footprint; cropland wetlands were 15% smaller than their hydric footprint. However, removing the cropland outlier (NWI indicated that the wetland was 91% larger than its hydric footprint), cropland wetlands have lost 25% the area of their original hydric footprint.

DISCUSSION

Within the Great Plains, annual precipitation can be highly variable and has profound effects on playa hydroperiod (Smith and Haukos 2002) and thus on ecosystem services provided such as waterbird habitat at migratory stopover sites (Smith et al. 2011). In years of increased rainfall, migratory waterfowl, shorebirds, and geese have a wider availability of wetlands from which to choose than in drought years. During the wet year (2008), restored wetlands were predicted to have a greater abundance of dabbling and diving ducks and provide habitat for an additional 8 species of waterbirds per wetland compared to cropland wetlands; however, this was predominately due to an area effect. In the year of reduced rainfall (2009), RWB wetlands primarily available for stopover sites include reference and restored wetlands and few cropland wetlands. Although there were no statistical differences in projected waterbird abundance among the 3 land use treatments in 2009, restored and reference land use wetlands were 4 times more likely to contain water in drier years than cropland wetlands. Restoring cropland wetlands provided additional habitat needed to support waterbird populations during migration (O'Neal et al. 2008), especially in years of reduced rainfall. Within the RWB, most restored (100% of the sampled wetlands in 2008; 89% of the sampled wetlands in 2009) and a some reference wetlands (36% sampled in 2008 and 2009) had water control structures such as pumps. However, none of the wetlands sampled in our study had water pumped into them and therefore modeling results for individual wetlands were not influenced by this management activity.

When we removed some wetlands to make area more similar among the three land use treatments, there were no statistical differences among bird metric during the year of increased or reduced precipitation. However, in the year of increased precipitation, restored wetlands were projected to provide habitat for twice as many diving ducks and dabbling ducks and nearly three times the amount of habitat for geese than reference wetlands. Restored wetlands were also projected to provide migratory habitat for 5 more species of waterbirds than reference wetlands and 11 more than cropland wetlands. In years of reduced precipitation, restored wetlands were projected to provide nearly twice the number of diving ducks, 1.5 times the number of dabbling ducks, 25% more shorebirds, and over 10 times the number of geese than reference wetlands. In addition, restored wetlands are projected to provide habitat for 4 more species of waterbirds than reference wetlands. These results may indicate that restored wetlands have the ability to provide more suitable habitat variables for migrating waterfowl, shorebirds (in dry years), and geese than reference and cropland wetlands. Restored wetlands ability to achieve greater predicted abundances of waterfowl, shorebirds, and geese may be attributed to: 1) wetland area; 2) reduced vegetation cover; and 3) increased water depth/ ability to contain water in dry years.

Wetland Area

After taking area into account, restored wetlands in our study were still on average 5 ha larger than reference and cropland wetlands in 2008 and were 3 ha larger in 2009. Larger area is a positive predictor of increased abundance and species richness (MacArthur and Wilson 1967; Rosenzweig 1995) and had the biggest influence on bird metrics in Webb et al.’s (2010) models. Thus restoring cropland wetlands with large hydric footprints should be a primary objective of restoration if the goal is to provide increased migratory habitat. Due to sedimentation and hydrology alteration (installation of pits) cropland wetlands in the RWB average 25% smaller than their hydric footprint and the cropland wetlands that have been lost or fossilized have been primarily seasonal and temporary wetlands (LaGrange 2005). The majority of semi-permanent cropland wetlands that remain may function like semi-permanents in some respects (e.g., dominated by perennial plant communities), however they may have water holding capabilities more similar to seasonal and temporary wetlands. Decreased precipitation further exasperates the problem of cropland wetland when they do not have the ability to pond water, thus concentrating waterfowl, shorebirds, and geese on reference and restored wetlands. Greater densities of waterbirds on these wetlands can have negative consequences leading to increased avian cholera outbreaks (Smith and Higgins 1990) and shorter stopover times (Webb et al. 2010).

Vegetation

When analyzing vegetation (percent emergent), Webb et al.'s (2010) models puts the largest emphasis on the hemi-marsh condition. Species richness and dabbling duck densities have been shown to be greatest in wetlands with intermediate vegetation cover and decrease with sparse or dense vegetative cover on breeding grounds, (Weller and Spatcher 1965; Weller and Fredrickson 1974; VanRees-Siewert and Dinsmore 1986), wintering grounds (Smith et al. 2004), and at migrating stopover sites (Webb et al. 2010). In both years, restored wetlands had a more equal amount of vegetation to water ratio compared to reference wetlands; however, hemi-marsh conditions were more pronounced in wet years for restored wetlands, containing a 3:2 ratio of emergent vegetation to water compared to 3:1 for reference and 4:1 for cropland wetlands. The hemi-marsh condition of restored wetlands may likely be associated with hydrologic restoration and management activities. Most restored and reference wetlands in the RWB are periodically managed through grazing (Davis and Bidwell 2008), prescribed burning (Brennan et al. 2005), and disking (Davis and Bidwell 2008) to help reduce vegetative cover (especially of invasive species) and provide areas of open habitat. These management activities along with deeper water levels (see below) created by hydrologic restoration at restored wetlands promoted submergents and larger amounts of open water areas for waterfowl, shorebirds, and geese.

Shorebird abundance at stopover sites is greatest in wetlands with sparse to intermediate vegetation cover (Davis and Smith 1998; Webb et al. 2010). No wetlands within our study had sparse vegetative cover. However, restored wetlands in our study had less cover than reference and cropland wetlands. Intermediate vegetation cover of restored wetlands allows for increased foraging opportunities compared to densely vegetated reference and cropland wetlands and may also be correlated with greater invertebrate densities (Davis and Bidwell 2008). In addition, shorebird abundance can be limited by dense stands of emergent vegetation due to limited predator detection, mobility, and feeding activity (Metcalfe 1984; De Leon and Smith 1999).

The presence of inner marsh vegetation is a positive predictor of diving duck abundance. Inner marsh vegetation comprises a majority of several diving duck species diets (Moore et al. 1998) and also usually includes areas of deeper open water more suitable for diving duck species. Restored wetlands had similar amounts of inner marsh vegetation compared to reference wetlands in 2008; however, in 2009, reference wetlands contained twice the amount compared to restored wetlands. Even though the amount of inner marsh vegetation in reference wetlands was equal to or surpassed that of restored wetlands, restored wetlands ability to contain areas of deeper water (see below) was a more important predictor for diving duck species allowing restored wetlands to have a greater predicted abundance.

Increased emergent vegetative cover is a negative indicator of geese abundance in the models (Webb et al. 2010). Cropland wetlands were predicted to have the greatest abundance of geese (when area between land use groups is equivalent) in wet years; however, this is partially due to one cropland wetland predicted to have geese abundance in excess of 11,000 birds. Removing this cropland wetland, restored wetlands were predicted to have the greatest abundance of geese during wet years and provide the most suitable habitat for geese in drier years. Row crop production surrounding cropland wetlands may provide better feeding habitat for geese than native grasslands of reference and restored wetlands, however, dense stands of *Typha*, *Scirpus fluviatilis*, and *Phalaris arundinacea* in cropland wetlands during wet years and lack of water in dry years (in addition to *Typha* ect.) most likely limits geese feeding within the wetland.

Water Depth

For both years of the study, restored wetlands were deeper than reference and cropland wetlands. This is the result of removing up to 30 cm of sediment from the center of the basin. In doing so, restored wetlands are able to hold water for longer amount of time. This is of particular importance in dry years when water sources for waterbirds may be scarce.   In addition, restored wetlands' gradation from the center of the wetlands to the perimeter may allow for multiple feeding depths that can support a greater abundance and diversity of dabbling ducks and shorebirds than reference and cropland wetlands, however further investigation is needed.

Limitations of models and analyses

The waterbird abundance and species richness models developed by Webb et al. (2010) were developed based on data collected during 3 years of below average precipitation. In drought conditions birds have to choose what is available (any wetland with water will provide some habitat) which may negate the influence of other environmental variables associated with the wetland. With fewer playas in which to chose, differences between reference, restored, and cropland wetlands may have been less discernable because cropland wetlands would have virtually be eliminated from the original models due to dry playas not being sampled in the study by Webb et al. (2010). Moreover, within wetland land use groups, wetlands were highly variable for all waterbird models. This resulted in large standard errors for all waterbird models among each land use treatment and possible lack of significance. Also, for all models, area was the largest predictor for waterbird abundance and restored wetlands tended to be the largest. It was difficult to find wetlands of similar sizes within each category. Many wetlands that tend to be hydrologically restored across the entire wetland basin are large cropland wetlands that landowners enroll into a conservation program. Smaller cropland wetlands can be farmed in drier years or have been hydrologically altered to allow farming and are often not enrolled into conservation programs.

CONCLUSIONS

Restored wetlands provide additional migratory habitat needed to support populations of migratory waterbirds during spring migration in a heavily fragmented ecosystem. When comparing restored cropland wetlands to cropland wetlands, restored wetlands have the ability to support larger abundances of waterfowl, shorebirds, and geese. Wetland restoration should focus on restoring cropland wetlands with the largest wetland footprint in order to provide habitat to support large migratory waterbird populations. In years of low precipitation, cropland wetlands that hold water are limited on the landscape leaving only reference and restored wetlands as migratory stopover habitat.

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Table 3.1: Number of parameters (*K*) and the weight of the models (AIC_c) hypothesized to predict dabbling duck abundance in Rainwater Basin wetlands during spring migration. Models with larger AIC_c have more substantial support.

Model[a]	**K**	**AIC_C Weight**
5.138 + (SinEmerg*1.655) - (Asp10*0.0003) + (LnArea*1.001) - (Hunting*0.544)	5	0.48
4.705 + (SinEmerg*1.646) - (Asp10*0.0004) + (LnArea*1.036)	4	0.23
9.092 + (SinEmerg*1.565) - (Hunting*0.523) - (Asp10*0.0004) + (LnArea*0.991) - (LnAg10*0.370)	6	0.16
-1.028 + (SinEmerg*1.546) - (ASP10*0.0004) + (LnArea*0.994) + (LnAg10*0.576)	5	0.08
5.008 + (SinEmerg* 1.777) - (Hunting*0.616) + (LnArea*0.917)	4	0.03
6.130 + (SinEmerg*1.698) - (Hunting*0.589) + (LnArea0.895) - (LnAg10*0.097)	5	0.01
4.483 + (SinEmerg*1.783) + (LnArea*0.947)	3	0.01

[a] Model parameters (n=5) include sin transformed percent emergent vegetation (SinEmerg), area of semipermanent wetlands within 10 km (Asp10), log transformed wetlands area (LnArea), open to hunting (Hunting), and log transformed area of cropland within 10 km (LnAg10).

Table 3.2: Number of parameters (*K*) and the weight of the models (AIC_c) hypothesized to predict diving duck abundance in Rainwater Basin wetlands during spring migration. Models with larger AIC_c have more substantial support.

Model[a]	**K**	**AIC_C Weight**
1.609 - (0.910*LnArea) + (0.717*LnWaterD) + (0.544*LnIM)	4	0.72
1.610 - (0.909*LnArea) + (0.711*LnWaterD) + (0.015*LnRiv5) + (0.546*LnIM)	5	0.24
1.116 + (0.782*LnArea) + (0.621*LnIM)	3	0.02
0.998 + (0.779*LnArea) + (0.082*LnRiv5) + (0.628*LnIM)	4	0.01

[a] Model parameters (n=4) include log transformed wetland area (LnArea), log transformed max water depth (LnWaterD), log transformed percent inner marsh vegetation (LnIM), and log transformed area of riverine habitat within 5 km (LnRiv5).

Table 3.3: Number of parameters (*K*) and the weight of the models (AIC_c) hypothesized to predict shorebird abundance in Rainwater Basin wetlands during spring migration. Models with larger AIC_c have more substantial support.

Model[a]	**K**	**AIC_C Weight**
4.285 + (SinEmerg*1.001) + (Pwet10*0.001) + (Area*0.011) - (MDWater*0.019)	5	0.52
4.792 + (SinEmerg*1.201) + (Area*0.0136) - (MDWater*0.019)	4	0.33
4.587 + (Pwet10*0.001) + (Area*0.015) - (MDWater*0.019)	4	0.06
4.325 + (SinEmerg*1.371) + (Pwet10*0.001) - (MDWater*0.019)	4	0.05
5.257 + (Area*0.018) - (MDWater*0.020)	3	0.03
5.002 + (SinEmerg*1.666) - (MDWater*0.020)	3	0.02

[a] Model parameters (n=4) include sin transformed percent emergent vegetation (SinEmerg), number of wetlands within 10 km (Pwet10), area of wetland (Area), and mean water depth (MDWater).

Table 3.4: Number of parameters (K) and the weight of the models (AIC_c) hypothesized to predict geese abundance in Rainwater Basin wetlands during spring migration. Models with larger AIC_c have more substantial support.

Model[a]	**K**	**AIC_C Weight**
6.741 + (Area*0.063) - (LinEmerg*0.029) - (MDWater*0.021) - (Hunting*1.281)	5	0.49
5.797 + (Area+0.061) + (Aag5*0.0001) - (LinEmerg*0.028) - (MDWater*0.021) - (Hunting*1.181)	6	0.17
7.569 + (Area*0.066) - (LinEmerg*0.023) - (MDWater*0.860)	4	0.13
3.438 - (LinEmerg*0.026) + (Aag5*0.0002) + (Area*0.056) - (MDWater*0.022)	5	0.09
6.016 - (LinEmerg*0.026) + (Area*0.066) - (Hunting*1.459)	4	0.05
4.429 + (Aag5*0.0001) - (LinEmerg*0.025) + (Area*0.061) - (Hunting*1.283)	5	0.02
5.752 - (MDWater*0.019) + (Area*0.059) - (Hunting*1.121)	4	0.01
1.788 - (LinEmerg*0.023) + (Area*0.056) - (Aag5*0.0003)	4	0.01
4.877 - (LinEmerg*0.024) + (Area*0.067)	3	0.01
4.995 - (MDWater*0.021) + (Area*0.060)	3	0.01

[a] Model parameters (n=5) include wetland area (Area), percent emergent vegetation (LinEmerg), mean water depth (MDWater), open to hunting (Hunting), and area of cropland within 5 km (Aag5).

Table 3.5: Number of parameters (*K*) and the weight of the models (AIC_c) hypothesized to predict species richness of waterbirds in Rainwater Basin wetlands during spring migration. Models with larger AIC_c have more substantial support.

Model[a]	**K**	**AIC_C Weight**
22.639 + (SinEmerg*4.030) - (LnAsp10*2.187) + (LnArea*4.354) + (LnWaterD*0.747)	5	0.55
24.349 - (LnAsp10*2.384) + (LnArea*4.802) + (LnWaterD*0.827)	4	0.20
21.617 - (LnAsp10*2.185) + (LnArea*4.347) + (LnWaterD*0.747) + (SinEmerg*4.021) + (LnGrass5*0.166)	6	0.18
22.965 + (LnGrass5*0.225) - (LnAsp10*2.381) + (LnArea*4.792) + (LnWaterD*0.826)	5	0.07

[a] Model parameters (n=5) include sin transformed percent emergent vegetation (SinEmerg), log transformed area of semipermanent wetlands within 10 km (LnAsp10), log transformed wetland area (LnArea), log transformed max water depth (LnWaterD), and log transformed area of grassland within 5 km (LnGrass5).

Table 3.6: Local and landscape level variables among wetland land use treatments in the Rainwater Basin. The 2008 and 2009 all sampled wetlands were analyzed without taking into account differences in area between land use treatments. In 2008, 12 reference, 11 restored and 11 cropland wetlands were sampled. In 2009, 11 reference, 11 restored, and 12 cropland wetlands were sampled. The 2008 and 2009 subset of sampled wetlands categories were calculated after area differences among land use treatments were taken in account by removing the 3 smallest reference and cropland wetlands and the 3 largest restored wetlands in 2008. In 2009, the 2 smallest reference, 3 smallest cropland, and two largest restored wetlands were removed.

	Reference		Restored		Agriculture			
2008 All Sampled Wetlands	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
Area (ha)	21.33	4.57	43.22	10.44	20.57	9.75	2.49	0.0993
Average water depth (cm)	15.39	3.41	23.78	4.3	2.68	5.39	1.14	0.3329
Max water depth (cm)	24.85	5.17	35.57	6.05	7.21	6.79	0.8	0.458
Composition of emergent vegetation	85.37	4.36	73.91	7.44	81.83	4.18	1.26	0.297
Composition of inner marsh vegetation	8.94	3.27	10.45	3.97	3.83	1.52	1.33	0.2791
2008 Subset of Sampled Wetlands								
Area (ha)	26.69	4.84	31.56	7.96	25.93	13.3	0.13	0.8772
Average water depth (cm)	18.3	3.99	25.31	4.64	21.16	7.41	0.5	0.6105
Max water depth (cm)	29.1	6.04	38.6	6.53	35.72	9.4	0.52	0.6013
Composition of emergent vegetation	77.69	6.12	63.9	9.31	80.27	6.71	1.48	0.2478
Composition of inner marsh vegetation	8.81	3.4	11.1	5.9	4.51	2.82	0.53	0.5952
2009 All Sampled Wetlands								
Area (ha)	22.7	4.78	41.53	10.87	21.17	22.35	9.04	0.1732
Average water depth (cm)	4.73	2.57	5.72	2.55	1.53	1.59	1.06	0.3572
Max water depth (cm)	7.4	4.33	12.13	5.31	1.95	2.03	1.78	0.1854
Composition of emergent vegetation	96.48	1.04	92	3.3	91.63	4.45	0.7	0.5026
Composition of inner marsh vegetation	6.41	2.99	2.08	0.91	1.61	1.41	1.98	0.1546

Table 2.6 (cont.)

	Reference		Restored		Agriculture			
	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
2009 Subset of Sampled Wetlands								
Area (ha)	26.67	4.85	29.49	8.42	26.69	11.72	0.04	0.9626
Average water depth (cm)	5.78	3.07	5.62	3	2.04	2.16	0.65	0.5296
Max water depth (cm)	9.05	5.21	11.32	5.98	2.6	2.75	0.98	0.3907
Composition of emergent vegetation	93.45	2.24	87.73	5.58	97.84	0.7	2.37	0.1155
Composition of inner marsh vegetation	8.08	4.23	3.07	1.99	2.58	2.18	1.17	0.3279

Table 3.7: Predicted abundance of diving and dabbling ducks, shorebirds, geese, and species richness of waterbirds at reference, restored, and crop land use wetlands during spring migration in the Rainwater Basin in 2008. Results are from all sampled wetlands regardless of size difference between land use treatments.

	Reference		Restored		Agriculture			
	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
Divers[a]	82.06	23.71	235.62	82.19	71.01	23.27	3.66	0.0374
Dabblers[b]	5109.44	1554.22	13672.20	3520.90	3601.57	1181.96	6.07	0.0060
Shorebirds	180.84	25.02	240.63	43.05	186.52	28.43	1.11	0.3433
Geese	101.74	41.49	1378.76	827.26	1066.33	1078.45	0.86	0.4320
Species Richness[c]	25.97	2.09	33.74	2.64	25.11	1.61	5.22	0.0111

[a] Significant difference between restored and crop land use wetlands

[b] Significant difference between restored and reference land use wetlands; significant difference between restored and reference land use wetlands

[c] Significant difference between restored and reference land use wetlands; significant difference between restored and reference land use wetlands

Table 3.8: Predicted abundance of diving and dabbling ducks, shorebirds, geese, and species richness of waterbirds at reference, restored, and crop land use wetlands during spring migration in the Rainwater Basin in 2008. Results are from wetlands sampled after taken into account difference in area.

	Reference		Restored		Agriculture			
	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
Divers	106.80	26.77	200.84	76.48	89.66	29.74	1.55	0.2331
Dabblers	6640.18	1802.48	11270.80	3263.00	4252.52	1606.19	2.48	0.1059
Shorebirds	201.25	30.57	196.16	33.30	185.57	38.61	0.06	0.9415
Geese	130.31	52.87	357.78	163.19	1294.49	1510.80	0.85	0.4418
Species Richness	28.68	2.01	33.66	3.29	22.36	1.84	3.19	0.0599

Table 3.9: Predicted abundance of diving and dabbling ducks, shorebirds, geese, and species richness of waterbirds at reference, restored, and crop land use wetlands during spring migration in the Rainwater Basin in 2009. Results are from all sampled wetlands regardless of size difference between land use treatments.

	Reference		Restored		Agriculture			
	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
Divers	35.08	20.24	63.47	31.38	11.81	12.33	1.50	0.2379
Dabblers	2064.68	1117.52	3247.95	1761.75	219.57	229.34	1.89	0.1683
Shorebirds	64.91	29.50	88.46	38.80	8.57	8.95	2.45	0.1029
Geese	36.10	19.89	397.53	300.32	2.75	2.87	1.84	0.1761
Species Richness	11.33	5.11	15.18	5.76	2.42	2.53	2.29	0.1186

Table 3.10: Predicted abundance of diving and dabbling ducks, shorebirds, geese, and species richness of waterbirds at reference, restored, and crop land use wetlands during spring migration in the Rainwater Basin in 2009. Results are from wetlands sampled after taken into account difference in area.

	Reference		Restored		Agriculture			
	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
Divers	42.88	24.36	62.80	37.42	15.75	16.70	0.83	0.4490
Dabblers	2523.49	1337.36	2083.22	1244.30	292.76	310.52	1.37	0.2727
Shorebirds	79.34	34.63	68.98	33.01	11.42	12.12	1.85	0.1783
Geese	44.12	23.85	134.23	95.21	3.66	3.89	1.56	0.2302
Species Richness	13.85	6.00	14.65	6.49	3.23	3.43	1.52	0.2385

APPENDIX

Appendix A: Wetland names, land use category, wetland type, and location of wetlands sampled during the 2008 – 2009 field season.

Wetland Name	Land Use	Area	Wetland Type	Latitude	Longitude
Clay #116	Cropland	6.71	Seasonal	Private	Private
Clay #117	Cropland	21.11	Seasonal	Private	Private
Clay #158	Reference	6.33	Semi-permanent	Private	Private
Clay #21	Cropland	6.92	Semi-permanent	Private	Private
Clay #216	Reference	4.87	Seasonal	Private	Private
Clay #29	Cropland/Restored	8.27	Seasonal	Private	Private
Clay #30	Cropland	6.46	Temporary	Private	Private
Clay #33	Cropland	8.64	Seasonal	Private	Private
Clay #38	Cropland	9.96	Seasonal	Private	Private
Clay #75	Reference	19.54	Seasonal	Private	Private
Clay #79	Reference	50.92	Semi-permanent	Private	Private
Alberding WPA	Reference	14.87	Semi-permanent	40.490186	-97.989144
Bluebill A (North)	Restored	5.59	Temporary	40.640338	-97.702143
Bluebill B (South)	Restored	8.23	Seasonal	40.634556	-97.703007
Brinkerhoff	Reference	19.67	Seasonal	Private	Private
Bulrush WMA	Restored	60.36	Semi-permanent	40.390981	-98.082194
Deepwell WMA	Restored	18.95	Seasonal	40.842209	-98.218911
Eckhadt WPA	Reference	29.77	Semi-permanent	40.463739	-97.905435
Fillmore #11	Cropland	20.85	Seasonal	Private	Private
Gadwall WMA	Restored	15.59	Seasonal	40.940638	-98.035952
Gleason WPA	Reference	35.86	Seasonal	40.436198	-99.024983
Greenhead WMA	Restored	26.96	Semi-permanent	40.444090	-97.940212
Hultquist	Cropland	5.60	Temporary	Private	Private
Kissinger WMA	Restored	103.23	Seasonal	40.445534	-98.102868
Krause WPA	Cropland	111.75	Semi-permanent	40.473080	-97.797308
Lindau WPA	Reference	39.38	Seasonal	40.402863	-99.036343
Meadowlark WPA	Reference	6.16	Semi-permanent	40.472426	-97.998571
Moger WPA	Reference	23.94	Semi-permanent	40.482401	-97.992891
Morphy WPA	Cropland	34.27	Semi-permanent	40.610023	-97.732966
Renquist WMA	Restored	50.47	Seasonal	41.030002	-97.700055
Sandpiper WPA	Restored	31.41	Semi-permanent	40.500141	-97.715647

Appendix A (cont.)

Wetland Name	Land Use	Area	Wetland Type	Latitude	Longitude
Spikerush WMA	Restored	66.53	Seasonal	40.909766	-97.486020
TRPE	Cropland	1.97	Temporary	Private	Private
Verona WPA	Reference	4.77	Seasonal	40.549900	-97.960198
West Sac. WMA	Restored	88.22	Semi-permanent	40.361005	-99.308859
York #21	Cropland	20.00	Semi-permanent	40.714300	-97.528386

Appendix B: Plant guilds and scientific names for common species that germinated from the seed bank of Rainwater Basin wetlands.

Species	Abbreviation	Plant Guild
Abutilon theophrasti	Abuttheo	Upland
Agrostis hyemalis	Agrohyem	Wet Prairie Perennial
Alisma triviale	Alistriv	Shallow Emergent Perennial
Amaranthus retroflexus	Amarretr	Mudflat Annual
Ambrosia artemisifoliia	Ambrarte	Wet Prairie Perennial
Ambrosia grayi	Ambrgray	Mudflat Annual
Ammania robusta	Ammarobu	Mudflat Annual
Aster ericoides	Asteeric	Wet Prairie Perennial
Aster lanceolatus	Astelanc	Sedge Meadow Perennial
Bacopa rotundifolia	Bacorotu	Shallow Emergent Perennial
Boltonia asteroids	Boltaste	Sedge Meadow Perennial
Capsella bursa-pastoris	Capsburs	Mudflat Annual
Carex pellita	Carepell	Sedge Meadow Perennial
Chenopodium album	Chenalbu	Mudflat Annual
Chenopodium leptophyllum	Chenlept	Mudflat Annual
Conyza Canadensis	ConyCana	Mudflat Annual
Coreopsis tinctoria	Coretinc	Mudflat Annual
Echinochloa crus-galli	Echicrus	Mudflat Annual
Eleocharis acicularis	Eleoacic	Mudflat Annual
Eleocharis compressa	Eleocomp	Shallow Emergent Perennial
Eleocharis erythropoda	Eleoeryt	Shallow Emergent Perennial
Eleocharis palustris	Eleopalu	Shallow Emergent Perennial
Erechtites hieraciifolia	Erechier	Mudflat Annual
Hedeoma hispida	Hedehisp	Upland
Helianthus annuus	Heliannu	Mudflat Annual
Hordeum jubatum	Hordjuba	Mudflat Annual
Leersia oryzoides	Leer ory	Wet Prairie Perennial
Lepidium densiflorum	Lepidens	Mudflat Annual
Mentha spp.	Mint	Upland
Mollugo verticillata	Mollvert	Mudflat Annual
Pascopyrum smithii	Pascsmit	Wet Prairie Perennial
Phalaris arundinacea	Phalarun	Shallow Emergent Perennial
Polygonum amphibium	Polyamph	Shallow Emergent Perennial
Polygonum bicorne	Polybico	Mudflat Annual
Polygonum pensylvanicum	Polypens	Mudflat Annual
Polygonum ramosissimum	Polyramo	Mudflat Annual
Potamogeton nodosus	Potanodo	Submerged Aquatic
Potentilla norvegica	Potenorv	Mudflat Annual

Appendix B cont.

Species	Abbreviation	Plant Guild
Rorippa palustris	Roripalu	Mudflat Annual
Rumex altissimus	Rumealti	Sedge Meadow Perennial
Rumex crispsus	Rumecris	Sedge Meadow Perennial
Runuculus spp.	Rununcu	Mudflat Annual
Sagittaria calysina	Sagicaly	Mudflat Annual
Schoenoplectus tabernaemontani	Schotabe	Deep Emergent Perennial
Solanum rostratum	Solarost	Upland
Solidago missouriensis	Solimiss	Upland
Sparganium eurycarpum	Spareury	Shallow Emergent Perennial
Teucrium canadense	Teuccana	Shallow Emergent Perennial
Trifolium repens	Trifrepe	Upland

Appendix C. A list of the waterbird species that were encountered by Webb et al. (2010). All species listed were included in the species richness model. Bird model designation is listed next to the species that were included in that particular model. Species with no bird model designation were only included in the species richness model.

Family	Bird Model	*Scientific name*	Common name
Anatidae	Geese	*Anser albifrons*	Greater white-fronted goose
	Geese	*Chen* sp.	Snow and Ross' geese
	Geese	*Branta* sp.	Canada goose complex
		Cygnus buccinator	Trumpeter swan
	Dabbling	*Aix sponsa*	Wood Duck
	Dabbling	*Anas strepera*	Gadwall
	Dabbling	*Anas penelope*	Eurasian wigeon
	Dabbling	*Anas americana*	American wigeon
	Dabbling	*Anas platyrhynchos*	Mallard
	Dabbling	*Anas discors*	Blue-winged teal
	Dabbling	*Anas cyanoptera*	Cinnamon teal
	Dabbling	*Anas clypeata*	Northern shoveler
	Dabbling	*Anas acuta*	Northern pintail
	Dabbling	*Anas crecca*	Green-winged teal
	Diving	*Aythya valisineria*	Canvasback
	Diving	*Aythya americana*	Redhead
	Diving	*Aythya collaris*	Ring-necked duck
	Diving	*Aythya affinis*	Lessur scaup
	Diving	*Bucephala albeola*	Bufflehead
	Diving	*Bucephala clangula*	Common goldeneye
	Diving	*Lophodytes cucullatus*	Hooded merganser
	Diving	*Mergus merganser*	Common merganser
	Diving	*Mergus serrator*	Red-breasted merganser
	Diving	*Oxyura jamaicensis*	Ruddy duck
Podicipedidae		*Podilymbus podiceps*	Pied-bill grebe
		Podiceps auritus	Horned grebe
		Podiceps nigricollis	Eared grebe
		Aechmophorus clarkii	Clark's grebe
Pelecanidae		*Pelecanus occidentalis*	American white pelican
Phalacrocoridae		*Phalacrocorax auritus*	Double-crested cormorant
Anhingidae		*Anhinga anhinga*	Anhinga
Ardeidae		*Botaurus lentiginosus*	American bittern
		Ixobrychus exilis	Least bittern
		Aredea herodias	Great blue heron

Appendix C (cont)

Family	Bird Model	*Scientific name*	Common name
		Aredea alba	Great egret
		Egretta thula	Snowy egret
		Bubulcus ibis	Cattle egret
		Butorides virescens	Green heron
		Nycticorax nycticorax	Black-crowned night heron
Threskiornithidae		*Plegadis chihi*	White-faced ibis
Rallidae		*Porzana carolina*	Sora
		Fulica americana	American coot
Gruidae		*Grus canadensis*	Sandhill crane
Charadriidae	Shorebird	*Pluvialis squatarola*	Black-bellied plover
	Shorebird	*Pluvialis dominica*	American golden-plover
	Shorebird	*Charadrius semipalmatus*	Semipalmated plover
	Shorebird	*Charadrius melodus*	Piping plover
	Shorebird	*Charadrius vociferous*	Killdear
Recurvirostridae	Shorebird	*Himantopus mexicanus*	Black-necked stilt
	Shorebird	*Recurvirostra americana*	American avocet
Scolopacidae	Shorebird	*Tringa melanoleuca*	Greater yellowlegs
	Shorebird	*Tringa flavipes*	Lesser yellowlegs
	Shorebird	*Tringa solitaria*	Solitary sandpiper
	Shorebird	*Catoptrophorus semipalmatus*	Willet
	Shorebird	*Actitis macularius*	Spotted sandpiper
	Shorebird	*Bartramia longicauda*	Upland sandpiper
	Shorebird	*Limosa Haemastica*	Hudsonian godwit
	Shorebird	*Limosa fedoa*	Marbled godwit
	Shorebird	*Arenaria interpres*	Ruddy turnstone
	Shorebird	*Calidris canutus*	Red knot
	Shorebird	*Calidris alba*	Sanderling
	Shorebird	*Calidris pusilla*	Semipalmated sandpiper
	Shorebird	*Calidris mauri*	Western sandpiper
	Shorebird	*Calidris minutilla*	Least sandpiper
	Shorebird	*Caladris fuscicollis*	White-rumped sandpiper
	Shorebird	*Tryngites subruficollis*	Buff-breasted sandpiper
	Shorebird	*Limnodramus* sp.	Dowitcher complex
	Shorebird	*Gallinago delicata*	Wilson's snipe
	Shorebird	*Phalaropus tricolor*	Wilson's phalarope
	Shorebird	*Phalaropus lobatus*	Red-necked phalarope

Appendix D: Predicted values after area has been removed from the models. Results do not indicate predicted abundances for each waterbird group, however, results take into account differences in area among land use treatments. Results indicate a relative abundance in comparison to local and landscape level variables. Higher values indicate more suitable waterbird habitat.

	Reference		Restored		Agriculture			
2008 All Wetlands Sampled	Mean	SE	Mean	SE	Mean	SE	F-value	P-value
Divers	5.75	1.70	7.39	2.30	5.19	1.18	0.62	0.5461
Dabblers	212.83	44.86	317.28	52.04	223.45	34.19	1.82	0.1789
Shorebirds	133.55	12.30	127.54	9.18	142.48	18.39	0.32	0.7297
Geese	17.07	2.54	22.76	3.61	17.98	3.40	1.00	0.3785
Species Richness	13.59	1.36	18.46	2.97	13.81	1.33	2.35	0.1118
2009 All Wetlands Sampled								
Divers	2.63	1.52	2.08	0.82	1.02	0.82	0.63	0.5386
Dabblers	119.86	19.42	171.99	28.57	126.36	27.12	1.33	0.2784
Shorebirds	120.30	3.18	138.15	22.20	143.91	18.44	0.31	0.7381
Geese	13.98	0.57	18.41	3.78	16.25	3.11	0.64	0.5362
Species Richness	11.22	1.62	15.98	2.44	10.90	1.10	2.80	0.0764

VITA

Benjamin James Beas

Candidate for the Degree of

Doctor of Philosophy

Thesis: THE EFFECTS OF WETLAND SEDIMENTATION AND SEDIMENT REMOVAL ON RAINWATER BASIN VEGETATION, SEED BANK AND WATERBIRD COMMUNITIES

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Pages in Study: 111   Candidate for the Degree of Doctor of Philosophy

Major Field: Zoology

Scope and Method of Study: The purpose of this study was to assess the effects of restoration, by sediment removal, on wetlands located in the Rainwater Basin region of NE. Assessment included 3 objectives: extant vegetation (2008 & 2009), seed banks (2009), and using models to predict waterbird use (2008 & 2009) among reference, restored, and cropland wetlands.

Findings and Conclusions: Objective 1:For both years of the study, restored wetlands had more similar plant community characteristics (e.g., species richness) to reference wetlands than to cropland wetlands. Even though restored wetlands had similar plant community characteristics to reference wetlands, CCA results depicted that reference and restored wetlands are associated with differing plant species and guilds. In addition, restored wetland plant communities do not appear to be acting as intermediates between reference and cropland conditions or be on a trajectory to reach reference condition. Objective 2:There was no difference in the germinating seed bank plant community of reference, restored, and cropland wetlands. Cropland wetlands contained the greatest proportion of exotic species compared to reference and restored wetlands. Availability of seeds after removing 30 cm of sediment was low indicating that the seed bank was not the primary mechanism for plant recolonization of restored wetlands. Sediment removal was successful in removing weedy and invasive species from the seed bank. Objective 3:During wet years, restored wetlands were predicted, although not statistically significant, to have nearly twice the abundance of dabbling and diving ducks as reference and cropland wetlands and twice as many geese and contain 5 more species of waterbirds compared to reference wetlands. In dry years, restored wetlands were predicted, although not statistically significant, to have the greatest abundance of dabbling ducks, diving ducks, shorebirds, and geese among the 3 wetland types. In years of low precipitation, reference and restored wetlands are the primary habitat available for waterbirds during migration because most cropland wetlands are dry. Models predicted that restored wetlands within the Rainwater Basin will provide improved habitat needed for migrating waterbirds during spring migration.

ADVISER'S APPROVAL: ______________________________

Dr. Loren M. Smith