Biological Evaluation of Central Platte River Slough Wetland Restorations

A final Report to:

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and

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by

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Chapter I

PLANT COMMUNITIES AND ECOSYSTEM FUNCTION IN NATURAL AND RESTORED PLATTE RIVER WETLANDS

Introduction

Wetlands perform numerous ecosystem services, ranging from biological (e.g., providing habitat for various species)(e.g., Reineke and Krapu 1986, Snodgrass et al. 2000, Whiles and Goldowitz 2005), to hydrological (e.g., playing roles in groundwater discharge/recharge and surfacewater filtration)(e.g., Barber et al. 2001, Baldwin and Hodaly 2003, Bullock and Acreman 2003, Berthold et al. 2004, Tennant-Wood 2004), to biogeochemical (e.g., nutrient removal or transformation)(e.g., Morris 1991, Schlesinger 1997, Cirmo 1998, Hamersley and Howes 2002) processes. Freshwater wetlands are also some of the most diverse and productive ecosystems on earth (e.g., Mitsch and Gosselink 2000). Despite the inherent value of these systems, over half of the original wetland area that once existed within the contiguous 48 states of the United States has been lost (Vileisis 1997).

In areas such as the Great Plains of the United States, much of the land area that once existed as wetlands has been drained, or "reclaimed" for crop production and other developments. For example, between 1938 and 1989, 23-45% of wet meadows in the central Platte River valley, Nebraska were drained for crop production and development (Currier 1997). Depending on the river reach, 20-70% of natural wet meadow habitat has been destroyed, and it currently comprises <5% of the land area in the Platte River valley (Sidle et al. 1989, U.S. Fish and Wildlife Service 1997). The extensive degradation of wetland habitat in this region is of particular concern because it is a critical component of the central waterfowl flyway. Seven to nine million individuals of 300 waterfowl species, including endangered Interior Least Terns (*Sterna antillarum*) and Whooping Cranes (*Grus americana*), migrate annually through this region (Krapu et al. 1984, Richter and Powell 1996), relying on the area to acquire energy and nutrients essential for migration and reproduction (U.S. Fish and Wildlife Service 1997).

Worldwide interest in wetlands and awareness of threats to wetland health have resulted in increasing efforts to protect and properly manage remaining systems and to facilitate recovery of degraded systems through restoration. According to Bradshaw (1996), restoration is the return of a degraded system to its "original state", or at least to a "healthy or vigorous state"(Figure 1). Following disturbance, recovery of some or all aspects of a system can potentially occur through natural, successional processes. However, these processes often require much more time than is considered acceptable (e.g., Zedler and Calloway 1999). Therefore, restoration typically involves some facilitation of natural recovery processes by manipulation of habitat, increased dispersal of primary colonizers, or other practices that attempt to catalyze the return of a degraded system to an "original state". In some cases, recovery is on a trajectory toward the original state, but has not yet reached full recover (Figure 1). This type of recovery is termed "rehabilitation" (Bradshaw 1996). In other cases, manipulation of habitat results in a system that has "recovered" relative to the degraded system, but might differ from the original state in structure and/or function, in a process termed, "replacement" (Bradshaw 1996)(Figure 1). If degradation of a system is too severe, true restoration may never occur, because recovery of some aspects of structure and/or function are simply not possible (Bradshaw 1996). In these situations, although restoration of some aspects (e.g., biodiversity, productivity) may occur, rehabilitation or replacement may be the most realistic and attainable goals. The process of restoration typically involves three considerations; a remodeling of the physical habitat characteristics (e.g., soil structure); chemical characteristics (e.g., nutrients); and a replacement of missing species (or in some cases removal of exotics, or both)(Bradshaw 1996).

The plant community is often a focus for wetland restorations (e.g., Willard et al. 1990, Reinartz and Warne 1993, Galatowitsch and van der Valk 1996a-c, Montalvo et al. 1997, Budelsky and Galatowitsch 2000, Wilkins et al. 2003), not only because of its importance to wetland function, but also its role in the definition and delineation of wetlands (Federal Interagency Committee for Wetland Delineation 1989). Although natural plant communities can potentially reestablish in a degraded area following removal of anthropogenic disturbances (e.g., ceasing crop production), recovery can be accelerated through seeding (e.g., Rienartz and Warne 1993). Plant diversity is relatively high in Platte River valley wet meadows, with over 200 species (Currier 1998), and restoration typically involves an effort to seed native vegetation to increase the similarity of restored areas to natural systems (Currier 1994, 1998). Although seeding may enhance plant diversity in wetlands, Currier (1998) found that, despite seeding, 70% of native wetland species in central Platte valley restorations were still absent after several years of development. Currier (1998) also found that plant species associated with disturbed habitats were more prevalent in restored sites than natural wetlands, suggesting full recovery of plant communities may never happen. Vegetation may not always follow trajectories of recovery (as in Bradshaw 1996) or, in some cases, recovery may take longer than usual monitoring periods (Zedler and Calloway 1999). These observations underscore the need for assessing wetland restoration practices within the central Platte River valley, including studies that consider long-term patterns.

Wetland restoration and creation efforts in the central Platte River valley by groups such as the Nebraska Game and Parks, The Nature Conservancy, and the Platte River Whooping Crane Maintenance Trust continue to increase. These groups have been implementing slough wetland restorations for over 12 years through land contouring and seeding with species mixes found in nearby natural systems (Currier 1998). However, because the focus of many of these conservation groups is implementation rather than evaluation, and limited resources exist, restorations have not been adequately evaluated in terms of recovery of community structure, and particularly in terms of function. In order to assess and improve techniques, current restoration practices need to be fully evaluated to assess whether systems are being restored, rehabilitated, or replaced (as in Bradshaw 1996). Further, if successes are documented, this information can be used to justify further restoration efforts. Therefore, our overall objective was to evaluate wetland restorations in the central Platte River valley through the following specific objectives: 1. to measure and compare plant richness, diversity and community structure in nearby old restorations (6-7yrs old), new restorations (1-2yrs old) and natural wetlands; 2. to estimate and compare measure of aboveground net primary productivity in natural wetlands, old restorations, and new restorations; and 3. to estimate and compare measures of root biomass in natural wetlands, old restorations and new restorations. In doing so, we quantified important components of ecosystem structure and function, allowing for a comprehensive evaluation of these systems.

Study Region

The study region lies within the central Great Plains along a ~90 km stretch of the central Platte River valley in south central Nebraska, roughly from 10 km east of Grand Island, in Hall County, to near Elm Creek, in Phelps County. The central Platte River valley in Nebraska consists of the braided channels of the Platte River and adjacent wet meadows. Wet meadow systems are a matrix of mesic prairie with meandering linear wetland sloughs in low areas. These sloughs range in hydroperiod from ephemeral to perennial; water levels are affected by precipitation and river discharge through groundwater (Whiles and Goldowitz 1998). Vegetation in the meadows is dominated by sedges (e.g., *Carex emoryi*) and grasses (e.g., *Spartina pectinata*), with a substantial component of forbs (e.g., *Verbena hastata*). Climate is temperate with warm summer temperatures (July mean temp.= 24^0C) and cold winter temperatures (January mean temp.=-7^0C), often resulting in ice covered surface waters from November through March. Mean annual precipitation is 63 cm/yr, most of which falls in May and June.

Study Sites

Nine wetland sites were chosen, including three natural wetlands, three older restored wetlands (restored ≥6 years before initiation of project), and three newly restored wetlands (restored 1-2 years before initiation of project) (Table 1). Study wetlands all had similar morphology, and they were intermittent with dry periods typically occurring in late summer. Restoration procedures and management histories were also similar among the sites. All restored sites were previously crop fields that were restored through land contouring to create sloughs, followed by seeding with material collected from nearby natural wetlands.

At each site, all measurements were taken within a designated 30-m representative reach. Each 30-m reach included three transects, one found along the deepest part of the wetland (slough habitat) and one following the contour of each margin edge of the wetland (margin habitat) (Figure 2). These transects were implemented because of the different microhabitat and topographic positions and correspondingly different plant communities in each of these locations (Meyer and Whiles, personal observation).

Methods and Materials

Diversity and community composition

Plant richness and diversity were measured during spring (June 16-19) and summer (August 4-8), 2003 by identification of all plant species found within multiple 0.25-m^2 subsamples. This procedure was performed twice during the growing season to assure that sampling included both spring ephemerals and others that often are missed during late summer sampling. Six sub-samples were taken at 5-m increments along each transect with 0.25-m^2 square sampling frame (Figure 2). Richness was estimated as the mean number of plant species found in each transect. Diversity was calculated using the Shannon-Weiner Index: $H' = -\sum (pi * \log pi)$, where pi is the proportion of the ith species.

Percent canopy cover of each species was estimated in spring and summer, and the higher value was used. Most plant taxa were identified to species. Some exceptions included *Eleocharis*, *Equisetum*, *Solidago*, and some *Polygonum*, which were identified to genus due to lack of flowering structures or other diagnostic characteristics needed to identify species. Some individual plants were unidentifiable because of lack of developed characteristics, and were therefore excluded from some analyses (richness, diversity, %annuals, %natives, average WIS). Plant taxa were assigned to vegetation groups (grasses, forbs, sedges/rushes, woody plants), and the mean relative cover of those groups was estimated.

Wetland indicator status (WIS) was assigned to each species (USDA, NRCS 2005), and mean WIS for each site was calculated according to Wentworth et al. (1988). Species were given WIS values as follows: obligate=1, facultative wet=2, facultative=3, facultative upland=4, upland=5. Additional measures of plant life history (% annuals) and nativity (% natives) were estimated using relative cover of each plant species.

Aboveground productivity and belowground biomass

Aboveground net primary productivity (ANPP) and belowground biomass were measured for each site in late September of 2002 and 2003 following peak biomass (Bernard 1974, Hoagland et al. 2001). ANPP was estimated by clipping all vegetation at ground level within an area defined by a 0.1-m^2 sampling frame. Six plots were used at randomly chosen points along the middle transect, and three in each lateral transect at each site (Fig. 2). Standing dead vegetation was separated from current year vegetation by visual inspection and was discarded. Samples were sorted by vegetation group (grasses, sedges and rushes, forbs, *Typha*, woody plants), dried at 50^0C for 1 week, and weighed to estimate total ANPP (Briggs and Knapp 1991). In some cases, categories were combined to give ANPP of certain groups. Grass ANPP was the combination of all Poaceae. Sedge and rush ANPP included all plants belonging Cyperaceae and Juncaceae. Forbs included all herbaceous dicots. Woody plants included *Populus* and *Salix*.

Root biomass was estimated by taking soil cores (5-cm diameter to 20-cm depth). Six cores were taken at random points along the middle transect, and three in each lateral transect (Figure 2). Contents of the cores were placed in whirlpacks and stored at 4^0C until processing. Soil from the samples was washed away from roots in nested sieves to collect coarse (>1 mm) and fine (1 mm<250 µm>) root fragments. Large pieces of roots found in the 250-µm sieve were picked, and remaining minute root fragments were picked for an additional 10 minutes per sample to standardize effort. Root samples were then dried at 50^0C for 1 week to estimate belowground biomass of roots at 20-cm depth.

Data Analyses

All compositional measures (richness, diversity, WIS, % annuals, % natives) were analyzed using analysis of variance (ANOVA) procedures. First a two-way ANOVA was performed to test for differences between transects (slough vs. margin), differences between wetland age classes (natural vs. old restorations vs. new restorations), and to assess interactions (see Appendix I). Subsequently, all measures were analyzed using one-way ANOVA (SAS 1988) procedures, by transect, with mean separation using Tukey's HSD (α=0.05). Mean separation tests were confined to major taxonomic groups and a priori comparisons. Because some plant community measures were significantly different in the two wetland habitats (margins and sloughs), and for ease of interpretation, data from the slough and margin transects are presented separately in all cases. Data were log-transformed where necessary to satisfy normality assumptions and to correct for heteroscedasticity. Percentages were arcsine transformed. Because of the limited replication and inherent variability in these systems, we considered tests with $p<0.10$ and >0.05 marginally significant.

We also compared vegetation community composition in wetland age classes with non-metric multidimensional scaling (NMDS), which is a robust ordination technique (Minchin 1987). NMDS creates an ordination of composition data quadrats in various dimensions and adjusts the position of quadrats to minimize stress. Stress is measured using badness-of-fit of rank order regressions of ordination distances on dissimilarities. We used relative cover data for plant species found in all sampling plots of margins and sloughs in Platte River wetlands and calculated dissimilarities using the Bray-Curtis Index (Bray and Curtis 1957), which has been shown to be one of the most effective techniques for ordination of community data (Faith et al.

1987). We pooled data points from sloughs and margins, because separate analyses of those habitats yielded effectively identical ordinations. We performed NMDS in one to six dimensions, and used 100 random starting configurations.

We performed vector fitting (Dargie 1984, Faith and Norris 1989, Kantvilas and Minchin 1989) to examine correlations between plant composition and explanatory variables. The explanatory variables we used were restoration age, restoration status (restored sites vs. natural sites), old restoration recovery (old restorations vs. natural sites) and hydrology (number of months of standing water at a site). Vector fitting uses multiple linear regression to find the direction across ordination space that has the highest correlation between sample coordinates and a particular variable. To test the statistical significance of the correlation, values of the variable among quadrats are randomly permutated, simulating the null hypothesis of no trend. We performed all ordination and vector fitting techniques with DECODA software (Minchin 1989).

Productivity measures (ANPP and root biomass) were analyzed with repeated measures, using the mixed procedure in SAS (Little et al. 1996), and mean separation using Tukey's HSD. The three replicate sloughs of each age class (natural, old restorations, and new restorations) were considered subjects; the 2 consecutive years were repeated measures.

Results

Diversity and community composition

A total of 85 plant taxa were identified within the wetland sites, with an additional 9 that could not be identified. Dominant taxa in natural wetlands included *Carex emoryi, Spartina pectinata, Polygonum* spp., *Eleocharis* spp., and *Phyla lanceolata*, all of which are obligate or facultative wetland species. Dominant taxa within old restorations included *Spartina pectinata, Helianthus* *maximiliana, Eleocharis* spp., *Juncus torreyi,* and *Salix* spp. Four of these species are facultative wetland species, but *H. maximiliana* is an upland plant. Also, *Salix* is considered undesirable in these wetland systems. New restorations were dominated by *Populus deltoides, Desmanthus illinoiensis,* and *Andropogon gerardii,* three facultative or upland species, *S. pectinata,* and two wetland annuals, *Conyza Canadensis* and *Iva annua*.

Plant richness was significantly higher in margins of old restorations than either natural wetlands or new restorations (F=9.57; DF=2,6; p=0.0136)(Figure 3A). Similarly diversity was higher in old restoration margins than those of other age classes (F=4.81; DF=2,6; p=0.0567)(Figure 4A). However, no differences in richness or diversity were seen in sloughs between age classes (Figures 3B, 4B). Plant community percent similarity between new restorations and natural wetlands was relatively low in all cases, ranging from 13.5% in sloughs to 16% in margins (Table 2). Percent similarity between old restorations and natural wetlands was 29% and 33.7% in sloughs and margins, respectively (Table 2). In both margins and sloughs, old restorations were more similar to natural wetlands than new restorations, and this pattern was significant in margins (Table 2).

Relative cover of vegetation groups showed similar patterns in sloughs and margins. Sedges and rushes showed an increase with restoration age in both habitats, and had the highest relative cover in natural wetlands (Figures 5A,5B). Conversely, grasses showed a general decrease in cover from margins of new to old restorations, with lowest cover in margins of natural wetlands (Figure 5A). Within sloughs, relative cover of grasses was highest in new restorations and lowest in old restorations (Figure 5B). Woody vegetation cover was significantly different between age classes in both sloughs (F=6.07; DF=2,6; p=0.0362) and margins (F=5.67; DF=2,6; p=0.0414), and no woody vegetation was present in margins or sloughs of natural wetlands

(Figures 5A,5B). *Typha* was only found in natural and old restorations in both habitats (Figures 5A,5B). Relative cover of forbs was similar in all age classes in both margins and sloughs (Figures 5A,5B).

Percent native plant species ranged from 88% in new restoration margins to 99% in natural wetlands margins, and from 93% in new restoration sloughs to 99% in natural wetland sloughs (Figures 6A,6B). Percent of annual species in margins and sloughs showed no differences across age classes (Figures 7A,7B).

Average WSI values were significantly higher in natural wetland margins than margins of new restorations ($F=6.16$; $DF=2,6$; $p=0.0351$), and were intermediate in margins of older restorations (Figure 8A). WSI values appeared to decrease in sloughs with wetland age as well, but this pattern was not statistically significant (Figure 8B).

Two-dimensional NMDS ordination (stress=0.25, achieved from all 100 random starts) was used because it had relatively low stress compared to the one-dimensional solution (stress=0.40), and the addition of a third-dimensional solution did not cause an appreciable decrease in stress (stress=0.18), but would have decreased ecological interpretability. Fitted vectors of maximum correlation for restoration age ($r=0.69$), restoration status ($r=0.68$), old restoration recovery ($r=0.76$), and hydrology ($r=0.71$), were all highly significant ($p<0.0001$) and had similar correlations (r ranged from 0.68 to 0.76). The hydrology vector and the restoration age vector were only separated by 9.6° (Figure 9), because two of the natural restorations held water for 4 months, two of the older restorations held water for two months during 2003, and none of the new restorations had standing water (Table 1). Other vectors showed substantial separation (Figure 9). For example, the restoration status vector and the restoration age vector were separated by 46° (Figure 9). Also, the old restoration recovery vector and the restoration age vector were separated by 68°, suggesting that the trajectory of recovery from new to old restorations is not the same as the trajectory from restorations to natural wetlands.

Aboveground productivity and belowground biomass

Total ANPP in margins was significantly higher during 2003 than 2002 ($F=5.56$; $DF=1,6$; $p=0.0564$). Within wetland margins, total ANPP increased from new restorations to old restorations, and was highest in natural wetlands in both years (Figure 10-1A). In 2002, total ANPP in natural wetland margins was significantly higher than new restoration margins ($F=3.98$; $DF=2,6$; $p=0.0794$), and in 2003 new restoration margins had significantly lower total ANPP than margins of both natural wetlands and old restorations ($F=8.70$; $DF=2,6$; $p=0.0169$)(Figure 10-1A). Production patterns were similar, but not significant, in sloughs (Figure 10-1B).

Three vegetation groups showed significant differences in ANPP between wetland age classes. Although no differences were seen in wetland sloughs (Figure 10-4B), ANPP of sedges and rushes in margins was significantly lower in new restorations than either old restorations or natural wetlands in both 2002 ($F=4.32$; $DF=2,6$; $p=0.0688$) and 2003 ($F=3.99$; $DF=2,6$; $p=0.0792$)(Figure 10-4A). *Typha* ANPP was significantly higher in old restorations than the other age classes within sloughs in both 2002 ($F=4.77$; $DF=2,6$; $p=0.0575$) and 2003 ($F=5.68$; $DF=2,6$; $p=0.0412$)(Figure 10-3B). No differences in *Typha* ANPP were observed in wetland margins (Figure 10-3A), although new restorations had no *Typha* in either margins or sloughs. Woody vegetation was not present in any samples from sloughs or margins of natural wetlands, and was generally more abundant in restorations (Figures 10-2A,10-2B).

Root biomass in margins increased from new restorations to old restorations, and was highest in natural wetlands in both years (Figure 11A). New restoration margins had significantly lower root biomass than margins of natural wetlands or old restorations in 2002

(F=7.50; DF=2,6; p=0.0233), and natural wetland margins had significantly higher root biomass than new restoration margins in 2003 (F=5.29; DF=2,6; p=0.0473). No significant differences in root biomass in sloughs were evident across age classes (Figure 11B), but overall root biomass in sloughs was significantly higher in 2003 than 2002 (F=8.32; DF=1,6; p=0.0279).

Discussion

Differences between wetland habitats

Although many plant community parameters did not differ statistically between sloughs and margins in the wetlands we studied, recovery appeared to be proceeding at very different rates in the two habitats. Most measurements, including both measures of ecosystem structure (e.g., richness, diversity) and function (e.g., ANPP, root biomass), did not differ significantly among wetland age classes in sloughs. It is notable that these aspects of plant communities and ecosystem function did not differ in sloughs of wetlands that are only 1-2 years post-restoration and relatively undisturbed natural systems. However, this pattern is in sharp contrast to patterns in wetland margins, where measures of structure and function showed marked differences among age classes and a slower trajectory of recovery from new to old restorations and natural systems.

Differences in elevation between wetland margins and sloughs are likely linked to gradients in hydrology and soil properties, and in turn differential patterns of recovery. Depending on individual site morphology, margins and sloughs of our study sites differ in elevation from 10-25cm (Meyer 2005, unpublished data). Changes in hydrology and soil properties have been shown along elevational gradients in other wetlands (e.g., Reese and Moorhead 1996), and Bledsoe and Shear (2000) found that an elevation difference of only 10 cm resulted in a 20% difference in the frequency of surface flooding during the growing season in North Carolina alluvial swamps. Rheinhardt and Faser (2001) found that a difference in elevation of only 22-25cm in North Carolina swales was enough to cause a change in the vegetation community from emergent wetlands to scrub-shrub wetlands. Edwards and Proffitt (2003) compared wetland plant succession in several restored and natural Louisiana salt marshes and found that a site at higher elevation had a different successional trajectory than sites at lower elevations. Our results add evidence that subtle elevational gradients in wetlands are linked to significant changes in ecosystem structure and function, and thus differential recovery patterns following restoration. Our slough transects were located in the lowest, wettest area of each site. Wetter conditions in sloughs may have increased potential for seed germination and seedling survival during recovery, processes that are crucial to establishment of wetland vegetation (e.g., Morgan 1990, van der Valk et al. 1999), and could in turn affect overall succession within the wetlands. Regardless of the mechanisms, our results show clearly different recovery patterns across elevational gradients in these systems, and they suggest that restoration efforts may need to consider elevational differences and that recovery of higher areas is a longer process.

Comparisons between age classes

Plant community metrics that we measured were similar to other freshwater wetlands in the region. For example, Reinartz and Warne (1993) found 142 species of plants in depressional wetlands in southeastern Wisconsin, compared to the ~95 species that we found.

Many of the plant community metrics that we measured showed patterns of recovery from degradation, particularly in wetland margins. For example, richness and diversity were higher in margins of old restorations compared to new restorations. This pattern has been seen in other studies. For example, Galatowitsch and van der Valk (1996a) compared vegetation communities in created and natural prairie pothole wetlands and found that after 3 years of recovery, created wetlands still had only ~1/2 as many species as natural wetlands. In a similar study, Seabloom and van der Valk (2003) found lower richness in created prairie potholes than in natural sites. In these studies, however, created wetlands were left to processes of self-design, rather than being seeded.

Although diversity and richness increased from new to old restorations in our sites, values for natural systems were actually lower than those of the old restorations, and thus not different from new restorations. Higher richness and diversity in old restorations compared to new restorations and natural wetlands suggests that with seeding and increased time since restoration, there may be an overshooting of diversity (e.g., Baldwin 2004).

Although most patterns that we observed support an expected recovery trajectory (e.g., richness and diversity increasing with age), we cannot rule out the influence of weather patterns that prevailed during restorations. For example, the low richness and diversity that we measured in new restorations could be a result of the drought cycle that began in the region in 2000, coinciding with restoration activities in the new restorations (National Weather Service 2005). This lack of moisture during a critical time in restoration may have resulted in decreased seed germination and seedling survival of many species (e.g., van der Valk 1999), particularly in the drier margins where differences in richness and diversity were more pronounced. In contrast, relatively wet years during seeding and establishment of the old restorations (1995-1996) (National Weather Service 2005) could have resulted in higher germination success and survival, ultimately leading to the higher richness and diversity that we measured in those systems.

In addition to potential weather influences, we noted a difference in water retention capabilities among some restorations that might also have influenced recovery patterns. In particular, two of the old restorations held water for some period during 2003, while none of the new restorations did (Table 1). There is high variability in site-specific wetland hydrology throughout the Platte River valley (Whiles and Goldowitz 2001), and the hydrologic patterns that we observed may be related to some aspect of the restoration process or location (e.g., differences in morphology, depth to groundwater, or soil structure in restoration sites), resulting in drier restorations. Alternatively, development of organic materials and soil horizons during recovery may influence the capacity of a site to hold water and thus these differences may be a natural, and potentially limiting component, of recovery. The lack of differences in richness or diversity in sloughs among sites, as opposed to margins, supports the idea that moisture plays an important role in the recovery of these systems.

Hypotheses regarding wetland succession predict that new restorations will have a high proportion of annuals and a low proportion of native species compared to old restorations or natural systems (van der Valk 1981, Thompson 1992). Thus, we were surprised to find no significant differences in percent annuals or percent natives between wetland age classes. Seabloom and van der Valk (2003) found that naturally colonizing prairie pothole wetland restorations in the upper Midwestern United States were higher in annual species, whereas perennials dominated established systems. Similarly, Reinitz and Warne (1993) found that natural depressional wetlands in Wisconsin were dominated by native species. However, in a study comparing created and natural freshwater wetlands in Virginia, DeBerry and Perry (2004) found that many perennials had colonized the created wetland, suggesting that some perennial species may also utilize bare or "annual" habitats, as may be the case in the systems that we examined. Additionally, some of the species in our sites could only be identified to genus and these may have included annual or exotic species that we did not account for. For example, some species of *Polygonum* are annuals, and some species of *Amaranthus* are not native to the

Great Plains (USDA 2005). Further, seeding practices in the restorations that we examined may have enhanced recruitment of perennial species relative to some annual "weedy" species typically associated with wetland colonization.

Estimates of ANPP and root biomass in our study were similar to other studies in the region, and provide further evidence that freshwater wet meadow systems are highly productive. Auclair et al. (1976) estimated end of season aboveground plant production was 845g/m^2 in a natural *Scirpus-Equisetum* marsh complex in southern Quebec. Hoagland et al. (2001) measured above and belowground biomass in an Illinois wetland that was created to filter water draining from agricultural fields. Biomass peaked in September, and was 2300 g/m^2 belowground and 700 g/m^2 aboveground. These estimates are much higher than the systems that we examined, but this difference is likely due to the wetter conditions and higher nutrients from the agricultural fields that would increase production in the Illinois systems.

Results of this study suggest that total ANPP and root biomass in wetland margins are increasing with age of restoration, and may eventually approach the levels found in natural wetlands. Similarly, Craft et al. (1999) found that, although some aspects of restoration may take longer, productivity parameters such as ANPP recovered within five to ten years in North Carolina *Spartina* marshes. A similar recovery pattern has been shown in other systems such as tallgrass prairie. For example, Baer et al. (2002) measured ANPP and root biomass in a chronosequence of restored Conservation Reserve Program grasslands in Nebraska, and found that although ANPP did not show a clear pattern of increase with time since restoration because of site-specific management practices, root biomass showed a positive linear increase with time since restoration age.

Although recovery trajectories of biomass and productivity in the restorations we examined are encouraging, others have cautioned against using productivity measures as indicators of restoration success. For example, Whigham et al. (2002) examined restored depressional wetlands in Maryland and found that ANPP in sites was highly variable and more closely linked with precipitation than with time since restoration. Similarly, Platte River wetlands in our study showed an increase in both ANPP (in margins) and root biomass (in sloughs) from 2002 to 2003, and this was possibly linked to precipitation. Although snow and rainfall in both years was well below average for the region, precipitation in 2003 was over 10 cm higher than 2002, and most of that occurred during the growing season (Figure 12), which likely caused the observed increase in ANPP (e.g., Gross et al. 1990, Dunton et al. 2001).

In addition to total ANPP, specific groups of vegetation showed patterns of recovery in wetland margins. In particular, sedge and rush ANPP was highest in natural wetlands and lowest in new restorations. Sedges and rushes are an important component of freshwater wetlands, but are often slow to establish in restorations (e.g., Galatowitsch and van der Valk 1996a-c). For example, Mulhouse and Galatowitsch (2003) surveyed restored prairie pothole restorations and found a substantial component of aquatic species, but a distinct lack of sedge-meadow species. Difficulty in sedge establishment may be related to specific requirements of germination and survival of sedges (van der Valk et al. 1999, Budelsky and Galatowitsch 2000). Other vegetation groups such as forbs and grasses showed no distinct differences between wetland age classes, suggesting that sedges may be driving the differences in wetland ANPP. Patterns that we observed also provide further evidence that sedges and rushes may be good focal groups for assessments of wetland restoration status or recovery.

Although woody vegetation was not prevalent in the natural wetlands we examined, it was a common feature restorations. Severe changes in river discharge in the past century and a half due to impoundments on upstream reaches of the river, subsurface pumping, and diversion of water for irrigation have severely altered seasonal flow patterns, reduced annual peak and mean flows, and have ultimately resulted in woody encroachment onto previously denuded sandbars and riparian areas in the central Platte River valley (Williams 1978, Eschner et al. 1983, U.S. Fish and Wildlife Service 1981, Sidle et al. 1989, Richter and Powell 1996, Currier 1997). Increased numbers of mature trees in riparian habitats in this region represent abundant local sources for propagules of these historically uncommon wetland components. Land contouring for wetland restoration also exposes soils that are then readily colonized by woody species (Currier 1997).

The use of both univariate and multivariate analyses in our study enhanced our comparisons, as each technique showed some different patterns. For example, the ordination plot showed good separation of new restorations, old restorations, and natural wetlands, indicating differences in vegetation communities, but these patterns were not immediately evident from standard diversity analyses or ANOVA results. More importantly, although univariate analyses reflected recovery patterns for a variety of metrics in wetland margins, ordination results showed that older restorations are not on a trajectory toward natural systems. Rather, vectors showing the direction of plant community change indicate that the plant community change from new restorations to old restorations is moving in a different direction than from restorations to natural systems. This, along with some differences in measures of ecosystem function, suggests that restoration practices may not be driving recovery of the plant community in a trajectory toward natural systems. In other words, true “restoration”, as defined by Bradshaw (1996) may not be occurring in the central Platte River valley wetlands of central Nebraska.

Although some aspects of structure and function in restorations reflected natural conditions, particularly in sloughs as opposed to margins, our results indicate that the restored wetlands in this region are primarily replacement habitats, rather than true restorations. It is not clear whether persistent differences between restored and natural systems in this region are linked to shortcomings in restoration practices, or possibly because the recent drought cycle in the region limited recovery potential. Nonetheless, differences between restorations and natural systems still exist after up to 7 years. Future studies that examine longer time scales of recovery will reveal if recovery to natural conditions is possible in existing restorations. Likewise, investigations of the roles of other potentially important factors that we identified in this study (e.g., soil development and hydrology) may allow for modification of restoration and management practices to enhance recovery.

Acknowledgements

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Table 1. List of study sites, included age class, year of restoration, age of restored sites in 2003, and location. All sites are linear wet meadow slough wetlands located within the central Platte River Valley in south central Nebraska.

Site Name	Land Owner[1]	Age Classes	Year of Restoration	Age[2]	Hydroperiod[3]	Location
Mormon East	PRWCMT	Natural	----	----	4	Hall County
Mormon Middle	PRWCMT	Natural	----	----	4	Hall County
Mormon West	PRWCMT	Natural	----	----	0	Hall County
Uridil	PRWCMT	Older Restored	1995	7	0	Hall County
Johns Restoration	PRWCMT	Older Restored	1996	6	2	Phelps County
Studnicka	TNC	Older Restored	1996	6	2	Adams County
Speidell 2000	TNC	Newer Restored	2000	2	0	Kearney County
Speidell 2001	TNC	Newer Restored	2001	1	0	Kearney County
Derr	TNC	Newer Restored	2000	2	0	Adams County

[1] PRWCMT= Platte River Whooping Crane Maintenance Trust, Wood River, Nebraska
TNC= The Nature Conservancy, Central Nebraska Projects Office, Aurora, Nebraska

[2] Age= age in years of the restored wetland at the beginning of the study, in 2002

[3] Hydroperiod= number of months during 2003 that each site had standing water

Table 2. Percent similarity of plant communties in Platte River valley wetlands, sampled in spring and summer, 2003. Means are averages (standard error) of similarity comparisons of three sites in each of three age classes; natural wetlands=non-degraded wetlands, old= 6-7yr old restorations, and new=1-2yr old restorations. Significance values are the result of t-test comparing natural vs. old and natural vs. new restorations in both sloughs and margins.

	Sloughs	Margins
Natural vs. Old	29.1% (6.4)	33.7% (3.9)
Natural vs. New	13.5% (6.1)	16.0% (5.0)
p value=	0.15	0.05

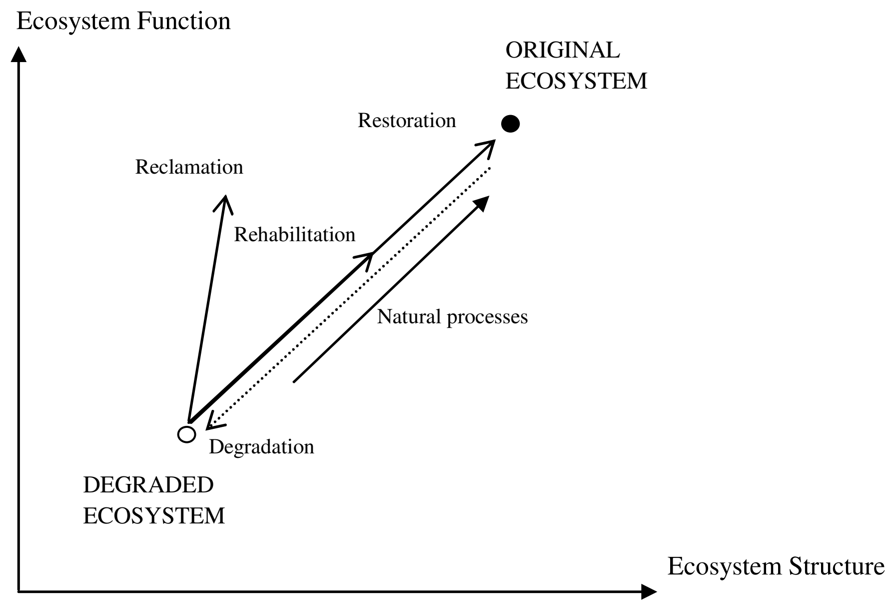


Figure 1. Options of recovery in ecological remediation, as expressed by Bradshaw (1996). Degraded ecosystems have decreased ecosystem structure (diversity, complexity, etc.) and function (biomass, productivity, etc.) relative to the original ecosystem. Restoration is the full recovery of the system back to its original structure and function. In rehabilitation, recovery is occuring in the trajectory of natural processes, but has not yet occured in full. In reclamation, the system may have increased structure or function relative to the original ecosystem, but recovery has created a different system (also called "replacement").

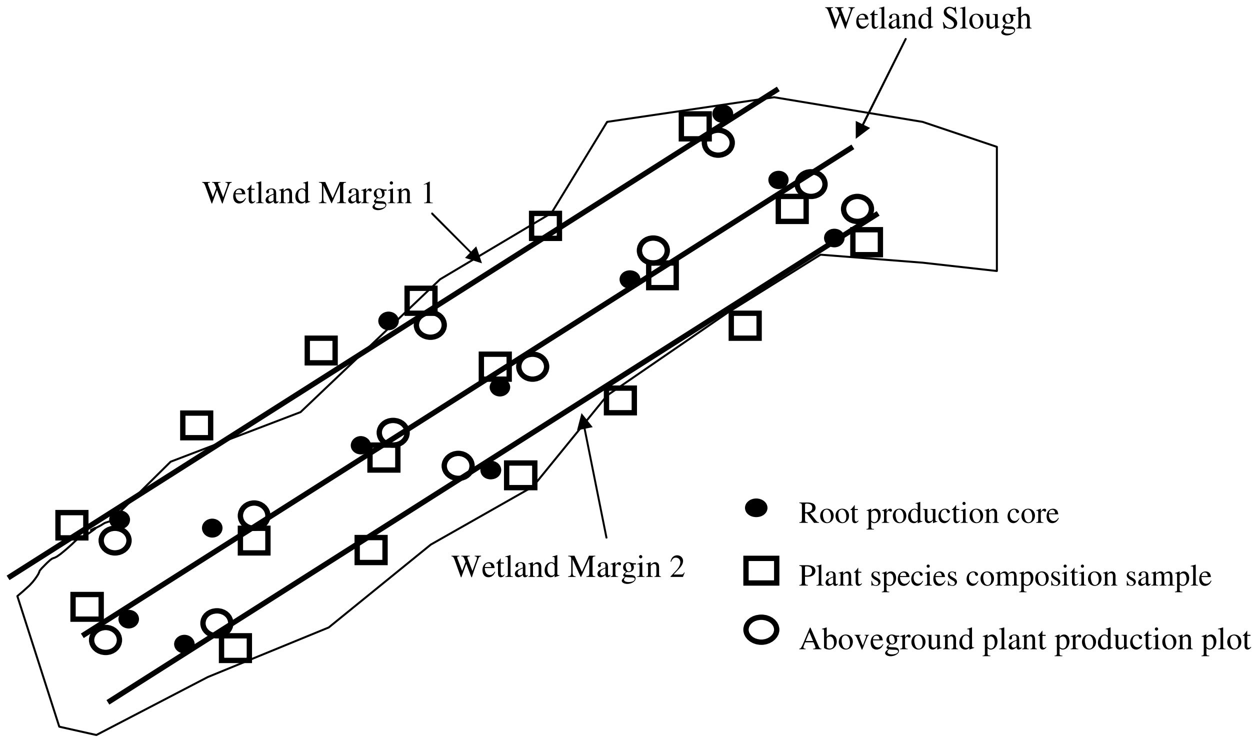
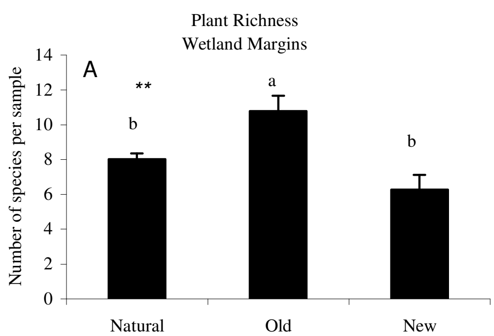


Figure 2. Sampling strategy for plant communities in Platte River wetlands. Diagram is of a typical wetland slough reach; lines represent wetland transects (one in slough, and two in margins) and symbols indicate approximate sampling locations for roots, plant cover (species composition), and aboveground plant production.



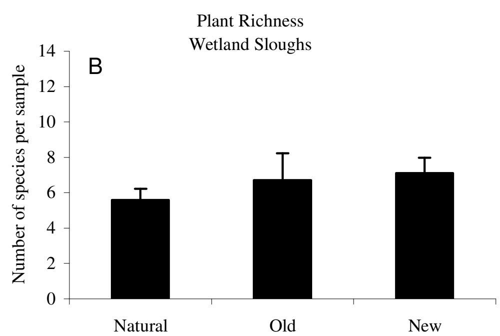


Figure 3. Average richness (no. of distinct taxa/sample) of plants found within margins (A) and sloughs (B) of Platte River wetlands sampled spring and summer, 2003. Bars are means of three sites in each of three age classes; natural= non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Error bars are one standard error. **Means with different letters are significantly different (p<0.05).

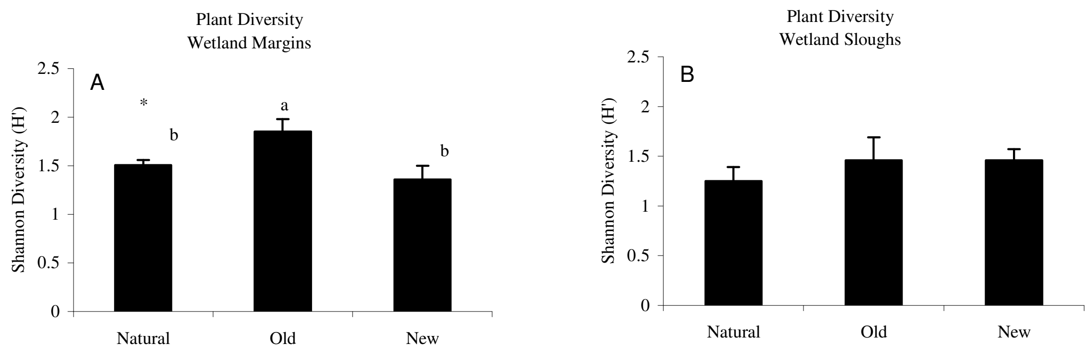


Figure 4. Plant diversity (Shannon Diversity) found within margins (A) and sloughs (B) of Platte River wetlands sampled spring and summer, 2003. Bars are means of three sites in each of three age classes; natural=non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Error bars are one standard error. *Means with different letters are significantly different ($p<0.10$).

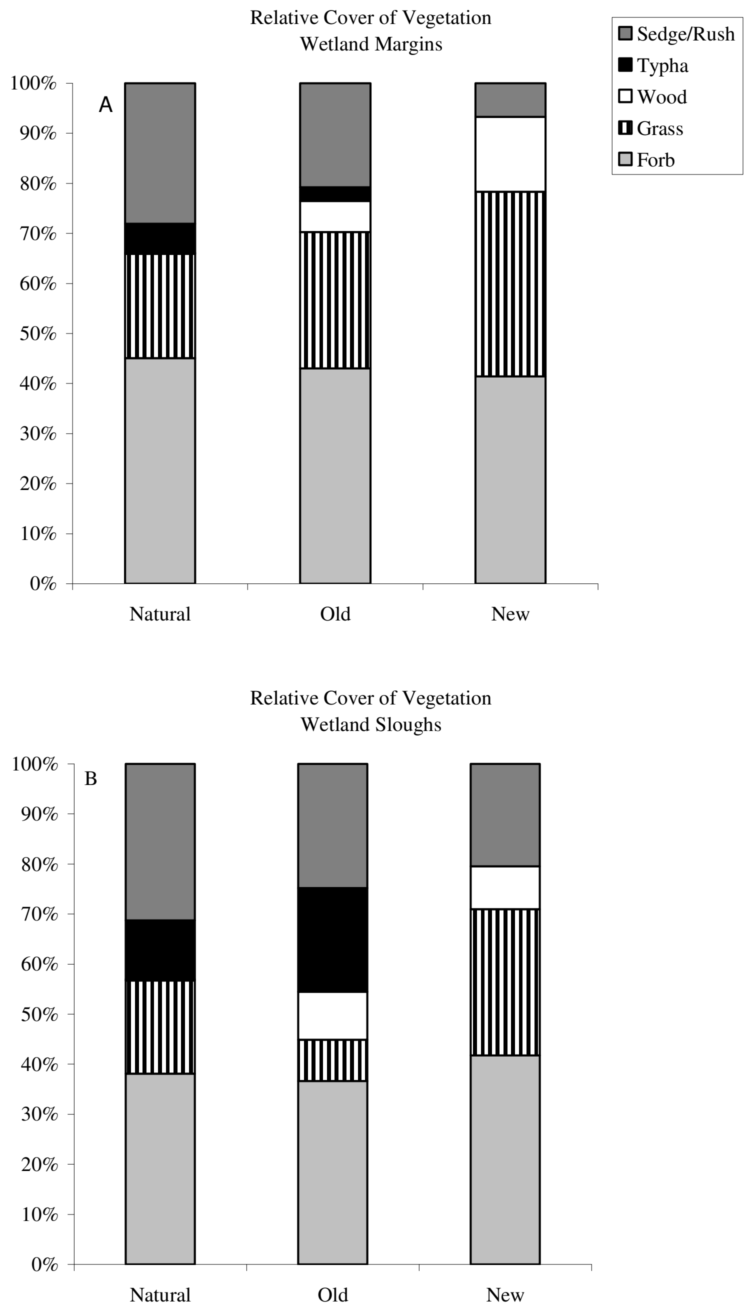


Figure 5. Relative cover of vegetation groups found within margins (A) and sloughs (B) of Platte River wetlands sampled spring and summer, 2003. Stacked bars are means of three sites in each of three age classes; natural= non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Within sloughs relative cover of woody vegetation was significantly higher in natural wetlands than new restorations ($F=6.07$; $DF= 2,6$; $p=0.036$). Relative cover of wood within margins significantly higher in new restorations than in natural wetlands and old restorations ($F=5.67$; $DF=2,6$; $p=0.041$).

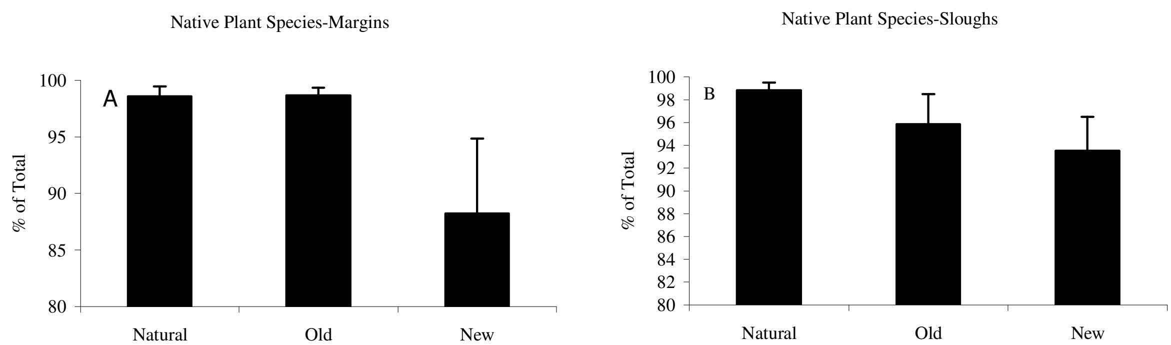


Figure 6. Percent of native species found within margins (A) and sloughs (B) of Platte River wetlands sampled spring and summer, 2003. Bars are means of three sites in each of three age classes; natural= non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Error bars are one standard error.

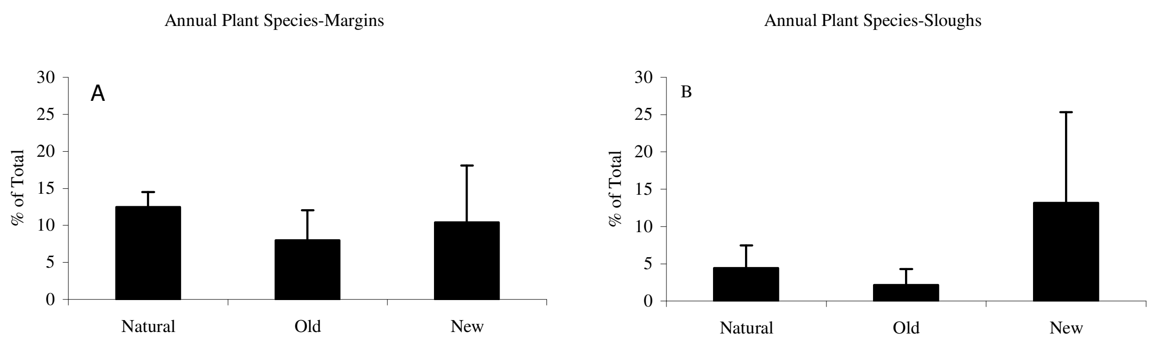


Figure 7. Percent of annual species found within margins (A) and sloughs (B) of Platte River wetlands sampled spring and summer, 2003. Bars are means of three sites in each of three age classes; natural= non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Error bars are one standard error.

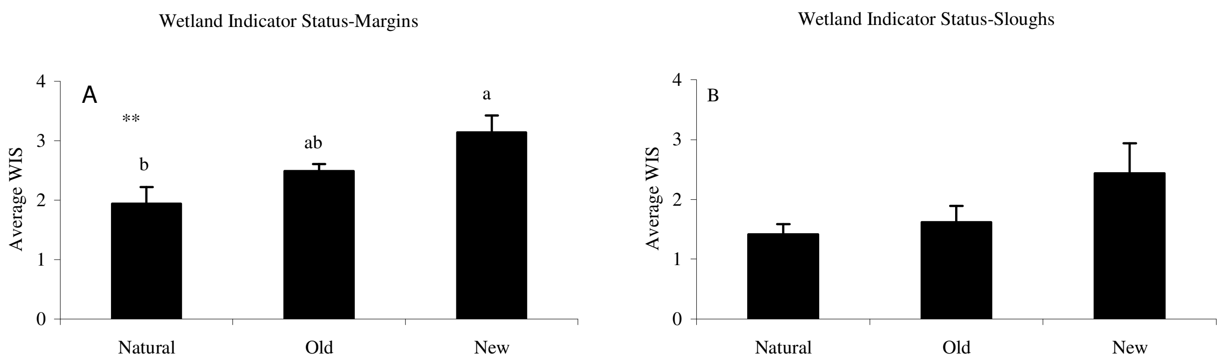


Figure 8. Average wetland indicator status (WIS) of plants found within margins (A) and sloughs (B) of Platte River wetlands sampled spring and summer, 2003. Averages are calculated based on WIS of plants as follows: obligates= 1, facultative wet= 2, facultative= 3, facultative upland= 4, upland= 5. Bars are means of three sites in each of three age classes; natural= non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Error bars are one standard error. **Means with different letters are significantly different (p<0.05).

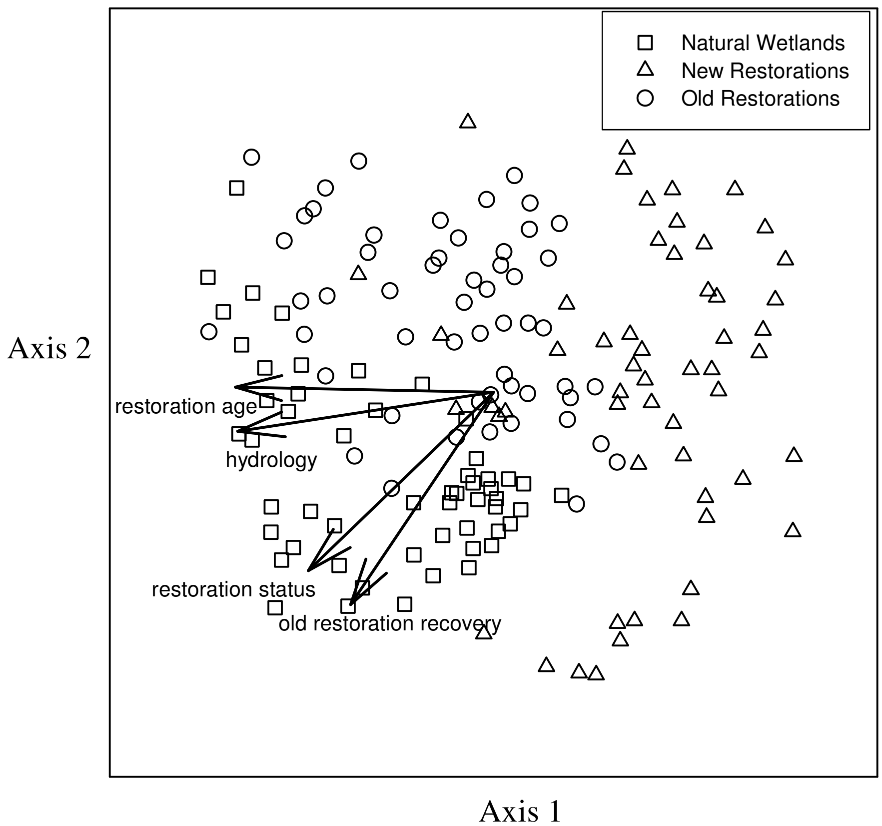


Figure 9. Two-dimensional NMDS Ordination Plot (stress=0.25) based on relative cover of vegetation using the Bray-Curtis dissimilarity index. Scatterplots consist of 108 plant composition plots; 18 in each of nine wetland sites in each of three ageclasses, natural= non-degraded wetlands, old=6-7yr old restorations, and new=1-2yr old restorations. Significant vectors ($p<0.0001$) were fitted for the following vectors: restoration age ($r=0.69$), restoration status ($r=0.68$, old restoration recover ($r=0.76$), and hydrology ($r=0.71$).

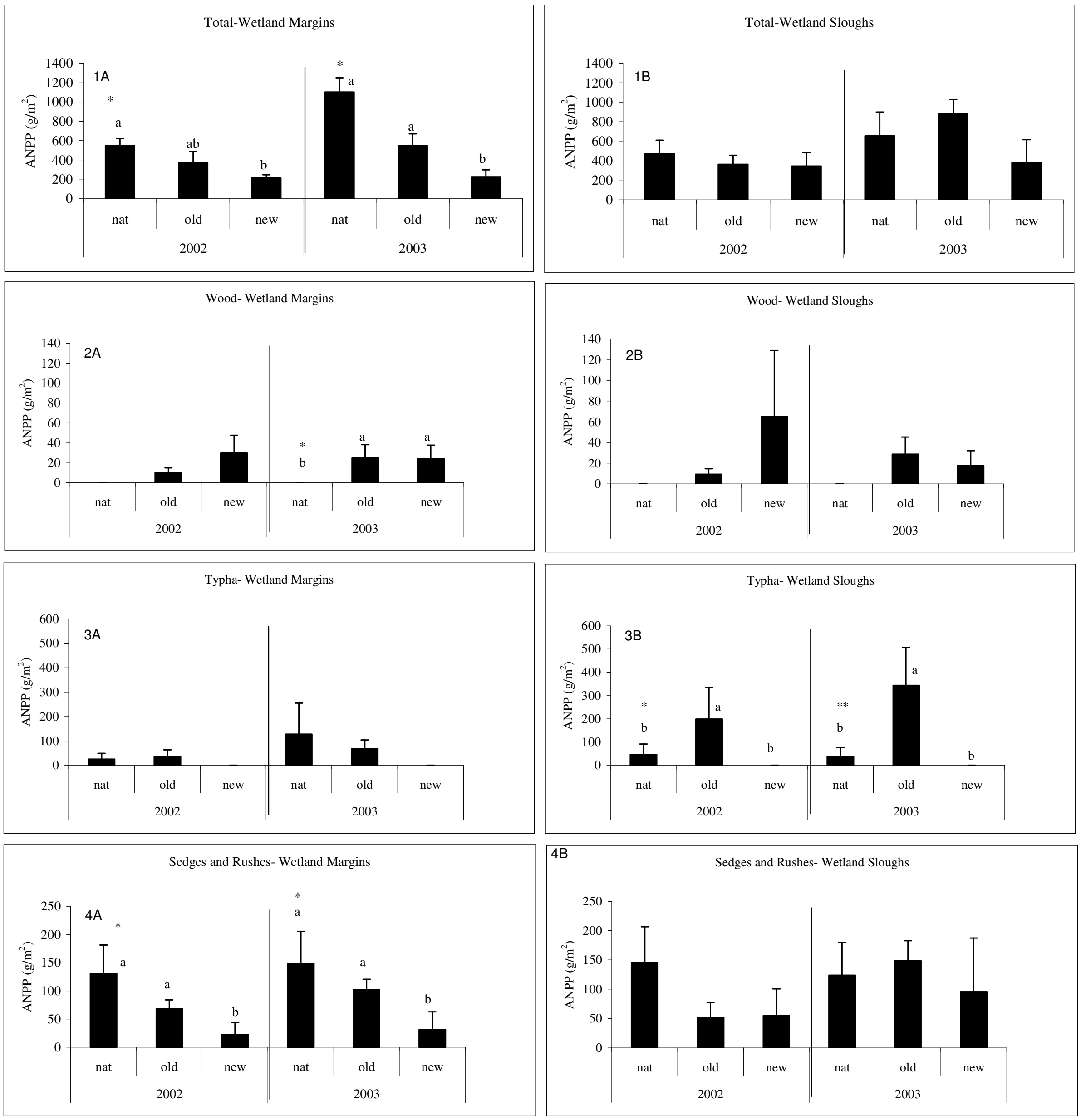


Figure 10. Aboveground Net Primary Productivity (g/m2) of total plant community (1), woody vegetation (2), typha (3), and sedges/rushes (4) found within margins (A) and sloughs (B) of Platte River wetlands sampled September, 2002 and 2003. Bars are means of three sites in each of three age classes; natural= non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Error bars are one standard error. Means with different letters are significantly different within year only. ** indicates significance at the p<0.05 level, and * indicates significance at the p<0.10 level.

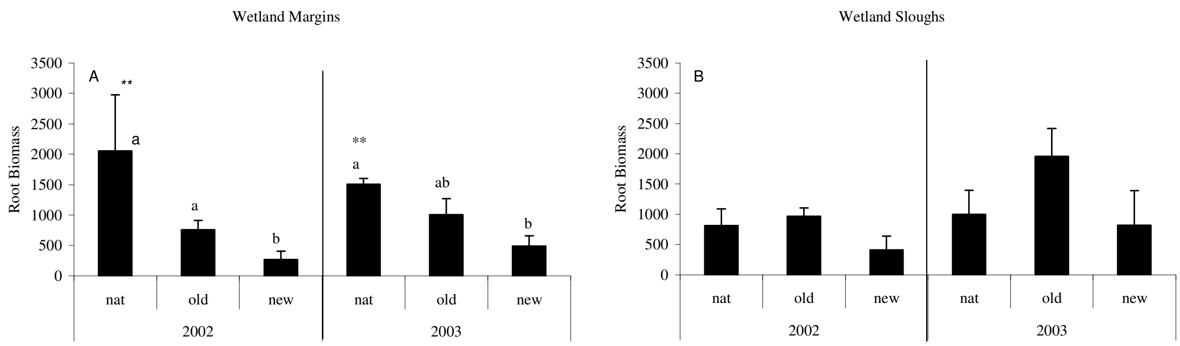


Figure 11. Root biomass (g/m2 at 20cm depth) found within margins (A) and sloughs (B) of Platte River wetlands sampled September, 2002 and 2003. Bars are means of three sites in each of three age classes; natural= non-degraded wetlands, old= 6-7yr old restored wetlands, new= 1-2yr old restored wetlands. Error bars are one standard error. **Means with different letters are significantly different within year only ($p<0.05$). Within wetland sloughs, root biomass was significantly higher in 2003 than 2002 ($F=8.32$; $DF=1,12$; $p=0.0279$).

Appendix 1. Anova tables for tests for differences between transects (margin vs. slough), and tests for interaction between transect location and ageclass (natural vs. old restoration vs. new restoration).

	Test between transects Margin vs. Slough				Test for ageclassxtransect interaction			
	p	F Statistic	Numerator DF	Denominator DF	p	F Statistic	Numerator DF	Denominator DF
Compositional Measures								
Richness	0.0270	6.34	1	12	0.0578	3.65	2	12
Diversity	0.1421	2.47	1	12	0.2471	1.57	2	12
% Annuals	0.3256	1.05	1	12	0.5952	0.54	2	12
% Natives	0.8278	0.05	1	12	0.7706	0.27	2	12
Ave. Wetland Indicator Status	0.0083	9.94	1	12	0.7854	0.25	2	12
Productivity Measures								
2002								
Total ANPP	0.9573	0.00	1	12	0.7658	0.27	2	12
Wood ANPP	0.7178	0.14	1	12	0.9664	0.03	2	12
Sedge and Rush ANPP	0.8001	0.07	1	12	0.6695	0.42	2	12
Typha ANPP	0.4027	0.75	1	12	0.5819	0.57	2	12
Root Biomass	0.9836	0.01	1	12	0.1626	2.12	2	12
2003								
Total ANPP	0.9096	0.01	1	12	0.2943	1.36	2	12
Wood ANPP	0.7134	0.14	1	12	0.9651	0.04	2	12
Sedge and Rush ANPP	0.8750	0.03	1	12	0.6473	0.45	2	12
Typha ANPP	0.6059	0.28	1	12	0.6003	0.53	2	12
Root Biomass	0.8669	0.03	1	12	0.5418	0.65	2	12

Chapter II

MACROINVERTEBRATE ABUNDANCE, BIOMASS, AND COMMUNITY STRUCTURE IN RESTORED AND NATURAL SLOUGH WETLANDS: EVALUATION OF RESTORATION PRACTICES IN THE CENTRAL PLATTE RIVER VALLEY

Introduction

Wetlands perform numerous valuable ecosystem services including flood control, groundwater discharge or recharge, and removal or transformation of nutrients such as nitrogen through denitrification and microbial immobilization (Schlesinger 1997). Because of their unique role in global biogeochemical cycles, wetlands are disproportionately important considering the total area they occupy (Schlesinger 1997).

Wetlands also provide important habitat for fish and wildlife, and harbor diverse biological communities, many of which are unique to intermittent aquatic habitats (e.g., Williams 1996, Semlitsch and Bodie 1998, Batzer et al. 1999). For example, one third of all North American bird species rely on wetland-derived resources (Keddy 2000). The importance of wetland habitats is now magnified and wetlands are now generally considered critical habitats, as much of the area that once existed as wetlands in North America has been destroyed or altered for agriculture since European settlement. Mitsch and Gosselink (2000) estimated that 53% of wetlands in the United States were lost between 1780's and 1980's alone.

The central Platte River basin in Nebraska consists of the braided channels of Platte River, which carries water from the North and South Platte drainages downstream of their convergence near North Platte, NE, and adjacent wet sedge meadows. Wet meadow systems are a matrix of mesic prairie with meandering linear depressional wetland sloughs found in low-lying areas. These sloughs range in hydroperiod from ephemeral to perennial, and water levels are affected by precipitation and river discharge through groundwater (Whiles and Goldowitz 1998).

This region is a focus for conservation efforts because of its importance to migratory birds, including the federally endangered Interior Least Tern (*Sterna antillarum*) and Whooping Crane (*Grus americana*), which utilize the area during migrations to acquire foods essential for successful migration and nesting (U.S. Fish and Wildlife Service 1997). Eighty percent of the continental population of Sandhill Cranes (*Grus canadensis*) also migrates annually through central Nebraska en route to arctic breeding grounds (Krapu et al. 1984). An added 7-9 million individuals of 300 other waterfowl species migrate annually through this region as well (Richter and Powell 1996). In addition to these high profile species, the central Platte Valley and its associated wetlands are also home to a diverse community of plants (up to 300 species) and other animals, some of which are species of concern in Nebraska (e.g., Woodhouse's toad [*Bufo woodhousii*], plains topminnow, [*Fundulus sciadicus*], brook stickleback [*Culaea inconstans*]). Wetlands in the Platte Valley are also home to a diverse and productive invertebrate community (Gordon et al. 1990, Davis and Vohs 1993, Whiles et al. 1999, Whiles and Goldowitz 2001) that includes a recently discovered caddisfly (*Ironoquia plattensis*), currently known only from these Platte River wetlands (Alexander and Whiles 2000).

Despite their importance to ecosystem integrity and wildlife, wetlands in the Platte basin have become highly endangered systems. Between 1938 and 1989 alone, 23-45% of wet meadows were drained for crop production and development (Currier 1997). Wet meadow habitat currently comprises <5% of the land area in the Platte River valley (Sidle et al. 1989, U.S. Fish and Wildlife Service 1997). Impoundments on upstream reaches of the river as well as subsurface pumping and diversion of water into canals for irrigation began as early as 1870 (Williams 1978), and have severely altered seasonal flow patterns by shifting annual peaks by as much as 29-38 days (Richter and Powell 1996), reducing annual peak and mean flows (Eschner et al. 1981), and have resulted in a ~70% reduction in discharge (Sidle et al. 1989). This diminished discharge has resulted in decreased scouring and shifting of streambed alluvium (Williams 1978, U.S. Fish and Wildlife Service 1981), increased depth to groundwater (Eschner et al. 1981, Currier 1997), declines in wetland hydroperiod, and encroachment of woody vegetation onto previously denuded sandbars and riparian areas of the Platte (Williams 1978, Eschner et al. 1981, U.S. Fish and Wildlife Service 1981). The combined effect of these alterations is substantial changes in Platte River hydrology and morphology, and in turn serious degradation of wetland and wet meadow habitats.

Recent heightened awareness of the importance of wetlands in the region has resulted in increased wetland restoration and creation efforts by groups such as the Nebraska Game and Parks, The Nature Conservancy, and the Platte River Whooping Crane Maintenance Trust. These groups have been implementing slough wetland restorations for over 12 years in this region by land contouring and seeding with species mixes found in nearby natural systems. Although they are an increasingly common feature in the region, and substantial amounts of time and money have gone into them, these restorations have not been adequately evaluated in terms of their hydrology, habitat quality, biotic communities, and overall ecosystem function relative to natural systems.

Because of their utility for biological assessments and their importance to ecosystem function, macroinvertebrates are both a logical focus of wetland restoration and a useful tool to assess restoration success. The use of macroinvertebrates as bioindicators of ecosystem integrity in aquatic systems in general, and in wetlands (Batzer et al. 1999, Spieles and Mitsch 2000), is well established because of their known sensitivities to changes in physical factors associated with disturbance and pollution (Hilsenhoff 1987, 1988, Resh et al. 1996, Karr and Chu 1998) and their intimate associations with other biota (e.g., plant communities [De Szalay and Resh 2000]). Whereas measures of physicochemical parameters often provide only a "snapshot" of baseline aquatic habitat condition, inhabitant biota can give a dynamic assessment of ecosystem health and integrate conditions over time (Resh et al. 1996, Barbour et al. 1999). Equally as important as their usefulness as bioindicators, invertebrates also play vital roles in processes such as energy flow and nutrient cycling and they represent important food for higher order consumers (Wallace and Webster 1996).

The main objective of this study was to compare macroinvertebrate communities (abundance, diversity, functional feeding groups, biomass) in restored and natural wetlands in order to assess whether restored wetlands can support the communities present in natural systems. Macroinvertebrates can colonize and recover relatively quickly after wetland restoration in some systems (e.g., Brown et al. 1997), and we predicted that many would do so in these systems because there are numerous nearby source areas for colonists and many taxa (e.g., most insects) have high vagility. However, habitat features and quality in restored sloughs (e.g., hydrology, substrates), and recovery of other groups such as plants, will also influence macroinvertebrate communities and we predicted they would limit some aspects of recovery. These limitations, when present, will likely be reflected in measures of taxonomic composition and functional structure, rather than total abundance and biomass, and differences may persist in older restorations unless physical factors such as hydrology and plant communities are very similar to natural sites.

Study Region

The study region lies within the central Great Plains, and is found along a 90km stretch of the central Platte River in south central Nebraska, roughly from 10km east of Grand Island, in Hall County, to near Elm Creek, in Phelps County. The landscape is a series of wet meadows with linear sloughs in low areas. Vegetation within the meadows is dominated by sedges (e.g., *Carex* spp., *Scirpus* spp., etc.) and grasses (e.g., *Spartina pectinata*, etc.), with a substantial component of forbs (e.g., *Verbena hastata*, *Alisma* spp., etc.). Climate is temperate with warm summer temperatures (July mean temp.= 24^{0}C) and cold winter temperatures (January mean temp.=-7^{0}C), often resulting in ice covered surface waters from November through March. Mean annual precipitation is 63cm/yr, most of which falls in May and June.

Study Sites

Four natural wetland sites and four restored sites (ranging from 5 to 11 years old at onset of study) were been chosen for comparison (see Table 1). Management histories (e.g., grazing and burning) were similar among all sites chosen and restoration procedures (e.g., land contouring and seeding) used for the restored sites were all similar. At each wetland site, all sampling and measurements were taken within a permanently marked 20m representative reach. All sloughs chosen for this study were intermittent, with dry periods typically occurring in late summer. However, this area has been under drought conditions for the past 3-4 years, which resulted in increased frequencies and durations of dry periods during this study.

Methods

Physical habitat parameters

Staff gauges were installed in the deepest points in each slough and read at ~bi-weekly intervals when water was present. Reference transects for wetted width and depth measurements were established at 10m intervals perpendicular to the slough within the 20m reach (at 0m, 10m, and 20m). Along each perpendicular transect, width of wetted area was measured, and depth was measured with a meter stick at 0.5 or 1m intervals, depending on slough width. Water volume was estimated from wetted width and depth transects, which were taken during a variety of hydrologic conditions (high and low extremes). Volume estimates were regressed against staff gauge readings to develop predictive equations of wetted volume. A thermograph was placed in each site to record temperature at 30min intervals when water was present. Spot measurements of dissolved oxygen, conductivity, specific conductance were measured at monthly intervals, and substrate composition was visually estimated in three random locations during invertebrate sampling at each site during each month when water was present.

Benthic invertebrate sampling and processing

Three macroinvertebrate samples were taken from random locations in each site seasonally when water was present during March, June, September, and December 2003. Samples were taken using a dipnet with a 655cm^2 aperture, and 0.5mm Nytex™ mesh. Sampling invertebrates in wetlands and other heavily vegetated aquatic habitats with any sampling device can be problematic. Turner and Trexler (1997) compared the performance of various wetland invertebrate sampling devices and found that a dipnet captured the highest diversity of invertebrates and showed lowest variability between samples. Additionally, to standardize the area sampled and minimize invertebrate avoidance of the net, a drop trap (1m tall, 0.43m wide, and 0.5m long), covered on four sides with 0.5mm Nytex™ mesh, was used. During sampling, the net was bounced along the substrate within the trap in alternating directions for a total of 5 sweeps. Calibration of this sampling technique in our sites showed that 5 sweeps captured 80-90% of invertebrates in the droptrap (Figure 1). Samples were elutriated through the net with clean water, placed inside pollination bags, labeled, and preserved in 8% formalin.

Samples were processed in the laboratory by washing through nested sieves to separate into coarse (>1mm) and fine (<1mm, >0.5mm) fractions. Invertebrates in coarse fractions were separated from debris with forceps under a dissecting microscope. Invertebrates in fine fractions were subsampled with a Folsom wheel sample splitter, usually to 1/4-1/16 of the original volume. Samples containing voluminous amounts of filamentous algae that could not be split with the Folsom wheel were split in sorting trays. For this procedure, the sample spread evenly into a white picking tray. Algae and associated materials were then subsampled by placing a steel ring (3cm dia. = 1/10 of surface area of the tray) with a sharp edge in a random location within the tray and forcing it through the materials to the bottom of the tray. All material was then removed from within the interior of the ring and invertebrates were separated with forceps. Ring subsamples were taken until 100-200 invertebrates were removed.

Invertebrates were measured (total body length) and identified to the lowest practical taxonomic level. For most insects and other taxa, this was the generic level. Problematic taxa that were not identified to genus include Oligochaeta, Crustacea, and Chironomidae. Functional assignments were based on information in Merritt and Cummins (1996), Smith (2001), or our knowledge of local fauna.

Analysis of macroinvertebrate data

Literature values of length-specific biomass (Bottrell et al. 1976, Benke et al.1999) were used to estimate biomass of individual taxa. Abundance and biomass data were analyzed using repeated measures ANOVA (SAS Institute 1988), with each site considered a subject, and each month representing a repeated measure of that subject. There were four replicates of each wetland treatment (natural vs. restored). Whenever water was not present at a site on a given date, that site was not included in analysis for that date. Whenever data did not meet the assumptions of ANOVA (i.e., normality, equality of variances), values were log-transformed.

Results

Macroinvertebrate Abundance and Diversity

Seventy five macroinvertebrate taxa were collected at all sites combined during this study. Annual average macroinvertebrate abundance was 5864 individuals/m^2 in natural wetlands and 16,239 individuals/m^2 in restored wetlands, but was not significantly different between the two wetland types (Table 2). Although again not statistically significant, there was a similar trend of lower annual average total biomass in natural wetlands (1,382.9 mg DM/m^2) compared to restored wetlands (2,102.1 mg DM/m^2) (Table 2). Macroinvertebrate abundance peaked in March in restored sites (Fig. 2A) and in June in natural sites (Fig. 2A). Biomass peaked in both wetland types in September (Fig. 2B).

Total and average taxon richness and Shannon diversity were both slightly higher in restored sites than natural sites (Table 2). Natural sites all had at least 1 unique taxon (Wild Rose East=4, Mormon Mid=4, Mormon East=1), and restored sites ranged from 0 to 10 unique taxa (Nature Center= 10, Johns Restoration= 4, Johns Clearing= 0).

Invertebrate abundance was dominated by non-insect groups, which accounted for 64% and 83% of total abundance in natural and restored wetlands, respectively (Table 3). Annelids and crustaceans were the most well-represented non-insect groups in both wetland types (Table 3). Insects accounted for 38% of total wetland invertebrates in natural wetlands, and only 16% of abundance in restored wetlands.

In contrast to abundance, insects and mollusks constituted the bulk of biomass in natural and restored wetlands (Table 4). Mollusks contributed 23% of total biomass in natural wetlands and consisted mostly of fingernail clams (Sphaeridae) followed by the snail genera *Physella* and *Fossaria*. In contrast, mollusks dominated biomass in restored wetlands (61%) and consisted almost solely of *Physella* and *Fossaria*. Annelid biomass in natural and restored wetlands represented 3% and 15% of total biomass respectively. Only oligochaetes were found in restored wetlands, whereas natural wetlands also supported leeches (*Erbopdella* and *Placobdella*). Crustacean biomass in natural wetlands consisted mostly of amphipods (*Crangonyx* and *Hyalella*), while in restored wetlands ostracods contributed over half of crustacean biomass (Table 4). Planarians (*Dugesia*) accounted for 10% of total biomass in natural wetlands, and only ~1 percent in restored wetlands.

Insect biomass accounted for 55% of total biomass in natural wetlands (Table 4), and was dominated by beetles (mostly Dytiscids and Hydrophilids) and flies (mostly chironomids, and *Culex* in one natural site). In restored wetlands, insects accounted for only 22% of total biomass, which consisted of chironomids, beetles, and odonates (mostly *Enallagm*a and *Anax*).

Macroinvertebrate Functional Structure

Functional structure based on abundance showed subtle differences between natural and restored wetlands. All sites we sampled were dominated by collector-gatherers, which, on average, represented 73% and 95% of total abundance in natural and restored wetlands, respectively (Table 2). Collector-filterers were significantly more abundant in natural wetlands ($p=0.0007$), and averaged 29% of total abundance in natural wetlands and <1% in restored wetlands (Table 2).

Based on biomass, functional structure was more evenly distributed among groups in both natural and restored wetlands (Table 2). In both types of wetlands, collector-gatherers dominated biomass (Table 2). In natural wetlands, predators had the second highest contribution to biomass, followed by herbivores and collector-filterers, which had significantly higher biomass in natural wetlands than in restored wetlands (p=0.0038) (Table 2). Restored wetland biomass consisted mostly of collector-gatherers and scrapers, with the third highest contribution to biomass from predators (Table 2).

Discussion

Platte River slough assemblages

In general, macroinvertebrate densities and assemblage structure in the Platte River valley wetlands that we examined were similar to other wetland communities in the Great Plains region (Whiles and Goldowitz 2001, Euliss et al. 1999, Hall et al. 1999, Lovvorn et al. 1999, Gordon et al. 1990). These similarities are likely related to similarities in the habitat template among wetlands. Because of the intermittent nature of wetland habitats, wetland invertebrate assemblages are typically composed of taxa with drought-resistant life stages (e.g., resistant eggs or pupae) and/or life history adaptations to escape drying (e.g., mechanisms for migrating to inundated areas, rapid life cycles), as well as adaptations to living in the low oxygen conditions that are typical of shallow, temporary waters (e.g., air-breathing, presence of hemoglobin) (Williams 1996, Wiggins et al. 1980, Sharitz and Batzer 1999, Whiles et al. 1999). Invertebrate abundance and biomass in our sites was dominated by non-insect groups such as crustaceans, oligochaetes, and gastropods, and insect groups with substantial contributions to abundance and biomass included chironomids, hydrophilids, dytiscids, and corixids, all of which are common wetland inhabitants. One notable exception was the abundance of a CPOM-shredding caddisfly, *I. plattensis*, in one of our natural sites. The presence of this caddisfly is unusual because shredders are often underrepresented in wetland habitats (Wissinger 1999), and it is currently only known to exist in a handful of intermittent Platte River wetlands (Alexander and Whiles 1998, Whiles et al. 1999), including one of our natural study sites.

Restored versus natural sites

This study represents the first intensive investigation of invertebrate communities in restored wetlands in the central Platte River basin. As such, results of this study are an important step toward assessing the effectiveness and success of current restoration practices. While some metrics that we examined (e.g., total abundance, diversity) suggest that restorations are similar to natural sites, others (e.g., assemblage and functional structure) indicate persistent differences, suggesting that recovery in restorations may take longer than the time frame considered in our study.

Our comparison of natural and restored wetlands yielded few differences in terms of total abundance, biomass, and diversity. Although not statistically significant, average mean abundance and biomass during 2003 was actually higher in restored wetlands than natural systems, and this pattern has been observed in other wetland restoration studies. For example, Levin et al. (1996) found that macroinvertebrate richness in a reconstructed salt marsh was similar to natural salt marsh richness within 4 years. Craft et al. (1999) found that 5 years following restoration of a *Spartina* marsh in North Carolina, macrophyte communities and primary productivity had recovered to levels found in natural marshes, and density and richness of benthic fauna exceeded that of natural sites after 15-25 years. Given the extent to which our restored sites had been degraded prior to the restoration process (e.g., row-cropped fields prior to restoration; Currier 1997, Sidle et al. 1989, USFW 1981), it was surprising that some metrics (e.g., total abundance, biomass, and diversity) showed recovery patterns within a similar time frame as has been seen in studies of other restored wetland systems (Levin et al. 1996, Craft et al. 1999).

Functional structure

Although some aspects of wetland structure and function may recover quickly, others may recover more slowly (e.g., McKenna 2003). Whereas total abundance and biomass showed patterns suggesting recovery in our restored sites, we observed differences in functional feeding groups, particularly in terms of biomass, between natural and restored sites. These differences are potentially important, as macroinvertebrate functional structure is closely linked to ecosystem function in freshwater systems (e.g., Wallace and Webster 1996), and thus may reflect persisting differences in ecosystem processes and function between restorations and natural sites.

Differences in functional structure between natural and restored sites were particularly evident and statistically significant with collector-filterers, which accounted for an average of over ¼ of total macroinvertebrate abundance and 14% of biomass in natural wetlands, but were less than 1% of either in restorations. These patterns were driven by two taxa. Fingernail clams, *Pisidium*, were found in much higher numbers in natural sites, perhaps because of their limited dispersal ability compared to insects with flying adult stages (e.g., Brown et al. 1997). *Culex*, a mosquito, was common in two of the natural sites (Mormon East, Mormon Mid), but not at any of the restored sites. Reasons for the absence of *Culex* in restorations are less clear, and are likely related to habitats specifics that we did not measure.

Differences in functional structure between natural and restored sites were also evident in predator biomass. Predatory taxa such as odonates and dytiscid beetles were the second highest contributors to biomass in natural sites, accounting for a much higher proportion of total biomass than in restorations. Others have noted a lag period for recovery of predator richness in temporary waters (Williams 1996), and this is likely linked to the relatively larger size, longer life cycles, and naturally lower population densities of predators compared to lower trophic levels.

Scrapers also differed between restorations and natural sites, as they accounted for almost half the biomass in restored wetlands and only 10% in natural sites. This pattern was mostly driven by two genera, *Physella* and *Fossaria*, which were found in exceptionally large numbers in the Johns Clearing restoration. The abundance of these snails, particularly the physids, in restorations may have been related to lower levels of predators, as they are particularly vulnerable to a variety of predatory invertebrates that can control their populations (Alexander and Covich 1991, Tripet and Perrin 1994, Gray and Dodds 1998, Aditya and Raut 2002).

Drought conditions during our study period greatly reduced our sampling effort in the sites because hydroperiods were significantly shortened. For example, in a previous study of emergence production in several natural wetland sloughs within the region, including all of the natural sites included in the current study (Whiles and Goldowitz 2001), the Mormon East site held water for ~9 months during 1997, compared with 4 months of inundation during 2003. Additionally, although the natural site at Wild Rose East had historically been considered permanently wet (Whiles and Goldowitz 2001 and B.Goldowitz, personal communication) the site dried for ~3 weeks during Sept. – Oct. during our investigation.

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Table 1. Physical characteristics of the four natural and four restored wetland sites sampled durng 2003. All sites are representative 20m reaches of linear sloughs. Average or maximum water depths are readings averaged from March-December, 2003. In some cases, when water was only present for one month, the values are the same.

Natural Sloughs	Site Location	Year restored	Number of months* with water present	Avg. water depth at staff guage (cm)	Max. water depth at staff guage (cm)	DO (mg/L)	Conductivity (µS)	Water Temp (^oC)	pH
Mormon East	Hall County	———	4	51.4	65.2	8.3	1454.0	22.1	8.0
Mormon Mid	Hall County	———	4	21.4	42.1	5.7	1219.0	15.4	7.9
Wild Rose West	Hall County	———	1	3.7	3.7	13.0	1808.0	19.8	5.1
Wild Rose East	Hall County	———	10**	9.0	13.4	7.7	860.0	17.2	7.2
Restored Sloughs									
Johns Clearing	Phelps County	1992	10***	15.8	31.1	8.1	1110.0	16.2	8.4
Uridil	Hall County	1995	0	NA	NA	NA	NA	NA	NA
Johns Restoration	Phelps County	1996	2	11.6	11.6	3.4	935.5	17.4	NA
Nature Center	Hall County	1998	4	34.1	56.7	10.8	825.0	18.1	8.5

*Sample period spanned 10 months; March through December, 2003
** WR East went dry from ~October 27-November 3, 2003
***Johns Clearing was dry in 80% of the sample reach from Aug 15-Sep 11, 2003

Table 2. Macroinvertebrate mean abundance, mean biomass, functional structure, and measures of diversity at the four natural and four restored wetland sites in the central Platte River valley during 2003. Values in parentheses following abundance, biomass, and diversity metrics are 1 standard error. Values in parentheses following functional groups are percentages of total abundance or biomass. Numbers followed by different letters are significantly different at p=0.05, within functional feeding groups only.

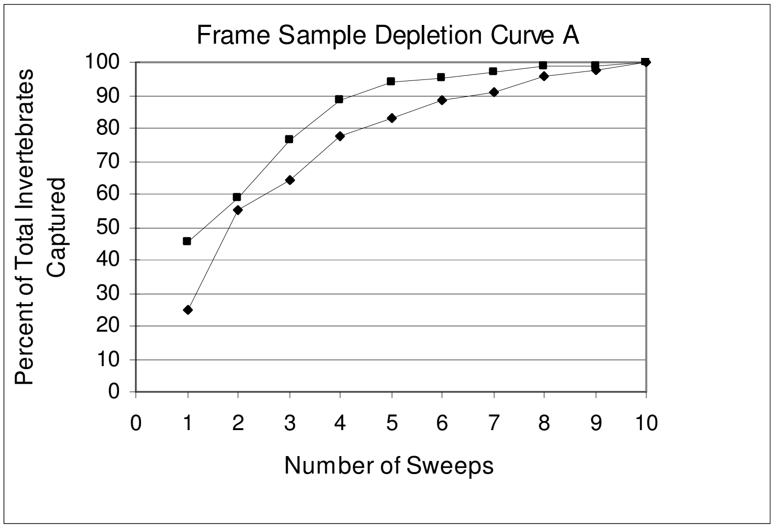
Metric	Natural	Restored
Abundance (no./m^2)	**5864.0 (1176.9)**	**16239.1 (5789.4)**
Collector-gatherers	4258.7 (73%)	15479.2 (95%)
Collector-filterers	1709.8[a] (29%)	0.4[b] (<1%)
Scrapers	305.8 (5%)	516.0 (3%)
Shredders	20.7 (<1%)	17.9 (<1%)
Herbivores	12.8 (<1%)	0.4 (<1%)
Predators	76.8 (1%)	363.4 (2%)
Biomass (mg DM/m^2)	**1382.9 (229.1)**	**2102.1 (445.0)**
Collector-gatherers	401.0 (29%)	994.8 (47%)
Collector-filterers	187.1[a] (14%)	0.1[b] (<1%)
Scrapers	141.8 (10%)	910.7(43%)
Shredders	20.5 (1%)	20 (1%)
Herbivores	241.0 (17%)	10.9 (1%)
Predators	391.6 (28%)	282.4 (13%)
Average taxon richness	**14.0 (0.6)**	**14.6 (1.1)**
Total taxon richness	**53.0**	**66.0**
Shannon Diversity	**1.3 (0.1)**	**1.5 (0.1)**

Table 3. Average abundance (no./m^2) and percent contribution of macroinvertebrate taxa in natural and restored wetlands in the central Platte River valley. Percent contribution of major groups is the percent of total macroinvertebrate abundance. Percent contribution of each taxon within a group is the contribution to that group.

	Natural		Restored	
Taxon	No./m^2	%	No./m^2	%
Tricladida	**293.7**	**6**	**44.3**	**<1**
Nematoda	**20.5**	**<1**	**49.1**	**<1**
Annelida	**920.9**	**18**	**5719.7**	**41**
Oligochaeta	904.9	98	5719.7	100
Hirudinea	16.0	2	0.0	0
Crustacea	**1192.6**	**23**	**4989.5**	**36**
Cladocera	62.0	5	234.9	5
Ostracoda	548.2	46	4003.5	80
Copepoda	91.7	8	727.3	15
Amphipoda	490.8	41	23.9	<1
Insecta	**1945.5**	**38**	**2373.3**	**17**
Collembola	0.5	<1	16.7	1
Ephemeroptera	0.0	0	39.5	2
Odonata	12.5	1	102.5	4
Hemiptera	8.4	<1	7.2	<1
Coleoptera	38.9	2	31.8	1
Trichoptera	10.9	1	1.8	<1
Diptera	1874.3	96	2173.8	92
Molluska	**811.4**	**16**	**779.1**	**6**
Lymnaeidae	96.2	12	413.2	53
Physidae	38.8	5	363.0	47
Planorbidae	74.7	9	2.6	<1
Sphaeriidae	601.7	74	0.3	<1

Table 4. Average biomass (mg/m^2) and percent contribution of macroinvertebrate taxa in natural and restored wetlands in the central Platte River valley. Percent contribution of major groups is the percent of total macroinvertebrate biomass. Percent contribution of each taxon within a group is the contribution to that group.

	Natural		Restored	
Taxon	mg/m^2	%	mg/m^2	%
Tricladida	**113.9**	**10**	**15.3**	**1**
Nematoda	**0.0**	**0**	**0.1**	**<1**
Annelida	**30.7**	**3**	**361.2**	**15**
Oligochaeta	26.8	87	361.2	100
Hirudinea	3.9	13	0.0	0
Crustacea	**98.6**	**9**	**43.7**	**2**
Cladocera	1.1	1	4.0	9
Ostracoda	3.2	3	23.1	53
Copepoda	0.1	<1	1.0	2
Amphipoda	94.3	96	15.7	36
Insecta	**595.4**	**55**	**533.7**	**22**
Collembola	0.0	0	0.7	<1
Ephemeroptera	0.0	0	12.9	2
Odonata	3.6	1	72.5	14
Hemiptera	21.7	4	9.5	2
Coleoptera	351.5	59	158.0	30
Trichoptera	1.8	<1	0.1	<1
Diptera	216.9	36	280.0	52
Molluska	**250.3**	**23**	**1519.1**	**61**
Lymnaeidae	67.5	27	701.7	46
Physidae	58.8	23	815.4	54
Planorbidae	15.6	6	1.9	<1
Sphaeriidae	108.6	43	0.1	<1



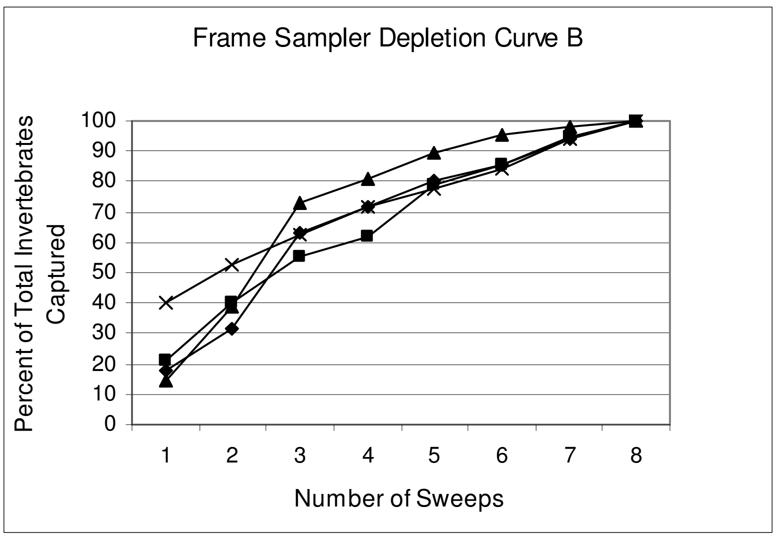


Figure 1. Depletion curve for a series of sweeps taken to calibrate a drop trap sampler in Platte River wetlands. Samples were taken at two different locations: Wild Rose East, a relatively non-vegetated site, in November of 2002 (A) and Mormon East, a vegetated site, in April 2003 (B). Each plotted line refers to a different location within the wetland slough. Drop traps were placed in an undisturbed location within the slough and a series of sweeps with a dip net were taken in alternating directions. Contents of the net were emptied and preserved after each sweep. In both locations, 5 sweeps resulted in 80-90% capture of macroinvertebrates.

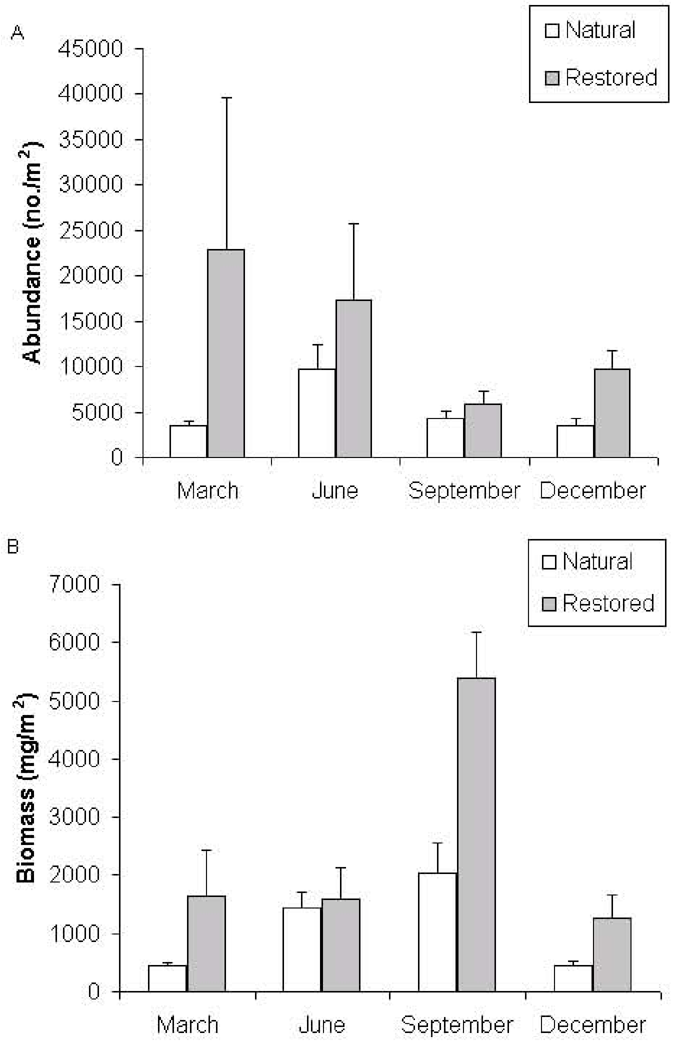


Figure 2. Seasonal average (+/-standard error) total macroinvertebrate abundance (A) and biomass (B) in the natural and restored wetland sites in the central Platte River Valley during 2003.