

RESPONSE OF NONGAME BIRDS AND TERRESTRIAL
INVERTEBRATES TO HIGH-DIVERSITY AND LOW-
DIVERSITY RESTORATION PLANTINGS ON
UPLAND GRASSLANDS IN THE
RAINWATER BASIN REGION,
NEBRASKA

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Bachelor of Science

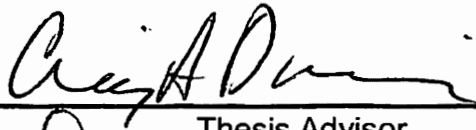
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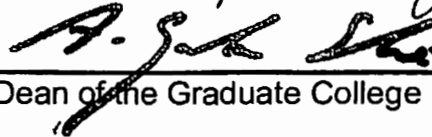
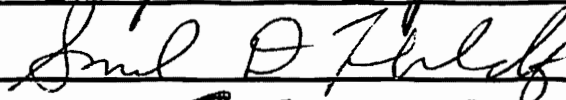
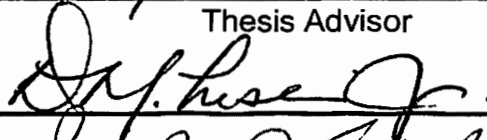
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CHAPTER 1: INTRODUCTION AND JUSTIFICATION

LITERATURE REVIEW

Grassland avifauna

Grassland birds have “shown more consistent and steeper, geographically widespread declines than any other grouping of North American species” (Samson and Knopf 1994:418). From 1969 to 1991, grassland bird declines (ranging from 24–91%) were greatest in Illinois, Minnesota, Wyoming, Nebraska, and Missouri (Samson and Knopf 1994). These declines have been attributed to loss and degradation of breeding and wintering habitat and encroachment of woody vegetation due to fire suppression and woody plantings. Additionally, brown-headed cowbirds (*Molothrus alter*) have benefited from the encroachment of woody vegetation because there are more perches available for female cowbirds to use for locating potential host nests, resulting in increased brood parasitism of grassland birds (Knopf 1986, Johnson and Temple 1990, Samson and Knopf 1994). With the loss of large grassland patches, habitat fragmentation has become another factor that has contributed to the decline in grassland bird populations (Johnson and Temple 1990, Herkert 1994a, Vickery et al. 1994, Helzer and Jelinski 1999, Winter and Faaborg 1999, O’Leary and Nyberg 2000, Johnson and Igl 2001). With increased fragmentation, there has been a concomitant increase in edge habitat, resulting in greater predation and parasitism (than in interior habitats) of grassland birds (Johnson and Temple 1986).

Habitat loss and degradation.—The primary cause for grassland bird declines is loss of habitat. Since the early 1800s, grasslands have been intensely altered due primarily by agricultural activities, and they are now among North America’s most

endangered ecosystems (Noss et al. 1995). Native prairies in some regions of North America have declined as much as 99.9% (Samson and Knopf 1994). Due to these habitat losses, one-third of North American grassland bird species have declined at statistically significant rates (Knopf 1996). Habitat degradation due to overgrazing also has been implicated in the decline of local and possibly regional grassland bird populations (Peterjohn and Sauer 1999). At high densities, grazing animals (especially cattle) can change vegetation structure, reduce biodiversity, and increase soil erosion (White et al. 2000). Although not all grassland birds are impacted negatively by intense grazing pressure (e.g., mountain plovers [*Charadrius montanus*], horned lark [*Eremophila alpestris*]), several studies have shown that some grassland birds are absent or dramatically decline in areas that are grazed heavily. For example, Henslow's sparrows (*Ammodramus henslowii*) are generally absent from heavily grazed grasslands because this species requires dense, undisturbed standing dead vegetation with a well-developed litter layer, which is usually absent from heavily grazed grasslands (Zimmerman 1988, Herkert 1994b, Herkert 1994c, Cully and Michaels 2000).

An increase in hay cropping also may have contributed to grassland bird declines. Hayfields are now cut more frequently and at earlier dates, which has a direct impact on many grassland species during the breeding season by destroying nests with eggs or nestlings and indirect impacts on grassland birds by making the hayfield unattractive for nesting because of the short plant height (Bryan and Best 1991, Peterjohn and Sauer 1999). Bollinger et al. (1990) noted that early hay-cropping resulted in the failure of 85% of bobolink (*Dolichonyx oryzivorus*) nests in hayfields in west-central New York. Bryan and Best (1991) found that mowing of grassed waterways in Iowa impacted nesting grassland birds by reducing vegetation height and density, which made the waterways unattractive to birds that prefer tall and dense cover, and by destroying a large number of the nests because mowing often coincided with peaking

nesting. Mowing accounted for the second greatest cause of nest failure in the waterways next to predation.

Effects of brood parasitism.—Brown-headed cowbirds parasitize nests of most grassland bird species (Peterjohn and Sauer 1999). Cowbirds are more effective brood parasites of grassland birds near wooded edge habitats (Johnson and Temple 1990). It is possible that elevated tree limbs along and in grasslands provides perches for cowbirds to more effectively locate and monitor host nests. With the increase in wooded edges in grasslands throughout the Great Plains, brood parasitism by cowbirds is becoming a growing threat to the nest productivity of many grassland bird species (Johnson and Temple 1990).

Parasitism rates of grassland birds are influenced by nest characteristics (e.g., nest substrate, height, concealment), habitat features (e.g., habitat type, distance of nest to habitat edge and to perches), host species defense, and distance from grazed areas (Shaffer et al. 2003). Because parasitism rates are influenced by many factors, parasitism rates of grassland birds throughout the Great Plains are highly variable. For example, grasshopper sparrows (*Ammodramus savannarum*) in Kansas had parasitism rates of 22–58% (Hill 1976, Klute et al. 1997), while grasshopper sparrows in North Dakota had parasitism rates of 2–26% (Koford et al. 2000, Granfors et al. 2001). Dickcissel (*Spiza americana*) parasitism rates were as low as zero in Illinois (Robinson et al. 2000) and as high as 95% in Kansas (Elliott 1978).

Effects of habitat fragmentation.—Habitat fragmentation also has been associated with grassland bird declines (Herkert 1994a, Johnson and Igl 2001). Habitat fragmentation involves separation of large areas of habitat into smaller areas that are isolated from each other resulting in reduced areas of core habitats for nesting birds, increased edge effects, and isolation from source populations (Johnson and Igl 2001). Johnson and Temple (1986) found that grassland birds that nested in remnants of tall-

grass prairie near wooded edges produced fewer young than birds that nested farther from wooded edges. Several studies also have reported that decreasing patch size negatively affects abundance and presence of many grassland bird species; however, most of those studies did not provide demographic data (e.g., nest success, fledgling rate, and survival), which is critical for determining productivity of these patches (Herkert 1994a, Vickery et al. 1994, Helzer and Jelinski 1999, Winter and Faaborg 1999, O'Leary and Nyberg 2000, Johnson and Igl 2001). A study by Johnson and Temple (1990) that did rely on demographic data found nest predation rates for 5 grassland bird species to be lower for nests on large fragments (≥ 130 ha) and in areas farther from a wooded edge (≥ 45 m), but those results are in question because data were pooled (Johnson and Temple 1990, Johnson 2001).

Others studies have used artificial nests to determine the influence of patch size and edge on nest success and have found that increased predation occurs closer to edges and within smaller grassland fragments (Burger et al. 1994, Davison and Bollinger 2000). Edge clearly can reduce abundance and nest success in grassland bird species through increased predation, changes in microclimate, and higher disturbance levels (Johnson and Temple 1986, Helzer 1996, O'Leary and Nyberg 2000, Johnson 2001). Several studies have shown that grassland bird abundance and nesting success are impacted negatively by edge. Helzer and Jelinski (1999) found that abundances of grasshopper sparrow and bobolink were higher >75 m from wooded edges than <50 m from wooded edges. O'Leary and Nyberg (2000) found that savannah sparrows (*Passerculus sandwichensis*), grasshopper sparrows, Henslow's sparrows, eastern meadowlarks (*Sturnella magna*), and bobolinks did not use edge as much as interior areas. Johnson (2001) reported that predation rates of Henslow's sparrow and dickcissel were higher near shrubby edges compared with core habitat farther from edges.

Although many studies have considered the small-scale effect of habitat fragmentation, few have addressed landscape-level effects (Herkert and Knopf 1998). A study by McCoy et al. (1999) determined whether Conservation Reserve Program (CRP) fields should be considered source or sink habitats for grassland bird species. A source habitat (represented by a stable population) is defined as a habitat having greater fecundity rates than mortality rates along with greater emigration than immigration, whereas a sink habitat (represented by a non-stable population) has greater mortality rates than fecundity rates along with greater immigration than emigration (Donovan et al. 1996). McCoy et al. (1999) found that source-sink dynamics of CRP habitat varied by species; CRP was a source habitat for 4 species (grasshopper sparrows, field sparrows [*Spizella pusilla*], eastern meadowlarks, American goldfinches [*Carduelis tristis*]) and a sink habitat for 2 species (red-winged blackbirds [*Agelaius phoeniceus*] and common yellowthroats [*Geothlypis trichas*]). Currently, there is little information guiding habitat managers in determining how large conservation areas should be or how they should be spatially arranged to benefit grassland birds (Herkert and Knopf 1998).

Restoration of grasslands

Response to grassland restoration practices.— The historical conversion of native grasslands to croplands has likely contributed to declines in grassland bird populations. Samson and Knopf (1994) reported that grassland coverage in North America is <0.1% of pre-agriculture conversion conditions. Due to this decline in grassland habitat, restoration of grasslands has become an important tool in the conservation of grassland birds and other wildlife. Several conservation and crop-reduction programs have been developed by the federal government to revert cropland to perennial grassland with the intension of reducing crop surpluses, preventing further soil erosion, reducing sedimentation, and improving wildlife habitat (Council for Agricultural Science and Technology 1990, Johnson and Schwartz 1993). One of these

long-term cropland acreage reduction programs is CRP. Landowners in this program are paid to plant and maintain perennial grasses or native prairie plant mixtures on their land for contracts of 10–15 years (Johnson and Schwartz 1993). In general, CRP lands are planted with a low-diversity mix of either introduced, cool season or native, warm season grass species. Although the primary goal of the program is to reduce wind and water erosion, a secondary goal is to provide quality habitats for wildlife (Council for Agricultural Science and Technology 1990).

Several studies have examined the effects of CRP on grassland bird species in the Great Plains. Some of these studies have found that grassland birds have benefited from CRP land by the increase in potential breeding habitats (Johnson and Schwartz 1993, Patterson and Best 1996, Best et al. 1997, Delisle and Savidge 1997, Herkert 1998). In general, CRP offers breeding habitat for several grassland species in decline and may have the potential to reverse these species' downward population trends (Johnson and Schwartz, 1993). However, without proper management (e.g., prescribed burning, grazing), many of the CRP fields may eventually have a negative impact on some grassland birds. Herkert (1998) recognized the importance of CRP to species that require taller grasses (e.g., Henslow's sparrow, dickcissel), but he concluded that species that require periodic disturbances (e.g., grasshopper sparrow, bobolink) could be negatively impacted if CRP fields are not managed properly.

Because few native grassland remnants remain in most of the midwestern and eastern portions of North America, effective grassland bird conservation will require protection and enhancement of the remaining native grasslands and restoration of grassland habitats. Herkert (1991) suggested that grassland restoration offers a promising avenue for reversing the decline in grassland bird species. Moreover, habitat restoration has become increasingly important for endemic grassland species and is the best option for long-term viability of this guild (Vickery et al. 1999). However, Herkert

and Knopf (1998) noted that there has been a lack of research examining effects of grassland restoration, enhancement, and creation on grassland birds. They stated that development of grassland restoration techniques has focused primarily on plants, without regard to habitat requirements of birds or other wildlife. With grassland bird populations continually declining, researchers must consider effects of different restoration techniques on this imperiled bird guild.

Grassland invertebrates

Importance as avian food source.—Invertebrates are the primary food source for most grassland bird species during the breeding season (Risser et al. 1981, Robel and Xiong 2001). Joern (1992) found that bird predation contributed significantly to population and community dynamics of dominant insect taxa such as grasshoppers (Acrididae). Some of the more important invertebrate taxa occurring in grassland bird diets include Lepidoptera, Orthoptera, Araneae, Homoptera, Coleoptera, Hymenoptera, Hemiptera, and Diptera (Risser et al. 1981, O'Leske et al. 1997). Because invertebrates are predominantly composed of proteins and other important nutrients, growth and development of young grassland birds are related strongly to invertebrate availability. For example, Whitmore et al. (1986) found that ring-necked pheasant (*Phasianus colchicus*) chicks predominately selected large insects, presumably for the proteins and other nutrients required for rapid growth and development.

Importance as indicator species.—Invertebrates play a key role in many ecosystem processes (e.g., water and nutrient cycling and energy flow) (Pik et al. 2002). Therefore, many invertebrate taxa can be used as indicator species to reflect changes or influences of management or restoration practices (Kremen et al. 1993, Arenz and Joern 1996, Pik et al. 2002). Additionally, terrestrial invertebrate response to management practices or anthropogenic threats can be monitored easily and extrapolated to other biota, such as birds and mammals (Arenz and Joern 1996). Because many

invertebrates have a shorter response time to environmental change than vertebrate species, invertebrates may provide potential early warnings of environmental change, which also can be useful in monitoring the status of restorations (Kremen et al. 1993, Pik et al. 2002).

Effect of restoration.—Few studies have evaluated the effect of grassland restoration on invertebrate communities. Hull et al. (1996) found that despite increased forb species in CRP fields, there was no difference in invertebrate biomass from non-CRP fields. Leathers (2003) found that interseeded CRP fields supported greater invertebrate biomasses than control CRP fields. However, further study is needed to evaluate effects of grassland restoration on different invertebrate communities.

JUSTIFICATION

The Rainwater Basin Region (RWBR) encompasses a 6,720-km² drainage area in south-central Nebraska (Erickson and Leslie 1987). This region is considered one of the most endangered wetland systems in North America (Smith 1998). Prior to settlement, the RWBR contained >3,900 wetlands that occupied about 38,000 ha (Gersib et al. 1992). However, since settlement, most of these major wetlands have been drained and converted to cropland. Schildman and Hurt (1984) estimated that only 10% (nearly 400 wetlands) of the original major RWBR wetlands remained by 1982. Because most of the watersheds in the RWBR are intensively cultivated, these remaining wetlands have been impacted severely by agricultural influences. Sedimentation from surrounding croplands is a serious threat to existing wetlands. Sedimentation of RWBR wetlands not only reduces the functional size of the wetland but also alters hydrology of the wetland (Smith 1998). Additionally, modification of RWBR wetlands for storage of irrigation water and diversion of water away from wetlands have greatly reduced the hydrologic function of these wetlands.

Although most of the wetlands in the RWBR have been lost, this region is still an important region for migratory birds, especially waterfowl. Pederson et al. (1989) noted that RWBR wetlands are likely the most critically important palustrine wetlands in the Northern Great Plains for waterfowl. During spring, 10–15 million ducks and geese use RWBR wetlands to replenish depleted nutrient reserves needed for migration and reproduction (Gersib et al. 1992). The region is considered especially critical for providing habitat to migrating greater white-fronted geese (*Anser albifrons*) and northern pintails (*Anas acuta*) (Pederson et al. 1989). About 90% of the mid-continent population of greater white-fronted geese and 30% of the continental population of northern pintails use RWBR wetlands during spring (Gersib et al. 1992). Additionally, several million lesser snow geese (*Chen caerulescens*) use RWBR wetlands as temporary stopovers during spring. Compared with waterfowl, less is known about the importance of RWBR wetlands to nongame birds. However, the contribution of nongame birds to the overall biodiversity of the RWBR is considerable. For example, a minimum of 200,000–300,000 migrant shorebirds may use RWBR wetlands as temporary stopover sites during spring when wetland conditions are optimal (LaGrange 1997). Moreover, >257 bird species are known to use RWBR wetlands during at least some part of their annual cycle (LaGrange 1997).

In recognition of the importance of RWBR wetlands in providing migratory habitat for waterfowl, especially greater white-fronted geese and northern pintails, the region was designated as a Joint Venture under the North American Waterfowl Management Plan in 1991 (Gersib et al. 1992). One of the objectives of the Rainwater Basin Joint Venture is to protect, restore, and create an additional 10,117 ha of wetland habitat along with another 10,117 ha of upland habitat adjacent to wetlands (Gabig 2000). During the last 10 years, U.S. Fish and Wildlife Service (USFWS), Nebraska Game and Parks Commission (NGPC), and other cooperators have restored about

5,000 ha of wetland and upland habitat within the RWBR (J. Drahota, USFWS, personal communication). Although the primary goal of restoring upland habitat adjacent to wetlands is to reduce sedimentation and pollution into the wetlands, these restored uplands also may provide important nesting habitat for breeding grassland birds.

Grassland Birds

The importance of these restored uplands to grassland birds may be even greater, given that 70% of the 29 prairie bird species found in North America have declined during the last 35 years (North American Breeding Bird Survey Data; Sauer et al. 1999). Knopf (1994) noted that grassland birds have exhibited the greatest decline of any terrestrial bird guild in North America. Additionally, grassland birds are an excellent indicator of grassland ecosystem structure and function (Askins 2000). Currently, little is known about the importance of RWBR uplands to nesting grassland birds.

Smith (1998) noted that studies examining avian community-wetland habitat associations throughout the annual cycle are needed in the RWBR. Currently, most research on avian species in the RWBR has focused on waterfowl during the migration period. Little is known about the avian community in the RWBR during other seasons (e.g., breeding season). Moreover, information on responses of the avian community to management along with the concomitant changes in the vegetation (species composition and structure) is needed. An investigation of the avian community at different spatial scales (i.e., size of wetland basin and proximity to other wetland basins or grassland habitats [Brown and Dinsmore 1986]) also is needed. Such information would be very useful for developing management guidelines for different bird guilds on a seasonal basis. Additionally, managers could use this information to target acquisition and restoration efforts based on landscape-scale concerns (e.g., proximity to existing habitats) and management priorities (e.g., waterfowl or biodiversity) (Smith 1998).

Grassland Invertebrates

There is a paucity of information about invertebrate communities within the RWBR. Gordon et al. (1990) provided baseline information on the aquatic invertebrate community in 8 RWBR wetlands, but no studies have examined the terrestrial invertebrate community inhabiting upland grasslands. Terrestrial invertebrate communities play a critical role in many grassland ecosystem processes and are an integral part of a self-sustaining ecosystem (Bradshaw 1983). Hence, an examination of the terrestrial invertebrate community not only will provide an indication of grassland ecosystem health but also provide insight into a possible limiting factor to grassland bird productivity.

Because invertebrates play an important role in ecosystem health, they are also beneficial in measuring the overall success of ecosystem restorations. Changes to ecosystem processes are often not reflected in plant communities as quickly as in invertebrate communities (Jonas et al. 2002, Pik et al. 2002). Therefore, invertebrates are often useful in indicating the status of a restoration due to their rapid response to environmental change (Pik et al. 2002).

STUDY AREA

This study was conducted in the RWBR of south-central Nebraska on uplands and wetlands managed by USFWS. The climate in this area is dry to subhumid, with precipitation 50–60 cm/year (Pederson et al. 1989). The soils are generally mollisols with scattered patches of entisols, alfisols, and vertisols (Pederson et al. 1989). The predominant land-use in the region is agricultural cropland, mainly corn and soybeans. Due to the large-scale conversion of native grasslands to agricultural cropland, there is very little native grassland remaining in the region. The average size of wetland basins and their adjacent uplands in the RWBR is 324 ha. Generally, the basins and uplands are managed using prescribed fire, grazing, and haying. The frequency of these

management techniques varies depending on management goals for the individual basin.

I selected 12 rainwater basins composed of 6 replicate uplands of 2 treatments (high-diversity seeding [> 20 plants] and low-diversity seeding [≤ 5 plants]) for this study. The seeding mix used for the low-diversity planting was similar to CRP warm season seed mixes with the major grass species being big bluestem (*Andropogon gerardii*), indiangrass (*Sorghastrum nutans*), little bluestem (*Schizachyrium scoparium*), switchgrass (*Panicum virgatum*), and sideoats grama (*Bouteloua curtipendula*) (Tom Koerner, USFWS, personal communication). In an attempt to return the plant community on many basins to as close to preconversion condition as possible, the USFWS began high-diversity restoration seedings in 1999. The goal of the high-diversity seedings was to add an additional 10–50 species of grass and forb species to each site. I selected basins based on the size of the basin (≥ 4 ha), proximity to other basins and grassland habitats, age of restoration (≥ 3 years), and current and past management history.

OBJECTIVES

The 3 objectives of my study were:

- 1) To evaluate the response of the grassland bird community to different restoration techniques (high-diversity and low-diversity plantings) on upland grasslands in the RWBR of south-central Nebraska.
- 2) To determine grassland bird habitat-use and nest productivity in RWBR grasslands in south-central Nebraska relative to different spatial scales (i.e., landscape scale vs. local scale) and habitat characteristics.

- 3) To evaluate the response of the terrestrial invertebrate community to different restoration techniques (high-diversity and low-diversity plantings) on upland grasslands in the RWBR of south-central Nebraska.
- 4) To evaluate the response of the terrestrial invertebrate community to vegetation characteristics of restored grasslands in the RWBR in south-central Nebraska.

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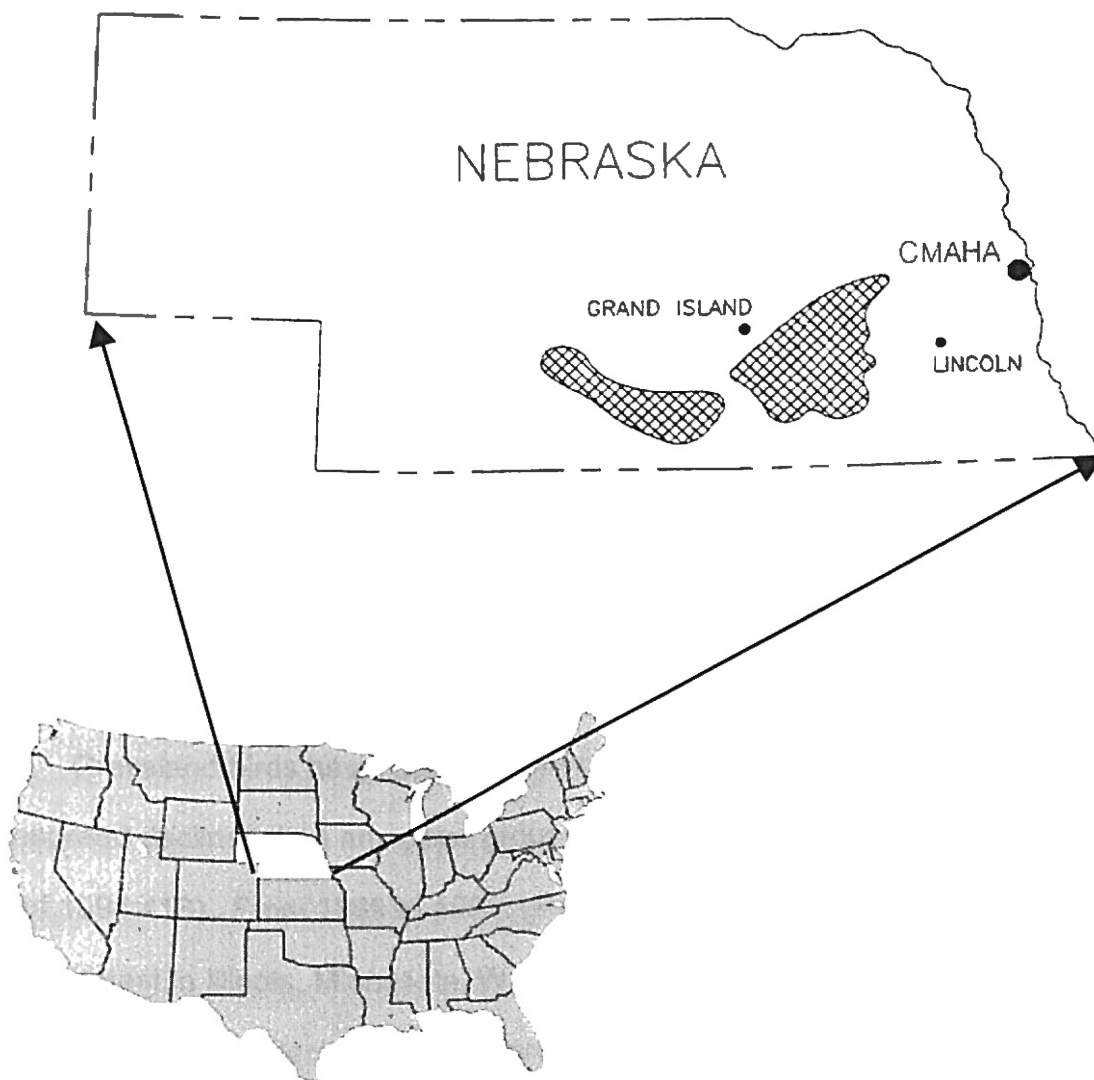
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Figure 1. Location of Rainwater Basin Region (area with crosshatch) within Nebraska, USA.



CHAPTER 2: RESPONSE OF NONGAME BIRDS TO HIGH-DIVERSITY AND LOW-DIVERSITY PLANTINGS ON UPLAND GRASSLANDS IN THE RAINWATER BASIN REGION, NEBRASKA

INTRODUCTION

Native prairies in some regions of North America have declined as much as 99.9% (Samson and Knopf 1994). This decline began in the early 1800s, with intense alteration of grasslands by agricultural activities. Due to this dramatic decline, grasslands are now among North America's most endangered ecosystems (Noss et al. 1995). Moreover, as a result of these habitat losses, one-third of North American grassland bird species have declined at statistically significant rates (Knopf 1996).

Grassland birds have "shown more consistent and steeper, geographically widespread declines than any other grouping of North American species" (Samson and Knopf 1994:418). From 1969 to 1991, grassland bird declines (ranging from 24–91%) were greatest in Illinois, Minnesota, Wyoming, Nebraska, and Missouri (Samson and Knopf 1994). These declines have been attributed predominately to habitat loss and degradation but also to encroachment of woody vegetation due to fire suppression and woody plantings.

As a result of this dramatic decline in grassland bird populations, there has been increasing pressure to restore grassland habitats in North America. Herkert (1991) suggested that grassland restoration offers a promising avenue for reversing the decline in grassland bird species. Moreover, habitat restoration has become increasingly important for endemic grassland species and is the best option for long-term viability of

this guild (Vickery et al. 1999). However, Herkert and Knopf (1998) noted there has been a lack of research examining effects of grassland restoration, enhancement, and creation on grassland birds. They stated that development of grassland restoration techniques has focused primarily on plants, without regard to habitat requirements of birds or other wildlife. With grassland bird populations continually declining, researchers must consider effects of different restoration techniques on this threatened bird guild.

There have been a number of studies that have evaluated the response of grassland avifauna to habitat enhancement, especially to CPR restorations. Many of these studies have found that grassland birds have benefited from CRP land by the increase in potential breeding habitats (Johnson and Schwartz 1993, Patterson and Best 1996, Best et al. 1997, Delisle and Savidge 1997, Herkert 1998). In general, CRP offers breeding habitat for several grassland species in decline and may have the potential to reverse these species' downward population trends (Johnson and Schwartz 1993). However, CRP plantings typically only include 4-5 grass species; there has been little research on the response of grassland bird communities to restoration plantings that include more diverse plantings with both grass and forb species.

Due in part to the loss of grassland habitat in the RWBR in south-central Nebraska, the USFWS and NGPC began restoring grasslands in the 1980s. Initially, they restored these grasslands using low-diversity plantings, but in 1999, they began using high-diversity plantings, which include 10–50 more grass and forb species than in the low-diversity plantings. The goal of increasing plant diversity in restored grasslands was to return the plant community to as close to pre-agriculture conversion conditions as possible. The increase in plant community diversity is thought to increase invertebrate food resources needed by breeding grassland birds and increase grassland bird-use of those habitats (Weathers 1992, Hull et al. 1996, Jamison et al. 2002).

The objectives of my study were to evaluate the response of the grassland bird community to different restoration techniques (high-diversity and low-diversity plantings) on upland grasslands in the RWBR of south-central Nebraska and to determine grassland bird habitat-use and nest productivity in these grasslands relative to different spatial scales (i.e., landscape scale vs. local scale) and habitat characteristics. It is crucial to assess the response of the grassland bird community to restoration treatments in the RWBR to better guide managers in habitat conservation for this declining avian guild.

METHODS

Data Collection

Bird surveys.—I surveyed breeding birds using methods similar to those employed by Stewart and Kantrud (1972) and Igl and Johnson (1997). One or two (depending on the size of grassland and wetland habitat in each basin) 150-m long transects were established in both upland and wetland areas of each basin site. Transects ≥ 100 m of the basin edge on all sides of basin (Best et al. 1997) to decrease the probability of edge (bordered by forests, roads, or agricultural crops) affecting the occurrence of most of the grassland species in the study sites (Helzer 1996). The size of each transect (150-m long by 100-m wide) was chosen primarily for feasibility and to reduce the chance of passive sampling (i.e., increased patch size will increase measurement of species richness simply due to this larger size; Connor and McCoy 1979, Helzer and Jelinski 1999). Determination of patch size effect on grassland avifauna can be impacted by passive sampling (Connor and McCoy 1979, Helzer and Jelinski 1999, Johnson 2001). To account for this problem, I sampled equal-sized transects in all uplands and wetlands of different sizes (Johnson 2001).

I surveyed each basin for avian abundance at least once during each of 3 periods (15 May–15 June, 15 June–15 July, and 15 July–15 August) (Best et al. 1997,

McCoy et al. 2001). I recorded avian abundance by counting all birds seen and/or heard within 50 m of each transect while I walked along the transect. I recorded the species and habitat associations (upland vs. wetland) of each bird observed. Those birds that were observed either flying over the transect without landing or outside the transect were recorded but not included in the species richness tally for each habitat. Migrant flocks (in direct flight), individuals of species that do not generally breed in the RWBR (Mollhoff 2001) vagrant water birds, and wide-ranging colonial water birds observed passing overhead were excluded from surveys. I conducted surveys between sunrise and 3 hours after sunrise on days with little to no wind (< 20 km/hour), no rain, and no fog (Ralph et al. 1993).

Nest searches and monitoring.—I conducted nest searches 2–3 times on a rotational basis from late May to late July within established 4-ha plots. Nest searches were conducted using 3 methods: 1) observing birds engaged in suspicious behavior indicating nest building, incubating eggs, or presence of nestlings (Martin and Geupel 1993, Giuliano and Daves 2002), 2) random searching by observers placed 10–15 m apart while walking through plots, and 3) using rope dragging to flush females from nests (Wiens 1969). Additional nests were located incidentally while conducting other data collections.

Each nest location was flagged 25 m north and east of the nest, which appears to be the appropriate distance to prevent increased predator detection (Martin and Geupel 1993). I also recorded Universe Transverse Mercator (UTM) coordinates for each nest location using a Global Positioning System (GPS) unit. I monitored nests every 3–5 days until nest fate was known. Outcome was recorded as successful (≥ 1 young fledged), failed (no young fledged due to depredation, nest abandonment, cowbird parasitism, weather, or unknown cause), or undetermined. When a nest was empty, we concluded that it was successful if there was evidence that the nestlings had

fledged (i.e., feces in nest, side of nest flattened down, or fledglings observed in area). Nest-success probabilities for species with large enough sample sizes (≥ 10 nests) were determined using the Mayfield method (Mayfield 1975).

Vegetation sampling.—Four sampling points for vegetation measurements were located at 30-m, 60-m, 90-m, and 120-m intervals along each bird survey transect and one point was located at each nest. I sampled vegetation 3 times (early June, early July, and early August) during each field season at each sampling point and at nests soon after nest fate was known.

At each sampling point and nest, I recorded percent vegetation cover, horizontal visual obstruction, maximum vegetation height, and litter depth. I estimated percent vegetation cover (grass, forbs, shrub, dead material, and bare ground) using a 20 x 50 cm Daubenmire (1959) frame. Measurements of horizontal visual obstruction and vegetative height were recorded using a Robel pole (Robel et al. 1970). Visual obstruction readings were taken at a height of 1 m above and 4 m from each cardinal direction of the Robel pole, which was placed near the Daubenmire frame. Litter depth and maximum vegetative height (living and standing dead vegetation) were measured within 50 cm of the Robel pole from each cardinal direction. For each vegetation characteristic that was measured, I calculated the mean and coefficient of variation (CV) for the 4 sampling point measurements taken along each transect. CV provides an estimate of “patchiness” or heterogeneity of each characteristic (Roth 1976).

Landscape-level effects.—I used Geographic Information System (GIS; ArcView Version 3.2, Environmental Systems Research Institute, Inc., Redlands, California, USA) to determine various landscape characteristics for each study basin. Specifically, I used GIS to determine the size and perimeter of each contiguous upland habitat and the size of the entire basin complex (including upland and wetland habitat) from digital orthophoto quadrangles for south-central Nebraska.

Statistical Analyses

Prior to conducting statistical analyses, I determined total bird and individual species relative abundances for each restoration treatment by averaging the count data from each survey for each year. Only birds that potentially bred in grassland habitat in south-central Nebraska were included in analyses (Ducey 1988, Mollhoff 2001). I calculated overall bird diversity for each restoration treatment using Shannon-Weiner diversity index (Krebs 1999). I used Shapiro-Wilks' test to test for normality (SAS Institute 1989) and Levene's test to test for homogeneous variances of the avian variables (Zar 1999). I used a square-root transformation to correct for non-normality in the total bird relative abundance data set. Because individual species relative abundance, species richness, and species diversity data sets did not meet assumptions of normality and homogeneity, I rank-transformed those data sets (Conover and Iman 1981). I used a 2-way analysis of variance (ANOVA) to examine differences in avian variables (i.e., relative abundances [total and individual species], species richness, and species diversity) between restoration treatments and years (SYSTAT 1998). Only individual bird species that were observed in >1% of surveys were included in analyses. I also used a 1-way ANOVA to compare vegetation variables between successful and unsuccessful nests of all nongame grassland birds, dickcissels, and individual species with >10 nests necessary for analysis.

Most of the vegetation variables did not meet assumptions of normality and homogeneity. Maximum vegetation height, percent grass coverage, and percent dead material were square-root transformed to correct for non-normality and heteroscedacity. Visual obstruction, litter depth, and percent coverage of forbs, bare ground and sedges did not meet assumptions following transformations and thus, were rank-transformed (Conover and Iman 1981). I used a 2-way analysis of covariance (ANCOVA) with period as a covariate to examine differences in vegetation variables (i.e., % vegetation cover,

visual obstruction, litter depth, and maximum vegetation height) between restoration treatments and years (SYSTAT 1998). For all analyses performed, I inferred significance level at $P \leq 0.10$. I selected $P \leq 0.10$ to reduce the chance of making a Type II error (the probability of not rejecting a false null hypothesis).

I used multiple logistic regressions to develop predictive models for the occurrence of individual grassland bird species on restored grasslands based on vegetation characteristics (i.e., mean and CV of forb cover, grass cover, sedge cover, bare ground, dead material, visual obstruction, litter depth, and maximum height) and landscape characteristics (i.e., area, perimeter, and perimeter-area ratio of contiguous upland habitat and area, perimeter, and perimeter-area ratio of entire basin [upland and wetland habitats combined]). I selected logistic regression over linear regression because the individual species abundance data were heavily weighted with zeros and violated assumptions of linear regression (Zar 1999). I used univariate tests to reduce number of variables for inclusion in multiple logistic regression models with $P < 0.25$ as inclusion criteria (Hosmer and Lemeshow 1989). Variables that met the inclusion criteria were used in a backward-elimination routine to create the best multivariable model for each individual bird species' presence. With this routine, a variable was eliminated from the model if its observed significance level for the regression coefficient (based on Wald chi-square significance) was $P > 0.05$. I used the Hosmer and Lemeshow (1989) test to assess the goodness-of-fit of the model. Only common grassland birds occurring in >1% of individual surveys were included in the analyses.

RESULTS

Grassland Bird Community

I observed 13 grassland bird species in each restoration during both years. Ring-necked pheasants (*Phasianus colchicus*) and common yellowthroats were the only species unique to each restoration. Ring-necked pheasants were only found in high-

diversity restorations, while common yellowthroats were only found in low-diversity restorations. Mean total grassland bird abundance and species richness were similar between high-diversity and low-diversity restorations and between years (Table 1). There was a treatment x year interaction for species diversity ($F_{1,20} = 2.91$, $P = 0.10$). In 2002, species diversity was higher in low-diversity restorations than high-diversity restorations ($F_{1,10} = 3.46$, $P = 0.09$), but in 2003, species diversity did not significantly differ between high-diversity and low-diversity restorations.

During this study, 2 species were significantly more abundant in low-diversity restorations than high-diversity restorations, and the relative abundances of 10 species were similar between high-diversity and low-diversity restorations (Table 2). Two species had significantly higher abundances in 2002 than 2003, 1 species had significantly higher abundances in 2003 than 2002, and 9 species had similar abundances in both years (Table 2). There were no significant treatment x year interactions for any species. Dickcissels and common yellowthroats were more abundant in low-diversity restorations than high-diversity restorations. Dickcissels and western meadowlarks had greater abundances in 2002 than 2003, whereas sedge wrens (*Cistothorus platensis*) were more abundant in 2003 than 2002. Overall, dickcissels, grasshopper sparrows, and bobolinks were the most abundant species over both years and in each of the restorations, accounting for 74% of the total bird abundance.

Nesting Success

I located 84 nests (41 nests in 2002, 43 nests in 2003) during the study. Of those 84 nests, 30 nests of 7 species were in high-diversity restorations, and 54 nests of 8 species were in low-diversity restorations. Of the nests where fate could be determined, 31% of the nests were successful and 69% were unsuccessful. Of the unsuccessful nests, 66% failed due to predation, 24% due to brown-headed cowbird

parasitism, and 10% due to abandonment. In high-diversity restorations, 43% were successful, and 56% were unsuccessful, while in low-diversity restorations 24% were successful and 76% were unsuccessful. The major factor for failure in both high-diversity and low-diversity restorations was predation. Using the Mayfield method (1975), I found that the nest-success probability of all species was higher in high-diversity than low-diversity plantings (Table 3). For dickcissels, nest-success probabilities were similar for high-diversity and low-diversity restorations (Table 3).

Overall, most nest-site vegetation characteristics were similar between successful and unsuccessful nests for grassland birds (Table 4). Visual obstruction, litter depth, and maximum vegetation height differed between successful and unsuccessful nests with each characteristic being higher at unsuccessful nests than successful nests (Table 4). For dickcissel nests, only litter depth and maximum vegetation height differed between unsuccessful and successful nests (Table 5). Litter depth and maximum vegetation height were higher at unsuccessful nests than successful nests. For grasshopper sparrow nests, percent forb cover was higher for successful nests than unsuccessful nests, and visual obstruction was higher for unsuccessful nests than successful nests (Table 6).

Vegetation Characteristics

I found few differences in vegetation characteristics between high-diversity and low-diversity restorations (Table 7). Visual obstruction and maximum vegetation height were the only vegetation characteristics that differed between high-diversity and low-diversity restorations; both characteristics were higher in low-diversity restorations than high-diversity restorations (Table 7). Percent grass cover and maximum vegetation height differed between years, and both were higher in 2003 (grass cover: $\bar{x} = 52.13\%$, $SE = 2.23$; maximum vegetation height: $\bar{x} = 63.62$ cm, $SE = 2.70$) than 2002 (grass

cover: $\bar{x} = 43.08\%$, SE = 2.28; maximum vegetation height: $\bar{x} = 55.79$ cm, SE = 2.00).

There were no treatment x year interactions except for percent forb cover ($F_{1,67} = 3.42$, $P = 0.07$). In 2002, there was no difference in forb cover between restoration treatments, but percent forb cover in 2003 was greater for high-diversity than low-diversity restorations ($F_{1,33} = 3.56$, $P = 0.07$).

Landscape and Local Influences

Overall, there was very little influence of landscape variables on the occurrence of most grassland bird species in RWBR restored grasslands (Table 8). Dickcissels, bobolinks, and red-winged blackbirds were the only species with a significant goodness-of-fit according to the Hosmer and Lemeshow (1989) goodness-of-fit test. Basin perimeter was the only significant predictor variable for the occurrence of dickcissels and red-winged blackbirds. Both species were negatively associated with basin perimeter (Table 8). Bobolink occurrence was best predicted by 3 landscape variables: basin perimeter, basin perimeter-area ratio, and basin size. Bobolink occurrence was associated positively with basin perimeter and associated negatively with basin perimeter-area ratio and basin size.

There was considerable variation among grassland bird species in terms of the influence of vegetation variables on the occurrence of these species in RWBR restored grasslands. Ten of the 12 most common grassland bird species had significant predictive models (Table 9). Occurrence of 7 of the grassland bird species was associated with a single predictor variable. Occurrences of grasshopper sparrows and red-winged blackbirds were best predicted by percent forb cover. Grasshopper sparrow occurrence was predicted by decreasing percent forb cover; red-winged blackbird occurrence was predicted by increasing percent forb cover. Maximum vegetative height was the best predictor for occurrences of western meadowlarks (*Sturnella neglecta*) and

sedge wrens (Table 9). Occurrence of western meadowlarks was predicted by decreasing vegetative height, and occurrence of sedge wrens was predicted by increasing vegetative height. Occurrence of dickcissels was associated positively with increasing patchiness of dead material. Occurrence of brown-headed cowbirds in restored grasslands was associated with increasing litter depth patchiness, and American goldfinches were associated with increasing visual obstruction.

Bobolinks were influenced by 2 predictor variables, and eastern kingbirds (*Tyrannus tyrannus*) and common yellowthroats were influenced by 3 predictor variables (Table 9). Occurrence of bobolinks was best predicted by increasing percent grass cover and decreasing visual obstruction. Occurrence of eastern kingbirds was best predicted by increasing visual obstruction and litter depth and decreasing percent grass cover. Occurrences of common yellowthroats was best predicted by increasing visual obstruction and percent forb cover and decreasing percent grass cover (Table 9).

DISCUSSION

In general, restored grasslands in the RWBR provided breeding habitat for a wide variety of grassland birds. I recorded 13 grassland species in restored grasslands, and many of those species have exhibited declining populations in Nebraska and continentally (Herkert 1996). Analysis of North American Bird Survey data between 1966 and 1993 showed that many grassland bird species have declined, especially grasshopper sparrows, western meadowlarks, and bobolinks (Herkert 1995). Moreover, these habitats also are important to many grassland species of greatest conservation concern in the Midwest. Based on conservation priority rankings by Herkert et al. (1996), 5 of the top 9 grassland bird species of conservation concern in the Midwest occurred on my study sites: dickcissel, bobolink, sedge wren, grasshopper sparrow, and upland sandpiper (*Bartramia longicauda*).

Given that the RWBR landscape is dominated by agriculture and agricultural fields provide minimal habitat for grassland birds, these restored grasslands along with the dried basin wetlands provide critical habitat to grassland birds (Appendices A and B). Although I did not directly compare avian abundance between restorations and row crops, several other studies have showed the importance of restorations relative to row crops. Best et al. (1997) found that overall bird abundance levels were 1.4–10.5 times greater in CRP fields than row crops throughout the Midwest. Additionally, they found 3 bird species of conservation concern on CRP fields compared with none on row crops. Patterson and Best (1996) also found a greater total bird abundance for CRP fields than row crops in Iowa and found significantly greater numbers of species of management concern (sedge wrens, dickcissels, grasshopper sparrows, and bobolinks) on CRP fields than on row crops. Bryan and Best (1991) compared grassland bird abundance between grass waterways and crop fields and found that overall grassland bird abundance in grassed waterways to be 3 times that in crop fields.

The USFWS and NGPC began high-diversity restoration plantings, which include 10–50 more grass and forb species than in the low-diversity plantings, in 1999 to return the plant community to as close to pre-agriculture conversion conditions as possible. The increase in plant community diversity is thought to increase invertebrate food resources needed by breeding grassland birds and increase grassland bird-use of those habitats (Weathers 1992, Hull et al. 1996, Jamison et al. 2002). However, I found little difference in grassland bird communities between high-diversity and low-diversity grassland restorations. Moreover, I found little difference in terrestrial invertebrate communities between high-diversity and low-diversity restorations (Chapter 3). Jamison et al. (2002) also found bird and invertebrate abundances to be related, with lesser prairie-chickens (*Tympanuchus pallidicinctus*) being more abundant in areas with greater invertebrate abundance. Consequently, the lack of difference in bird communities

between restorations in my study may be explained partially by the lack of difference in invertebrate communities between restorations.

Although bird use of grasslands may be influenced by availability of invertebrate foods, the bird carrying capacity in tallgrass prairie generally is set by the physical structure of vegetation rather than food resources (Kaufman et al. 1998). Patterson and Best (1996) found that the diversity in vegetation cover and structure in CRP fields in central Iowa resulted in differences in bird species communities using those fields. In contrast, I found that overall bird abundance, most individual bird abundances, and bird species diversity were similar between restorations. Dickcissels and common yellowthroats were the only species that exhibited differences in abundances between high-diversity and low-diversity restorations. Unlike Patterson and Best (1996), I found few differences in vegetation characteristics between high-diversity and low-diversity restorations. Because vegetation differences correlate with bird community differences (Best et al. 1997), it is not surprising that I found few differences between overall bird abundance, individual abundances, and species diversity between treatments.

I did not detect many differences in vegetation characteristics between restorations, except for greater maximum vegetation height and visual obstruction in low-diversity restorations. There are many possible reasons for few of the vegetation characteristics differing between the 2 restorations. Factors such as site preparation prior to planting (mechanical and chemical applications), seed bank viability, age of restoration, and management regime (e.g., grazing, burning) could have contributed to the few differences in vegetation characteristics between high-diversity and low-diversity restorations. For most of the sites used, the specific site preparation technique was unknown. However, in many of the restorations there were large patches of smooth brome grass (*Bromus inermis*), a cool season exotic grass that forms monotypic stands. Occurrence of smooth brome grass could be due to either lack of chemical and

mechanical site preparation prior to planting of the restoration or the presence smooth brome grass in the initial seed bank. The management technique used on restored grasslands also varied by site; many sites were idle for several years, which could have promoted exotics and woody species, while other sites were burned and grazed. In the absence of disturbance (i.e., fire or grazing), extensive invasion of woody plant species can occur, especially in smaller patches (Herkert et al. 1996). Occurrence of woody plant species in high-diversity and low-diversity restorations in the RWBR was very similar. Brye et al. (2002) found that vegetation characteristics stabilized 19 years post-grassland restoration. Although many of my low-diversity restorations are >20 years in age since being converted, most of my high-diversity restorations are at most 5 years old (Tom Koerner, USFWS, personal communication). Hence, vegetation characteristics may be similar between high-diversity and low-diversity restorations because high-diversity restorations likely did not have enough time for many of the plant species to become well-established.

In restored grasslands of the RWBR, a variety of landscape and local (i.e., vegetation characteristics) factors influenced grassland bird presence. Most common bird species were influenced by vegetation factors, but only a few species appeared to be influenced by landscape factors. This could be due to the fact that species were not affected consistently by landscape factors throughout all sites. For example, Johnson and Igl (2001) found differences in area sensitivities for the same bird species over different counties. Their study demonstrates that factors that influence species' presence in one area, such as the RWBR, do not necessarily apply to other areas.

Dickcissel presence was associated with decreasing total basin perimeter, meaning that at least in the RWBR this species is associated with less edge habitat. However, Herkert (1994a) found that dickcissels have no positive area relationship, and Helzer (1996) found this species to be still common in small patches. Dickcissels also

were associated positively with percent dead material cover CV, which may indicate that this species may prefer grasslands with patchy areas of dead material.

Presence of grasshopper sparrows was determined by landscape and local variables; however, their presence was likely more influenced by local variables. Although the overall landscape model for grasshopper sparrow presence was not significant, individual landscape factors, perimeter-area ratio and upland perimeter, were significant predictors of presence. Presence of grasshopper sparrows was associated negatively with perimeter-area ratio and associated positively with upland perimeter. Helzer (1996) also found that grasshopper sparrows have a negative correlation with perimeter-area ratio. I also found that presence of grasshopper sparrow was associated negatively with forb cover. This negative association could be with exotic forbs (rather than native species), such as Canada thistle (*Cirsium arvense*) or leafy spurge (*Euphorbia esula*), which are commonly found in many restorations.

Presence of bobolinks also was influenced by landscape and local factors. I found presence of bobolinks to be correlated negatively with basin perimeter-area ratio, which was similar to Helzer and Jelinski's (1999) results. Initially, this suggests that patch characteristics such as shape and core area influence presence of bobolinks. However, presence of bobolink also was linked negatively to basin area and associated positively with basin perimeter, which seems to contradict their negative association with basin perimeter-area ratio. Yet, perimeter-area ratio is a more effective measurement of habitat patch quality than area because it reflects both size and shape (Helzer and Jelinski 1999). Therefore, it appears that bobolinks in the RWBR prefer basins with a high perimeter-area ratio. Bobolinks were associated positively with grass cover and associated negatively with visual obstruction. In general, bobolinks occurred in restored grasslands that contained grassy areas that were less dense. Madden et al. (2000) also found that presence of bobolinks was linked with grass cover, and Helzer and

Jelinski (1999) found that presence of bobolinks was correlated negatively with horizontal visual obscurity in 1 of 2 years.

For the landscape variables, presence of common yellowthroats was associated negatively with basin perimeter, but for the vegetation variables, presence of common yellowthroats was associated positively with forb CV and visual obstruction and associated negatively with grass cover in restored grasslands. Even though the common yellowthroat had a significant negative relationship with increased basin perimeter, it was a very weak model overall; therefore, it appears as though vegetation characteristics influence their presence more than landscape variables. Johnson and Igl (2001) reported similar results in that common yellowthroats selected habitat features rather than keying in on the size of the grassland. Common yellowthroats often are abundant in fields with high forb cover (Herkert 1991, Patterson and Best 1996). Similarly, I found that common yellowthroats were affected positively by patchiness of forb cover in restored grasslands.

Red-winged blackbirds had a positive relationship with forb cover and a negative association with basin perimeter. As for vegetation factors, red-winged blackbirds in my study were found predominantly at sites with tall forbs, which are commonly used for nest placement (Ducey 1988). For landscape factors, there have been varied results as far as response of red-winged blackbird. Herkert (1994b) found that red-winged blackbirds were more common on small prairies, while Johnson and Igl (2001) found varying results depending on location.

Eastern kingbirds, sedge wrens, western meadowlarks, brown-headed cowbirds, and American goldfinches (*Carduelis tristis*) were only associated with vegetation characteristics. Eastern kingbirds were associated predominately with increased visual obstruction and litter depth. The sites that eastern kingbirds were observed in also had shrubs and trees along grassland edges, which is the type of habitat in which they often

breed in (Murphey 1996). Herkert (1994*b*) found that sedge wrens were a “vegetation restricted species” that only occurred in transects according to vegetation features rather than landscape variables (e.g., patch size). However, Johnson and Igl (2001) reported that sedge wrens were an area-sensitive species. Sedge wrens were not associated with any landscape variables but were associated with increased vegetation height, which is similar to other findings (Herkert et al. 2001). I found that western meadowlarks were associated negatively with maximum vegetation height. My results were comparable to results from other studies that found that they prefer areas that are dominated by dense stands of short grass due to long-term moderate grazing (Wiens 1969, Klute et al. 1997). Presence of brown-headed cowbirds was associated positively with litter depth in my study. Similarly, Schneider (1998) reported that occurrence of brown-headed cowbirds in south-central and northwestern North Dakota was related to litter depth. Presence of American goldfinches was associated with increasing visual obstruction. The increased visual obstruction may represent the increased shrubby vegetation and trees that occurred at 2 grasslands at which I observed American goldfinches. American goldfinches prefer shrubby vegetation for nesting (Middleton 1993), which may explain the relationship with increased visual obstruction.

Nest success of grassland birds is severely limited by high predation rates and cowbird brood parasitism (Johnson and Temple 1990, Burger et al. 1994). In my study, the nest success rate for all grassland bird species was 26%, which is similar to results from other studies examining nesting success in restored grasslands. Grassland bird nests found in CRP fields throughout the Midwest had a success rate of 40% (Best et al. 1997). McCoy et al. (2001) found individual grassland bird nest success rates ranged from 19 to 47% in CRP CP2 (warm-season grass) fields. Nest success rates found by Giuliano and Daves (2002) were higher at 60% in warm-season grass fields and 48% in cool-season grass fields, which may be due to many different factors (e.g., different bird

species, less threat from predators and brood parasites, larger grassland area, and different surrounding land practices).

Predation was the primary cause of nest failure in restored grasslands, accounting for 41% of all nest failures. Predators also caused the majority of nest failures of grassland birds nesting in CRP fields throughout the Midwest (Patterson and Best 1996, Best et al. 1997). Although there are many potential predators of grassland birds, many studies have found snakes to be the dominant predator of nests in grasslands and shrub habitats (Best 1978, Thompson et al. 1999, Davison and Bollinger 2000). Although I did not directly observe snake predation, many of my nests showed no signs of disturbance other than egg removal, which would suggest snake predation (Davison and Bollinger 2000). Davison and Bollinger (2000) found that 83% of the nests depredated by mammals or birds showed some type of sign of disturbance besides egg removal.

After predation, brown-headed cowbird parasitism was the next highest source of nest failure during my study with a rate of 16% for all nests. Similar results to mine occurred in CRP fields in Iowa; the incidence of parasitism for red-winged blackbirds, dickcissels, and grasshopper sparrows was 25, 33, and 9%, respectively (Patterson and Best 1996). In warm-season grass fields in southwestern Pennsylvania, 2 of 70 grassland bird nests (3%) were parasitized by brown-headed cowbirds (Giuliano and Daves 2002); however, those results were based on assortment of bird nests, several of which differed from those found in this study. Best et al. (1997) reported a parasitism rate of 3% for grassland bird nests in CRP fields in Nebraska, although those results should be viewed with caution because this was from a sample size of only 9 nests. The rate of parasitism for dickcissels breeding in native prairie fragments in southwestern Missouri was also relatively low at 9.6% (Winter 1999). Compared with some of these studies, the occurrence of brood parasitism was relatively high for my study. The higher

rate of parasitism in my study may be due to most of the restored grasslands being very small in size and often bordered by eastern red cedar (*Juniperus virginiana*) windbreaks, which could provide elevated perches for brown-headed cowbirds to search for nests.

Previous studies have found that dickcissel nest success was affected only by vegetation characteristics, rather than landscape factors, within the fields they nested in (Herkert et al. 1993, Hughes et al. 1999). I found that success of dickcissels nests was related to only increased vegetation height and litter depth. In areas with taller vegetation, dickcissels were often nesting higher in the vegetation, which could have made their nests more visible to predators and brown-headed cowbirds.

Dickcissels nesting in high-diversity restorations had a nest-success rate of 14.7% (Mayfield nest-success probability estimate), compared to 12.8% in low-diversity restorations. Dickcissels nesting in CRP fields in Iowa had a similar nest- success rate at 14% (Patterson and Best 1996), but dickcissels nesting in native grassland fragments in southwestern Missouri had a higher nest success rate at 29.7% (Mayfield estimate) (Winter 1999). Again, due to the small size of many of these restored grasslands, the associated increased effects of predation and brown-headed cowbird parasitism likely decreased nest success.

Successful grasshopper nests in restored grasslands had less visual obstruction and higher forb cover than unsuccessful nests. Patterson and Best (1996) reported similar results in that grasshopper sparrow nests were associated less with vertical vegetation cover than all other breeding grassland birds. Grasshopper sparrows generally establish territories in areas with moderate forb cover and high vegetative diversity (Wiens 1969, Klute et al. 1997). It is possible that when grasshopper sparrows nested near forbs, it was more difficult for predators and brood parasites to locate the nest because it was better concealed.

CONSERVATION AND RESEARCH IMPLICATIONS

According to my results, it appears as though low-diversity and high-diversity plantings provide comparable habitat for grassland birds. Both restorations are beneficial to grassland birds of management concern by providing breeding habitat (Herkert et al. 1996). However, although my results show that these restoration plantings may be similar, they should be viewed with caution because many of the high-diversity restorations have not been fully established. The high-diversity restorations were restored up to 20 years after the low-diversity restorations had been restored, and therefore, may have not yet had time to become as well established as the low-diversity restorations. Because bird abundance was similar between the 2 restorations, this might imply to managers that low-diversity plantings are sufficient for providing breeding habitat for grassland birds. However, there are many factors such as age of restoration, basin size, management history, and surrounding landscape that likely influenced bird abundance and species richness in restored grasslands of RWBR.

Although I did not examine whether these restored grasslands act as source or sink habitats for grassland birds, it is possible that these grassland restorations may act as sink habitat rather than source habitat. A source habitat (represented by a stable population) is defined as having greater fecundity rates than mortality rates along with greater emigration than immigration rates, whereas a sink habitat (represented by a non-stable population) has greater mortality rates than fecundity rates (Donovan et al. 1996). Perkins et al. (2003) found that core areas provided source habitat for grassland bird species and edge habitat provided sink habitat. There are very few restored grasslands in the RWBR that are large enough to have core area not affected by edge, which could suggest that the majority of these grasslands are functioning as sink habitat to grassland birds.

In future studies, it is crucial to take into account both density and demographic factors when assessing importance of grasslands as breeding habitat for grassland birds. I found that different factors could affect certain species differently with regard to density and demographics. For example, I found that presence of grasshopper sparrows was not associated positively with forb cover; however, successful grasshopper sparrow nests had significantly greater forb cover around the nest than unsuccessful nests. Bird density and nesting success must both be taken into consideration, because census data alone might not detect sensitivity to different variables (e.g., vegetation characteristics, patch size, proximity to edge, landscape characteristics) (Winter and Faaborg 1999).

One of the problems with my study was that most of the high-diversity restorations were relatively new (<5 yrs old). To fully assess responses of grassland birds to high-diversity restorations, long-term monitoring of grassland birds in these restorations will be needed. When future restorations are created, site preparation techniques, cropping history, specific species planted at each site, and seed-spreading rate should be recorded in detail. Site preparation also should include chemical or mechanical techniques that rid uplands of exotics, such as smooth brome grass, which were often observed in large patches in high-diversity and low-diversity restorations. Finally, if possible larger areas should be restored instead of small, fragmented areas, since restorations that are heterogeneous and large in size can be more effective in attracting large populations of grassland birds (Herkert et al. 1996).

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TABLE 1. Total relative abundance^a, species richness^b, and species diversity^c of grassland birds in high-diversity and low-diversity grassland restorations during the breeding season in the Rainwater Basin Region, Nebraska, 2002-2003.

	Year	High-diversity (<i>n</i> = 12)		Low-diversity (<i>n</i> = 12)		<i>P</i> ^d	
		Mean	S.E.	Mean	S.E.	Treatment	Year
Total abundance (no./ transect)	2002-2003	3.35	0.43	4.65	0.68	0.20	0.33
Species richness	2002-2003	3.83	0.32	3.58	0.38	0.81	0.51
Species diversity	2002	0.46	0.02	0.54	0.04	0.81	0.07
	2003	0.44	0.04	0.35	0.08		

^a Mean abundance from 3 sampling periods during each year.

^b Number of grassland-associated birds counted during each year.

^c Shannon-Weiner diversity index used to calculate bird species diversity (Krebs 1999).

^d *P* = *P*-value for treatment (high-diversity vs. low-diversity) and year (2002 vs. 2003) effects from 2-way analysis of variance. No interaction effects (treatment x year) occurred, except for species diversity ($F_{1,20} = 2.91$, $P = 0.10$). Means are reported separately for each year for species diversity

TABLE 2. Relative abundance^a (no./ transect) of grassland bird species that occurred in >1% of all surveys in high-diversity and low-diversity grassland restorations in the Rainwater Basin Region, Nebraska, 2002-2003.

Species	High-diversity (n = 12)		Low-diversity (n = 12)		<i>P</i> ^b	
	Mean	S.E.	Mean	S.E.	Treatment	Year
Dickcissel (<i>Spiza americana</i>)	0.93	0.14	1.72	0.39	0.10	0.07
Grasshopper sparrow (<i>Ammodramus savannarum</i>)	1.36	0.28	0.92	0.26	0.30	0.98
Bobolink (<i>Dolichonyx oryzivorus</i>)	0.57	0.34	0.64	0.44	0.95	1.00
Eastern kingbird (<i>Tyrannus tyrannus</i>)	0.13	0.04	0.14	0.10	0.24	0.62
Western meadowlark (<i>Sturnella neglecta</i>)	0.08	0.04	0.11	0.07	0.54	0.06
Brown-headed cowbird (<i>Molothrus ater</i>)	0.08	0.04	0.33	0.19	1.00	0.21
American goldfinch (<i>Carduelis tristis</i>)	0.01	0.01	0.25	0.19	0.25	0.90
Common yellowthroat (<i>Geothlypis trichas</i>)	0.00	0.00	0.15	0.10	0.08	0.54
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	0.03	0.03	0.22	0.14	0.27	0.37
Sedge wren (<i>Cistothorus platensis</i>)	0.01	0.01	0.08	0.06	0.48	0.08
Upland sandpiper (<i>Bartramia longicauda</i>)	0.03	0.03	0.03	0.03	1.00	1.00
Mourning dove (<i>Zenaida macroura</i>)	0.06	0.06	0.03	0.03	0.95	0.17

^a Mean abundance from 3 sampling periods during each year.

^b *P* = *P*-value for treatment (high-diversity vs. low-diversity) and year (2002 vs. 2003) effects from 2-way analysis of variance. No treatment x year interaction effects occurred.

TABLE 3. Nest-success probabilities^a for all grassland birds and dickcissels in the Rainwater Basin Region, Nebraska, 2002-2003.

Species	High-diversity			Low-diversity		
	Incubation (%)	Nestling (%)	Overall (%)	Incubation (%)	Nestling (%)	Overall (%)
All grassland bird species	39.68	63.38	25.15	29.77	53.92	16.05
Dickcissel	30.86	47.73	14.73	23.95	53.26	12.76

^a Determined using Mayfield (1975) method.

TABLE 4. Vegetation characteristics for successful and unsuccessful nests of grassland birds in the Rainwater Basin Region, Nebraska, 2002-2003.

Vegetation characteristic	Successful (<i>n</i> = 25)		Unsuccessful (<i>n</i> = 62)		<i>P</i> ^a
	Mean	S.E.	Mean	S.E.	
Forb (%)	12.40	1.96	15.43	2.07	0.78
Grass (%)	56.60	3.40	56.42	2.92	0.75
Sedge (%)	2.50	0.00	2.72	0.22	0.52
Dead material (%)	53.00	3.16	47.96	1.86	0.19
Bare ground (%)	7.40	1.35	5.56	0.66	0.11
Shrub (%)	3.40	0.90	10.17	3.09	0.32
Visual obstruction (cm)	23.00	2.86	38.88	2.61	< 0.001
Litter depth (cm)	0.56	0.03	0.66	0.03	0.096
Maximum vegetation height (cm)	59.15	3.55	79.32	3.19	< 0.001

^a *P* = *P*-value for nest fate effects from 1-way analysis of variance.

TABLE 5. Vegetation characteristics for successful and unsuccessful nests of dickcissels in the Rainwater Basin Region, Nebraska, 2002-2003.

Vegetation characteristic	Successful (<i>n</i> = 12)		Unsuccessful (<i>n</i> = 36)		<i>P</i> ^a
	Mean	S.E.	Mean	S.E.	
Forb (%)	13.13	2.81	17.85	2.94	0.99
Grass (%)	60.83	3.36	53.61	3.95	0.50
Dead material (%)	43.33	2.97	47.22	2.36	0.47
Bare ground (%)	8.54	2.76	4.86	0.57	0.39
Shrub (%)	2.50	0.00	14.86	4.83	0.16
Sedge (%)	2.50	0.00	2.50	0.00	1.00
Visual obstruction (cm)	33.75	3.81	42.47	2.30	0.14
Litter depth (cm)	0.55	0.04	0.72	0.05	0.06
Maximum vegetation height (cm)	69.44	5.52	83.16	3.04	0.06

^a *P* = *P*-value for nest fate effects from 1-way analysis of variance.

TABLE 6. Vegetation characteristics for successful and unsuccessful nests of grasshopper sparrows in the Rainwater Basin Region, Nebraska, 2002-2003.

Vegetation characteristic	Successful (<i>n</i> = 8)		Unsuccessful (<i>n</i> = 6)		<i>P</i> ^a
	Mean	S.E.	Mean	S.E.	
Forb (%)	12.50	3.98	5.00	1.12	0.08
Grass (%)	51.25	6.80	58.33	5.58	0.50
Sedge (%)	2.50	0.00	2.50	0.00	1.00
Dead material (%)	66.25	5.49	56.67	3.07	0.91
Bare ground (%)	6.25	0.82	5.83	1.05	0.76
Visual obstruction (cm)	11.25	1.25	17.50	0.65	0.002
Litter depth (cm)	0.56	0.06	0.60	0.04	0.70
Maximum vegetation height (cm)	47.28	2.32	55.75	4.86	0.11

^a *P* = *P*-value for nest fate effects from 1-way analysis of variance.

TABLE 7. Vegetation characteristics for high-diversity and low-diversity grassland restorations in the Rainwater Basin Region, Nebraska, 2002-2003.

Vegetation characteristic	Year	High-diversity (<i>n</i> = 36)		Low-diversity (<i>n</i> = 36)		<i>P</i> ^a	
		Mean	S.E.	Mean	S.E.	Treatment	Year
Forb (%)	2002	11.32	1.60	13.09	2.06	0.34	0.91
	2003	16.01	2.36	11.94	2.46		
Grass (%)	2002-2003	45.28	2.33	49.93	2.37	0.15	0.01
Sedge (%)	2002-2003	2.55	0.04	2.50	0.00	0.15	0.15
Dead material (%)	2002-2003	64.07	2.68	58.15	2.56	0.12	0.73
Bare ground (%)	2002-2003	11.36	2.04	11.53	1.27	0.23	0.70
Visual obstruction (cm)	2002-2003	18.50	1.92	21.66	1.48	0.03	0.14
Litter depth (cm)	2002-2003	0.68	0.05	0.64	0.05	0.59	0.74
Maximum vegetation height (cm)	2002-2003	56.28	2.41	63.13	2.39	0.02	0.01

^a *P* = *P*-value for treatment (high-diversity vs. low-diversity) and year (2002 vs. 2003) effects from 2-way analysis of covariance. No interaction (treatment x year) effects occurred, except for forb cover ($F_{1,67} = 3.42$, $P = 0.07$). Means are reported separately for each year for forb cover.

TABLE 8. Logistic regression models for landscape variables that best predicted grassland bird presence in high-diversity and low-diversity grassland restorations in the Rainwater Basin Region, Nebraska, 2002-2003. Variables were selected from a set of 6 landscape variables using a backward-elimination routine.

Species	Fitted logistic model ^{ab}	<i>P</i> ^c
Dickcissel (<i>Spiza americana</i>)	2.64 – 0.0002 (basin perimeter)	0.834
Grasshopper sparrow (<i>Ammodramus savannarum</i>)	– 6.04 + 0.004 (upland perimeter) – 1180.4000 (basin perimeter:area)	0.005
Bobolink (<i>Dolichonyx oryzivorus</i>)	168.9000 + 0.0151 (basin perimeter) – 65,947.3000 (basin perimeter:area) – 0.3869 (basin size)	0.148
Eastern kingbird (<i>Tyrannus tyrannus</i>)	No significant factor found	
Western meadowlark (<i>Sturnella neglecta</i>)	No significant factor found	
Brown-headed cowbird (<i>Molothrus ater</i>)	No significant factor found	
American goldfinch (<i>Carduelis tristis</i>)	No significant factor found	
Common yellowthroat (<i>Geothlypis trichas</i>)	0.5522 – 0.0004 (basin perimeter)	0.001
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	1.5238 – 0.0006 (basin perimeter)	0.539
Sedge wren (<i>Cistothorus platensis</i>)	No significant factor found	
Upland sandpiper (<i>Bartramia longicauda</i>)	No significant factor found	
Mourning dove (<i>Zenaida macroura</i>)	No significant factor found	

Table 8. Continued.

^a Logit [presence(x)] = $\ln[\text{presence}(x)/\text{absence}(x)] = b_0 + b_1(x_1) + b_2(x_2) \dots + b_i(x_i)$,

$\text{Presence}(x) = 1/[1 + \exp\{-(b_0 + b_1(x_1) + b_2(x_2) \dots + b_i(x_i))\}]$, and $\text{Presence}(x) = 1 - \text{Presence}(x)$.

^c $P = P$ -value for Hosmer and Lemeshow goodness-of-fit test of overall model.

^b basin size = upland and wetland combined area, upland perimeter = upland perimeter, basin perimeter:area = ratio perimeter to area for upland and wetland combined basin, and perimeter = perimeter upland and wetland combined.

TABLE 9. Logistic regression models for vegetation variables that best predicted grassland bird presence in high-diversity and low-diversity grassland restorations in the Rainwater Basin Region, Nebraska, 2002-2003. Variables were selected from a set of 16 vegetation variables using a backward-elimination routine.

Species	Fitted logistic model ^{ac}	<i>P</i> ^b
Dickcissel (<i>Spiza americana</i>)	– 1.1406 + 0.0805 (dead CV)	0.319
Grasshopper sparrow (<i>Ammodramus savannarum</i>)	1.6088 – 0.0823 (forb mean)	0.504
Bobolink (<i>Dolichonyx oryzivorus</i>)	– 3.9789 + 0.0972 (grass mean) – 0.1311 (vo mean)	0.419
Eastern kingbird (<i>Tyrannus tyrannus</i>)	– 4.0321 + 0.0762 (vo mean) + 3.0122 (litter depth mean) – 0.0634 (grass CV)	0.872
Western meadowlark (<i>Sturnella neglecta</i>)	1.6736 – 0.0689 (height mean)	0.919
Brown-headed cowbird (<i>Molothrus ater</i>)	– 3.9489 + 0.0565 (ld CV)	0.842
American goldfinch (<i>Carduelis tristis</i>)	– 4.0704 + 0.0715 (vo mean)	0.860
Common yellowthroat (<i>Geothlypis trichas</i>)	– 2.6202 + 0.1567 (vo mean) + 0.0850 (forb CV) – 0.2725 (grass mean)	0.996
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	– 4.7202 + 0.1179 (forb)	0.669
Sedge wren (<i>Cistothorus platensis</i>)	– 17.4118 + 0.1919 (height mean)	0.999
Upland sandpiper (<i>Bartramia longicauda</i>)	No significant factor found	
Mourning dove (<i>Zenaida macroura</i>)	No significant factor found	

Table 9. Continued.

^a Logit [presence(x)] = $\ln[\text{presence}(x)/\text{absence}(x)] = b_0 + b_1(x_1) + b_2(x_2) \dots + b_i(x_i)$,

$\text{Presence}(x) = 1/[1 + \exp\{-(b_0 + b_1(x_1) + b_2(x_2) \dots + b_i(x_i))\}]$, and $\text{Presence}(x) = 1 -$

$\text{Presence}(x)$

^b $P = P$ -value for Hosmer and Lemeshow goodness-of-fit test of overall model

^c dead CV = dead material cover coefficient of variance, forb = forb cover, grass = grass cover, bare = bare ground cover, height = maximum vegetation height, forb CV = forb cover coefficient of variance, ld CV = litter depth coefficient of variance, vo = visual obstruction, and height CV = maximum vegetation height coefficient of variance.

CHAPTER 3: RESPONSE OF TERRESTRIAL INVERTEBRATES TO GRASSLAND RESTORATIONS IN THE RAINWATER BASIN REGION, NEBRASKA

INTRODUCTION

Grassland cover in North America has declined over 99% from pre-agriculture conversion conditions (Samson and Knopf 1994). Due to this dramatic decline, grasslands are now among North America's most endangered ecosystems (Noss et al. 1995). Consequently, restoration of grasslands in the Great Plains has become an important tool in the conservation of grassland birds and other wildlife that rely on these endangered habitats.

An important component of grassland ecosystems is the terrestrial invertebrate community. Terrestrial invertebrates play a key role in many grassland ecosystem processes (e.g., water and nutrient cycling and energy flow) (Pik et al. 2002) and are critical food resources during summer to many grassland birds, especially to their young (Risser et al. 1981, Robel and Xiong 2001). Because of their importance to grassland ecosystem health and as food resources, many terrestrial invertebrate taxa have been used as bioindicators to reflect changes or influences of restoration practices (Kremen et al. 1993, Arenz and Joern 1996, Pik et al. 2002). Additional benefits of using terrestrial invertebrates to monitor restorations are that their response to changes in environment conditions can be monitored easily and they typically have a shorter response time to environmental change than most vertebrate and plant species (Arenz and Joern 1996, Jonas et al. 2002, Pik et al. 2002). In the Great Plains, terrestrial invertebrates have only recently been used as bioindicators to evaluate grassland restorations, with most focusing on CRP fields (Hull et al. 1996, Leathers 2003). Little attention has been given to using terrestrial invertebrates to evaluate other types of grassland restorations.

During the last 20 years, the USFWS and NGPC have been restoring upland habitats in the RWBR of south-central Nebraska using low-diversity plantings and high-

diversity plantings. Low-diversity plantings are similar to CRP warm season seed mixes with the major grass species being big bluestem (*Andropogon gerardii*), indiagrass (*Sorghastrum nutans*), little bluestem (*Schizachyrium scoparium*), switchgrass (*Panicum virgatum*), and sideoats grama (*Bouteloua curtipendula*), while high-diversity plantings are composed of >20 plants that include a mix of grasses and forbs (Tom Koerner, USFWS, personal communication). Currently, there is a lack of information with regard to the effect of restoration plantings on terrestrial invertebrate communities in the RWBR and general knowledge of invertebrate taxa inhabiting these restored grasslands. Additionally, information on how the terrestrial invertebrate communities respond to these restorations is needed because they provide habitat for many breeding grassland bird species (Chapter 2) that rely on these invertebrates for food.

The objectives of this study were to evaluate the response of the terrestrial invertebrate community to different restoration techniques (high-diversity and low-diversity plantings) on upland grasslands in the RWBR of south-central Nebraska and to evaluate the response of the terrestrial invertebrate community to various vegetation characteristics. By assessing the response of the terrestrial invertebrate community to restoration treatments, managers will be able to determine the current status of these restorations and better plan future restorations in the RWBR.

METHODS

Data Collection

Invertebrate sampling.—I used pitfall traps and sweep nets to collect terrestrial invertebrates from 6 high-diversity and 6 low-diversity restorations during summer 2002–2003. I randomly located 4 pitfall trap arrays that consisted of 5 pitfall traps placed about

30 cm apart and connected by plastic edging along a 210 m transect that traversed each restoration. I used pitfall trap arrays rather than a single pitfall trap because arrays increase the probability of collecting a more accurate sample of the invertebrate community from each restoration. Pitfall traps, similar to those used by Morrill (1975), consisted of 2, 470 ml Solo[®] plastic cups with 1 cup placed inside of the other. Plastic lids, supported by 2, 9-cm nails, were placed 1–2 cm above the cups to protect contents from rainfall and reduce evaporative losses of preservative solutions and capture of nontarget organisms (e.g., small mammals). The inside cup was filled with a 1:1 mixture of environmentally friendly ethylene glycol and ethyl alcohol.

I collected pitfall and sweep net samples in mid-late June and mid-late July during each year. After the pitfall traps were set for 48 hours, I collected the contents from each trap and preserved them in 80% ethyl alcohol. For sweep net samples, I collected invertebrates by making 50 full sweeps through the vegetation (O'Leske et al. 1997). I collected sweep net samples 0.5 m above ground on days with little to no wind (< 20 km/hour) and no rain. After each sweep net collection, I first placed collected invertebrates in plastic bags and then placed the collected invertebrates in jars with 80% ethyl alcohol.

Invertebrate taxa were identified according to published descriptions (Borrer et al. 1970, Borror et al. 1989). For the most part, all taxa were identified to the family level. I counted all invertebrates and oven-dried them at 60°C for at least 24 hours to a constant mass (0.001 g). Voucher specimens were stored at the Department of Zoology, Oklahoma State University, Stillwater, Oklahoma.

Vegetation sampling.—For vegetation measurements, I located 4 sampling points at 30 m, 60 m, 90 m, and 120 m along an established transect in close proximity to the collection sites for the pitfall trap arrays and sweep net samples. I collected

vegetation data during early July and early August of each year. At each sampling point, I recorded percent vegetation cover, horizontal visual obstruction, maximum height, and litter depth. I estimated percent vegetation cover (grass, forbs, shrub, dead material, and bare ground) using a 20 x 50 cm Daubenmire (1959) frame. Horizontal visual obstruction and vegetation height measurements were recorded using a Robel pole (Robel et al. 1970). Visual obstruction readings were taken at a height of 1 m above and 4 m from each cardinal direction of the Robel pole, which was placed near the Daubenmire frame. Litter depth and maximum vegetative height (living and standing dead vegetation) was measured within 50 cm of the Robel pole from each cardinal direction.

Statistical Analyses

For each restoration, I calculated invertebrate abundance, familial richness, and Shannon-Weiner diversity index (Krebs 1999) from pitfall trap samples based on the 4 pitfall arrays along each transect during each sampling period. For 2 of the restorations, 3 pitfall arrays were used instead of 4 arrays in the analyses because of collection difficulties experienced in the field. For sweep net samples, I calculated abundance, familial richness, and Shannon-Weiner diversity index for each restoration based on each sweep net sample collected along each transect. I calculated familial richness as the number of invertebrate families found in each restoration during each sampling period. To meet assumptions for parametric tests, I square-root transformed total invertebrate abundance and familial richness for the pitfall data and total invertebrate abundance for the sweep net data (Zar 1999). I rank-transformed diversity index values from sweep net data and abundance for individual families from both data sets because assumptions for parametric tests were not met (Conover and Iman 1981). I used a 3-way ANOVA to examine differences in invertebrate variables (i.e., total invertebrate and individual taxa abundance, familial richness, and diversity index) separately for pitfall

trap and sweep net data between restorations, sampling periods, and years (SYSTAT 1998). For analysis of individual taxa, I only analyzed those taxa that accounted for $\geq 5\%$ of the total number of invertebrates in each of the sample types.

I used Morisita's index to determine similarities between invertebrate communities in high-diversity and low-diversity restorations. The equation for Morisita's index is: $C_\lambda = 2 \sum x_i y_i / (l_1 + l_2) N_1 N_2$, where x_i equals the number of individuals in species i for community 1, y_i is the abundance of species i , l_1 is Simpson's dominance index for community 1, l_2 is Simpson's dominance index for community 2, N_1 is the total number of individuals in community 1, and N_2 is the total number of individuals in community 2. C_λ ranges from 0 (indicating no similarity) to 1 (complete similarity) (Brower and Zar 1977). Morisita's index indicates the probability that individuals randomly drawn from each of the 2 communities compared will belong to the same taxa (i.e., the probability of randomly selecting a pair of individuals of the same taxa from one of the communities) (Brower and Zar 1977).

To examine differences in vegetation variables (i.e., percent vegetation cover, visual obstruction, litter depth, maximum vegetation height) between restorations, sampling periods, and years, I used a 3-way ANOVA. For vegetation data that did not meet assumptions of parametric tests, I used a rank-transformation (Conover and Iman 1981). For all analyses performed, I inferred a significance level at $P \leq 0.10$. I selected $P \leq 0.10$ to reduce the chance of making a Type II error (the probability of not rejecting a false null hypothesis).

I used Canonical Correspondence Analysis (CCA) to relate the terrestrial invertebrate community to quantitative environmental variables (i.e., percent cover [grass, forbs, dead material, bare ground, and sedges], maximum vegetation height, litter depth, and visual obstruction) (CANOCO version 4.5; ter Braak and Šmilauer 2002). CCA is a direct gradient analysis that relates community composition to known

variation in the environment. In CCA, axes are chosen that are linear combinations of environmental variables (ter Braak 1986). I performed a principal components analysis (PCA) prior to CCA to reduce the dimensionality of the explanatory variables (i.e., vegetation characteristics). I performed partial CCAs with nominal variables (treatment, period, and year) set as covariables and blocks on square-root transformed data based on results from the PCA (i.e., variables that were most highly correlated with first, second, and third axes were included in the analysis). I performed these analyses separately for pitfall and sweep net data and ran 499 permutations of Monte Carlo test to determine the significance of invertebrate community patterns for each data set.

RESULTS

Invertebrate Abundance, Richness, and Diversity

Overall, I identified 79 recognizable taxa and 10,103 individuals in pitfall and sweep net samples from restored grasslands during 2002 and 2003 (Table 1). From pitfall traps, I identified 62 recognizable taxa and 8,192 individuals, while from sweep nets, I identified 67 recognizable taxa and 1,191 individuals. Sixty-one of the taxa were unique to pitfall samples, while 58 taxa were unique to sweep net samples. Overall, the most abundant invertebrate taxon in restored grasslands was Formicidae, which accounted for 41.6% of the total number of invertebrates collected. In pitfall samples, the most abundant taxa were Carabidae, Araneae, Gryllidae, Formicidae, Acrididae, and Silphidae, accounting for 85.2% of the total number of invertebrates collected (Table 1). In sweep net samples, the most abundant taxa were Araneae, Acrididae, Cicadellidae, Tettigoniidae, Muscidae, Pentatomidae, Formicidae, Pyralidae, and Chrysomelidae, accounting for 73.8% of the total number of invertebrates collected.

Pitfall trap samples.---Overall, I collected 5,145 individuals from high-diversity restorations and 3,767 individuals from low-diversity restorations. I identified 25 invertebrate taxa from high-diversity restorations and 31 taxa from low-diversity

restorations. Most of the taxa were common in high-diversity and low-diversity restorations; however, a few taxa were unique to each of the restorations (Table 1). Taxa unique to high-diversity restorations included Acarina, Blattidae, Cleridae, Dermestidae, Sphaeritidae, Chironimidae, Dolichopodidae, Delphacidae, Nabidae, Megachilidae, Platystomatidae, Berytidae, and Ichneumonidae, while taxa unique to low-diversity restorations included Culicidae, Heleomyzidae, Dictyopharidae, Cynipidae, Chrysopidae, and Oligochaetae.

There were no differences in total invertebrate abundance between treatments, years, or periods ($P > 0.10$; Figs. 1A-3A), but there was a period x year interaction ($F_{1,38} = 16.92$, $P < 0.0001$). Therefore, I continued analyses by year to evaluate differences in total invertebrate abundance between periods and treatments within each year. In 2002, total invertebrate abundance was greater during period 1 ($\bar{x} = 278.00$ individuals/transect, S.E. = 35.16) than period 2 ($\bar{x} = 145.58$ individuals/transect, S.E. = 19.10; $F_{1,19} = 11.54$, $P = 0.003$), while in 2003, total invertebrate abundance was greater in period 2 ($\bar{x} = 223.45$ individuals/transect, S.E. = 36.17) than period 1 ($\bar{x} = 133.42$ individuals/transect, S.E. = 32.13; $F_{1,19} = 4.27$, $P = 0.053$). Total invertebrate abundance did not differ between treatments within each year.

Overall, invertebrate family richness was greater in high-diversity plantings than low-diversity plantings ($F_{1,38} = 3.98$, $P = 0.053$) and greater in 2002 than 2003 ($F_{1,38} = 4.55$, $P = 0.039$) (Figs. 1B and 2B). Invertebrate family richness did not differ between periods (Fig. 3B). There was a period x year interaction ($F_{1,38} = 43.93$, $P < 0.0001$); therefore, I further examined invertebrate family richness within years. Similar to results for total invertebrate abundance, invertebrate family richness was greater during period 1 ($\bar{x} = 17.27$ individuals/transect, S.E. = 0.81) than period 2 ($\bar{x} = 11.67$

individuals/transect, S.E. = 0.68) in 2002 ($F_{1,19} = 28.17$, $P < 0.0001$) and greater for period 2 ($\bar{x} = 15.54$ individuals/transect, S.E. = 1.22) than period 1 ($\bar{x} = 10.25$ individuals/transect, S.E. = 0.58) in 2003 ($F_{1,19} = 17.61$, $P < 0.0001$). Invertebrate family richness did not differ between treatments within each year.

Invertebrate diversity did not differ between treatments, years, or periods ($P > 0.10$; Figs. 1C-3C). There were no interactions for invertebrate diversity. Invertebrate communities from high-diversity plantings and low-diversity plantings were very similar ($C_\lambda = 0.98$).

Of the 5 most abundant taxa, Silphidae was the only taxa that differed between restorations (Table 2). Silphidae abundance was higher in low-diversity plantings than high-diversity plantings. Silphidae abundance was also higher in 2003 than in 2002 and higher in period 2 than period 1. There was a treatment x period ($F_{1,38} = 5.43$, $P = 0.025$) and a year x period ($F_{1,38} = 33.30$, $P = < 0.001$) interaction for Silphidae abundance. In 2002, there was a treatment ($F_{1,19} = 3.32$, $P = 0.084$) and period ($F_{1,19} = 103.93$, $P < 0.001$) effect, but not in 2003. Silphidae abundance was higher in high-diversity plantings ($\bar{x} = 1.42$ individuals/transect, S.E. = 1.33) than low-diversity plantings ($\bar{x} = 0.00$ individuals/transect, S.E. = 0.00) and higher in period 2 ($\bar{x} = 1.42$ individuals/transect, S.E. = 1.33) than period 1 ($\bar{x} = 0.00$ individuals/transect, S.E. = 0.00) during 2002.

Carabidae and Gryllidae abundances differed between periods (Table 2). For Carabidae, abundance was higher in period 1 ($\bar{x} = 25.00$ individuals/transect, S.E. = 6.38) than period 2 ($\bar{x} = 11.61$ individuals/transect, S.E. = 1.80) (Table 2). However, there was a treatment x period ($F_{1,38} = 3.10$, $P = 0.087$) and a year x period interaction ($F_{1,38} = 3.06$, $P = 0.088$). In 2002, Carabidae abundance was greater during period 1 (\bar{x}

= 36.00 individuals/transect, S.E. = 12.27) than period 2 (\bar{x} = 10.17 individuals/transect, S.E. = 10.07; $F_{1,19} = 8.31$, $P = 0.010$) and there was no difference between treatments ($F_{1,19} = 0.08$, $P = 0.780$); while in 2003, there was no difference in either period or treatment (period: $F_{1,19} = 0.05$, $P = 0.830$; treatment: $F_{1,19} = 1.43$, $P = 0.249$). For Gryllidae, abundance was greater for period 2 (\bar{x} = 20.00 individuals/transect, S.E. = 6.67) than period 1 (\bar{x} = 5.65 individuals/transect, S.E. = 1.72), but there was a treatment x period interaction ($F_{1,38} = 3.86$, $P = 0.057$). Therefore, I further examined treatment effects within each period. For period 1, abundance was greater for low-diversity (\bar{x} = 23.27 individuals/transect, S.E. = 3.59) than high-diversity (\bar{x} = 13.87 individuals/transect, S.E. = 3.44; $F_{1,21} = 3.57$, $P = 0.07$) restorations, but there was no difference between restorations for period 2 ($F_{1,21} = 1.11$, $P = 0.310$).

Abundance for both Araneae and Formicidae differed between years (Table 2). Both taxa had higher abundances for 2002 (Araneae: \bar{x} = 32.39 individuals/transect, S.E. = 4.84; Formicidae: \bar{x} = 113.08 individuals/transect, S.E. = 20.44) than 2003 (Araneae: \bar{x} = 22.65 individuals/transect, S.E. = 2.61; Formicidae: \bar{x} = 67.61 individuals/transect, S.E. = 19.24). Araneae abundance also differed between periods; abundance was greater for period 1 (\bar{x} = 36.22 individuals/transect, S.E. = 4.52) than period 2 (\bar{x} = 18.83 individuals/transect, S.E. = 2.26). Because there was a treatment x year x period ($F_{1,38} = 4.25$, $P = 0.046$) interaction for Araneae abundance, I further examined treatment and period effects within each year. In both years, Araneae abundance differed between periods (2002: $F_{1,19} = 7.79$, $P = 0.012$; 2003: $F_{1,19} = 15.79$, $P = 0.001$) with abundances being greater during period 1 (2002: \bar{x} = 45.27 individuals/transect, S.E. = 8.05; 2003: \bar{x} = 27.92 individuals/transect, S.E. = 3.34) than period 2 (2002: \bar{x} = 20.58 individuals/transect, S.E. = 3.07; 2003: \bar{x} = 16.91

individuals/transect, S.E. = 3.40). There was no difference between restorations for 2002 ($F_{1,21} = 1.19$, $P = 0.290$) or 2003 ($F_{1,21} = 0.07$, $P = 0.790$). However, there was a treatment x period interaction ($F_{1,19} = 5.36$, $P = 0.032$) in 2003. During period 1 in 2003, Araneae abundance was greater for low-diversity sites than high-diversity sites ($F_{1,10} = 3.84$, $P = 0.079$), while during period 2, Araneae abundance did not differ between treatments ($F_{1,10} = 1.92$, $P = 0.199$). There were no significant interaction effects for Formicidae ($P > 0.10$).

Sweep net samples.— Overall, I collected 505 individuals from high-diversity restorations and 686 individuals from low-diversity restorations (Table 1). I identified 44 invertebrate taxa from high-diversity restorations and 46 taxa from low-diversity restorations. Most of the taxa were found in both high-diversity and low-diversity restorations; however, there were some taxa that were unique to each restoration (Table 1). Unique taxa from high-diversity restorations included Acarina, Blattidae, Cleridae, Dermestidae, Sphaeritidae, Chironimidae, Elateridae, Meloidae, Sarcophagidae, Sciomyziidae, Berytidae, Cixiidae, Delphacidae, Dictyopharidae, and Coenagrionidae, and unique taxa from low-diversity restorations included Cantharidae, Carabidae, Endomychidae, Bombyliidae, Lochaeidae, Platystomatidae, Tachinidae, Rhopalidae, Halictidae, Sphecidae, and Tenthredinidae.

There was no difference in total invertebrate abundance between treatments or periods ($P > 0.10$; Figs. 1A and 3A), but there was a difference between years (Fig. 2A). Total invertebrate abundance was higher in 2003 than 2002 ($F_{1,39} = 6.22$, $P = 0.017$). There was a treatment x year interaction ($F_{1,39} = 4.45$, $P = 0.041$); therefore, I further examined the effects of treatment and period on total invertebrate abundance within each year. In 2002, total invertebrate abundance was greater on low-diversity plantings ($\bar{x} = 30.82$ individuals/transect, S.E. = 9.51) than high-diversity plantings ($\bar{x} = 11.17$

individuals/transect, S.E. = 3.11; $F_{1,19} = 4.67$, $P = 0.044$), but did not differ between periods ($F_{1,19} = 0.06$, $P = 0.800$). In 2003, there was no difference in total abundance between treatments ($F_{1,19} = 0.04$, $P = 0.850$), however, total abundance was greater for period 1 ($\bar{x} = 38.25$ individuals/transect, S.E. = 4.33) than period 2 ($\bar{x} = 21.67$ individuals/transect, S.E. = 3.09; $F_{1,20} = 8.96$, $P = 0.007$).

Invertebrate family richness did not differ between treatments ($F = 0.87$, $P = 0.360$) but did differ between years ($F_{1,39} = 23.92$, $P < 0.0001$) and periods ($F_{1,39} = 4.78$, $P = 0.035$). Invertebrate family richness was higher in 2002 than 2003 and higher in period 1 than period 2 (Figs. 2B and 3B). There were no third-order or second-order interactions. Invertebrate diversity did not differ between treatments or periods ($P > 0.10$) but did differ between years ($F_{1,39} = 31.39$, $P < 0.0001$). Invertebrate diversity was higher in 2003 than 2002 (Fig. 2C). No third-order or second-order interactions occurred. Similar to the pitfall trap samples, invertebrate communities in the high-diversity and low-diversity plantings were very similar ($C_A = 0.89$).

Of the 5 most abundant taxa, Araneae and Acrididae were the only taxa to differ in abundance between restorations, with both taxa having higher abundances in low-diversity sites than high-diversity sites (Table 3). Acrididae also differed between years and periods. Acrididae abundance was greater for 2002 ($\bar{x} = 5.61$ individuals/transect, S.E. = 2.84) than 2003 ($\bar{x} = 1.12$ individuals/transect, S.E. = 0.33) and greater for period 2 ($\bar{x} = 5.29$ individuals/transect, S.E. = 2.72) than period 1 ($\bar{x} = 1.26$ individuals/transect, S.E. = 0.43). There were no significant interaction effects for Acrididae ($P > 0.10$). Araneae abundance differed between periods; abundance was greater for period 1 ($\bar{x} = 4.43$ individuals/transect, S.E. = 0.75) than period 2 ($\bar{x} = 3.13$ individuals/transect, S.E. = 0.95). However, there was a year x period interaction for Araneae ($F_{1,39} = 6.63$, $P = 0.014$). In 2002, Araneae abundance was greater for low-

diversity sites ($\bar{x} = 5.54$ individuals/transect, S.E. = 1.80) than high-diversity sites ($\bar{x} = 1.33$ individuals/transect, S.E. = 0.31; $F_{1,19} = 8.39$, $P = 0.009$) but did not differ between periods ($F_{1,19} = 0.03$, $P = 0.862$). In 2003, there was no difference in Araneae abundance between treatments ($F_{1,20} = 0.47$, $P = 0.502$), but abundance was higher in period 1 ($\bar{x} = 6.17$ individuals/transect, S.E. = 1.07) than period 2 ($\bar{x} = 2.17$ individuals/transect, S.E. = 0.76; $F_{1,20} = 12.84$, $P = 0.002$).

Abundances of Cicadellidae, Tettigoniidae, and Chrysomelidae did not differ between high-diversity and low-diversity restorations (Table 3). However, Cicadellidae and Tettigoniidae abundances did differ between years, while Chrysomelidae did not differ between years. Abundances of Cicadellidae and Tettigoniidae were greater for 2003 (Cicadellidae: $\bar{x} = 6.75$ individuals/transect, S.E. = 1.24; Tettigoniidae: $\bar{x} = 2.75$ individuals/transect, S.E. = 0.67) than 2002 (Cicadellidae: $\bar{x} = 1.91$ individuals/transect, S.E. = 0.63; Tettigoniidae: $\bar{x} = 1.04$ individuals/transect, S.E. = 0.35). Cicadellidae abundance also differed between periods with abundance greater for period 1 ($\bar{x} = 6.13$ individuals/transect, S.E. = 1.30) than period 2 ($\bar{x} = 2.71$ individuals/transect, S.E. = 0.76). There was a year x treatment interaction for Cicadellidae ($F_{1,39} = 2.90$, $P = 0.097$). Therefore, I further analyzed data within each year. In 2002, Cicadellidae abundance was greater for period 1 ($\bar{x} = 3.00$ individuals/transect, S.E. = 1.14) than period 2 ($\bar{x} = 0.92$ individuals/transect, S.E. = 0.50; $F_{1,19} = 4.26$, $P = 0.053$) and greater for low-diversity sites ($\bar{x} = 3.09$ individuals/transect, S.E. = 1.18) than high-diversity sites ($\bar{x} = 0.83$ individuals/transect, S.E. = 0.37) ($F_{1,19} = 3.68$, $P = 0.07$). In 2003, there was no difference in abundance between periods ($F_{1,20} = 2.37$, $P = 0.139$) or treatments ($F_{1,20} = 0.16$, $P = 0.690$). There were no significant interaction effects for Tettigoniidae or Chrysomelidae ($P > 0.10$).

Vegetation Characteristics

In general, vegetation characteristics did not differ between high-diversity and low-diversity plantings; maximum vegetation height was the only characteristic that differed between plantings (Table 4; $F_{1,40} = 4.26$, $P = 0.046$). Maximum vegetation height was higher in low-diversity plantings than in high-diversity plantings. Percent grass cover ($F_{1,40} = 29.98$, $P < 0.0001$), visual obstruction ($F_{1,40} = 5.18$, $P = 0.028$), and maximum vegetation height ($F_{1,40} = 13.96$, $P = 0.001$) were the only characteristics that differed between years with each being higher in 2003 than in 2002. Percent forb cover ($F_{1,40} = 3.61$, $P = 0.065$), percent grass cover ($F_{1,40} = 10.08$, $P = 0.003$), and percent dead material ($F_{1,40} = 6.68$, $P = 0.014$) differed between periods. Percent forb cover and percent grass cover were higher in period 1 than period 2, and percent dead material was greater in period 2 than period 1 (Table 4). For percent grass cover, there were significant third-order ($F_{1,40} = 3.35$, $P = 0.075$) and second-order ($F_{1,40} = 4.30$, $P = 0.045$) interactions, and for visual obstruction, there was a significant second-order interaction ($F_{1,40} = 3.87$, $P = 0.056$). Therefore, I further evaluated treatment and year effects within each year for these variables. In 2002, percent grass cover was higher in period 1 ($\bar{x} = 47.95\%$, S.E. = 2.91) than period 2 ($\bar{x} = 32.45\%$, S.E. = 3.43; $F_{1,20} = 12.88$, $P = 0.002$) but did not differ between periods in 2003 ($F_{1,20} = 0.65$, $P = 0.429$). Percent grass cover also did not differ between treatments for each year (2002: $F_{1,20} = 0.27$, $P = 0.613$; 2003: $F_{1,20} = 0.24$, $P = 0.628$). For visual obstruction, I found no differences between periods (2002: $F_{1,20} = 1.45$, $P = 0.242$; 2003: $F_{1,20} = 2.81$, $P = 0.109$) or treatments (2002: $F_{1,20} = 0.29$, $P = 0.599$; 2003: $F_{1,20} = 0.73$, $P = 0.404$) for each year.

Invertebrate and Vegetation Relationships

Pitfall trap samples.----A partial CCA determined the effects of vegetation variables on individual invertebrate taxa abundance. In the partial CCA, I included maximum vegetation height, litter depth, forb cover, bare ground, and dead material (based on results from the PCA) in the analysis. The first 4 axes of the pCCA accounted for 11.8% of the variation (Table 5). Results from the Monte Carlo test showed that the observed relationships of the terrestrial invertebrate community with vegetation variables were not significantly different from random ($F = 1.17$, $P = 0.106$). The species-environment correlation coefficients for first, second, third, and fourth axes were 0.825, 0.727, 0.863, and 0.774, respectively, suggesting a stronger relationship between the invertebrate community and vegetation variables for the third and first axes (Table 5). Of the 5 vegetation variables, litter depth and bare ground had the highest correlation coefficients with the first axis. Litter depth was correlated negatively with the first axis, while bare ground was correlated positively. For axis 2, maximum vegetation height had the highest (positive) correlation coefficient. Axis 1 seems to indicate a ground cover gradient of bare ground to high litter cover (Table 5, Fig. 4). Invertebrate taxa associated with bare ground included Curculionidae, Gryllidae, and Carabidae. Those taxa associated with low litter depth included Tettigoniidae and Anthomyiidae, whereas Scarabaeidae was the only taxon highly associated with high litter depth (Fig. 4). Axis 2 seems to indicate a gradient from low vegetation height to high vegetation height. Cantharidae and Formicidae were the taxa most associated with low maximum vegetation height, while Lygaeidae and Silphidae were the taxa most associated with high vegetation height (Fig. 4).

Sweep net samples.----A partial CCA was also conducted on sweep net data to determine the effects of vegetation variables on individual invertebrate taxa abundance. In the partial CCA, I included maximum vegetation height, litter depth, forb cover, bare ground, and dead material (based on results from the PCA) in the analysis. The first 4

axes of the pCCA accounted for 12.3% of the variation (Table 6). Results from the Monte Carlo test showed that observed relationships of the terrestrial invertebrate community with vegetation variables differed from random ($F = 1.22$, $P = 0.07$). The species-environment correlation coefficients for first, second, third, and fourth axes were 0.843, 0.759, 0.874, and 0.823, respectively, suggesting a stronger relationship between the invertebrate community and vegetation variables for the third and first axes (Table 6). For axis 1, maximum vegetation height had the highest (positive) correlation coefficient, while for axis 2, dead material (positive) and bare ground (negative) had the highest correlation coefficients. Axis 1 seems to indicate a gradient from high to low vegetation height (Table 6, Fig. 5). Chrysomelidae was the taxon most associated with low vegetation height, and Ichneumonidae and Gryllidae were most associated with high vegetation height (Fig. 5). Axis 2 seems to indicate a gradient from high amounts of dead matter and high litter depth to high bare ground (Table 6, Fig. 5). Asilidae, Scutelleridae, Cleridae, Lygaeidae, and Coccinellidae were most associated with high litter depth and dead material, and Curculionidae, Anthomyiidae, and Pyralidae were associated with higher amounts of bare ground (Fig. 5).

DISCUSSION

The most significant contributors to the diversity of a grassland ecosystem, comprising most of the species, are invertebrates (Arenz and Joern 1996). Risser et al. (1981) reported that the above-ground invertebrate community of a native tallgrass prairie in Oklahoma contained 131 families of invertebrates, while Kaufman et al. (1998) reported that the above-ground invertebrate community of a native tallgrass prairie in Kansas contained at least 200 families of insects. Jonas et al. (2002) found a total of only 26 taxa in a native prairie in Kansas; however, most of the invertebrate families in that study were grouped together into orders. By comparison, I identified 79 terrestrial invertebrate taxa from pitfall and sweep net samples collected from restored grasslands

in the RWBR. Although these restored grasslands seem to support a fairly diverse invertebrate community, the invertebrate communities of these grasslands are still considerably less diverse than those found in native grasslands. However, there are several factors that may have contributed to lower invertebrate diversity in these restored grasslands compared with native grasslands. First, overall, plant communities in my restorations were not very diverse, especially compared with native prairies that may have over 100 different plant species (J. Sporrang, Oklahoma State University, personal observation). Second, I collected invertebrates during 2 sample periods only, which may have resulted in some taxa, whose life cycle was completed before, after, or between collection times, not being collected. Third, although all sampling techniques have associated biases, the actual number of pitfall arrays or sweep net samples I collected may not have been as numerous as needed to properly determine the number of invertebrate taxa inhabiting these restorations. *Finally, these restorations may not have been established long enough for the full complement of terrestrial invertebrates to colonize them.*

In a study of breeding grassland birds in a tallgrass prairie, Risser et al. (1981) reported that the diets of many of the birds included Carabidae, Curculionidae, Gryllidae, Acrididae, Formicidae, Araneida, Chrysomelidae, and Lepidoptera. In my study, all of these taxa occurred in the restored grasslands. Moreover, Carabidae, Gryllidae, Acrididae, Araneae, Formicidae, and Chrysomelidae were the most abundant invertebrates collected from pitfall and sweep net samples. Therefore, it appears that these restored grasslands provide adequate food resources for breeding grassland birds.

In general, I found that the invertebrate communities in high-diversity and low-diversity restorations were quite similar. Specifically, I found few differences in overall invertebrate abundance, individual taxa abundance, and invertebrate diversity between

high-diversity and low-diversity restorations for sweep net and pitfall samples; however, I did find a difference in family richness between high-diversity and low-diversity restorations for pitfall samples. Invertebrate diversity has often been associated with plant species diversity and structural diversity (Tscharntke and Greiler 1995, Siemann et al. 1998, Jonas et al. 2002). Jonas et al. (2002) found invertebrate taxa richness and diversity to be correlated with diversity and richness of plant species. Specifically, more diverse plant communities provide more niches for invertebrates. In my study, the lack of difference in invertebrate abundance (overall and most individual taxa) and diversity could be attributed to the lack of difference in vegetation characteristics between restorations. I did not detect many differences between restorations, except for greater maximum vegetation height in low-diversity restorations. Although many factors may have contributed to the restorations having few differences in the vegetation characteristics (e.g., inadequate site preparation prior to seeding; exotic, invasive plant seeds in the seed bank; inconsistent management regimes), one possible major reason for few differences in the vegetation characteristics may be that high-diversity restorations have not had much time to become well-established. Brye et al. (2002) found that vegetation characteristics stabilized 19 years post-grassland restoration. Although many of the low-diversity restorations are >20 years since being converted, most of the high-diversity restorations are at most 5 years old (Tom Koerner, USFWS, personal communication). Consequently, the similarity of the vegetation community between restorations is reflected in the similarity of the invertebrate community between the restorations.

Several other factors may have contributed to the lack of differences in the invertebrate community between restorations. One factor that might have attributed to the lack of differences is the innate patchiness of invertebrate populations. Specifically, habitats of most invertebrate taxa are not distributed uniformly over an area but instead

occur as separate habitat patches due to differing soil types, soil moisture, and food resources (Fahrig and Paloheimo 1988). This patchy distribution of invertebrates could have caused higher variability in my results. This lack of difference between restorations also could be due to biases associated with identification to the family level only. Although family-level identification often does allow for distinguishing differences between treatment effects, life-history strategies and habitat selection patterns of genera and species within families may differ considerably. For example, *Eritettix simplex* differs from other Acrididae species in its ability to acquire nutrition from plants unsuitable to other species (Jonas et al. 2002). Hence, identification at the family-level may mask differences in taxa at the genera or species level between restorations. Finally, limitations and biases of each sampling technique may have resulted in high variability in data. Pitfalls more accurately estimate species richness of Coleoptera and Araneae taxa (predatory taxa), whereas sweep nets are more efficient at estimating species richness of Diptera and Hemiptera taxa (herbaceous or parasitic taxa) (Standen 2000). Pitfalls often over-represent larger Coleoptera, especially families that are attracted to decomposing invertebrates (e.g., Silphidae) and male spiders (Araneae). Pitfall traps also can be influenced heavily by weather conditions (Jonas et al. 2002). Sweep net sampling also has some shortcomings. For example, sweep nets may not capture invertebrates that have the ability to avoid the sweep net and effectiveness of sweep netting can be influenced by vegetation density and structure (Thompson 1987).

Although abundance and diversity did not differ between restorations, family richness for pitfall samples was greater for high-diversity than low-diversity restorations. Invertebrates often respond more rapidly to environmental change than vegetation (Arenz and Joern 1996, Jonas et al. 2002, Pik et al. 2002), and although I did not see a difference in vegetation characteristics, greater invertebrate family richness may indicate that high-diversity restorations are beginning to provide habitat for a greater number of

invertebrate taxa than low-diversity restorations. However, it also is possible that because I identified invertebrates only to the family level rather than species-level, that this difference between treatments for pitfall samples would not be present if a species-level approach was taken.

Of 9 common invertebrate taxa found in pitfall and sweep net samples, only 3 differed between high-diversity and low-diversity restorations. Acrididae, Araneae, and Silphidae were more abundant in low-diversity than high-diversity sites. Results from the CCA showed that these taxa were all associated with greater maximum vegetation height, which was characteristic of low-diversity restorations. The reason for this association was not abundantly clear, but this association may be related indirectly to some other factor such as soil moisture, vegetation cover, or food preferences. In the case of Acrididae, greater abundance on low-diversity restorations may be related to their food preference (Jonas et al. 2002). Jonas et al. (2002) suggested that acridids may prefer exotic grasses, such as smooth brome grass, as a food source. Although I did not specifically measure the amount of smooth brome grass on restorations, it was observed in many restorations and it is possible that it is more abundant in low-diversity restorations.

Weather conditions during the 2 years of my study were quite different with precipitation in 2002 being less than in 2003 (National Oceanic and Atmospheric Administration 2002, 2003). Drought conditions in 2002 likely affected the invertebrate community. For statistical analyses of both the pitfall and sweep net data, I had numerous year-by-treatment and year-by-period interactions, which were likely caused by differences in precipitation between the 2 years. In general, I observed differences in overall invertebrate abundance, family richness, and abundance of some individual taxa between years. These differences in the invertebrate community between years may be attributed directly to drought conditions or indirectly to the drought through its effect on

the vegetation community. With drought, there is a decrease in density of vegetation which will continue until the drought ends and soil is again sufficiently moist to support the growth of vegetation (Albertson et al. 1957). In my study, percent cover of grass and forbs and maximum vegetation height were greater in 2003 than 2002.

During my study, Acrididae, Formicidae, and Araneae were greater in 2002 than in 2003, while Cicadellidae, Tettigoniidae, and Silphidae were greater in 2003 than 2002. Responses of terrestrial invertebrates to seasonal and annual drought conditions are likely not uniform across and within functional groups (e.g., herbivores, predators, detritivores) (Blair et al. 2000). Hence, the inconsistencies in year effects that I observed among the different taxa may partially be explained by different responses to drought. For example, Acrididae nymphs and adults require warm, dry, sunny conditions, while their overall fecundity is reduced by cool, damp weather (Curry 1994). Considering this requirement, it is reasonable that Acrididae abundance was greater during 2002 (drought conditions). Formicidae are similarly found in drier, warmer, climates (Curry 1994), which agrees with my results; Formicidae were more abundant in 2002 than 2003. It also is possible that the effects of drought on Acrididae, Formicidae, and Araneae were not realized until 2003, with decreased abundances due to suppressed reproduction from the drought in 2002. In terms of Cicadellidae and Tettigoniidae, the reduction in plant biomass caused by the drought most likely affected their numbers during 2002. Cicadellidae and Tettigoniidae are both herbivores, which are often affected by decreased plant productivity (Blair et al. 2000). As for Silphidae, it is likely that reduced abundance was due to increased temperatures and drier overall conditions during 2002. Researchers have found that the burying beetle (*Nicrophorus marginatus*) (Silphidae) is highly susceptible to death from desiccation (to a greater degree than beetles in other families) during dry summer months (Bedick et al. 2004).

As for period differences in common taxa, Carabidae and Cicadellidae were more abundant during period 1 than period 2, while Acrididae and Gryllidae were more abundant during period 2 than 1. It is possible that herbivores (e.g., Cicadellidae) were less abundant during period 2 due to decreased plant production (Blair et al. 2000). I observed less grass and forb cover in period 2 than period 1. Additionally, the breeding and hatching phenology of some taxa (e.g., Carabidae) may have influenced their numbers between period 1 and period 2. For example, some of the taxa may have completed their life cycle before the second period. In the case of Acrididae and Gryllidae, it is most likely that sampling period 2 (late July) coincided with peak abundances for Acrididae and Gryllidae, because the nymph and adult life stages of these taxa are associated with hot, dry conditions (Curry 1994).

Patterns observed for the invertebrate communities in these restored grasslands can be explained partially by patterns observed in the vegetation. For the most part, invertebrate taxa appeared to be distributed along gradients according to their associations with percent bare ground, litter depth, and vegetation height. For some of these species, associations appear to be related to their foraging guild, while for others they appear to be related to habitat preference. For example, Scarabaeidae, which are predominantly detritivores, were related strongly to litter depth (Ritcher 1958), while Carabidae, which are predominantly predators that chase their prey, were strongly related to bare ground (Curry 1994). However, conclusions from the CCA of the pitfall data should be viewed with some reservations because the invertebrate community–vegetation characteristic relationships were not significantly different from random. In contrast, results from the CCA of the sweep net data for the invertebrate community–vegetation characteristic relationships were different from random.

In a CCA, interpretability of results depends on the environmental variables that are chosen. There are environmental variables that I did not collect that may play an

important role in how these invertebrate communities are composed and structured. Although vegetation characteristics that were used in direct gradient analyses did affect overall variance in species composition, there was still a large amount of variance that was not explained. In addition to the vegetation characteristics I measured, other factors such as soil moisture, habitat size, surrounding habitat and land-use practices, and plant species composition also should be taken into consideration to better understand the invertebrate communities in these grasslands. In particular, plant species diversity may play a major role in the composition of the invertebrate communities in these restored grasslands. I only measured vegetation cover rather than measuring plant species diversity. As mentioned earlier, several studies have shown that invertebrate diversity is related to plant species diversity (Tscharntke and Greiler 1995, Siemann et al. 1998, Jonas et al. 2002).

CONSERVATION IMPLICATIONS

Terrestrial invertebrate communities of high-diversity and low-diversity restorations in the RWBR were quite similar in my study. However, although these results show invertebrate communities in restoration plantings to be similar, they should be viewed with caution because at this time many of the high-diversity restorations have not been fully established. The high-diversity sites were restored up to 2 decades after the low-diversity sites, and therefore have not yet had time to become as well-established as the low-diversity sites. Additionally, there are many other factors such as age of restoration, basin size, management history, and surrounding landscape that likely influenced vegetation characteristics and invertebrate communities within the two restoration treatments.

Often times, one sampling technique is not effective at collecting the entire invertebrate community. In my study, I found several unique invertebrate taxa and different common invertebrate taxa between pitfall and sweep net samples. Standen

(2000) found pitfall samples and swish net/ D-vac suction trap samples to both be effective at capturing certain invertebrate taxa with each sampling method contributing different species assemblages to the overall assessment of invertebrate species assemblages in grasslands. Because there are different invertebrate communities captured by pitfall traps versus sweep nets, it is important that both of these techniques be used in future evaluations of grassland restorations. Although both pitfall traps and sweep nets are standard sampling techniques for terrestrial invertebrates, researchers and managers should be aware of the problems associated with these techniques (e.g., biased collections by attracting or repelling invertebrates, weather influences on capture rates, and limitations of effectiveness in tall and dense vegetation) and realize that these problems may affect the quality of the data they collect.

In previous studies, terrestrial invertebrate taxa have been used as indicator species to reflect changes or influences of management or restoration practices (Kremen et al. 1993, Arenz and Joern 1996, Pik et al. 2002). My study also showed that using terrestrial invertebrates as an indicator of restoration status is appropriate. Although there were very few differences in invertebrate abundance or diversity between high-diversity and low-diversity restorations, there was greater family richness for high-diversity restorations (for pitfall samples). Terrestrial invertebrates could be indicators of high-diversity restorations becoming more "diverse," because they may be more suitable to a wider range of invertebrate taxa.

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TABLE 1. Total number of invertebrates collected in pitfall trap ($n = 24$ for high-diversity plantings and $n = 22$ for low-diversity plantings) and sweep net samples ($n = 24$ for high-diversity plantings and $n = 23$ for low-diversity plantings) from high-diversity and low-diversity grassland restorations in the Rainwater Basin Region, Nebraska during June and July, 2002-2003.

Order	Family	Sweep net		Pitfall	
		High-diversity	Low-diversity	High-diversity	Low-diversity
Acarina		1	0	3	0
Araneae		62	115	633	633
Blattaria	Blattidae	---	---	2	0
Coleoptera	Cantharidae	0	4	88	9
	Carabidae	0	2	341	501
	Chrysomelidae	28	39	6	4
	Cicindelidae	---	---	10	4
	Cleridae	7	2	1	0
	Coccinellidae	13	11	2	4
	Curculionidae	8	5	11	26
	Dermestidae	---	---	2	0
	Elateridae	1	0	11	8
	Endomychidae	0	1	93	5
	Meloidae	2	0	5	5
	Melyridae	3	2	---	---
	Mordellidae	0	2	---	---
	Nitidulidae	---	---	34	30
	Scarabaeidae	---	---	68	29
	Silphidae	---	---	397	94
	Sphaeritidae	---	---	7	0
	Staphilinidae	---	---	20	7
Diplopoda		---	---	81	42
Diptera	Anthomyiidae	3	5	39	20
	Asilidae	4	3	---	---
	Bombyliidae	0	1	---	---
	Calliphoridae	1	1	4	51
	Chironimidae	---	---	2	0
	Chloropidae	1	1	---	---
	Culicidae	---	---	0	1
	Dolichopodidae	6	10	1	0
	Heleomyzidae	---	---	0	1
	Lauxaniidae	3	3	6	1
	Lochaeidae	0	1	---	---
	Megachilidae	---	---	1	0
	Muscidae	20	19	9	14
	Otitidae	3	1	---	---

Table 1. Continued.

	Platystomatidae	0	1	2	0
	Sarcophagidae	4	0	8	17
	Sciomyzidae	1	0	----	----
	Syrphidae	4	11	----	----
	Tachinidae	0	1	2	7
Ephemeroptera		1	0	----	----
Hemiptera	Berytidae	4	0	1	0
	Coreidae	----	----	1	2
	Lygaeidae	10	10	19	30
	Miridae	7	8	2	6
	Nabidae	13	18	1	0
	Pentatomidae	15	22	18	5
	Reduviidae	4	4	----	----
	Rhopalidae	0	8	2	2
	Scutelleridae	1	8	----	----
Homoptera	Cercopidae	15	5	16	4
	Cicadellidae	104	102	66	62
	Cixiidae	1	0	1	3
	Delphacidae	2	0	1	0
	Dictyopharidae	4	0	0	2
Hymenoptera	Apidae	----	----	3	9
	Braconidae	2	3	3	1
	Chalcididae	----	----	2	1
	Chalcididae	----	----	2	1
	Cynipidae	----	----	0	1
	Formicidae	12	36	2,500	1,656
	Halictidae	0	1	----	----
	Ichneumonidae	4	3	1	0
	Pompilidae	----	----	9	8
	Sphecidae	0	2	----	----
	Tenthredinidae	0	1	----	----
Lepidoptera	Noctuidae	14	8	53	50
	Pyrilidae	34	24	7	9
	Undetermined	0	1	0	1
Neuroptera	Chrysopidae	3	3	0	1
Odonata	Coenagrionidae	1	0	----	----
Oligochaetae		----	----	0	1
Orthoptera	Acrididae	33	123	172	108
	Gryllidae	4	5	320	270
	Tettigoniidae	40	50	56	21
	Total	505	686	5,145	3,767

TABLE 2. Mean abundance^a (number per transect) for 5 invertebrate taxa collected from pitfall traps in high-diversity and low-diversity restored grasslands during mid-June and mid-July in the Rainwater Basin Region, Nebraska, 2002-2003.

Invertebrate taxa	High-diversity (<i>n</i> = 24)		Low-diversity (<i>n</i> = 22)		<i>P</i> ^b		
	Mean	S.E.	Mean	S.E.	Treatment	Year	Period
Carabidae	14.21	2.52	22.77	6.57	0.286	0.403	0.044
Araneae	26.38	4.11	28.77	3.90	0.284	0.081	<0.001
Gryllidae	13.33	5.65	12.27	4.34	0.538	0.233	0.007
Formicidae	104.1	23.15	75.27	15.91	0.614	0.034	0.963
	7						
Silphidae	16.54	8.38	4.27	3.81	0.025	<0.001	<0.001

^a Mean abundance from 2 sampling periods during each year

^b *P* = *P*-value for treatment (high-diversity and low-diversity), year, and period effects from 3-way analysis of variance. Interaction effects for Carabidae included treatment x period ($F_{1,38} = 3.10$, $P = 0.087$) and year x period ($F_{1,38} = 3.06$, $P = 0.088$), for Araneae included treatment x year x period ($F_{1,38} = 4.25$, $P = 0.046$), for Gryllidae included treatment x period ($F_{1,38} = 3.86$, $P = 0.057$), and for Silphidae included treatment x period ($F_{1,38} = 5.43$, $P = 0.025$) and year x period ($F_{1,38} = 33.30$, $P = <0.001$).

TABLE 3. Mean abundance^a (number per transect) for 5 invertebrate taxa collected from sweep nets in high-diversity and low-diversity restored grasslands during mid-June and mid-July in the Rainwater Basin Region, Nebraska, 2002-2003.

Invertebrate taxa	High-diversity (<i>n</i> = 24)		Low-diversity (<i>n</i> = 22)		<i>P</i> ^b		
	Mean	S.E.	Mean	S.E.	Treatment	Year	Period
Araneae	2.58	0.60	5.00	1.03	0.013	0.281	0.026
Acrididae	1.37	0.51	5.35	2.83	0.025	0.071	0.050
Cicadellidae	4.33	1.23	4.43	0.98	0.253	<0.001	0.014
Tettigoniidae	1.67	0.51	2.17	0.63	0.529	0.047	0.238
Chrysomelidae	1.17	0.42	1.70	1.17	0.529	0.552	0.452

^a Mean abundance from 2 sampling periods during each year.

^b *P* = *P*-value for treatment (high-diversity and low-diversity), year, and period effects from 3-way analysis of variance. Interaction effects for Araneae included *year x period* ($F_{1,39} = 6.63$, *P* = 0.014) and for Cicadellidae included *year x treatment* ($F_{1,39} = 2.90$, *P* = 0.097).

TABLE 4. Comparison of vegetation characteristics for high-diversity and low-diversity grassland restorations in the Rainwater Basin Region, Nebraska during mid-June (Period 1) and mid-July (Period 2), 2002-2003.

Vegetation characteristic	Restoration treatment					Year				
	High-diversity (<i>n</i> = 24)		Low-diversity (<i>n</i> = 24)		<i>P</i> ^a	2002 (<i>n</i> = 24)		2003 (<i>n</i> = 24)		<i>P</i> ^a
	Mean	S.E.	Mean	S.E.		Mean	S.E.	Mean	S.E.	
Forb cover (%)	13.23	1.73	11.59	1.91	0.288	10.43	1.32	14.46	2.13	0.412
Grass cover (%)	47.01	3.03	48.88	2.88	0.481	39.86	2.76	55.94	2.01	<0.001
Sedge cover (%)	2.58	0.06	2.50	0.00	0.122	2.50	0.00	2.58	0.06	0.122
Dead material cover (%)	61.98	2.80	58.74	2.43	0.429	62.11	3.09	58.76	2.08	0.383
Bare ground cover (%)	11.45	2.71	11.36	1.80	0.495	13.74	3.12	9.08	0.85	0.802
Visual obstruction (cm)	22.11	2.49	24.05	2.07	0.355	20.50	1.95	25.57	2.53	0.028
Litter depth (cm)	0.61	0.03	0.59	0.04	0.744	0.57	0.04	0.64	0.03	0.320
Maximum vegetation height (cm)	62.17	2.70	69.67	2.56	0.046	59.48	2.37	72.03	2.44	0.001

^a *P* = *P*-value for treatment (high-diversity vs. low-diversity), year (2002 vs. 2003), and period (period 1 vs. 2) effects from analysis of variance; interaction effects included treatment × year × period ($F_{1,40} = 3.35$, $P = 0.075$) and period × year ($F_{1,40} = 4.30$, $P = 0.045$) for percent grass cover, and period × year ($F_{1,40} = 3.87$, $P = 0.056$) for visual obstruction.

TABLE 4. Continued.

Vegetation characteristic	Period				<i>P</i> ^a
	Period 1 (<i>n</i> = 24)		Period 2 (<i>n</i> = 24)		
	Mean	S.E.	Mean	S.E.	
Forb cover (%)	13.87	1.90	11.02	1.69	0.065
Grass cover (%)	52.99	2.20	42.81	3.22	0.003
Sedge cover (%)	2.58	0.06	2.50	0.00	0.122
Dead material cover (%)	55.95	2.42	64.91	2.54	0.014
Bare ground cover (%)	10.91	1.88	64.91	2.54	0.466
Visual obstruction (cm)	23.17	2.17	22.90	2.47	0.977
Litter depth (cm)	0.61	0.04	0.59	0.03	0.880
Maximum vegetation height (cm)	64.29	2.62	67.22	2.86	0.375

TABLE 5. Results of partial canonical correspondence analysis for terrestrial invertebrate taxa collected by pitfall traps and intraset correlation coefficients between selected vegetation variables and CCA axes for terrestrial invertebrate taxa from restored grasslands in the Rainwater Basin Region, Nebraska in 2002 and 2003.

Axes	1	2	3	4
Eigenvalues:	0.067	0.051	0.041	0.032
Species-environment correlations	0.825	0.727	0.863	0.774
Cumulative percentage variance of species data	4.1	7.3	9.8	11.8
% cover forb	-0.119	-0.084	-0.151	0.439
% cover dead material	0.047	-0.195	0.490	-0.545
% cover bare ground	0.529	0.289	0.147	0.485
Maximum vegetation height	-0.143	0.377	-0.568	-0.192
Litter depth	-0.698	0.136	0.222	-0.167

TABLE 6. Results of partial canonical correspondence analysis for terrestrial invertebrate taxa collected by sweep net and intraset correlation coefficients between vegetation variables and CCA axes for terrestrial invertebrate taxa from restored grasslands in the Rainwater Basin Region, Nebraska in 2002 and 2003

Axes	1	2	3	4
Eigenvalues:	0.153	0.120	0.113	0.100
Species-environment correlations:	0.843	0.759	0.874	0.823
Cumulative percentage variance of species data:	3.9	6.9	9.8	12.3
% cover forb	0.219	-0.152	0.548	-0.539
% cover dead material	-0.175	0.554	-0.070	-0.292
% cover bare ground	0.409	-0.394	0.319	0.378
Maximum vegetation height	0.641	0.044	-0.246	-0.099
Litter depth	-0.065	0.592	0.373	-0.006

Fig. 1. Mean \pm SE A) total invertebrate abundance, B) invertebrate family richness, and C) Shannon-Wiener diversity index for pitfall trap and sweep net samples collected from high-diversity ($n = 24$ for both samples) and low-diversity ($n = 22$ for pitfall trap and $n = 23$ for sweep net samples) plantings in the Rainwater Basin Region, Nebraska during 2002 and 2003. An asterisk indicates a significant difference ($P < 0.10$).

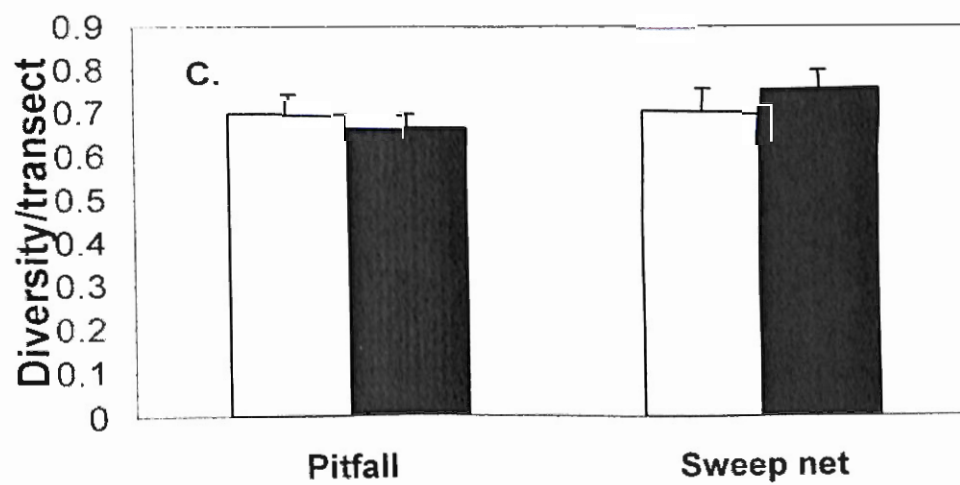
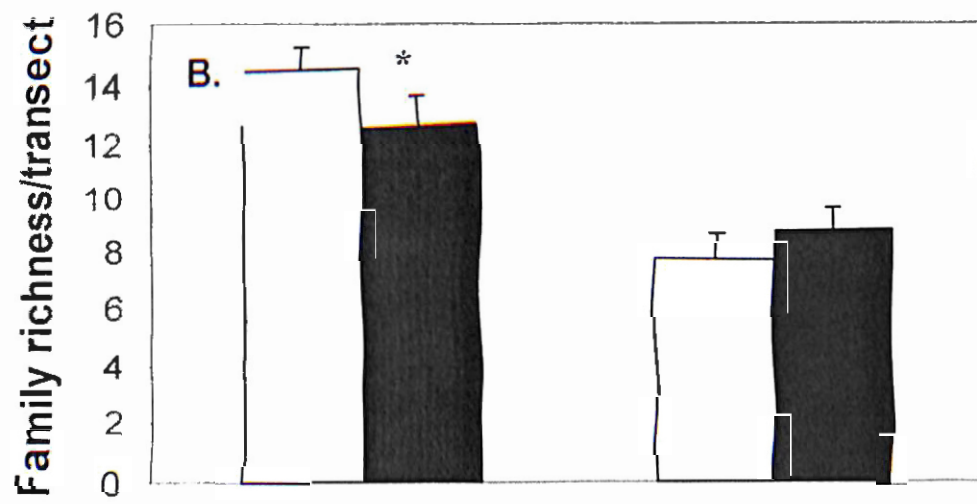
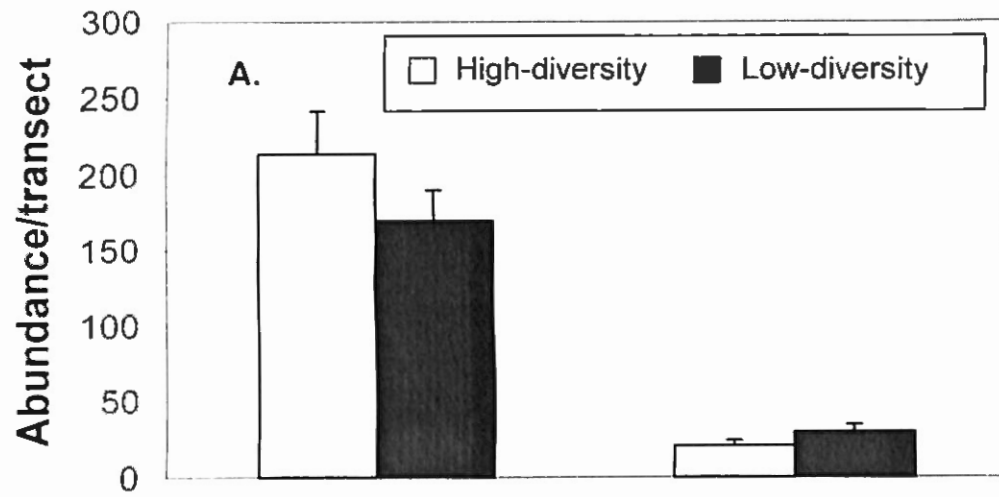


Fig. 2. Mean \pm SE A) total invertebrate abundance, B) invertebrate family richness, and C) Shannon-Wiener diversity index for pitfall trap and sweep net samples collected in 2002 ($n = 23$ for both samples) and 2003 ($n = 23$ for pitfall trap and $n = 24$ for sweep net samples) from restoration plantings in the Rainwater Basin Region, Nebraska. An asterisk indicates a significant difference ($P < 0.10$).

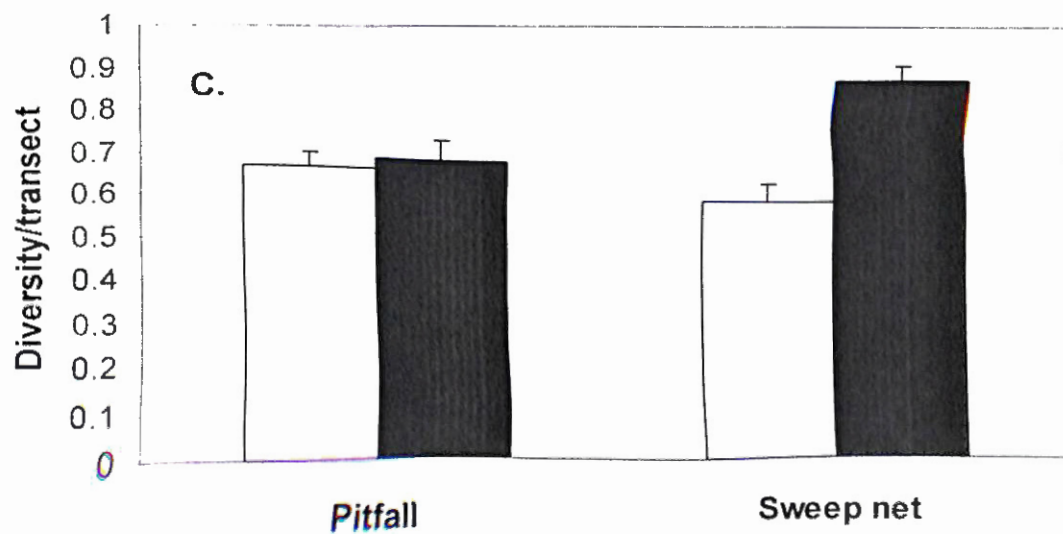
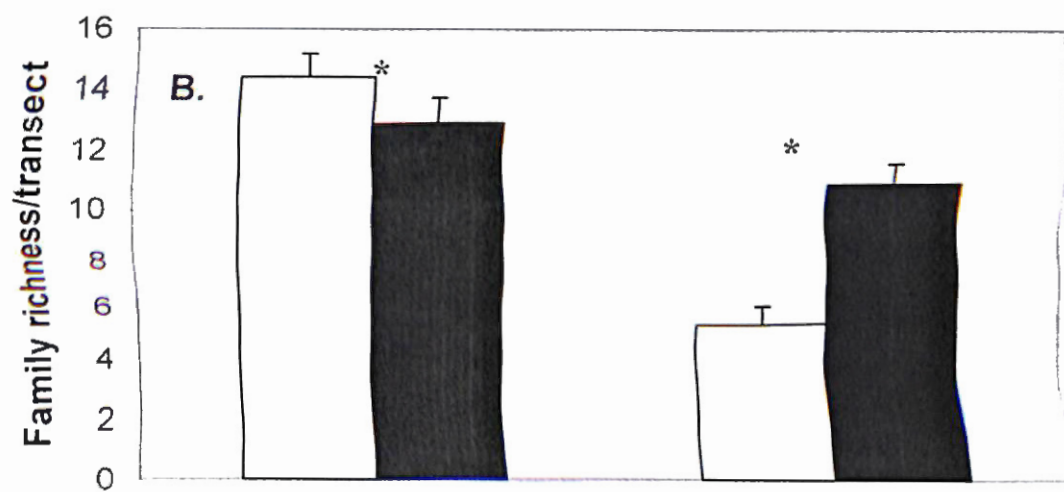
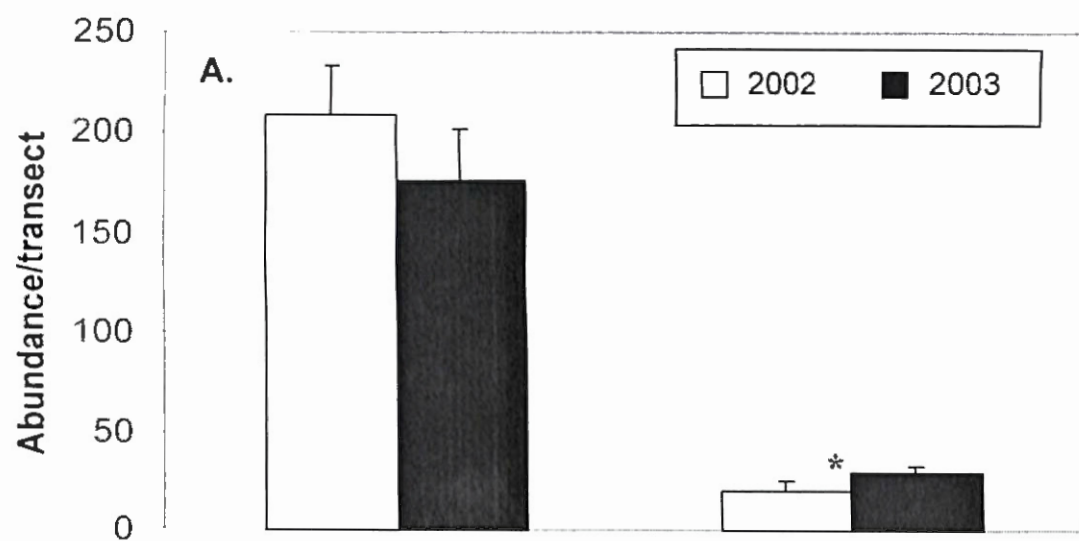


Fig. 3. Mean + SE A) total invertebrate abundance, B) invertebrate family richness, and C) Shannon-Wiener diversity index for pitfall trap and sweep net samples collected during mid-June (Period 1: $n = 23$ for both samples) and mid-July (Period 2: $n = 23$ for pitfall trap and $n = 24$ for sweep net samples) from restoration plantings in the Rainwater Basin Region, Nebraska, 2002 and 2003. An asterisk indicates a significant difference ($P < 0.10$).

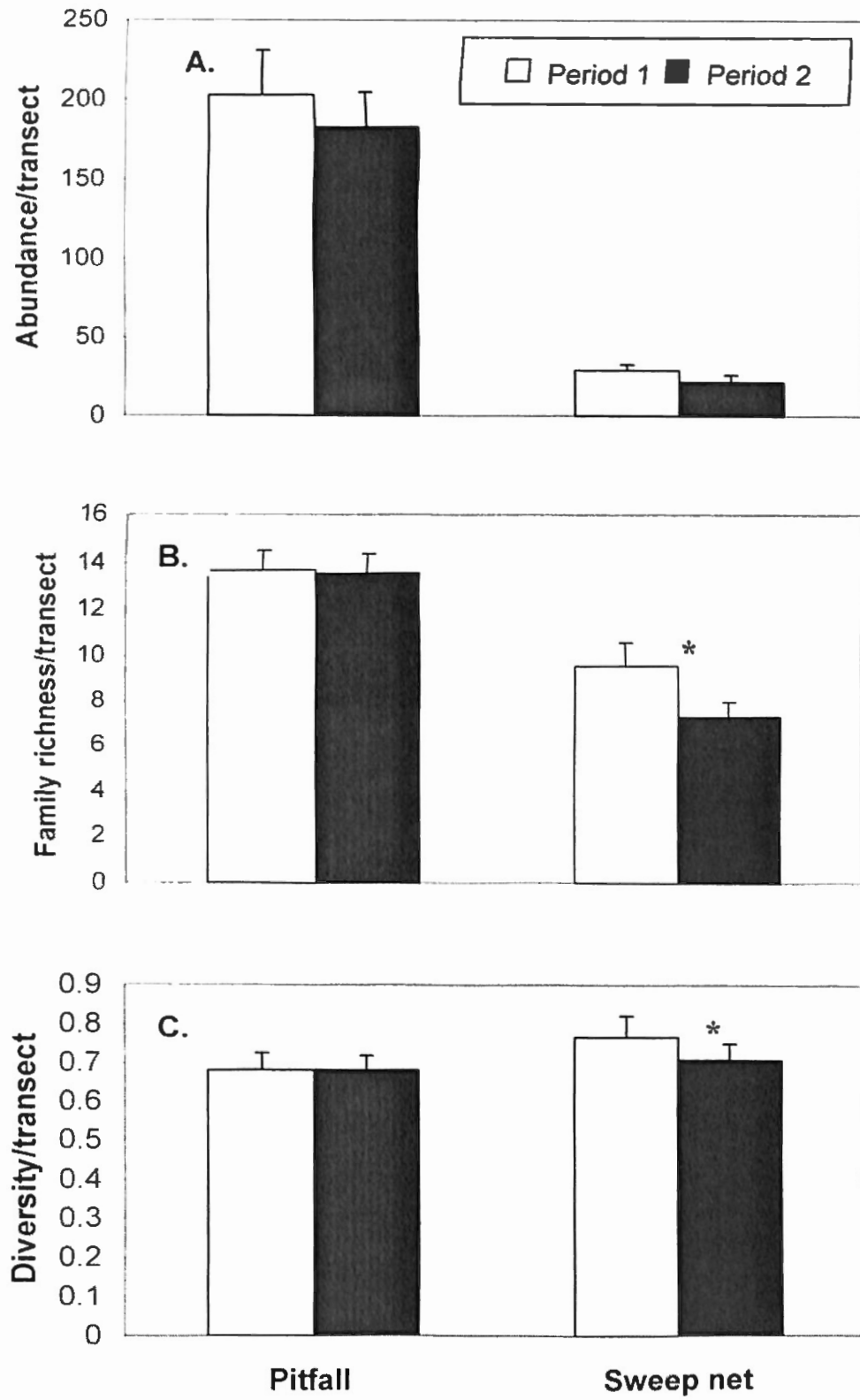


Fig. 4. Biplot of invertebrate taxa abundances collected by pitfall trap in restored grasslands in the Rainwater Basin Region, Nebraska in 2002 and 2003 and selected vegetation variables using partial canonical correspondence analyses. For each analysis, treatment, period, and year were set as covariables and blocks. Vegetation variables are represented by vectors and invertebrate taxa by unfilled triangles. Only taxa representing >5% of total number of individuals are included in plot. Individual taxa are designated by the first 6 letters of their name (see Table 1 for full taxa names and Table 4 for vegetation variable names).

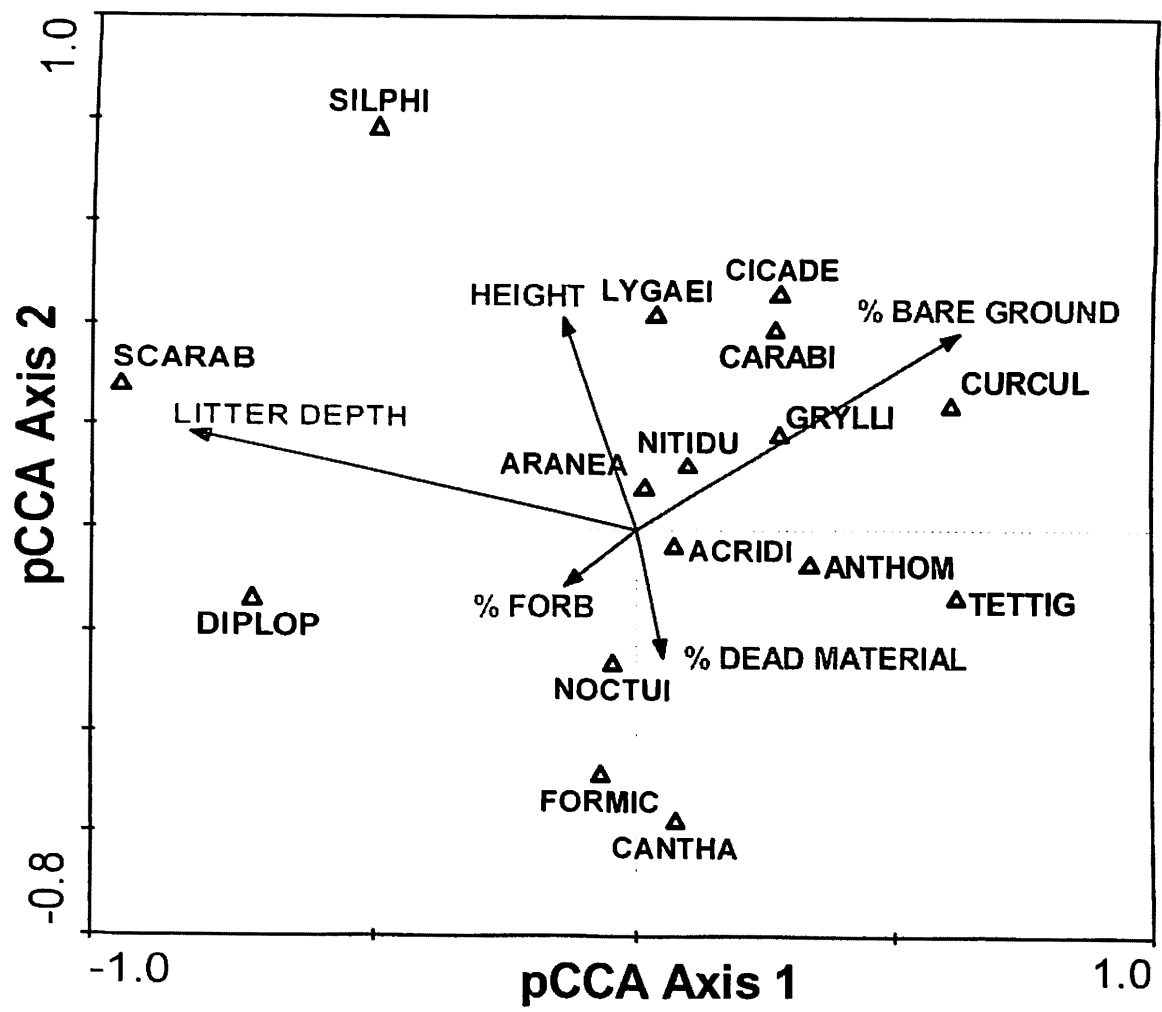


Fig. 5. Biplot of invertebrate taxa abundances collected by sweep net in restored grasslands in the Rainwater Basin Region, Nebraska in 2002 and 2003 and selected vegetation variables using partial canonical correspondence analyses. For each analysis, treatment, period, and year were set as covariables and blocks. Vegetation variables are represented by vectors and invertebrate taxa by unfilled triangles. Only taxa representing >5% of total number of individuals are included in plot. Individual taxa are designated by the first 6 letters of their name (see Table 1 for full taxa names and Table 4 for vegetation variable names).



APPENDIX A

Relative abundance^a (no./ transect) of grassland bird species that occurred in Waterfowl Production Areas (WPA) in western basin wetlands in the Rainwater Basin Region, Nebraska, 2002-2003.

Species	WPA					
	Atlanta	Cottonwood	Jensen	Peterson	Prairie Dog	Quadhammer
Dickcissel (<i>Spiza americana</i>)	1.17	0.50	1.67	0.50	1.83	2.17
Grasshopper sparrow (<i>Ammodramus savannarum</i>)	1.17	2.67	0.00	0.83	1.00	1.17
Common yellowthroat (<i>Geothlypis trichas</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Brown-headed cowbird (<i>Molothrus ater</i>)	0.83	0.50	0.17	0.50	1.67	0.00
Western meadowlark (<i>Sturnella neglecta</i>)	0.67	0.17	0.17	0.17	0.50	0.00
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	0.50	4.67	3.00	0.67	2.50	2.17
Mourning dove (<i>Zenaidura macroura</i>)	0.33	0.00	1.00	0.00	0.00	0.67
Upland sandpiper (<i>Bartramia longicauda</i>)	0.00	0.00	0.33	0.00	0.17	0.00
Eastern kingbird (<i>Tyrannus tyrannus</i>)	0.00	0.00	0.33	0.17	0.00	0.00
Barn swallow (<i>Hirundo rustica</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Bobolink (<i>Dolichonyx oryzivorus</i>)	0.00	0.00	0.00	0.50	0.00	0.00
Orchard oriole (<i>Icterus spurius</i>)	0.00	0.00	0.00	0.17	0.00	0.00
American goldfinch (<i>Carduelis tristis</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Mallard (<i>Anas platyrhynchos</i>)	0.00	0.33	0.00	0.00	0.00	0.00
Song sparrow (<i>Melospiza melodia</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Ring-necked pheasant (<i>Phasianus colchicus</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Sedge wren (<i>Cistothorus platensis</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Killdeer (<i>Charadrius vociferus</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Yellow-headed blackbird (<i>Xanthocephalus xanthocephalus</i>)	0.00	0.33	0.00	0.00	0.00	0.00

APPENDIX B

Relative abundance^a (no./ transect) of grassland bird species that occurred in Waterfowl Production Areas (WPA) in eastern basin wetlands in the Rainwater Basin Region, Nebraska, 2002-2003.

Species	WPA					
	Hultine	Mallard Haven	Massie	McMurtrey	Springer	Verona
Dickcissel (<i>Spiza americana</i>)	0.17	0.00	1.00	0.50	1.33	0.67
Grasshopper sparrow (<i>Ammodramus savannarum</i>)	1.50	0.00	0.00	0.33	0.33	0.50
Common yellowthroat (<i>Geothlypis trichas</i>)	0.00	0.67	0.67	0.00	0.17	0.00
Brown-headed cowbird (<i>Molothrus ater</i>)	0.00	0.00	1.50	1.67	0.00	1.00
Western meadowlark (<i>Sturnella neglecta</i>)	0.17	0.00	0.00	0.33	0.17	0.17
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	0.33	3.50	2.83	2.67	4.67	4.50
Mourning dove (<i>Zenaida macroura</i>)	0.00	0.00	0.00	0.00	0.00	0.17
Upland sandpiper (<i>Bartramia longicauda</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Eastern kingbird (<i>Tyrannus tyrannus</i>)	0.00	0.00	0.00	0.33	0.17	0.00
Barn swallow (<i>Hirundo rustica</i>)	0.00	0.00	0.33	0.00	0.17	0.00
Bobolink (<i>Dolichonyx oryzivorus</i>)	0.00	0.00	0.00	0.00	0.00	0.50
Orchard oriole (<i>Icterus spurius</i>)	0.00	0.00	0.00	0.00	0.00	0.00
American goldfinch (<i>Carduelis tristis</i>)	0.00	0.00	0.00	0.00	0.17	0.00
Mallard (<i>Anas platyrhynchos</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Song sparrow (<i>Melospiza melodia</i>)	0.00	0.17	0.00	0.00	0.00	0.00
Ring-necked pheasant (<i>Phasianus colchicus</i>)	0.00	0.17	0.00	0.00	0.00	0.00
Sedge wren (<i>Cistothorus platensis</i>)	0.00	0.00	0.00	0.00	0.00	0.17
Killdeer (<i>Charadrius vociferus</i>)	0.67	0.00	0.00	0.00	0.00	0.00
Yellow-headed blackbird (<i>Xanthocephalus xanthocephalus</i>)	0.00	1.50	0.00	0.00	0.00	0.00

APPENDIX B

Relative abundance^a (no./ transect) of grassland bird species that occurred in Waterfowl Production Areas (WPA) in eastern basin wetlands in the Rainwater Basin Region, Nebraska, 2002-2003.

Species	WPA					
	Hultine	Mallard Haven	Massie	McMurtrey	Springer	Verona
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Grasshopper sparrow (<i>Ammodramus</i> <i>savannarum</i>)	1.50	0.00	0.00	0.33	0.33	0.50
Common yellowthroat (<i>Geothlypis trichas</i>)	0.00	0.67	0.67	0.00	0.17	0.00
Brown-headed cowbird (<i>Molothrus ater</i>)	0.00	0.00	1.50	1.67	0.00	1.00
Western meadowlark (<i>Sturnella neglecta</i>)	0.17	0.00	0.00	0.33	0.17	0.17
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	0.33	3.50	2.83	2.67	4.67	4.50
Mourning dove (<i>Zenaida macroura</i>)	0.00	0.00	0.00	0.00	0.00	0.17
Upland sandpiper (<i>Bartramia longicauda</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Eastern kingbird (<i>Tyrannus tyrannus</i>)	0.00	0.00	0.00	0.33	0.17	0.00
Barn swallow (<i>Hirundo rustica</i>)	0.00	0.00	0.33	0.00	0.17	0.00
Bobolink (<i>Dolichonyx oryzivorus</i>)	0.00	0.00	0.00	0.00	0.00	0.50
Orchard oriole (<i>Icterus spurius</i>)	0.00	0.00	0.00	0.00	0.00	0.00
American goldfinch (<i>Carduelis tristis</i>)	0.00	0.00	0.00	0.00	0.17	0.00
Mallard (<i>Anas platyrhynchos</i>)	0.00	0.00	0.00	0.00	0.00	0.00
Song sparrow (<i>Melospiza melodia</i>)	0.00	0.17	0.00	0.00	0.00	0.00
Ring-necked pheasant (<i>Phasianus colchicus</i>)	0.00	0.17	0.00	0.00	0.00	0.00
Sedge wren (<i>Cistothorus platensis</i>)	0.00	0.00	0.00	0.00	0.00	0.17
Killdeer (<i>Charadrius vociferus</i>)	0.67	0.00	0.00	0.00	0.00	0.00
Yellow-headed blackbird (<i>Xanthocephalus</i> <i>xanthocephalus</i>)	0.00	1.50	0.00	0.00	0.00	0.00