ECOLOGICAL RESTORATION OF TALLGRASS PRAIRIE: GRAZING MANAGEMENT BENEFITS PLANT AND BIRD COMMUNITIES IN UPLAND AND RIPARIAN HABITATS

by

TRACEY N. JOHNSON

B.S., Texas A&M University at College Station, 2001

A THESIS

submitted in partial fulfillment of the requirements for the degree

MASTER OF SCIENCE

Division of Biology
College of Arts and Science

KANSAS STATE UNIVERSITY
Manhattan, Kansas

2006

Approved by:

Major Professor
Dr. Brett K. Sandercock

Abstract

Cattle-grazing is a dominant land use in the United States, with more than 300 million hectares of land grazed each year. The habitat changes facilitated by cattle grazing can influence resource availability and habitat selection for associated wildlife. To investigate the potential for changes in traditional livestock management to restore native grassland and riparian habitat, we evaluated biological community responses to winter-grazing and livestock exclusion at the Kansas Army Ammunition Plant in southeastern Kansas. In grassland habitats, we combined winter-grazing by domestic cattle and discontinued fertilization in an attempt to restore pastures dominated by tall fescue to native tallgrass prairie and improve habitat for grassland-breeding birds. We observed a decrease in tall fescue and an increase in native, warm-season grasses in winter-grazed pastures compared to fertilized, year-round grazed pastures. Grassland-breeding bird responses to winter-grazing were species-specific. Dickcissels preferred winter-grazed pastures, while Eastern Meadowlarks and Grasshopper Sparrows tended to prefer year-round grazed pastures. Dickcissels were negatively correlated with the presence of cattle during the breeding season and the abundance of tall fescue. Grasshopper Sparrows were negatively correlated with native, warm-season grass abundance and visual obstruction, but were positively correlated with forb abundance. Henslow's Sparrows and Common Yellowthroats were detected breeding in low numbers on pastures that had been winter-grazed for five years. Our results suggest that winter-grazing and discontinued fertilization of agricultural grasslands can direct semi-natural plant communities toward tallgrass prairie and benefit some grassland-breeding birds.

In riparian habitats, livestock were excluded from 1996 to 2005. We measured bird community responses in grazed and ungrazed sites using baseline data collected in 1996-97 and post-treatment data collected in 2004-05. Riparian bird community data were analyzed using robust design mark-recapture models that allowed us to evaluate changes in bird species richness while accounting for differences in detectability among species. We detected increases in species richness in both ungrazed and grazed treatments. We observed few differences in community vital rates between treatments; however, we did detect differences in guild responses. The changes observed within both grazed and ungrazed riparian bird communities were likely influenced by regional fluctuations in species richness and composition.

Table of Contents

List of Figures

List of Tables

Acknowledgements

A number of people deserve my thanks for their advice and support throughout my master's program. My advisor, committee members, friends, and fellow graduate students helped me to see this project through to completion.

For funding and logistical support, I would like to thank the Division of Biology at Kansas State University, the Department of Defense, the Kansas Cooperative Fish and Wildlife Research Unit, Kansas Department of Wildlife and Parks, Prairie Biotic Inc., the Kansas Ornithological Society, The Kansas Chapter of the Wildlife Society, Terry C. Johnson Center for Basic Cancer Research, the Kansas State Graduate Student Council, and the Biology Graduate Student Association. I would also like to thank all the wonderful people at the Kansas Army Ammunition Plant for making my time there so enjoyable, especially Chris Deurmyer. I would especially like to thank my advisor, Dr. Brett Sandercock, for all the help, advice, and expertise he contributed to this project. Dr. Sandercock was an excellent role model through his work ethic, attitude, and experience, and made my graduate experience at Kansas State successful and enjoyable.

I thank Drs. Dave Hartnett, John Blair, and Tony Joern for serving on my supervisory committee, and for the thoughts, advice, and time they contributed to this project. I thank Dr. Phil Gipson, Roger Applegate, and David Hoover for sharing data and providing logistical support and ideas. I would also like to thank all the other members of the Avian Ecology Lab, as well as the wonderful cohort of graduate students I was lucky enough to encounter while at

Kansas State. All of these people created a positive environment in which I worked on a daily basis, and for that I am grateful.

I would also like to thank Carol Blocksome, Max Goode, Gene Towne, and Tara Conkling for help with plant identification and vegetation work in the field. I am also indebted to John and Nell Ball for welcoming me into their home for two summers.

I am most thankful for my husband and colleague, Karl Kosciuch for his love, support, and encouragement. He has provided field help, logistical help, and editorial comments, and has shown patience through it all. I am forever grateful.

CHAPTER 1 - Introduction

Anthropogenic activities have altered biological communities worldwide, resulting in many species at risk of local or global extinction (Hooper et al. 2005). One important human activity known to affect biological communities is domestic livestock grazing (Cole and Landres 1996). Livestock grazing is a dominant land use in the United States, with more than 300 million hectares of land grazed each year (Lubowski et al. 2006). Grazing by livestock causes changes in vegetation species composition and cover for animal populations, soil compaction, changes in geomorphology, nutrient redistribution, declines in stream stability and water quality, changes in forest cover, and habitat fragmentation through the construction of fences and water developments (West 1993, Cole and Landres 1996). The habitat changes facilitated by cattle grazing can directly and indirectly influence resource availability and habitat selection for associated wildlife. In North America, the loss of high-quality grassland habitat from livestock grazing has contributed to severe population declines of several species of grassland breeding bird (Evans et al. 2005).

Many conservation plans for declining species include habitat restoration as a means of increasing population size and, thus, decreasing the risk of extinction; therefore, developing effective ecological restoration techniques is integral to mitigating recent declines in biodiversity (Schrott et al. 2005). Better approaches to restoration may be achieved through the development of an increased understanding of species responses to restoration techniques, and the development of effective restoration techniques based on existing agricultural practices will likely be the most successful conservation initiatives for species declining in the Great Plains (Larison et al. 2001, Peterjohn 2003).

The goals of this thesis were to integrate rangeland management with ecological restoration in an effort to improve habitat for breeding birds. More specifically, our goals were to determine 1) if winter-grazing by livestock can decrease the abundance of an introduced plant and restore native tallgrass prairie plant species, 2) if winter-grazing by livestock can improve grassland quality for breeding birds, and 3) if short-term livestock exclusion can increase breeding bird species richness in riparian habitats. The results from our research are applicable to current issues in restoration, range, and community ecology.

This thesis is organized into two core chapters. In chapter two, we evaluate community responses to winter-grazing and the suspension of fertilization as a potential restoration technique. We assess the effect of our restoration method on plant and bird species richness and community composition. In chapter three, we evaluate riparian bird community responses to livestock exclusion as a passive restoration method. We use robust design mark-recapture models for mixtures to estimate riparian bird species richness in fenced and grazed riparian sites. Additionally, we quantify community dynamics in both fenced and grazed sites to determine if there were differences in local extinction, colonization, or species turnover since the time of livestock exclusion. Chapter four is a synthesis of the major conclusions from our studies.

References

Cole, D.N., and P.B. Landres. 1996. Threats to wilderness ecosystems: impacts and research needs. Ecological Applications **6**: 168-184.

Evans, D.M., S.M. Redpath, S.A. Evans, D.A. Elston, and P. Dennis. 2005. Livestock grazing affects the egg size of an insectivorous passerine. Biology Letters **1**: 322-325.

Hooper, D.U., F.S. Chapin III, J.J. Ewel, A. Hector, P. Inchausti, S. Lavorel, J.H. Lawton, D.M. Lodge, M. Loreau, S. Naeem, B. Schmid, H. Setälä, A.J. Symstad, J. Vandermeer, and D.A. Wardle. 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecological Monographs **75**: 3-35.

Larison, B., S.A. Laymon, P.L. Williams, and T.B. Smith. 2001. Avian responses to restoration: nest-site selection and reproductive success in Song Sparrows. Auk **118**: 432-442.

Lubowski, R.N., M. Vesterby, S. Bucholtz, A. Baez, and M.J. Roberts. 2006. Major uses of land in the United States, 2002. Economic Information Bulletin No. EIB-14. U.S. Department of Agriculture, Economic Research Service. Washington D.C.

Peterjohn, B.G. 2003. Agricultural landscapes: can they support healthy bird populations as well as farm products? Auk **120**: 14-19.

CHAPTER 2 - Pasture to prairie? Restoration of grassland plant and bird communities in fescue pastures

Tracey N. Johnson and Brett K. Sandercock

Abstract

More than 95% of native tallgrass prairie has been lost to fragmentation or conversion, with detrimental effects on associated plant and animal communities. One important cause of grassland habitat loss has been the introduction of tall fescue, an exotic grass used as cattle forage. The restoration of tall fescue-dominated grasslands to native tallgrass prairie usually requires expensive, labor intensive methods such as burning, herbicide application, and reseeding. We attempted to restore tallgrass prairie using a combination of winter-grazing and cessation of fertilizer addition at the Kansas Army Ammunition Plant, Kansas, U.S.A. In 2004-05, we compared responses of grassland plant and bird communities in a chronosequence of pastures winter-grazed from 1-5 years to pastures grazed year-round (as a restoration starting point), and to native prairie remnants (as a restoration endpoint). The abundance and biomass of native, warm-season grasses was higher in winter-grazed pastures and tended to increase over time. Tall fescue abundance and biomass declined with successive years of winter-grazing; however, mean tall fescue abundance did not differ among grazing treatments. Grassland birds showed variable responses to winter-grazing: Dickcissel densities increased, but Eastern Meadowlark and Grasshopper Sparrow densities decreased with years of exposure to winter-grazing. Our experimental restoration method had positive effects on some tallgrass prairie plant and bird species in less than five years, although evaluation of long-term effects of winter-

grazing would be beneficial. Winter-grazing of tall fescue-dominated pastures could be an effective, low-cost method of tallgrass prairie restoration that allows continued utilization of rangelands while improving habitat quality for associated bird communities.

Introduction

Grasslands are one of the most endangered ecosystems in North America (Noss et al. 1995, Van Dyke et al. 2004). Within the Great Plains region of North America, more than 95% of native tallgrass prairie has been converted to rowcrop agriculture, fragmented, or modified by rangeland management practices (Samson and Knopf 1994). Habitat loss and changes in plant community composition and structure threaten grassland-dependent animal communities (Askins 1993, Horncastle et al. 2005). For example, the loss of quality grassland habitat has contributed to the decline of many grassland birds, including Grasshopper Sparrow (*Ammodramus savannarum*), Henslow's Sparrow (*A. henslowii*), Dickcissel (*Spiza americana*) and Upland Sandpiper (*Bartramia longicauda*; Askins 1993, Van Dyke et al. 2004, Sauer et al. 2005).

One important cause of grassland habitat loss has been the introduction of agricultural crop plants and exotic grasses used as cattle forage (Vickery et al. 1999*a*). A common exotic grass is *Lolium arundinaceum* (tall fescue), a cool-season perennial originally introduced from Europe in the 1800's (Clay and Holah 1999, Barnes 2004). Tall fescue is planted extensively for use as turf grass or cattle forage, and widely used in the revegetation of Conservation Reserve Program lands and reclaimed mine lands, covering more than 14 million ha in the United States (Buckner et al. 1979, Ball et al. 1993, Barnes 2004). Tall fescue is state-listed as an invasive species in ten states, but is widespread throughout the U.S. (Swearingen 2005, Walsh 1995). In the southern Great Plains and southeastern U.S., over-seeding of native grasslands with cool-season forage grasses, continuous cattle grazing, and annual nitrogen fertilization has resulted in the conversion of many native plant communities to pastures dominated by tall fescue (Mitchell et al. 1985, Barnes et al. 1995, Brummer and Moore 2000).

Restoration of tallgrass prairie requires development of effective, low cost ways to reduce the dominance of tall fescue. Fescue is an aggressive invader of native grasslands and reduces overall diversity in plant communities because it is tolerant to drought and grazing, and has potential for allelopathy (Barnes et al. 1995, Clay and Holah 1999, Renne et al. 2004, but see Spyreas et al., 2001). Fescue negatively affects animal populations in at least three ways. First, fescue can exclude native grasses and forbs that provide food for native wildlife. Second, tall fescue can alter vegetation structure, ultimately affecting resource availability for animal populations that require grassland habitats for food, nesting substrate, or escape cover (Scheiman et al. 2003). Last, if fescue is infected with the symbiotic fungal endophyte *Neotyphodium coenophialum* (Clavicipitaceae), it can be toxic to animals that consume vegetative parts or seeds (Barnes et al. 1995, Conover and Messmer 1996*a*,*b*, Durham and Tannenbaum 1998, Tannenbaum et al. 1998).

Habitat restoration is one of the most effective ways of reversing the loss of native grasslands and mitigating population declines for associated animals (Vickery et al. 1999*b*). The restoration of agricultural grasslands from semi-natural communities dominated by tall fescue to native warm-season grasses often involves some combination of herbicide application, prescribed burning, clipping to defoliate the tall fescue, or over-seeding with native grass species (Washburn et al. 2002, Wilson and Pärtel 2003, Barnes 2004). However, most of these methods are costly and labor-intensive, and herbicides can have detrimental effects on native grassland plants (Lawrence et al. 1995, Sheley and Krueger-Mangold 2003). Some researchers have suggested that livestock grazing can be used to control exotic or invasive plants, and ultimately direct plant communities toward more desirable species composition (Walker 1995, Olson 1999, Huwer et al. 2005, Lym 2005). Selective grazing by large herbivores influences competitive interactions among plant species, reducing competitively dominant species and allowing subordinate species to increase in abundance (Collins et al. 1998, Knapp et al. 1999, Chase et al. 2002). In environments with a seasonal component in which plant species density and palatability changes through time, grazing should exert a greater influence over species coexistence due to diet-switching that coincides with temporal changes in resource density over the growing seasons. In a seasonal system, selective grazing can ultimately lead to a reversal in the competitively dominant species (Hamback 1998).

In addition to the potential effects of selective grazing on returning agricultural grasslands to tallgrass prairie, manipulation of soil fertility can be useful to restoration efforts, especially when the nutrient requirements of exotic plants are known (D'Antonio and Meyerson 2002). Improvement of soil nutrient levels through fertilization allows exotic species to replace slow-growing native species that thrive under low nutrient conditions (Maron and Jeffries 2001). Some restorationists have had success restoring tallgrass prairie species solely by manipulating N availability in former agricultural fields (Baer et al. 2003, Averett et al. 2004, Kulmatiski and Beard 2006, but see Corbin et al. 2004).

Restoration in an agricultural setting can result in suitable conditions for breeding birds, and development of effective restoration techniques that are based on present agricultural practices will most likely be the key to successful conservation initiatives (Fletcher and Koford 2003, Peterjohn 2003). Renfrew and Ribic (2002) found that pastures can provide valuable habitat for grassland birds, and recommended that grazed pastures be managed for species of conservation concern. In light of the precipitous decline of many species of grassland birds, coupled with the extensive area within the United States that has been over-seeded with tall fescue, understanding bird-fescue relationships is important, especially in regions dominated by agriculture.

Our experimental restoration was an attempt to alter the competitive dynamics of plants within fescue-dominated pastures. We grazed cattle during the cool-season growing period of tall fescue and rested the pastures during the warm-season growing period to confer a competitive advantage to native warm-season grasses. Tall fescue is dependent on nitrogen addition to sustain rapid growth, and we combined winter grazing with suspension of bi-annual fertilization (Mitchell et al. 1985, Mazzanti et al. 1994). Our three main objectives were: 1) to evaluate the effects of winter-grazing by cattle on the abundance and biomass of tall fescue and native warm-season grasses, 2) to determine if densities of grassland songbirds were higher in winter-grazed rather than year-round grazed pastures, and 3) to identify which habitat variables are important in predicting the density of breeding grassland birds in tall fescue-dominated pastures. We predicted that if winter-grazing reduces the abundance or biomass of tall fescue and increases the abundance or biomass of native warm-season grasses thus restoring some of the native vegetation structure, breeding bird densities would be positively affected.

Methods

Study site

The study was conducted May-August of 2004-05 at the Kansas Army Ammunition Plant (KSAAP) in north central Labette County, Kansas (37°18'N, 95°10'W). KSAAP is a 5,555 ha military installation historically dominated by big bluestem (*Andropogon gerardii*), little bluestem (*Schizachrium scoparium*), switchgrass (*Panicum virgatum*), and indiangrass (*Sorghastrum nutans*) in upland areas, with deciduous and cross timbers forest along creeks and river valleys (Eifler et al. 1995). In the 1940's, pastures with native vegetation were over-seeded and replaced by tall fescue (*Lolium arundinaceuma*). Fescue pastures are presently managed for cattle production with year-round or seasonal grazing and bi-annual nitrogen fertilization. Fertilization consists of urea, potassium carbonate, and diammonium phosphate added in early spring in variable amounts based on soil tests for each pasture. Woody encroachment in grazed pastures is manually removed with a brush-hog once every five years (C. Deurmyer 2004, Kansas Army Ammunition Plant, Parsons, KS, personal communication). The study site contains silt loam surface soil with a clay or loam subsoil, and slopes range from 0-3% (Owens et al. 1990). In the two years of the study, total annual precipitation was 995 and 830 mm, of which 543 and 562 fell from April through September, respectively.

Experimental design and restoration approach

Our restoration approach involved top-down control of plant community structure through herbivory to create higher quality habitat for grassland-breeding birds. We attempted to reduce tall fescue and increase native warm-season grass abundance and biomass in experimental pastures by: 1) eliminating fertilizer addition, and 2) grazing fescue with cattle during its cool-season growing period (hereafter, winter-grazing). Cattle were allowed to graze from 1 October to approximately 1 May in all years of the study, which coincides with the growing period of tall fescue in southeastern Kansas. In the same pastures, cattle were excluded for the remainder of each year, 1 May to 1 October, which is the growing period of native warm-season grasses. Winter-grazing was initiated in pastures on the KSAAP beginning in 2000, and additional pastures entered the winter-grazing regime in subsequent years until we began evaluating the community responses in 2004. This created a chronosequence of pastures that had been winter-grazed for 1-2 (Winter1/2), 2-3 (Winter2/3), or 4-5 (Winter4/5) years, which allowed us to evaluate the short-term responses of grassland communities to winter-grazing. From our set of experimental pastures, we selected six pastures in which to quantify plant and breeding bird responses to winter-grazing. Two pastures were replicates of the 1-2 year treatment (winter-grazing initiated in 2003), two were replicates of the 2-3 year treatment (winter-grazing initiated in 2002), and two were replicates of the 4-5 year treatment (winter-grazing initiated in 2000). Pastures were chosen as replicates within each time step of the chronosequence based on the similarity of stocking rates between them. All grazed pastures included in the study averaged 94.7 ha (± 17.6 ha), and stocking rates averaged 2.2 ha per animal unit (Table 2.1). To minimize the potentially confounding effect of fire on vegetation community composition, and because prescribed burning was controlled in the vicinity of munitions production facilities at our study site, fire was excluded from all pastures from 2000-05. Fire typically increases the abundance of many native, warm-season grasses and would likely benefit our restoration efforts. However, we were interested in quantifying the efficacy of winter-grazing and cessation of fertilizer addition as a low-cost, relatively low-maintenance restoration method without the requirement of prescribed burns.

In addition to the six experimentally winter-grazed pastures, our sampling scheme included four sites that represented restoration endpoints against which changes in plant and bird communities could be measured. Two sites were pastures that were grazed year-round and fertilized every other year in early spring, a management practice common in southeastern Kansas. These pastures served as baselines against which changes in fescue abundance and biomass and breeding bird densities in winter-grazed pastures could be evaluated. Additionally, remnants of native vegetation still exist on the KSAAP in the form of right-of-ways and small hay meadows. Management of native prairie sites includes annual or bi-annual spring burning and cutting for hay in late July to maintain original native prairie plant communities. In 2005, we included two of these native prairie sites in our sampling scheme to serve as a reference for local abundances of native grassland plant species. Native prairie sites served as benchmark endpoints for our restoration efforts. The two native prairie sites chosen for the study were 10.3 and 6.9 ha in size, and were smaller than the experimental pastures, but were the largest remnants of native prairie locally available. Due to the size disparity among native prairie remnants and all other pastures included in the study and the potential area effects on grassland bird densities, we do not attempt to compare native prairie bird densities to those in other treatments (Johnson and Igl 2001, Winter et al. 2006). Native prairie sites were not grazed or cut for hay in either 2004 or 2005.

Vegetation sampling

To determine the number of transects necessary to adequately sample pasture plant communities, we first sampled three 50 m transects in a four year winter-grazed pasture with the highest apparent plant diversity and created species accumulation curves using program EstimateS (Colwell 2004). We sampled additional transects one at a time and added species to our accumulation curve until curves became asymptotic at 46 species for nine transects. All other pastures included in the study had nine transects, and all transects were established in a stratified random design so that we could identify as many species present in each pasture as possible. All transects were placed at least 50 m from the edge of the pasture to minimize potential edge effects on the plant community. Transects were marked with metal T-posts and mapped with a hand-held GPS unit.

Plant communities were sampled twice each growing season: once in early summer (late May to early June), and once in late summer (late July to mid-August) to account for differences in phenology among plant species. We used a modified step-point method to quantify plant community composition in all pastures. Species identities were recorded at one-meter intervals on each transect (Owensby 1973). To survey for rare or infrequent forb species, we used a nested sampling technique (Hickman et al. 2004). The closest rooted plant and forb to each sampling point were identified to species and recorded. This method yielded two sets of data for each transect; one set included all species detected and one set included forb species only. At each sampling point on a transect, we also categorized ground cover using four cover classes: live vegetation, litter, bare ground, or rock.

We quantified aboveground biomass in early August using nine 0.25 m^2 quadrats per pasture. Quadrats were placed one meter to either side of each vegetation transect, and all aboveground biomass within the quadrat was removed. In the second year of the study, the opposite side of each transect was used for biomass collection. In 2005 we built nine 0.5 m^2 cattle exclosures in year-round grazed pastures in an attempt to account for biomass removal by cattle during the warm-season growing period to better identify any differences in fescue biomass among grazing treatments. Biomass samples were sorted to three categories (tall fescue, native grasses, and forbs), dried in a drying oven for 48 hours at 60°C, and weighed.

To examine the potential effects of the fungal endophyte *Neotyphodium coenophialum*, we quantified the infection rates for fescue-dominated pastures in our study. We collected seeds from a minimum of 50 different individual fescue plants from each pasture from 16-20 June 2004. Seeds were collected by systematically walking a zig-zag pattern through the entire pasture and collecting seeds from the nearest plant at ten meter intervals, avoiding ditches, fence rows, and cattle tanks within the pasture. All seed samples were tested for the presence of *Neotyphodium coenophalium* at the Plant Disease Diagnostic Laboratory, Department of Plant

Pathology, Kansas State University. Infection rates per pasture represent the number of seeds out of a subset of random seeds (*n* = 33) that tested positive for the fungus.

Bird surveys

Within each pasture, we established two 250 m bird transects, each of which incorporated one vegetation transect. The starting point of each bird transect was placed at least 200 meters from habitat edges in order to minimize possible edge effects on bird communities. We conducted bird surveys from dawn to four hours after dawn from 16 May to 15 July in 2004 and 2005. We did not conduct surveys during periods of rain, fog, or wind in excess of 16 kph. The observer (T.N.J.) slowly walked transects while recording all birds seen or heard. The radial distance from the observer to the initial point of detection of each bird was measured to the nearest meter with a laser range finder, along with the angle of detection from the bird to the transect line. We conducted four bi-weekly bird surveys at each transect in 2004-05. We systematically varied the start time of bird surveys at each transect to minimize any potential within-morning temporal effects on bird detections.

To help evaluate differences in vegetation structure among treatments that might influence bird densities, we quantified visual obstruction along each bird transect using a Robel pole (Robel et al. 1970). We recorded visual obstruction values for the entire length of each bird transect at 10 m intervals.

Statistical analysis

To evaluate differences in plant community composition, we used the maximum observed frequency of each species from the two sampling periods within a season (since cover changes throughout the growing season) to calculate relative frequencies of species occurrence (Hickman et al. 2004). We then averaged across all nine vegetation transects to obtain representative frequencies for each pasture. Results of vegetation analyses are presented by averaging across all 18 transects within a treatment.

We tested for differences in mean functional groups (grass vs. forb) and plant species abundance among year-round grazed pastures, winter-grazed pastures, and native prairie sites using Kruskal-Wallis nonparametric mean comparison tests. Plant biomass data met assumptions of normality and homogeneity of variances when arcsine-root transformed. Therefore, we tested for differences in biomass among treatments using analysis of variance. Additionally, because we were interested in the amount of variation in plant responses explained by the time since initiating the winter-grazing treatment, we treated the grazing regime as a continuous variable (year-round to winter-grazed for five years) and used simple linear regression. All statistical tests were performed using SAS System Version 9. An alpha level of 0.05 was used to determine statistical significance and all tests were two-tailed.

To estimate breeding bird densities while accounting for differences in detectability among species, we used Program Distance Ver. 5 Release 5 (Thomas et al. 2005). Distance observations of singing males only were retained for analyses due to differences in detectability among sexes of a species. The number of encounters for each bird species per transect were sparse. To increase the accuracy with which density was estimated, observations from multiple visits to each transect were pooled and the length of each transect was multiplied by the number of visits to that transect. Observations at the greatest distance from each transect were excluded from analysis to increase the precision of our density estimates. For each species of interest, we fitted detection curves by transect and then averaged across all transects to obtain a mean density estimate per treatment (*n* = 4). Candidate models consisted of half-normal, uniform, hazard rate, or negative exponential detection functions with a cosine series expansion. Model fit was based on Kolmogorov-Smirnov and Cramer-von Mises goodness-of-fit tests. We selected among candidate models with different numbers of adjustment terms using minimum ΔAIC_c values.

For transects with too few detections of a species to model density in Program Distance, we used an alternate method of density estimation. We assumed the average effective strip width (ESW) from other transects within the same treatment, and calculated density of singing males ($\hat{D}$) using the equation:

$$\hat{D} = \frac{d}{L \times 2\,(\text{ESW})}$$

where d is the number of singing males detected, and L is the effort (length × number of visits) per transect. This method assumes a uniform detection function within the sampled area. We then determined if grassland bird densities could be explained by our gradient of grazing treatments using linear regression.

To investigate bird-vegetation relationships in fescue-dominated pastures, we modeled each bird species' density using plant community composition, grazing status (grazed or rested during the breeding season), and visual obstruction measurements. Plant community composition variables included: percent fescue, all native warm-season grass species, forbs, litter, and bare ground. Since many habitat variables are correlated, we performed a principal components analysis on the correlated variables to reduce them to two major axes. We then used the two principal components and the presence or absence of cattle in the breeding season as predictor variables in a linear regression.

Results

Plant communities

We detected a total of 27 graminoid species and 70 forb species in 2004, and 26 graminoid species and 86 forb species in 2005. Mean species richness was higher in winter-grazed pastures than year-round grazed pastures in 2004 ($F_{3,68}$ = 4.4, P = 0.007) and 2005 ($F_{4,85}$ = 17.4, $P <$ 0.0001; Table 2). The increase in overall species richness in winter grazed pastures was not driven by an increase in exotic species. In 2004, the lowest native species richness was in the year-round grazing treatment ($\bar{x}$ = 27.0 ± 11.0 species) and the highest native species richness was in the four-year winter grazed treatment ($\bar{x}$ = 37.0 ± 6.0). Exotic species comprised 37.7% (± 5.3%) and 24.9% (± 1.0%) of year-round and winter grazed plant communities, respectively. In 2005, native prairie remnants had the highest overall species richness (Table 2.2). The lowest native species richness was in the year-round grazing treatment ($\bar{x}$ = 22.0 ± 7.0) and the highest native species richness was in native prairie sites ($\bar{x}$ = 50.0 ± 0.0). Exotic species comprised 35.8% (± 1.71%), 27.4% (± 1.09%), and 15.4% (± 0.40%) of year-round grazed, winter grazed, and native prairie plant communities, respectively.

Plant community composition differed among treatments. In 2004, the proportion of the plant community comprised of grasses differed among treatments (χ^2_3 = 11.5, P = 0.009). One-year winter-grazed pastures had a smaller grass component (mean = 63.5 ± 8.7% SE grass; n = 18 for each treatment) than two-year (mean = 91.8 ± 2.6% grass) or four-year (mean = 85.1 ± 3.0% grass) winter-grazed pastures, but all other treatments were not statistically different. Forb composition was similar among treatments, and ranged from a mean of 19.4 ± 3.7% in two-year winter-grazed pastures to a mean of 27.0 ± 2.8% in one-year winter-grazed pastures. In 2005, the proportion of grass differed among treatments (χ^2_4 = 12.5, P = 0.01). Native prairie sites had a larger grass component (mean = 93.8 ± 1.3% grass) than year-round grazed pastures (mean = 83.2 ± 2.8% grass). Additionally, the proportion of forbs was different among treatments (χ^2_4 = 23.0, $P = 0.001$). Year-round grazed pastures had a higher percentage of forbs (mean = 28 ± 3.5% forbs) than all other treatments.

The native species of warm-season grasses all showed increasing trends in winter-grazed pastures (W1-5) compared to year-round grazed pastures (YR) in both years of the study (Fig. 2.1). The abundance of three of five native warm-season grasses was ranked year-round < winter-grazed < native prairie (Little bluestem: χ^2_8 = 80.0, $P < 0.0001$; Switchgrass: χ^2_8 = 63.7, $P < 0.0001$; Indiangrass: χ^2_8 = 93.0, $P < 0.0001$). Big bluestem increased in abundance in winter-grazed pastures compared to year-round grazed pastures, but the difference was not statistically significant in 2004 or 2005 (Fig. 2.1). Tall dropseed was the only warm-season grass whose abundance was greater in winter-grazed pastures than native prairie sites, but the difference was not statistically significant. Mean abundance of tall fescue was not different among grazing treatments in 2004 or 2005 (χ^2_8 = 10.6, $P = 0.16$); but, in 2005 we detected a significant decreasing trend for tall fescue abundance in winter-grazed pastures (Fig. 2.1).

General ground cover (vegetation, litter, bare ground, or rock) was not different among treatments in 2004 or 2005. In both years, the mean proportion of points that struck basal vegetation ranged from 0.02-0.07 along each transect ($n = 50$) whereas the mean proportion of points that struck litter was high across all treatments, ranging from 0.83-0.95. The mean proportion of points that struck bare ground ranged from 0.03-0.11.

In 2004, native warm-season grasses accounted for 1-6% of the total biomass (live and dead plant material) in year-round grazed pastures and 4-27% of the total biomass in winter-grazed pastures. Warm-season grass biomass was higher in two- and four-year winter-grazed pastures than year-round or one-year winter-grazed pastures ($F_{3,68} = 6.2$, $P = 0.0009$; Fig. 2.2). Tall fescue accounted for 9-22% of total biomass in year-round grazed pastures, and 10-25% of total biomass in winter-grazed pastures. We detected no difference in mean fescue biomass among grazing treatments ($F_{3,68} = 0.9$, $P = 0.46$). Mean total biomass did not differ among year-round and winter-grazed pastures ($F_{3,68} = 2.3$, $P = 0.09$); but, we did detect differences in forb biomass among grazing treatments ($F_{3,68} = 3.1$, $P = 0.03$). Differences in mean forb biomass were significant between year-round grazed pastures and two-year winter-grazed pastures.

In 2005, native warm-season grasses accounted for 1-6%, 1-34%, and 31-34% of the total biomass in year-round grazed pastures, winter-grazed pastures, and native prairie sites, respectively. Warm-season grass biomass differed among the five treatments ($F_{4,85} = 31.2$, $P < 0.0001$; Fig. 2.2). Three-year winter-grazed pastures and native prairie sites had significantly higher warm season grass biomass than all other pastures. Moreover, tall fescue accounted for 22-40%, 11-24%, and 2-4% of the total biomass in year-round grazed pastures, winter-grazed pastures, and native prairie sites, respectively. We detected a difference in fescue biomass among year-round and winter-grazed treatments when we accounted for growing season biomass removal by cattle in year-round grazed pastures ($F_{4,85} = 12.3$, $P < 0.0001$; Fig. 2.2). Year-round grazed pastures had more fescue biomass than all winter-grazed pastures, and native prairie sites had less tall fescue biomass than year-round, one-year, and five-year winter-grazed pastures. Mean total biomass differed among grazing treatments in 2005 ($F_{4,85} = 3.1$, $P = 0.02$); this effect was driven by inclusion of the ungrazed, native prairie sites (Fig. 2.2). Mean forb biomass also differed among treatments ($F_{4,85} = 3.8$, $P = 0.007$). Differences in mean forb biomass were significant between two-year winter-grazed pastures and both three-year winter-grazed pastures and native prairie sites.

All grazed pastures included in the study had the tall fescue fungal endophyte present. Fungal infection rates ranged from 42-100% per pasture (*n* = 8), and the average infection rate was 80.8% (± 6.9%, SE). The highest rate of infection was in a two-to-three-year winter-grazed pasture, and the lowest rate of infection was in a one-to-two year winter-grazed pasture.

Bird communities

We detected 43 species of upland birds in 2004-05. In 2004, mean species richness per transect ranged from 7.25 species (± 1.44, SE, *n* = 4) in year-round grazed pastures to 13 species (± 1.41, *n* = 4) in four-year winter-grazed pastures. In 2005, mean species richness per transect ranged from 6.5 species (± 1.85, *n* = 4) in year-round grazed pastures to 11.5 species (± 0.87, *n* = 4) in five-year winter-grazed pastures. The most frequently encountered grassland-breeding species were: Dickcissel (*Spiza Americana*), Eastern Meadowlark (*Sturnella magna*), and Grasshopper Sparrow (*Ammodramus savannarum*; Table 2.3). We observed higher Dickcissel densities in winter-grazed pastures, although the trend was significant only in 2005 (Table 2.3). Eastern Meadowlarks had higher densities in year-round grazed pastures, but the trend was significant only in 2004. Grasshopper Sparrows tended to have higher densities in year-round grazed pastures; however, the trend was not significant in either year of the study. Brown-headed Cowbirds (*Molothrus ater*) were encountered more frequently in pastures grazed year-round than in winter-grazed pastures in 2004 ($\chi^2 = 9.0$, $P = 0.003$) and 2005 ($\chi^2 = 13.0$, $P = 0.0003$). Henslow's Sparrow (*Ammodramus henslowii*) and Common Yellowthroat (*Geothlypis trichus*) were detected breeding in low abundances only in native prairie remnants and pastures that had been winter-grazed for five years. The latter three species were detected too infrequently to model densities in any treatment.

Grassland bird-vegetation relationships varied among species. The principal component analysis reduced all habitat variables to two principal components that explained 78.2% and 81.4% of the variation in plant community data in 2004 and 2005, respectively (Table 2.4). Loadings on the principal components indicate tall fescue abundance is described by PC1 and native grass abundance and visual obstruction is described by PC2. The presence of cattle during the breeding season explained 62% of variation in Dickcissel density in 2004 (Table 2.5). In 2005, the presence of cattle and tall fescue abundance (PC1; Table 2.4) explained 75% of the variation in Dickcissel density. In 2004, native warm-season grass abundance and the abundance of forbs and bare ground (PC2; Table 2.4) explained 47% of the variation in Grasshopper Sparrow density. We did not identify any vegetation variables that significantly predicted Eastern Meadowlark density in either year of the study.

Discussion

Plant community responses

Plant species richness was higher in winter-grazed pastures than year-round grazed pastures and was highest in native prairie remnants. The increase in plant species richness was not due to a higher number of exotic species. Indeed, native species comprised a higher proportion of the plant community in winter grazed pastures than in year-round grazed pastures. Winter cattle-grazing and cessation of nitrogen addition was effective at increasing the abundance and biomass of several native, warm-season grass species compared to pastures grazed year-round. Additionally, we observed a decrease in tall fescue biomass in winter-grazed pastures compared to year-round grazed pastures. After five years of winter-grazing, we did not observe differences in mean tall fescue abundance among treatments; however, we did detect a trend for tall fescue abundance to decline over time.

The higher plant species richness we observed in unfertilized, winter-grazed pastures compared to fertilized, year-round grazed pastures is not surprising. Species richness in tallgrass prairies is strongly affected by nitrogen (N) availability. Most native prairie soils are characterized by low N availability, where species richness tends to be higher than in soils with high N availability (Foster and Gross 1998). Although we did not quantify soil N, there most likely was a decreased amount of total N present in unfertilized pastures compared to actively fertilized, year-round grazed pastures. Rapid increases in species richness have previously been observed after the discontinuation of fertilizer addition to grasslands, usually as a result of decreased productivity (Mountford et al. 1996, Bullock et al. 2001). Additionally, grazing can increase plant species richness in communities that have an evolutionary history with large ungulates, such as tallgrass prairie, especially through selective grazing (Collins et al. 1998, Olf and Ritchie 1998). If cattle were indeed selectively grazing tall fescue at our study site, then the observed increases in species richness could also be influenced by a decrease in competition between tall fescue, the dominant species within the community, and subordinate plant species.

Our results suggest that cessation of fertilizer application coupled with winter-grazing can direct fescue-dominated plant communities toward native tallgrass prairie. Our experimental restoration method allowed native warm-season grasses to significantly increase in abundance and biomass at our study site. Although these grasses were still a relatively small component of plant communities after five years of winter-grazing (generally < 10%), we believe the observed increases are important. Native warm-season grasses are long-lived perennials that require several years before achieving a large population size, especially if overseeding or transplanting with native species was not part of the restoration (Kulmatiski and Beard 2006). Many of the native grasses that occurred at our study site (e.g., Indiangrass) were never detected in year-round grazed pastures, or were detected in very low abundances. An increase in abundance of perennials after just five years is encouraging, considering the potential lag time that may be required for these species to reproduce. In addition to the increase in abundance, we observed an increase in native warm-season grass biomass. Biomass values for these species were similar to those for the dominant tall fescue after two to three years of winter-grazing. The low abundance but high biomass for these species suggests that once established in winter-grazed pastures, native warm-season grasses can be successful.

Cessation of fertilization and winter-grazing did not eliminate tall fescue over 5 years, which persisted as the dominant plant species in the community. However, we observed a decrease in the abundance and biomass of tall fescue over time at our study site despite extremely high levels of endophyte infection. Both discontinued fertilization and winter-grazing likely contributed to this result. Although few studies have evaluated the efficacy of cattle grazing as a potential restoration method, grazing can influence plant community dynamics in such a way that allows subdominant plants to increase in population size in mesic grasslands (Rambo and Faeth 1999, Wilson and Pärtel 2003, Kimball and Schiffman 2003). Management techniques that influence plant community composition are useful as a restoration tool if they facilitate changes in exotic-dominated communities, especially when the exotic plant is a persistent introduced perennial like tall fescue (D'Antonio and Meyerson 2002, Kulmatiski and Beard 2006). Tall fescue is a perennial grass that reproduces sexually and with rhizomes - traits which allow it to persist belowground and make it difficult to extirpate (Burchick 1993, Gibson and Newman 2001). Long-term monitoring will be necessary to determine if tall fescue can be eradicated with winter-grazing and discontinued fertilization, or if additional control must be implemented in combination with our restoration method to facilitate transition of fescue-dominated communities to tallgrass prairie. If total eradication of tall fescue at our study site is possible with discontinued fertilization and winter-grazing, it will most likely require repeated years of treatment to affect belowground reproductive parts.

Bird community responses

Grassland breeding bird community responses to winter-grazing varied among species. We detected a significant increase in Dickcissel densities in winter-grazed pastures over time and compared to year-round grazed pastures. Eastern Meadowlarks and Grasshopper Sparrows tended to have higher densities in year-round grazed pastures, although the trend was significant only for Eastern Meadowlarks in 2004. Grasshopper Sparrows were negatively associated with native warm-season grass abundance and positively associated with the abundance of forbs and bare ground. Henslow’s Sparrows and Common Yellowthroats were only detected breeding in winter-grazed pastures and native prairie remnants.

Consistent with similar studies on grassland bird responses to management, we observed variation in species’ responses to our restoration method (Walk and Warner 2000, Applegate et al. 2002, Fontaine et al. 2004, Van Dyke et al. 2004, Lueders et al. 2006, Powell 2006). In our study, Dickcissel density was higher in winter-grazed pastures. Breeding Dickcissel density was negatively associated with the presence of cattle, and likely increased in winter-grazed pastures due to decreased habitat disturbance by cattle during the breeding season. This response is consistent with other studies that found Dickcissels prefer to nest in prairie undisturbed during the breeding season (Temple et al. 1999, Dechant et al. 2003*a*). Additionally, Dickcissel density was positively influenced by a decrease in the fescue component of the plant community. Walk and Warner (2000) found that Dickcissels in Illinois had higher abundances in warm-season rather than cool-season grass dominated sites. Many investigators have found that Dickcissels are positively associated with a high forb component for nesting (Dechant et al. 2003*a*). Indeed, Dickcissels typically use forbs for nesting at our study site (personal observation). We observed higher Dickcissel density in winter-grazed pastures which had fewer forbs than year-round grazed pastures. However, forbs tend to increase in plant communities grazed by cattle, where herbivores preferentially remove grass biomass (Towne et al. 2005). Since both treatments include some grazing, forbs may not be a limiting factor for Dickcissels at our study site.

Eastern Meadowlarks tended to have lower densities in winter-grazed pastures than in year-round grazed pastures. We did not identify any specific habitat characteristics that were correlated with Eastern Meadowlark density. Eastern Meadowlark habitat preferences vary widely among regions; however, they do show a tendency to respond favorably to moderate grazing, especially in the southern Great Plains region (Hull 2003, Powell 2006).

Grasshopper Sparrows also tended to have lower densities in winter-grazed pastures than year-round grazed pastures, although the trend was not significant. This result is consistent with other studies that found Grasshopper Sparrows to be more abundant in pastures grazed during the breeding season, and is thought to be influenced by increased availability of bare ground produced by grazing (Temple et al. 1999, Dechant et al. 2003*b*, Powell 2006). Grasshopper Sparrows at our study site were negatively associated with native warm-season grass abundance and visual obstruction, and positively associated with forb abundance. This result is also consistent with other studies, which found pastures dominated by introduced forage species had higher Grasshopper Sparrow abundance (Davis and Duncan 1999, Dechant et al. 2003).

Henslow's Sparrows and Common Yellowthroats were both detected in very low abundances at our study site. We observed Henslow's Sparrows on four-year winter-grazed pastures in 2004 only during the early part of the breeding season; however, these individuals were transients and were not subsequently detected. In 2005, we observed both Henslow's Sparrows and Common Yellowthroats displaying territorial behavior in the same (then five-year) winter-grazed pastures. The only other plots in which we detected these species were native prairie remnants. The fact that Henslow's Sparrows were present in winter-grazed pastures is surprising, given that other studies have shown that grazing has a negative influence on Henslow's Sparrow populations (Herkert 2003, Powell 2006). However, Henslow's Sparrows could prefer winter-grazed pastures to year-round grazed pastures at our study site due to decreased habitat disturbance by cattle during the breeding season.

Brown-headed Cowbird abundance was lower in winter-grazed pastures than year-round grazed pastures. This result is not surprising because Brown-headed Cowbirds are strongly associated with the presence of livestock (Goguen and Mathews 2001). At our study site, livestock were present during the breeding season only in year-round grazed pastures. Parasitism rates by Brown-headed Cowbirds are highly correlated with cowbird abundance, and tend to be higher in grazed areas (Goguen and Mathews 2000, Jensen and Cully 2005, Patten et al. 2006). If parasitism rates are reduced at our study sites in winter-grazed pastures due to decreased cowbird abundance, winter-grazing could be indirectly beneficial to any grassland bird species by reducing brood parasitism rates.

Generally, winter-grazing of fescue-dominated pastures could be beneficial to some grassland bird populations in the Midwest. However, we must be cautious in interpreting higher population density as an indicator of increased habitat quality (Vickery et al. 1992). Animal population densities are dynamic and can change in response to many factors not included in our study, including patch size and habitat connectivity. Additionally, if bird densities are a function of settlement decisions based on vegetation conditions at the time of settlement, then fescue-dominated pastures could be an ecological trap because fescue cover peaks earlier than most of the native grasses and could be an unreliable indicator of habitat quality (Lloyd and Martin 2005, Shochat et al. 2005). If fescue-dominated pastures are ecological traps or population sinks for breeding birds, then any management technique that does not reduce tall fescue to a subdominant component of the plant community would not be adequate for improving grassland bird habitat. Habitat-specific demographic rates such as reproductive success or site propensity, and their ability to positively affect population growth, must be evaluated before we can conclude that winter-grazing is beneficial to any bird species (Smallwood 2001, Schrott et al. 2005).

Management recommendations

Loss of native grassland has occurred over such an extensive area that in many instances, habitat restoration is important for the persistence of threatened or endangered species. In many areas, effective conservation may ultimately depend on the creation or enhancement of artificial grassland habitats, which dictates a real need for effective restoration methods (Vickery et al. 1999*a*). Better approaches to restoration may be achieved through the development of land-management practices that support both wildlife and landowners, as well as increased understanding of plant and animal responses to restoration techniques (Samson and Knopf 1994, Larison et al. 2001, Murphy 2003). Our results contribute to the understanding of grassland plant and bird responses to an experimental restoration method which attempts to integrate multiple management goals that benefit grassland-dependent bird populations and allow utilization of pastureland. We hope that winter-grazing can be incorporated into future restoration efforts for native grasslands.

References

Applegate, R.D., B.E. Flock, and G.E. Horak. 2002. Spring burning and grassland area: effects on Henslow's Sparrow (*Ammodramus henslowii* [Audubon]) and Dickcissel (*Spiza americana* [Gmelin]) in eastern Kansas, USA. Natural Areas Journal **22**: 160-162.

Askins, R.A. 1993. Population trends in grassland, shrubland, and forest birds in eastern North America. Current Ornithology **11**: 1-33.

Averett, J.M., R.A. Klips, L.E. Nave, S.D. Frey, and P.S. Curtis. 2004. Effects of soil carbon amendment on nitrogen availability and plant growth in an experimental tallgrass prairie restoration. Restoration Ecology **12**: 568-574.

Baer, S.G., J.M. Blair, S.L. Collins, and A.K. Knapp. 2003. Soil resources regulate productivity and diversity in newly established tallgrass prairie. Ecology 84: 724-735.

Ball, D. M., J. F. Pederson, and G. D. Lacefield. 1993. The tall fescue endophyte. American Scientist **81**: 370-379.

Barnes, T.G., L.A. Madison, J.D. Sole, and M.J. Lackie. 1995. An assessment of habitat quality for Northern Bobwhite in tall-fescue dominated fields. Wildlife Society Bulletin **23**: 231-237.

Barnes, T. G. 2004. Strategies to convert exotic grass pastures to tallgrass prairie communities. Weed Technology **18**: 1364-1370.

Brummer, E.C. and K.J. Moore. 2000. Persistence of perennial cool-season grass and legume cultivars under continuous grazing by beef cattle. Agronomy Journal **92**: 466-471.

Buckner, R. C., J. B. Powell, and R. V. Frakes. 1979. Historical development. In: R. C. Buckner, L. P. Bush (Eds.), Tall Fescue. Agronomy Monograph 20:1-8. Madison, WI.

Bullock, J.M., J. Franklin, M.J. Stevenson, J. Silvertown, S.J. Coulson, S.J. Gregory, and R. Tofts. 2001. A plant trait analysis of responses to grazing in a long-term experiment. Journal of Applied Ecology **38**: 253-267.

Burchick, M. 1993. The problems with tall fescue in ecological restoration. Wetland Journal **5**: 16.

Chase, J.M., P.A. Abrams, J.P. Grover, S. Diehl, P. Chesson, R.D. Holt, S.A. Richards, R.M. Nisbet, and T.J. Case. 2002. The interaction between predation and competition: a review and synthesis. Ecology Letters **5**: 302-315.

Clay, K., and J. Holah. 1999. Fungal endophyte symbiosis and plant diversity in successional fields. Science **285**: 1742-1744.

Collins, S.L., A.K. Knapp, J.M. Briggs, J.M. Nlair, and E.M. Steinauer. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science **280**: 745-747.

Colwell, R. K. 2004. EstimateS: Statistical estimation of species richness and shared species from samples. Version 7.0. User's guide and application published at http://purl.oclc.org/estimates.

Conover, M.R., and T.A. Messmer. 1996*a*. Feeding preferences and changes in mass of Canada Geese grazing endophyte-infected tall fescue. Condor **98**: 859-862.

Conover, M.R., and T.A. Messmer. 1996*b*. Consequences for captive zebra finches of consuming tall fescue seeds infected with the endophytic fungus *Acremonium coenophialum*. Auk **113**: 492-495.

Corbin, J.D., C.M. D'Antonio, and S.J. Bainbridge. 2004. Tipping the balance in the restoration of native plants: experimental approaches to changing the exotic: native ratio in California grassland. Pages 154–179 in M.S. Gordon and S.M. Bartol, editors.

Experimental approaches to conservation biology. University of California Press, Berkeley.

D'Antonio, C.D. and L.M. Meyerson. 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restoration Ecology **10**: 703-713.

Davis, S.K. and D.C. Duncan. 1999. Grassland songbird occurrence in native and crested wheatgrass pastures of southern Saskatchewan. Studies in Avian Biology **19**: 211-218.

Dechant, J. A., M. L. Sondreal, D. H. Johnson, L. D. Igl, C. M. Goldade, A. L. Zimmerman, and B. R. Euliss. 2003. Effects of management practices on grassland birds: Dickcissel. Northern Prairie Wildlife Research Center, Jamestown, ND. Northern Prairie Wildlife Research Center Online. http://www.npwrc.usgs.gov/resource /literatr/grasbird /dick/dick.htm (Version 12DEC2003).

Dechant, J. A., M. L. Sondreal, D. H. Johnson, L. D. Igl, C. M. Goldade, M. P. Nenneman, and B. R. Euliss. 2003. Effects of management practices on grassland birds: Grasshopper Sparrow. Northern Prairie Wildlife Research Center, Jamestown, ND. Northern Prairie Wildlife Research Center Online. http://www.npwrc.usgs.gov/resource /literatr/grasbird/grsp/grsp.htm (Version 12AUG2004).

Durham, W.F., and M.G. Tannenbaum. 1998. Effects of endophyte consumption on food intake, growth, and reproduction in prairie voles. Canadian Journal of Zoology **76**: 960-969.

Fletcher, R.J. and R.R. Koford. 2003. Changes in breeding bird populations with habitat restoration in northern Iowa. American Midland Naturalist **150**: 83-94.

Fontaine, A.L., P.L. Kennedy, and D. H. Johnson. 2004. Effects of distance from cattle water developments on grassland birds. Journal of Range Management **57**: 238-242.

Foster, B.L., and K.L. Gross. 1998. Species richness in a successional grassland: effects of nitrogen enrichment and plant litter. Ecology **79**: 2593-2602.

Gibson, D.J. and J.A. Newman. 2001. *Festuca arundinacea* Schreber (*F. elatior* L. ssp. *arundinacea* (Schreber) Hackel). Journal of Ecology **89**: 304-324.

Goguen, C.B. and N.E. Mathews. 2000. Local gradients of cowbird abundance and parasitism relative to livestock grazing in a western landscape. Conservation Biology **14**: 1862-1869.

Goguen, C.B. and N.E. Mathews. 2001. Brown-headed Cowbird behavior in relation to livestock grazing. Ecological Applications **11**: 1533-1544.

Hamback, P.A. 1998. Seasonality, optimal foraging, and prey coexistence. American Naturalist **152**: 881-895.

Herkert, J. R. 2003. Effects of management practices on grassland birds: Henslow's Sparrow. Northern Prairie Wildlife Research Center, Jamestown, ND. Northern Prairie Wildlife Research Center Online. http://www.npwrc.usgs.gov/resource /literatr/grasbird/hesp/hesp.htm (Version 12DEC2003).

Hickman, K.R., D.C. Hartnett, R.C. Cochran, and C.E. Owensby. 2004. Grazing management effects on plant species diversity in tallgrass prairie. Journal of Range Management **57**: 58-65.

Horncastle, V. J., E. C. Hellgren, P. M. Mayer, A. C. Ganguli, D. M. Engle, and D. M. Leslie, Jr. 2005. Implications of invasion by *Juniperous virginiana* on small mammals in the southern Great Plains. Journal of Mammology **86**: 1144-1155.

Hull, S. D. 2003. Effects of management practices on grassland birds: Eastern Meadowlark. Northern Prairie Wildlife Research Center, Jamestown, ND. Northern Prairie Wildlife

Research Center Online. http://www.npwrc.usgs.gov/resource /literatr /grasbird/eame/eame.htm (Version 12DEC2003).

Huwer, R.K., D.T. Briese, P.M. Dowling, D.R. Kemp, W.M. Lonsdale, D.L. Michalk, M.J. Neave, A.W. Sheppard, and T.L. Woodburns. 2005. Can an integrated management approach provide a basis for long-term prevention of weed dominance in Australian pasture systems? Weed Research **45**: 175-192.

Jensen, W.E. and J.F. Cully. 2005. Geographic variation in Brown-headed Cowbird (*Molothrus ater*) parasitism on Dickcissels (*Spiza americana*) in great plains tallgrass prairie. Auk **122**: 648-660.

Johnson, D.H., and L.D. Igl. 2001. Area requirements of grassland birds: a regional perspective. Auk **118**: 24-34.

Kimball, S., and P. Schiffman. 2003. Differing effects of cattle grazing on native and alien plants. Conservation Biology **17**: 1681-1693.

Knapp, A.K., J.M. Blair, J.M. Briggs, S.L. Collins, D.C. Hartnett, L.C. Johnson, and E.G. Towne. 1999. The keystone role of bison in North American tallgrass prairie. BioScience **49**: 39-50.

Kulmatiski, A. and K.H. Beard. 2006. Activated carbon as a restoration tool: potential for control of invasive plants in abandoned agricultural fields. Restoration Ecology **14**: 251-257.

Larison, B., S.A. Laymon, P.L. Williams, and T.B. Smith. 2001. Avian responses to restoration: nest-site selection and reproductive success in Song Sparrows. Auk **118**: 432-442.

Lawrence, B.K., S.S. Waller, L.E. Moser, B.E. Anderson, and L.L. Larson. 1995. Weed suppression with grazing or atrazine during big bluestem establishment. Journal of Range Management **48**: 376-379.

Lloyd, J.D. and T.E. Martin. 2005. Reproductive success of chestnut-collared longspurs in native and exotic grasslands. Condor **107**: 363-374.

Lueders, A.S., P.L. Kennedy, and D.H. Johnson. 2006. Influences of management regimes on breeding bird densities and habitat in mixed-grass prairie: an example from North Dakota. Journal of Wildlife Management **70**: 600-606.

Lym, R.G. 2005. Integration of biological control agents with other weed management technologies: successes from the leafy spurge (*Euphorbia esula*) IPM program. Biological Control **35**: 366-375.

Maron, J.L. and R.L. Jeffries. 2001. Restoring enriched grasslands: effects of mowing on species richness, productivity, and nitrogen retention. Ecological Applications **11**: 1088-1100.

Mazzanti, A., G. Lemaire, and F. Gastal. 1994. The effect of nitrogen fertilization upon the herbage production of tall fescue swards continuously grazed with sheep. Grass and Forage Science **49**: 111-120.

Mitchell. R.L., A.L. Ewing, and W.E. McMurphy. 1985. N, P, and K fertilization of tall fescue (*Festuca arundinacea* Schreb.) overseeded range in eastern Oklahoma. Journal of Range Management **38**: 455-457.

Mountford, J.O., K.H. Lakhani, and R.J. Holland. 1996. Reversion of grassland vegetation following the cessation of fertilizer application. Journal of Vegetation Science **7**: 219-228.

Murphy, M.T. 2003. Avian population trends within the evolving agricultural landscape of eastern and central United States. Auk **120**: 20-34.

Noss, R. F., E. T. LaRoc, III, and J. M. Scott. 1995. Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation. Biological Report 28. Washington D.C.: National Biological Service, U.S. Department of the Interior.

Olf, H. and M.E. Ritchie. 1998. Effects of herbivores on grassland plant diversity. Trends in Ecology and Evolution **13**: 261-265.

Olson, B.E. 1999. Grazing and weeds. Pages 85-97 in R.L. Sheley and J.K. Petroff, editors. Biology and management of noxious rangeland weeds. Oregon State University Press, Corvallis, Oregon.

Owens, H.D., H.V. Campbell, E.L. Fleming, S.P.Graber, and D.W. Swanson. 1990. Soil survey of Labette County, Kansas. U.S. Department of Agriculture, United States Government Printing Office, Washington, D.C. 128pp.

Owensby, C.E. 1973. Modified step-point system for botanical composition and basal cover estimates. Journal of Range Management **26**: 302-303.

Patten, M.A., E. Shochat, D.L. Reinking, D.H. Wolfe, and S.K. Sherrod. 2006. Habitat edge, land management, and rates of brood parasitism in tallgrass prairie. Ecological Applications **16**: 687-695.

Peterjohn, B.G. 2003. Agricultural landscapes: can they support healthy bird populations as well as farm products? Auk **120**: 14-19.

Powell, A.F.L.A. 2006. Effects of prescribed burns and bison grazing on breeding bird abundances in tallgrass prairie. Auk **123**: 183-197.

Rambo, J.L. and S.H. Faeth. 1999. Effect of vertebrate grazing on plant and insect community structure. Conservation Biology **13**: 1047-1054.

Renfrew, R.B. and C.A. Ribic. 2002. Influence of topography on density of grassland passerines in pastures. American Midland Naturalist **147**: 315-325.

Renne, I.J., B.G. Rios, J.S. Fehmi, and B.F. Tracy. 2004. Low allelopathic potential of an invasive forage grass on native grassland plants: a cause for encouragement? Basic and Applied Ecology **5**: 261-269.

Robel, R.J., J.N. Briggs, A.D. Dayton, and L.C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management **23**: 295-297.

Samson, F. and F. Knopf. 1994. Prairie conservation in North America. BioScience **44**: 418-421.

Sauer, J.R., J.E. Hines, and J. Fallon. 2005. The North American Breeding Bird Survey, Results and Analysis 1966-2004. Version 2005. USGS Patuxent Wildlife Research Center, Laurel, MD.

Scheiman, D.M., E.K. Bollinger, and D.H. Johnson. 2003. Effects of leafy spurge infestation on grassland birds. Journal of Wildlife Management **67**: 115-121.

Schrott, G.R., K.A. With, and A.W. King. 2005. Demographic limitations of the ability of habitat restoration to rescue declining populations. Conservation Biology **19**: 1181-1193.

Sheley, R.L., and J. Krueger-Mangold. 2003. Principles for restoring invasive plant-infested rangeland. Weed Science **51**: 260-265.

Shochat, E., M.A., Patten, D.W. Morris, D.L. Reinking, D.H. Wolfe, and S.K. Sherrod. 2005. Ecological traps in isodars: effects of tallgrass prairie management on bird nest success. Oikos **111**: 159-169.

Smallwood, K.S. 2001. Linking habitat restoration to meaningful units of animal demography. Restoration Ecology **9**: 253-261.

Spyreas, G., D.J. Gibson, and B.A. Middleton. 2001. Effects of endophyte infection in tall fescue (*Festuca arundinacea: Poaceae*) on community diversity. International Journal of Plant Sciences **162**: 1237-1245.

Swearingen, J. 2005. Alien plant invaders of natural areas. Plant Conservation Alliance, Alien Plant Working Group. http://www.nps.gov/plants/alien/factmain.htm

Tannenbaum, M.G., S.L. Seematter, and D.M. Zimmerman. 1998. Endophyte-infected and uninfected fescue seeds suppress white-footed mouse (*Peromyscus leucopus*) reproduction. American Midland Naturalist **139**: 114-124.

Temple, S.A., B.M. Fevold, L.K. Paine, D.J. Undersander, and D.W. Sample. 1999. Nesting birds and grazing cattle: accommodating both on Midwestern pastures. Studies in Avian Biology **19**: 196-202.

Thomas, L., Laake, J.L., Strindberg, S., Marques, F.F.C., Buckland, S.T., Borchers, D.L., Anderson, D.R., Burnham, K.P., Hedley, S.L., Pollard, J.H., Bishop, J.R.B. and Marques, T.A. 2005. Distance 5.0. Release 5. Research Unit for Wildlife Population Assessment, University of St. Andrews, UK. http://www.ruwpa.st-and.ac.uk/distance/

Towne, E.G., D.C. Hartnett, and R.C. Cochran. 2005. Vegetation trends in tallgrass prairie from bison and cattle grazing. Ecological Applications **15**: 1550-1559.

Van Dyke, F., S.E. Van Kley, C.E. Page, J.G. Van Beek. 2004. Restoration efforts for plant and bird communities in tallgrass prairies using prescribed burning and mowing. Restoration Ecology **12**: 575-585.

Vickery, P.D., M.L. Hunter, Jr., and J.V. Wells. 1992. Is density an indicator of breeding success? Auk **109**: 706-710.

Vickery, P.D., P.L. Tubaro, J.M. Cardoso da Silva, B.G. Peterjohn, J.R. Herkert, and R.B. Cavalcanti. 1999*a*. Conservation of grassland birds in the western hemisphere. Studies in Avian Biology **19**: 2-26.

Vickery, P.D., J.R. Herckert, F. L. Knopf, J. Ruth, and C.E. Keller. 1999*b*. Grassland birds: an overview of threats and recommended management strategies IN Strategies for bird conservation: the partners in flight planning process. RMRS-P-16: U.S. Department of Agriculture, Forest Service, p. 74-77.

Walk, J.W., and R.E. Warner. 2000. Grassland management for the conservation of songbirds in the Midwestern USA. Biological Conservation **94**: 165-172.

Walsh, Roberta A. 1995. *Schedonorus arundinaceus*. In: Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available: http://www.fs.fed.us/database/feis/

Washburn, B.E, T.G. Barnes, C.C. Rhoades, and R. Remington. 2002. Using imazapic and prescribed fire to enhance native warm-season grasslands in Kentucky, USA. Natural Areas Journal **22**: 20-27.

Wilson, S.D., and M. Pärtel. 2003. Extirpation or coexistence? Management of a persistent introduced grass in a prairie restoration. Restoration Ecology **11**: 410-416.

Winter, M., D.H. Johnson, J.A. Shaffer, T.M. Donovan, and W.D. Svedarsky. 2006. Patch size and landscape effects on density and nesting success of grassland birds. Journal of Wildlife Management **70**: 158-172.

Table 2.1 Pasture, size, stocking rates, grazing treatment, and year treatment was initiated in experimental pastures on the Kansas Army Ammunition Plant, Kansas, U.S.A.

Pasture	***Area (ha)***	***Stocking rate*** [a]	***Treatment***	***Year Initiated***
9	64.75	4.05	Winter	2003
18	67.58	2.02	Winter	2000
22	134.36	1.21	Winter	2000
26	121.81	2.43	Winter	2003
30	71.63	3.24	Year-round	pre-1995[b]
1700	42.9	1.21	Winter	2002
1800	63.94	2.43	Year-round	pre-1995[b]
1900	190.61	0.81	Winter	2002
NP1.5	6.88	-	Ungrazed	-
NP2.5	10.93	-	Ungrazed	-
Average:	77.54	2.18		
SE:	17.99	0.39		

[a] Stocking rates are presented in hectares per animal unit. One animal unit is defined as a mature cow and her calf.

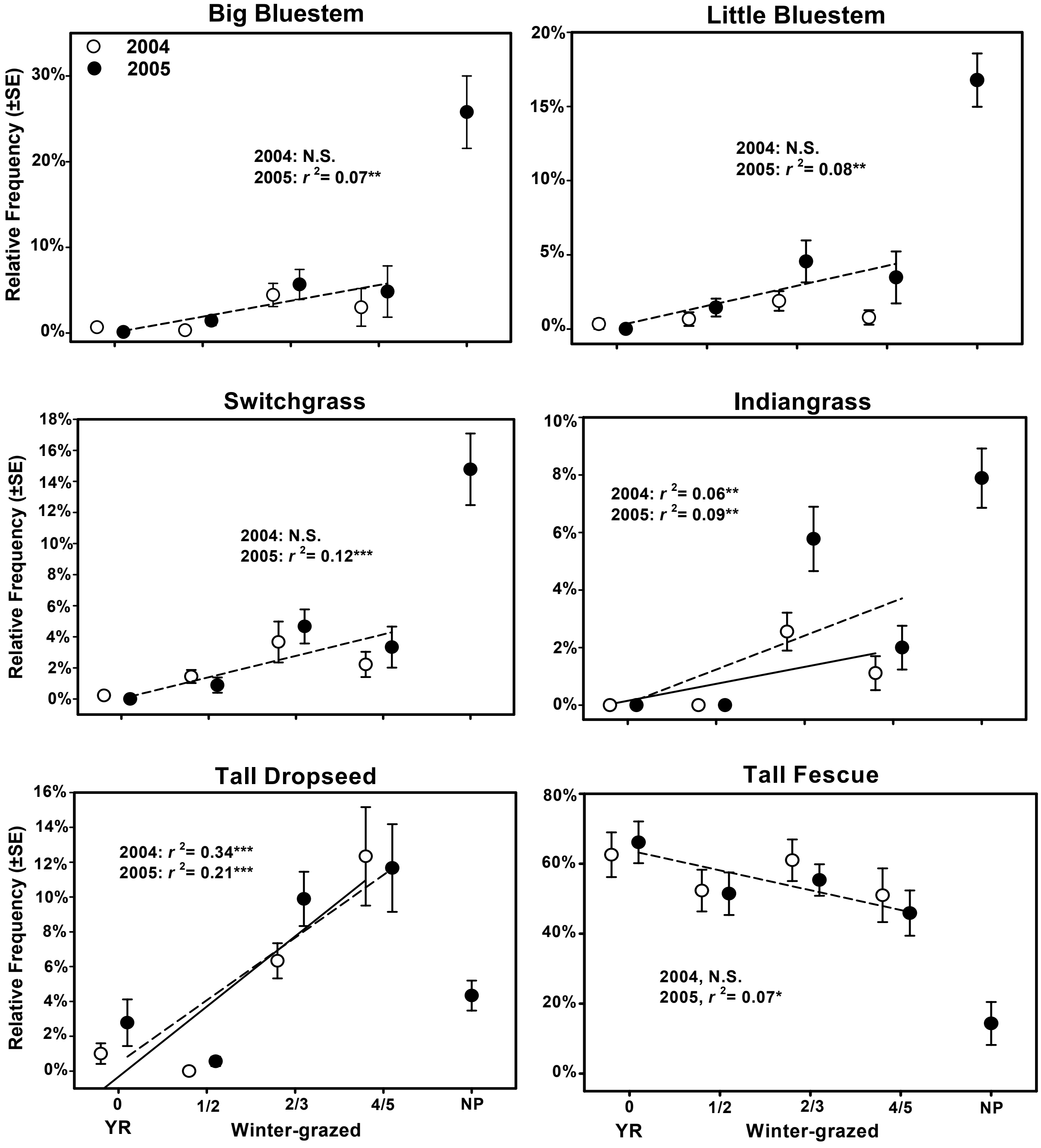
[b] Exact dates of initiation for year-round grazed pastures unknown but have been grazed year-round for at least ten years.

Table 2.2 Mean (±SE) plant species richness per transect (n = 9 per replicate) for year-round and winter-grazed fescue-dominated pastures on the Kansas Army Ammunition Plant, Kansas, U.S.A.

	Plant species richness per transect			
Treatment	2004	Pairwise[a] comparisons	2005	Pairwise comparisons
Year-round				
Rep 1	9.22 (±1.15)		11.00 (±0.77)	
Rep 2	18.11 (±1.87)		18.44 (±1.13)	
Mean (±SE):	13.67 (±1.52)	A	14.72 (±1.12)	A
Winter1/2				
Rep 1	19.67 (±1.09)		22.00 (±1.33)	
Rep 2	15.00 (±1.71)		17.56 (±0.99)	
Mean (±SE):	17.33 (±1.14)	B	19.78 (±0.96)	BC
Winter2/3				
Rep 1	20.89 (±1.82)		22.00 (±1.25)	
Rep 2	19.33 (±1.64)		23.11 (±1.84)	
Mean (±SE):	20.11 (±1.21)	BC	25.56 (±1.09)	B
Winter4/5				
Rep 1	17.44 (±1.73)		17.78 (±1.51)	
Rep 2	15.44 (±1.59)		21.22 (±1.30)	
Mean (±SE):	16.44 (±1.17)	AB	19.5 (±1.05)	C
Native prairie				
Rep 1	-		27.11 (±1.01)	
Rep 2	-		25.67 (±1.56)	
Mean (±SE):	-		26.39 (±0.92)	D

[a] Different letters indicate statistically different treatment means within a year.

Figure 2.1 Mean (± SE; *n* = 18) relative frequency per 50 m transect of native warm-season grasses and tall fescue in year-round grazed pastures (0), winter-grazed pastures (1/2 – 4/5), and ungrazed native prairie remnants (NP) on the Kansas Army Ammunition Plant, Kansas, U.S.A. in 2004-05. Statistical significance for treatment as a predictor variable is indicated with asterisks: = $P < 0.05$, *** = $P < 0.01$. The solid regression line is for 2004 values, and the dashed regression line is for 2005 values. Native prairie sites were not included in regressions and are presented for comparison only.**



Big Bluestem

Little Bluestem

Switchgrass

Indiangrass

Tall Dropseed

Tall Fescue

Figure 2.2 Biomass of tall fescue and native, warm-season grasses in year-round (YR), winter-grazed (1-5), and ungrazed native prairie remnants (NP) on the Kansas Army Ammunition Plant, Kansas, U.S.A. Statistical significance for treatment as a predictor variable is indicated with asterisks: = *P* < 0.05, *** = *P* < 0.01. The solid regression line is for 2004 values, and the dashed regression line is for 2005 values. Native prairie sites were not included in regressions and are presented for comparison only.**

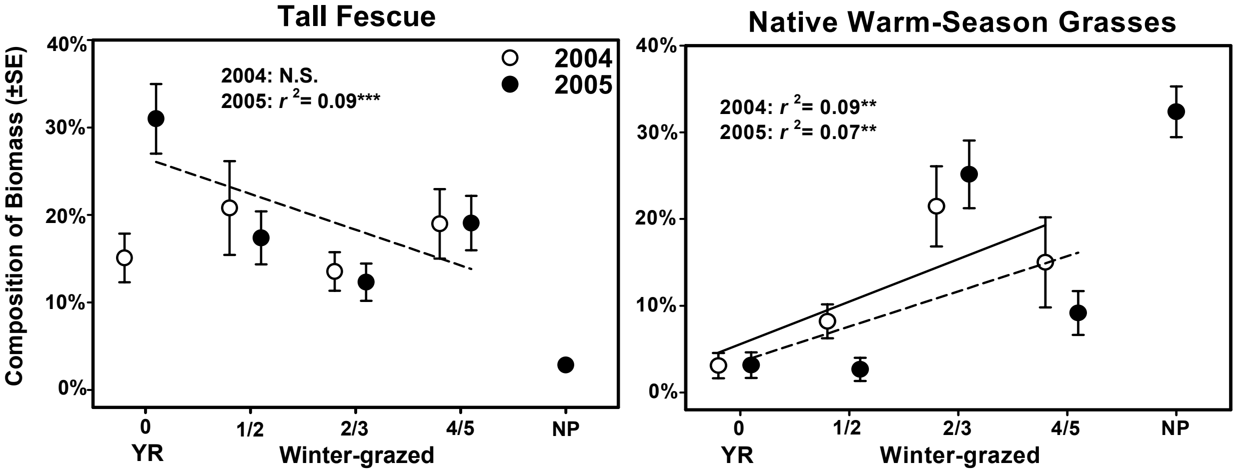


Table 2.3 Singing male birds per hectare (±SE) in winter-grazed and year-round grazed fescue-dominated pastures at the Kansas Army Ammunition Plant, Kansas, U.S.A, 2004-05.

	Grazing Treatment					
Species	*Year-round*	*Winter1/2*	*Winter2/3*	*Winter4/5*	*t**	*P* ≤
Dickcissel						
2004	0.22 (± 0.12)	1.51(± 0.20)	1.12 (± 0.39)	1.28 (± 0.23)	1.16	0.27
2005	0.34 (± 0.11)	1.25(± 0.12)	1.32 (± 0.29)	1.24 (± 0.18)	2.86	0.05
Eastern Meadowlark						
2004	0.86 (± 0.15)	0.51 (± 0.20)	0.75 (± 0.13)	0.29 (± 0.09)	-2.33	0.05
2005	0.31 (± 0.05)	0.15 (± 0.05)	0.31 (± 0.11)	0.11 (± 0.04)	-1.66	0.12
Grasshopper Sparrow						
2004	0.32 (± 0.20)	0.05 (± 0.05)	0.14 (± 0.06)	0.00	-1.77	0.10
2005	0.20(±0.14)	0.07 (± 0.04)	0.36 (± 0.17)	0.00	-0.82	0.43

* Statistical values are from a linear regression in which bird density was regressed on grazing treatment ar the t-statistic represents a test of the hypothesis that the slope = 0.

Table 2.4 Results of a principal components analysis of plant community composition on the Kansas Army Ammunition Plant, Kansas, U.S.A.

Eigenvectors	**PC1**	**PC2**
2004		
% fescue	**0.95**	-0.14
% native	-0.11	**0.48**
% forbs	-0.27	**-0.67**
% litter	0.05	0.20
% bare ground	-0.06	-0.27
visual obstruction	-0.05	**0.43**
% variation explained	**59.48**	**18.73**
2005		
% fescue	**0.99**	0.08
% native	-0.11	**0.91**
% forbs	-0.08	**-0.31**
% litter	0.02	0.03
% bare ground	0.02	-0.08
visual obstruction	-0.02	**0.26**
% variation explained	**45.11**	**36.31**

Table 2.5 Bird-vegetation relationships in year-round and winter grazed pastures on the Kansas Army Ammunition Plant, Kansas, U.S.A. See Table 4 for loadings from principal component analysis.

Species		Variable	β	$P \leq$
Dickcissel				
	2004	PC1	-0.008	0.08
		PC2	-0.009	0.34
		***grazed**	-1.281	**0.05**
	2005	PC1	-0.009	**0.05**
		PC2	0.000	0.91
		grazed	-0.935	**0.05**
Eastern Meadowlark				
	2004	PC1	0.000	0.91
		PC2	0.001	0.86
		grazed	0.363	0.15
	2005	PC1	0.002	0.14
		PC2	0.000	0.69
		grazed	0.149	0.15
Grasshopper Sparrow				
	2004	PC1	0.000	0.97
		PC2	-0.008	**0.05**
		grazed	0.108	0.41
	2005	PC1	0.003	0.31
		PC2	0.001	0.80
		grazed	0.889	0.63

* The variable grazed indicates the presence of cattle during the bird breeding season.

CHAPTER 3 - Avian community response to riparian restoration: a novel use of robust design mark-recapture models

Tracey N. Johnson, Roger D. Applegate, David E. Hoover, Phil S. Gipson, and Brett K. Sandercock

Abstract

The estimation of community attributes such as species richness and community turnover are critical when evaluating whether the goals of ecological restoration efforts have been met. Many estimates of species richness are underestimated due to variation in species detectability, but can be adjusted using mark-recapture approaches. We aimed to quantify the effects of livestock exclusion on riparian bird communities in southeast Kansas using robust design models for mixtures to account for heterogeneity in species detection. We estimated avian species richness within restored (ungrazed) and reference (grazed) communities, and used unbiased estimates of species richness to quantify community vital rates. We also examined qualitative changes in nesting guilds of breeding birds. We observed significant discrepancies between unadjusted species counts and richness estimates from robust design models. Species richness and community vital rates were similar among restored and reference communities after livestock exclusion; but we observed increased species richness in both communities over time. We detected changes within nesting guilds over the course of the study; however these changes likely had regional influences. The analysis presented here allowed us to compare standardized estimates between habitats, observers, and time periods by accounting for differences in detection rates. We suggest that use of robust design models to estimate community-level

parameters will facilitate more accurate assessments of the rate and trajectory of change following restoration efforts, and increased reliability in evaluating community responses to restoration efforts.

Introduction

Community attributes such as species richness, rates of local extinction and colonization, and community turnover are often used as metrics to evaluate the condition of natural systems (Nichols et al. 1998, McCoy and Mushinsky 2002). Estimation of community dynamics over time can be useful in identifying drivers of biodiversity loss, spatiotemporal changes in species distributions, or the importance of regional versus local processes in maintaining species richness (Griffiths 1997, Adler and Lauenroth 2003, Hansen et al. 2005). In conservation, knowledge of dynamic community processes is important in evaluating the potentially negative influences of land use change or habitat fragmentation on biodiversity, as well as the benefits of habitat remediation and ecological restoration (Balmford and Bond 2005). Conservation efforts for rare or declining species often rely on habitat restoration as a means of reversing or mitigating negative population trends (Schrott et al. 2005). For example, a decrease in the extinction probability for animal populations in recently restored habitat could serve as an indication of habitat improvement, and thus, a successful restoration. Similarly, increased persistence probabilities could indicate increased metapopulation connectivity for individuals in restored habitats. As restoration of populations and their habitats becomes increasingly central to conservation initiatives, the ability to accurately quantify the success of these efforts becomes essential (McCoy and Mushinsky 2002).

Despite the importance of estimating community-level parameters, methodological problems can result in misleading conclusions regarding community dynamics over time.

Observed species counts are often used to describe species richness, but can result in significant variation in richness estimates due to differences in the probability of detection among species (Nichols et al. 1998). Sampling situations in which species detection probabilities are less than one can bias the estimation of extinction or colonization probabilities, and consequently underestimate rates of community turnover (Nichols et al. 1998). Two methods that account for species detection probabilities when estimating species richness include species accumulation curves and mark-recapture models (Colwell and Coddington 1994, Boulinier et al. 1998).

Species detection probabilities are a particularly important characteristic to consider when evaluating a newly-restored community. In many types of communities, either naturally-occurring or anthropogenically created, many species are rare or occur at low abundance (Rabinowitz et al. 1986, Howe 1999, Maina and Howe 2000). In recently restored habitats, colonization will increase species richness, but colonizing species may be represented by only a few individuals. Individuals of relatively rare species will have detection probabilities lower than more abundant species, but recording the occurrence of colonizing species is important in characterizing changes in the community and evaluating the efficacy of restoration or remediation efforts. Consequently, the robust estimation of community dynamics that accounts for differences in detectability among species are crucial when evaluating community responses to changes in land use, management, or habitat improvements.

Assessment of animal community responses to riparian habitat restoration is a contemporary situation in which sampling issues can have strong effects on the observed results. Riparian habitats typically support a higher number of animal species than surrounding communities, especially those adjacent to grassland or agricultural lands (Kauffman and Krueger 1984). Riparian zones are particularly important as breeding and wintering sites for bird communities, and as stopover sites used during migration, especially in arid or semi-arid regions (Knopf and Samson 1994, Skagen et al. 1998, McComb et al. 2005). However, exposure of riparian habitats to livestock grazing can result in species-poor communities because cattle can negatively impact plant community structure, water quality, and other site characteristics (Kauffman and Krueger 1984). In the Great Plains region of the U.S., exclusion of livestock from riparian habitats is one means of habitat restoration. Typically, ungrazed riparian zones have a much higher volume of vegetation than the surrounding landscape, and can often have dense, scrubby undergrowth. The characteristics of undisturbed riparian habitats (high visual obstruction and number of bird species relative to the surrounding landscape) likely influence the probability of detecting all bird species present. When restored sites are compared to reference sites to assess whether restoration end-goals have been achieved, differences in habitat structure can potentially result in biased estimators of community recovery.

In this study, we evaluated bird community dynamics in response to livestock exclusion with a mark-recapture approach. Specifically, we used robust design models for mixtures to evaluate changes in species richness within grazed and restored riparian sites over time, while accounting for variation in detection probabilities among species and among habitats. Robust design models combine open and closed population estimators, and are generally applied to animal populations (Kendall et al. 1997, Sandercock in press). Here, we used them to estimate community-level parameters, as they have been previously used in three large-scale bird monitoring programs (the North American Breeding Bird Survey, the Swiss avian monitoring program, and the French Breeding Bird Survey; Boulinier et al. 1998, Kéry and Schmid 2004, Jiguet et al. 2005). The value of robust design models is that they offer unbiased estimates of species richness which can be used to calculate derived estimates of extinction probabilities, the number of colonizing species, turnover, and the rate of change in species richness (Nichols et al. 1998). Finally, to complement the investigation of quantitative changes in restored riparian communities, we assessed qualitative responses of avian nesting guilds to determine if changes in species composition occurred in response to livestock exclusion.

Materials and methods

Study area and field methods

We conducted this study in 1996-97 and 2004-05 at the Kansas Army Ammunition Plant (KSAAP) in Labette County, southeast Kansas (37°18'N, 95°10'W). This 5,555 ha military installation was historically tallgrass prairie dominated by big bluestem (*Andropogon gerardii)*, little bluestem (*Schizachyrium scoparium)*, switchgrass (*Panicum virgatum*), and indiangrass (*Sorghastrum nutans*) in upland sites, and riparian woodland along creeks and river valleys (Eifler et al. 1995). A large portion of KSAAP is leased to the public in the form of livestock grazing allotments. Grazing allotments at KSAAP riparian sites were grazed by cattle year-round with average stocking rates of 1.21 ha/animal unit prior to 1995 (C. Deurmyer, personal communication). In 1995, the average stocking rate in the riparian sites used in our study was reduced to 2.16 ha/animal unit (± 0.27 ha/animal unit, SE, $n = 6$).

No attempt was made to manage upland habitats separately from riparian habitats on the KSAAP prior to 1995, resulting in the continued presence and grazing by cattle in riparian sites. In March 1996, fences were erected around three riparian sites to permanently exclude cattle (hereafter, fenced sites), and three grazed riparian sites were paired with the three fenced sites to serve as reference sites (hereafter, grazed sites). Fenced sites ranged from 5.7 to 11.2 ha in size ($n = 3$), and grazed sites ranged from 6.5 to 11.8 ha in size ($n = 3$). Each grazed site was located adjacent to and downstream from its paired fenced site. Fenced and grazed sites had similar habitat width, stream length, and percent slope at the time cattle were permanently excluded (Hoover 1997). Grazing pressure in the 10-year period 1996-2005 varied slightly among replicates of reference areas. Grazed replicates 1 through 3 were grazed ten, eight, and nine years, respectively.

In 1996 and 1997, a single observer (D. Hoover) conducted ten-minute biweekly fixed-radius point counts in fenced and grazed sites. Counts were conducted from mid-May to mid-July to survey the breeding bird communities, visiting each site four times throughout the season. Survey points were located within 15 meters of the stream bank, and the first and last points between fenced and grazed sites were at least 300 meters apart. Each study site included three to five point count stations depending on the size of the site. All birds seen or heard within 50 meters of each survey point were recorded. Species encountered while the observer moved between points were recorded only if they were members of a species not previously encountered. All surveys were conducted between dawn and 10:00 C.S.T. throughout the course of the study, and surveys were never conducted in rain or winds in excess of 16 kph. In 2004 and 2005, a second observer (T. Johnson) followed the same protocols and conducted point counts two and three times, respectively, at the same study sites. In the first two years of the study, survey points were located at 150 meter intervals along the stream. In the last two years, we moved survey points to 200 meter intervals along the stream; however, six of the same survey points were used during both sampling periods.

A sampling design that included randomization of observers among time periods over the course of the study would have minimized potential observer bias. Such a design was not possible for our study because the original observer was not available to conduct surveys. However, a previous study with a similar sampling design found that detectability is more variable among years with the same observer than among observers with disparate survey experience (Kéry and Schmid 2004). If observer effects are important, we expected that changes in detection probability would coincide with a change in observer between 1996-97 and 2004-05.

We were interested in quantifying community dynamics of forest birds using riparian study sites for breeding. We included species that met the following criteria: birds listed as breeders within a ten-county region of southeast Kansas by the Kansas Breeding Bird Atlas (Busby and Zimmerman 2001), and species recorded on Breeding Bird Survey routes within the same region (2001-2005; Sauer et al. 2005). We discarded records of migratory species not known to breed in southeast Kansas, such as Mourning Warbler (*Oporornis philadelphia*), and Song Sparrow (*Melospiza melodia*); species only observed above the riparian canopy, such as Turkey Vulture (*Cathartes aura*), and Chimney Swift (*Chaetura pelagica*); and transient waterfowl, such as Canada Goose (*Branta canadensis*; see Table 3.6 for a list of all species included in the analysis).

Statistical analyses

Robust design models are based on a sampling design that includes primary sampling occasions that are subdivided into secondary sampling occasions (Kendall et al. 1997). Here, the avian community was assumed to be open to changes in composition among primary sampling occasions in different years, but closed to changes among secondary sampling occasions within a single breeding season. In this study, annual breeding seasons were the primary sampling occasions, and repeated visits to each site within a breeding season represent secondary sampling occasions. Our study included four primary sampling occasions, and two to four secondary sampling occasions per primary occasion (1996:4, 1997:4, 2004:2, and 2005:3).

Robust design mark-recapture models are typically applied to encounter histories of individuals to examine population dynamics. In this study, we used robust design models to examine community dynamics. Thus, encounter histories represent the detection or non-detection of avian species during visits to each study plot (Brose et al. 2003, Kéry and Schmid 2004). We created encounter histories for all landbird species over the entire sampling period of 1996 to 2005, and separated them by study site (Appendix Table A.1). Data were pooled from all point count stations within a site. Each encounter history contained information for thirteen visits total to each study site where detection and non-detection of a species were recorded as 1 and 0, respectively.

We initially used full heterogeneity models to analyze our data because heterogeneity in the detectability of species is a common feature of data from breeding bird surveys, and community studies of other terrestrial vertebrates (Wilson and Bart 1985, McShea and Rappole 1997, Boulinier et al. 1998, Herzog et al. 2002, Selmi and Boulinier 2003, Bailey et al. 2004). Full heterogeneity models estimate seven types of parameters in community studies. Our main interest was in the estimation of the two parameters: species richness (N) and species persistence (ϕ). The models also include five nuisance parameters: probability of emigration (γ''), probability of immigration (γ'), the proportion (mixture) of the community with a particular detection probability (π), initial detection probabilities for each mixture within the community (subset of the community with high and low detection rates; p_{π}, $p_{1-\pi}$, respectively), and subsequent detection probability (c). The global model for full heterogeneity model types (M_{tbh}, see Otis et al. 1978 for model notation) allows variation in each parameter estimate over time (t), with behavioral response to detection (b; this would suggest an increase or decrease in detection probability after the initial detection of a species), and heterogeneity of detection among individual species (h). However, we found that all the parameters for these models consistently collapsed into estimates that were constant for time and behavioral response to detection (results not shown). Furthermore, the associated standard errors of several parameters suggested that model M_{tbh} was over-parameterized. Thus, we opted to start with a global model that included only time and heterogeneity in detectability (M_{th}), which eliminated the probability of subsequent detection (*c*) from the model and reduced the number of parameter types to six.

We used closed-capture robust design models for mixtures in Program MARK (White & Burnham 1999) to compute parameter estimates separately for all six study sites. As implemented in Program MARK, closed-capture models are based on mixtures where different subsets of the community may vary in detectability (Norris and Pollock 1996, Pledger 2000). When applied to encounter histories for all species, these models estimate the six community-level parameters previously listed for the full heterogeneity models (Table 3.1). We used parameter estimates of $\hat{N}$ and $\hat{\phi}$ to calculate four community vital rates found in Nichols et al. (1998):

1) The probability of local extinction (*E*):

$$\hat{E} = 1 - \hat{\phi}_{ij}$$

where time $i <$ time j.

2) The probability of turnover among local species (*T*), which reflects the difference in communities from time *i* to time *j* and an estimate of one indicates all species present in time *j* are new species in the community:

$$\hat{T} = 1 - \hat{\phi}_{ji}$$

where $\hat{\phi}_{ji}$ is estimated by reverse-time modeling of the encounter histories.

3) We also estimated the number of local colonizers (B_{ij}), or the number of species present at time *j* but not at time *i*:

$$\hat{B}_{ij} = (\hat{N}_j \text{ - } \hat{\phi}_{ij} \, \hat{N}_i)$$

where $\hat{N}_i$ and $\hat{N}_j$ are the estimated number of species at times *i* and *j*, and $\hat{\phi}_{ij}$ is the estimated probability of a species persisting from time *i* to time *j*.

4) Last, we estimated the finite rate of community change ($\hat{\lambda}$) which in this case represents the rate of change in species richness between time *i* and time *j*:

$$\hat{\lambda} = \frac{\hat{N}_j}{\hat{N}_i}$$

where $\hat{N}_i$ and $\hat{N}_j$ are estimates of species richness at times *i* and *j*, respectively. Each of the four community vital rates was quantified for the 8-year interval between the 1996-97 and the 2004-05 sampling occasions.

To estimate the initial parameters, we considered a total of twelve candidate models. Candidate models included M_h (heterogeneity in detection) as suggested by Otis et al. (1978) and Boulinier et al. (1998) and M_o (all parameters constant), as well as ten different permutations of this model (e.g., one parameter was modeled as a function of time, while another parameter in the same model was held constant). We allowed each parameter to vary over time, or remain constant within a subset of the community (proportion of the community with a high or low detection probability), across subsets, within a sampling period, or across all sampling periods for the most simplified model M_o (Table 3.2). When the model did not fit the yearly encounter data, we examined a reduced model which allowed *N* to vary between the periods 1996-97 (first period) and 2004-05 (second period; see Kendall et al. 1997). This two-period model allowed us to compare estimated and observed species richness from the same years within each site to see if imperfect detection of species would have affected conclusions regarding the restoration of riparian bird communities in our study area.

Goodness-of-fit tests are not available for robust design models in Program MARK. To examine the sensitivity of our model conclusions to possible overdispersion, we manually increased the value of the variance inflation factor from $\hat{c}$ = 1 (no overdispersion) to 10 (high overdispersion) by increments of 0.5, and re-evaluated model rankings. Model fit was evaluated based on Akaike's Information Criterion values corrected for small sample sizes (AIC_c). Model selection was based on the differences in AIC_c values (ΔAIC_c) relative to the minimum AIC_c model where ΔAIC_c values ≤ 2 suggested models were equally parsimonious. The relative strength of support for each model was determined by the ratios of Akaike weights (w_i / w_j).

We tested for statistical significance between parameter estimates for fenced and grazed sites using Program Contrast (Ver. 2, Hines and Sauer 1989). All other statistical tests were performed using SAS System Version 8.01, and P-values were considered significant at $\alpha \leq 0.05$. To make statistical comparisons among derived parameters for communities at fenced and grazed sites, we used parametric bootstrapping to obtain variance estimators and confidence intervals in Program Matlab version 6.1 (Mathworks 2001). $\hat{N}$ was modeled as random draws from a normal distribution, whereas $\hat{\phi}$ was modeled as random draws from a beta distribution to bound draws between 0 and 1. Derived parameter standard errors were taken directly from the bootstrap distributions and associated values generated for the actual parameter estimates. All estimates of variance for derived parameters were based on 100,000 bootstrap replicates. To compensate for slight discrepancies between derived parameter values generated by the community vital rate equations and means of the bootstrap distributions, we calculated bias-corrected 95% confidence intervals as:

$$\text{Bias-corrected percentiles} = \Phi\,[2\,\Phi^{-1}(F) \pm 1.96]$$

where Φ is the normal cumulative distribution, Φ^{-1} is the inverse normal cumulative distribution, F is the fraction of bootstrap values smaller than the value from the community vital rates, and 1.96 is the critical value for the 95% confidence interval.

Guild analysis

We assigned bird species to one of four guilds based on the habitats primarily used for nesting. We defined guilds after Martin (1995) as: cavity nesters, ground nesters, shrub/low foliage nesters, and sub-canopy/canopy nesters. The habitats used by each species for nesting were determined following Martin (1995) and Ehrlich et al. (1988; see Table 3.6 for guild assignments). All species encountered in our study sites were included in the guild analysis except for the Brown-headed Cowbird (*Molothrus ater*) and Eastern Phoebe (*Sayornis phoebe*). Neither of these species could be assigned to a nesting guild and were excluded from the analysis. However, we did evaluate the changes in Brown-headed Cowbird abundance over time in fenced and grazed sites due to their close association with cattle (Goguen and Mathews 2001).

To estimate relative abundance of riparian birds, we used the mean number of detections of each species by pooling sampling points within a site and using the maximum number of individuals of that species detected during a given visit. We then averaged across plots within a treatment to get a mean number of detections within a treatment for each year. To evaluate differences in mean abundance for all guilds between treatments and to test for year by treatment interactions, we compared the summed responses of individuals within each guild using a mixed effects repeated measures analysis of variance, with year as the repeated variable. Covariance structure for measurements among time periods was initially unknown. Candidate model covariance structure included: compound symmetry, unstructured, autoregressive, heterogeneous autoregressive, and autoregressive moving average. We determined the most appropriate model covariance structure and improved model fit by comparing models using Akaike's Information Criterion adjusted for small sample sizes (AIC*c*). An alpha level of 0.05 was used to determine statistical significance.

Results

Observed species richness

We observed a total of 51 species of birds during the four years of the study: 43 species in fenced riparian sites and 44 species in grazed riparian sites. Observed species richness ranged from 19.0 ± 0.58 SE to 20.3 ± 2.34 species per year in fenced sites ($n = 3$) and from 17.6 ± 1.33 to 20.6 ± 1.77 species per year in grazed sites ($n = 3$), and was not significantly different between fenced and grazed sites in any year of the study (Wilcoxon two-sample test; $W \geq 10.0$, $P \geq 0.20$). Observed species richness was greater in 2005 than 1996 in two out of three fenced sites (11% and 16% increase), and in one out of three grazed sites (12% increase). Averages of observed species richness for consecutive years, 1996-97 and 2004-05, were not significantly different in either fenced ($W = 6.5$, $P = 0.10$) or grazed sites ($W = 9.0$, $P = 0.40$; Fig. 3.1a).

Estimated species richness

The top two models for each study site had the same model structure for all parameters (Table 3.2). Five of the six riparian study sites had the same best-fit model, which held parameters ϕ and π constant, allowed random exchange ($\gamma'' = \gamma'$), held p constant within a detection group, and held species richness (N) constant over the period 1996 to 2005 (model N_c; Table 3.2). Fenced replicate 2 had a different best-fit model than all other sites, but differed in model structure for only one parameter (N), which changed between the periods 1996-97 and 2004-05 (model N_{period}; Table 3.2). For all riparian sites, 2-3 models had ΔAIC_c values ≤ 2, and were an equally parsimonious fit to the data (Table 3.2). Subordinate model rank varied with site and treatment. The minimum AIC_c model remained top-ranked even when $\hat{c}$ was adjusted up to a value of ten. The top model for fenced replicate 2 became the second-ranked model when the value of $\hat{c}$ increased beyond one, and was replaced by the model that was top-ranked in all other study sites (see Table 3.2), which was robust to subsequent changes in $\hat{c}$ up to a value of ten.

The N_{period} model garnered a similar amount of support as the N_c model in all sites (Table 3.3), and was just as likely to be the top model based on strength of support for five of the six study sites. Parameter estimates are reported from the N_{period} model for comparison to observed species richness data for specific time periods. Parameter estimates and associated standard error and confidence intervals from the N_{period} model for each site are given in Table 3.4. Apparent species persistence was high across all sites ($\hat{\phi} > 0.8$), but tended to be lower in grazed sites. Estimates of temporary emigration were low across all sites ($\hat{\gamma}'' \leq 0.03$). The proportion of each community with a high detection rate was well below 1.0 at all sites, ranging from $\hat{\pi} = 0.53$ to 0.79. The detection probability for the high detection subset of the community ranged from $\hat{p}_{\pi} =$ 0.89 to 1.00, while detection probabilities for the low detection group were $\hat{p}_{1-\pi} = 0.18$ to 0.48. We detected no significant differences in estimates among fenced and grazed communities for any parameters (Table 3.4).

Estimates of N compared among fenced and grazed sites within the same time period were not different; however estimates of N compared between the two time periods within a site were different for fenced and grazed communities (Fig. 1.1b). Based on estimates from the robust-design models, fenced sites experienced a 14% - 42% increase in species richness, and grazed sites experienced a 16% - 22% increase in species richness over the course of the study.

Derived parameters

The probability of local extinction was low and tended to be higher in grazed ($\hat{E}$ = 0.07 – 0.14) than fenced sites ($\hat{E}$ = 0.06 – 0.08) from 1996 to 2005, but site differences were not significant. Turnover ($\hat{T}$ = 0.07 to 0.19) rates were not different among fenced and grazed communities. The number of colonizers ($\hat{B}_{ij}$) was highest in one of the three fenced riparian sites ($\hat{B}_{ij}$ = 11.9), but both fenced and grazed sites acquired five to twelve colonizers in the period from 1996 to 2005 (Table 3.5). The rate of change in species richness ($\hat{\lambda}$) was estimated to be greater than one for all riparian study sites, but the 95% confidence intervals included one for 5 of 6 sites, indicating no significant difference from a stationary community (Table 3.5). We detected no statistically significant differences in the rate of change in species richness between fenced and grazed communities.

Observed vs. estimated species richness

Species richness showed little change in both fenced and grazed sites from 1996-97 to 2004-05 based on observed species counts alone. Estimates of species richness generated from the robust-design models indicated a larger, and statistically significant, increase in species richness in fenced and grazed sites over the same period than estimates based on uncorrected counts (Fig. 1.1). In the 1996-97 period, observed and estimated species richness differed for two of three fenced sites, and for one of three grazed sites (Fig. 1.1). In the 1996-97 period, observed species counts resulted in species richness that was 0.0% to 34.5% lower than estimates of species richness, but observed and estimated values were not correlated (r = -0.11, P = 0.84, n = 6). In the 2004-05 period, observed and estimated species richness differed for all six study sites, and observed species counts were 9.7% to 61.2% lower than estimates of species richness. Again, observed and estimated values of species richness values were not correlated (r = 0.75, P = 0.08, *n* = 6). Overall, observed versus estimated species richness values for both time periods, however, were positively correlated (*r* = 0.62, *P* = 0.03, *n* = 12).

Guild responses

The riparian bird communities at our study site consisted of species represented by four different nesting guilds (Table 3.6). The shrub/low-foliage and sub-canopy/canopy guilds were the most diverse components of the riparian community, each consisting of 15 species. Based on detections alone, the cavity guild gained three new species between 1996-97 and 2004-05: Northern Flicker (*Colaptes auratus*), Pileated Woodpecker (*Dryocopus pileatus*), and Red-headed Woodpecker (*Melanerpes erythrocephalus*). The shrub/low foliage guild also gained three new species: Acadian Flycatcher (*Empidonax virescens*), Chipping Sparrow (*Spizella passerina*), and Dickcissel (*Spiza americana*). The sub-canopy/canopy guild gained four species: Fish Crow (*Corvus ossifragus*), Red-shouldered Hawk (*Buteo lineatus*), Ruby-throated Hummingbird (*Archilochus colubris*), Summer Tanager (*Piranga rubra*), but lost two: Baltimore Oriole (*Icterus galbula*) and Eastern Kingbird (*Tyrannus tyrannus*) in the 2004-05 study periods. The ground-nesting guild was least diverse, but increased in species richness from two to four species in 2004-05. The two colonizing species were Black-and-white Warbler (*Mniotilta varia*) and Kentucky Warbler (*Oporornis formosus*).

Six species were encountered only in fenced sites: of these six, four were in the cavity guild. Four species were encountered only in grazed sites: three of these species were in the sub-canopy/canopy guild. Most species occupied both fenced and grazed sites; however, there was variation in the number of individuals from each guild. A subset of species within each guild showed positive responses to livestock exclusion and were consistently encountered more frequently in fenced sites. These were as follows: cavity guild, four species (representing 27%

of the guild), ground guild, one species (25%), shrub/low-foliage guild, two species (13%), and sub-canopy/canopy, one species (0.7%). Alternatively, some species were encountered more frequently in grazed sites: ground guild, one species (25%), shrub/low foliage guild, one species (0.7%), and sub-canopy/canopy, five species (33%).

We detected no significant year by treatment interactions or between-treatment differences in abundance within the four guilds. Mean abundance of the cavity guild was higher in 2005 than all other years of the study in fenced and grazed sites ($\bar{X}$ = 16.0-23.3, $F_{3,12} = 3.39$, $P = 0.05$). Mean abundance of the shrub/low-foliage guild was lower in both fenced and grazed sites in 2005 than in 1996 ($\bar{X}$ = 13.0-21.3 shrub-nesters per site, $F_{3,4} = 113.07$, $P < 0.001$), and this difference was stronger in the grazed sites, although not significantly. Mean abundance in the ground nesting guild did not differ among years ($\bar{X}$ = 0.0-2.3 ground-nesters per site, $F_{3,4} = 6.16$, $P = 0.06$). Mean abundance in the canopy guild differed among years ($\bar{X}$ = 4.3-9.0 canopy-nesters per site, $F_{3,12} = 8.77$, $P = 0.002$), which was driven by a decrease in abundance in both fenced and grazed study sites between 1997 and 2004. We detected a year by treatment interaction effect for Brown-headed Cowbird abundance ($\bar{X}$ = 0.3-3.3 individuals per site, $F_{3,12} = 4.99$, $P = 0.02$). Brown-headed Cowbirds were more abundant at grazed sites than fenced sites until 2005, when they were more abundant at fenced sites (Table 3.6).

Discussion

Based on observed counts of species encountered during surveys, we found no differences in avian species richness between fenced and grazed sites, and no changes after cattle exclusion. However, when we accounted for variation in detection probabilities with robust design models, we detected an increase in species richness in both fenced and grazed sites over an 8-year period. Estimated species richness did not differ between fenced and grazed riparian sites despite eight years of cattle exclusion. Estimates of community vital rates in all study sites suggest that bird community dynamics over the study period were similar in both fenced and grazed riparian habitats. We detected new species from all nesting guilds in the 2004-05 sampling period, and found that Brown-headed Cowbird abundance was higher in grazed sites in the early years of the study, and became higher in fenced sites in the latter years of the study.

Livestock grazing of riparian zones can have negative effects on riparian vegetation and the associated vertebrate communities, and exclusion of livestock is often employed as a means of habitat restoration. Several studies evaluating the effects of livestock exclusion detected increased species richness, diversity, and abundance of riparian vertebrate communities in fenced habitats compared to grazed habitats (Szarro and Rinne 1988, Dobkin et al. 1998, Sarr 2002, and Giuliano and Homyack 2004). We expected a higher number of breeding bird species to occur in fenced riparian sites because sites which have eliminated livestock grazing usually have higher vegetation volume than grazed sites (Kauffman and Krueger 1984, Mills et al. 1991, Fleischman et al. 2003, Nemethova and Tirinda 2005). We observed increased bird species richness in 2004-05 when detection probabilities were accounted for, but species richness in exclosure sites did not differ from grazed sites. In southeastern Kansas, riparian habitats in which cattle have been excluded may require additional time or manual re-vegetation to see an increase in species richness beyond that of actively grazed sites.

In addition to local influences, regional processes likely contributed to the observed changes in both fenced and grazed communities. Unexpectedly, we observed an increase in estimated species richness in grazed sites and similar rates of local species turnover in both fenced and grazed sites over the course of this 10-year study. Increases in avian species richness in grazed riparian sites could be influenced by an overall reduction in stocking densities that occurred in 1995 on our study site in both riparian zones and surrounding upland pastures. Riparian vegetation and associated riparian bird communities can be significantly affected by differences in livestock stocking rates (Jansen and Robertson 2001*a*,*b*). Additionally, the landscape matrix in which riparian zones are embedded can exert strong influences on processes observed within riparian areas, affecting both habitat quality and species persistence (Dunford and Freemark 2004). Our study site is located at the interface of tallgrass prairie and eastern deciduous forest, an ecotone that coincides with the edge of many bird species' breeding distributions (e.g., Summer Tanager, Kentucky Warbler – two species that were detected during the latter portion of our study). A species' distribution is usually characterized by relatively lower abundance near boundary limits, and thus higher extinction and colonization rates as this portion of the distribution is dependent upon immigration for population persistence (Doherty et al. 2003, Karanth et al. 2006). If regional influences were a factor at our study site, spatiotemporal changes in species distributions could potentially obscure any differences in fenced and grazed riparian bird community dynamics. Furthermore, the increased variability in community dynamics that is characteristic of ecoregional boundaries is expected to result in lower mean species richness per year, but not result in lower species richness over all years due to the accumulation of new species over time (Boulinier et al. 2001, Karanth et al. 2006). In support of this hypothesis, we did not detect significant differences in species richness or guild abundance between treatments, but did detect increased species richness in all guilds over the course of the study in both treatments.

To test the prediction that there are differences between animal communities found in grazed or ungrazed riparian habitat, researchers have historically assumed that detection probabilities are similar among species, habitats, or time periods being compared. Our results suggest that significant heterogeneity exists at our study site among species in their probabilities of detection, as indicated by the large differences in detection probability among the high- and low-detection subsets of each community. However, detection probabilities within each subset did not change over the course of our study. The use of robust-design models for mixtures allowed for a standardized comparison across habitats with different detection probabilities among species (especially within the low detection subset of the community; see Table 3.4), as well as the evaluation of potential effects resulting from a change in observers. Detection probabilities were similar between the two time periods for all study sites, and the change in species richness could be attributed to some differences in habitat between the two time periods, albeit at a larger scale than our study was conducted.

A species' detection probability is dependent upon its abundance, which ultimately influences its extinction probability. Species that occur in low abundances tend to have a lower probability of detection and a higher probability of extinction (Alpizar-Jara et al. 2004). These relationships clearly have implications for ecological restoration efforts because the success of a restoration is often measured against some other reference community in time or space. Comparing samples from different communities in which detection probabilities have not been considered could result in the two communities appearing more similar than they actually are, and differences between them more difficult to detect. Additionally, the estimation of community dynamics accounting for the imperfect detection of species that occur in low abundances is necessary to accurately quantify extinction probabilities, or conversely, persistence probabilities, as well as the rate of change in species richness after habitat restoration. Accurate estimates of these community-level parameters will be critical in evaluating the suitability of restored habitats.

Our estimates of species richness demonstrate that unadjusted species counts can underestimate the number of avian species breeding in a Great Plains riparian community. When species richness is used as a variable to describe the state of a system, biased or inconsistent species counts will not be adequate for management or conservation objectives. The naïve estimators typically used to assess ecological restoration likely result in incorrect estimates of occupancy and species richness, and overestimated confidence in the results of faunal surveys (Wintle et al. 2004). Unbiased estimates of species richness are important because they allow ecologists to more accurately evaluate the efficacy of the restoration, the pace and trajectory of community recovery through accurate estimates of community-level vital rates, and to identify if more investment is needed in order to achieve restoration project end-goals. Future use of mark-recapture models for community data will likely contribute to increased accuracy and reliability in evaluating community responses to restoration efforts.

References

Adler, P.B., and Lauenroth, W.K. 2003. The power of time: spatiotemporal scaling of species diversity. Ecology Letters **6**: 749-756.

Alpizar-Jara, R., Nichols, J.D., Hines, J.E., Sauer, J.R., Pollock, K.H., and Rosenberry, C.S. 2004. The relationship between species detection probability and local extinction probability. Oecologia **141**: 652-660.

Bailey, L.L., Simons, T.R., and Pollock, K.H. 2004. Spatial and temporal variation in detection probability of *Plethodon* salamanders using the robust capture-recapture design. Journal of Wildlife Management **68**: 14-24.

Balmford, A., and Bond, W. 2005. Trends in the state of nature and their implications for human well-being. Ecology Letters **8**: 1218-1234.

Brose, U., Martinez, N.D., and Williams, R.J. 2003. Estimating species richness: sensitivity to sample coverage and insensitivity to spatial patterns. Ecology **84**: 2364-2377.

Boulinier, T., Nichols, J.D., Sauer, J.R., Hines, J.E., and Pollock, K.H. 1998. Estimating species richness: the importance of heterogeneity in species detectability. Ecology **79**: 1018-1028.

Busby, W.H., and Zimmerman, J.H. 2001. Kansas Breeding Bird Atlas. University Press of Kansas, Lawrence, Kansas.

Colwell, R.K., and Coddington, J.A. 1994. Estimating terrestrial biodiversity through extrapolation. Philosophical Transactions of the Royal Society of London B **345**: 101-118.

Dobkin, D.S., Rich, A.C., and Pyle, W.H. 1998. Habitat and avifaunal recovery from livestock grazing in a riparian meadow system of the northwestern Great Basin. Conservation Biology **12**: 209-221.

Doherty, P.F., Boulinier, T., and Nichols, J.D. 2003. Local extinction and turnover rates at the edge and interior of species' ranges. Annales Zoologici Fennici **40**: 145-153.

Dunford, W., and Freemark, K. 2004. Matrix matters: effects of surrounding land uses on forest birds near Ottawa, Canada. Landscape Ecology **20**: 497-511.

Ehrlich, P.R., Dobkin, D.S., and Wheye, D. 1988. The birder's handbook: a field guide to the natural history of North American birds. Fireside Press, New York.

Eifler, D.A., Busby, W.H., and Freeman, C.C. 1995. A survey for protected and rare species and exemplary natural areas on the Kansas Army Ammunition Plant in Labette County, Kansas. Kansas Biological Survey Rep. No. 84. University of Kansas. 184 pp.

Fleischman, E., McDonal, N., MacNally, R., Murphy, D.D., Walters, J., and Floyd, T. 2003. Effects of floristics, physiognomy and non-native vegetation on riparian bird communities in a Mojave Desert watershed. Journal of Animal Ecology **72**: 484-490.

Giuliano, W.M., and Homyack, J.D. 2004. Short-term grazing exclusion effects on riparian small mammal communities. Journal of Range Management 57: 346-350.

Goguen, C.B., and Mathews N.E. 2001. Brown-headed Cowbird behavior and movements in relation to livestock grazing. Ecological Applications **11**: 1533-1544.

Griffiths, D. 1997. Local and regional species richness in North American lacustrine fish. Journal of Animal Ecology **66**: 49-56.

Hansen, A.J., Knight, R.L., Marzluff, J.M., Powell, S., Brown, K., Gude, P.H., and Jones, A. 2005. Effects of exurban development on biodiversity: Patterns, mechanisms, and research needs. Ecological Applications **15**: 1893-1905.

Herzog, S. E., Kessler, M., and Cahill, T. M. 2002. Estimating species richness of tropical bird communities from rapid assessment data. Auk **119**: 749-769.

Hines, J.E., and Sauer, J.R. 1989. Program CONTRAST – A General Program for the Analysis of Several Survival or Recovery Rate Estimates. U.S. Fish and Wildlife Service, Fish and Wildlife Technical Report 24, Washington DC.

Hoover, D. 1997. Vegetation and breeding bird assemblages in grazed and ungrazed riparian habitats in southeastern Kansas. M. S. Thesis, Kansas State University.

Howe, H.F. 1999. Dominance, diversity, and grazing in tallgrass restoration. Ecological Restoration **17**: 59-66.

Jansen, A., and Roberston, A.I. 2001*a*. Relationships between livestock management and the ecological condition of riparian habitats along an Australian floodplain river. Journal of Applied Ecology **38**: 63-75.

Jansen, A., and Robertson, A.I. 2001*b*. Riparian bird communities in relation to land management practices in floodplain woodlands of south-eastern Australia. Biological Conservation **100**: 173-185.

Jiguet, F., Renault, O., and Petiau, A. 2005. Estimating species richness with capture-recapture models: choice of model when sampling in heterogeneous conditions. Bird Study **52**: 180-187.

Karanth, K.K., Nichols, J.D., Sauer, J.R., and Hines, J.E. 2006. Comparative dynamics of avian communities across edges and interiors of North American ecoregions. Journal of Biogeography **33**: 674-682.

Kauffman, J.B., and Krueger, W.C. 1984. Livestock impacts on riparian ecosystems and streamside management implications…a review. Journal of Range Management **37**: 430-438.

Kendall, W.L., Nichols, J.D., and Hines, J.E. 1997. Estimating temporary emigration using capture-recapture data with Pollock's Robust Design. Ecology **78**: 563-578.

Kéry, M., and Schmid, H. 2004. Monitoring programs need to take into account imperfect species detectability. Basic and Applied Ecology **5**: 65-73.

Knopf, F.L., and Samson, F.B. 1994. Scale perspectives on avian diversity in western riparian ecosystems. Conservation Biology **8**: 669-676.

Maina, G.G. and Howe, H.F. 2000. Inherent rarity in community restoration. Conservation Biology **14**: 1335-1340.

Martin, T.E. 1995. Avian life history evolution in relation to nest sites, nest predation, and food. Ecological Monographs **65**: 101-127.

McComb, B.C., Bilsland, D., and Steiner, J.J. 2005. Associations of winter birds with riparian condition in the lower Calapooia watershed, Oregon. Northwest Science **79**: 164-171.

McCoy, E.D., and Mushinsky, H.R. 2002. Measuring the success of wildlife community restoration. Ecological Applications **12**: 1861-1871.

McShea, W.J., and Rappole, J.H. 1997. Variable song rates in three species of passerines and implications for estimating bird populations. Journal of Field Ornithology **68**: 367-375.

Mills, G.S., Dunning Jr., J.B., and Bates, J.M. 1991. The relationship between breeding bird density and vegetation volume. Wilson Bulletin **103**: 468-479.

Nemethova, D., and Tirinda, A. 2005. The influence of intersections and dead-ends of line-corridor networks on the breeding bird distribution. Folia Zoologica **54**: 123-134.

Nichols, J. D., Boulinier, T., Hines, J.E., Pollock, K. H., and Sauer, J. R. 1998. Estimating rates of local species extinction, colonization, and turnover in animal communities. Ecological Applications **8**: 1213-1225.

Norris, J.L. and Pollock, K.H. 1996. Nonparametric MLE under two closed capture recapture models with heterogeneity. Biometrics **52**: 639-649.

Otis, D.L., Burnham, K.P., White, G.C., and Anderson, D.R. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs **62**: 21-50.

Pledger, S. 2000. Unified maximum likelihood estimates for closed capture-recapture models using mixtures. Biometrics **56**: 434-442.

Rabinowitz, D., Cairns, S., and Dillon, T. 1986. Conservation biology: the science of scarcity and diversity. Sinauer Associates, Sunderland, Maryland.

Sandercock, B.K. 2006. Estimation of demographic parameters from live encounter data: a summary review. Journal of Wildlife Management, in press.

Sarr, D.A. 2002. Riparian livestock exclosure research in the western United States: A critique and some recommendations. Environmental Management **30**: 516-526.

Sauer, J. R., Hines, J. E., and Fallon, J. 2005. The North American Breeding Bird Survey, Results and Analyses 1966 – 2004. Version 2005.2. USGS Patuxent Wildlife Research Center, Laurel, MD.

Schrott, G.R., With, K.A., and King, A.W. 2005. Demographic limitations of habitat restoration to rescue declining populations. Conservation Biology **19**: 1181-1193.

Selmi, S., and Boulinier, T. 2003. Does time of season influence bird species number determined from point-count data? A capture-recapture approach. Journal of Field Ornithology **74**: 349-356.

Skagen, S.K., Melcher, C.P., Howe, W.H., and Knopf, F.L. 1998. Comparative use of riparian corridors and oases by migrating birds in southeast Arizona. Conservation Biology **12**: 896-909.

White, G.C., and Burnham, K.P. 1999. Program MARK: Survival estimation from populations of marked animals. Bird Study 46 Supplement, 120-138.

Wilson, D.M., and Bart, J. 1985. Reliability of singing bird surveys: Effects of song phenology during the breeding season. Condor **87**: 69-73.

Wintle, B.A., McCarthy, M.A., Parris, K.M., and Burgman, M.A. 2004. Precision and bias of methods for estimating point survey detection probabilities. Ecological Applications **14**: 703-712.

Table 3.1 Parameters estimated by robust design closed-capture mixture models when applied to community-level data.

Parameters	Symbol	Definition
Apparent persistence	ϕ	Probability that a species is present from time i to time j [a]
Heterogeneity	π	Proportion of community with a high detection rate
Temporary emigration	γ''	Probability that a species is present at time i but absent at time j
Immigration	γ'	Probability that a species is absent at time i and remains absent at time j
High detection	p_{π}	Probability of detection for high detection group
Low detection	$p_{1-\pi}$	Probability of detection for low detection group
Species richness	N	Estimated number of species present

[a] Time i to time j represents intervals among primary sampling periods

Table 3.2 Delta Akaike Information Criterion (ΔAICc) values adjusted for small sample sizes for all robust design models (see Table 3.1 for parameter definitions).

Model structure[a]							Replicate 1		Replicate 2		Replicate 3	
ϕ	π	γ''	γ'	p	N	K[b]	fenced	grazed	fenced	grazed	fenced	grazed
c	c	random		mix	c	6	[c]**0.0**	**0.0**	**0.5**	**0.0**	**0.0**	**0.0**
c	c	random		mix	period	7	**0.2**	**0.1**	**0.0**	**1.4**	**0.5**	**0.5**
c	t	t	t	mix	t	16	29.1	14.7	13.0	8.2	13.7	26.3
c	t	t	t	c	t	15	51.7	21.8	29.1	31.8	40.3	45.3
c	c	random		mix	t	9	4.3	3.9	4.0	2.5	3.5	4.4
c	c	c	c	mix	c	7	**1.6**	2.2	2.3	2.2	2.2	**1.9**
c	c	c	c	mix	t	10	5.9	6.2	5.9	4.8	5.7	6.4
c	c	c	c	t	t	16	20.1	16.0	16.0	5.4	4.3	11.4
c	c	c	t	t	t	17	19.3	17.4	17.5	7.8	5.0	13.6
c	c	t	t	t	t	19	25.9	21.2	19.6	13.2	8.8	18.4
c	t	t	t	t	t	22	34.7	28.8	26.3	18.3	15.9	28.2
t	t	t	t	t	t	24	36.9	34.5	31.8	38.8	21.2	31.7

[a] Model notation: c = constant probability; random = γ'' and γ' are set equal, mix = two detection probabilities for the entire bird assemblage, constant within a primary sampling session; period = probabilities constant between 1996-97, varied between 1997-2004, and constant between 2004-05; and t = time or variation among primary sampling sessions.

[b] Number of parameters

[c] Values for most parsimonious models ($\Delta AIC_c \leq 2$) are in boldface.

Table 3.3 Akaike weights (wi) for candidate maximum-likelihood models that garnered > 10% of the support at any study site.

Model structure[a]						Replicate 1		Replicate 2		Replicate 3	
ϕ	π	γ''	γ'	p	N	fenced	grazed	fenced	grazed	fenced	grazed
c	c	random		mix	c	[b]**0.39**	**0.40**	**0.34**	**0.43**	**0.39**	**0.43**
c	c	random		mix	period	**0.37**	**0.39**	**0.44**	**0.21**	**0.31**	**0.33**
c	c	random		mix	t	0.05	0.06	0.06	0.12	0.07	0.05
c	c	c	c	mix	c	0.17	0.13	0.14	0.15	0.13	0.17

[a] See Table 2 for model notation.

[b] Weights for the most parsimonious models are in boldface.

Table 3.4 Community parameters obtained from robust design closed – capture maximum likelihood mixture models for riparian bird communities in southeast Kansas in the ten-year interval between 1996-97 and 2004-05.

Parameter *Treatment*	Replicate 1 ($\hat{\theta}^*$±SE)	 95% CI	Replicate 2 ($\hat{\theta}$ ±SE)	 95% CI	Replicate 3 ($\hat{\theta}$ ±SE)	 95% CI	χ^2_1	$P \leq$
Species persistence ($\hat{\phi}$)								
Fenced	0.93±0.05	0.72, 0.99	0.92±0.04	0.79, 0.97	0.94±0.06	0.67, 0.99	1.95	0.16
Grazed	0.86±0.06	0.71, 0.96	0.81±0.06	0.65, 0.91	0.93±0.06	0.72, 0.98		
Random emigration ($\hat{\gamma}''=\hat{\gamma}'$)								
Fenced	0.03±0.17	0.00, 1.00	0.00±0.00	0.00, 0.00	0.00±0.00	0.00, 0.00	0.00	1.00
Grazed	0.00±0.00	0.00, 0.00	0.03±0.08	0.00, 0.78	0.00±0.00	0.00, 0.00		
Proportion of community with a high detection rate ($\hat{\pi}$)								
Fenced	0.72±0.06	0.58, 0.82	0.79±0.06	0.66, 0.88	0.53±0.07	0.39, 0.66	0.01	0.91
Grazed	0.73±0.21	0.25, 0.96	0.69±0.07	0.54, 0.80	0.59±0.07	0.45, 0.72		
Detection probability for high detection group ($\hat{p}_{\pi}$)								
Fenced	0.97±0.05	0.41, 1.00	1.00±0.00	0.99, 1.00	0.91±0.03	0.83, 0.96	0.23	0.61
Grazed	0.89±0.14	0.33, 0.99	1.00±0.02	0.00, 1.00	0.91±0.04	0.81, 0.96		
Detection probability for low detection group ($\hat{p}_{1-\pi}$)								
Fenced	0.28±0.09	0.14, 0.48	0.44±0.05	0.35, 0.54	0.18±0.05	0.10, 0.30	1.36	0.24
Grazed	0.43±0.11	0.24, 0.64	0.48±0.06	0.37, 0.58	0.20±0.06	0.11, 0.33		

* parameter estimate

Table 3.5 Derived parameter estimates based on robust design closed – capture maximum likelihood mixture models for riparian bird communities in southeast Kansas in the ten-year interval between 1996-97 and 2004-05.

Parameter	Replicate 1		Replicate 2		Replicate 3			
Treatment	($\hat{\theta}^{*}$ ±SE)	95% CI	($\hat{\theta}$ ±SE)	95% CI	($\hat{\theta}$ ±SE)	95% CI	χ^2_1	$P \leq$
Extinction ($\hat{E}$)								
Fenced	0.07±0.05	0.01, 0.27	0.09±0.04	0.03, 0.22	0.06±0.06	0.00, 0.30	1.75	0.19
Grazed	0.14±0.06	0.06, 0.29	0.19±0.06	0.09, 0.34	0.07±0.06	0.01, 0.26		
Turnover ($\hat{T}$)								
Fenced	0.18±0.06	0.09, 0.31	0.15±0.05	0.07, 0.27	0.13±0.05	0.05, 0.27	0.02	0.90
Grazed	0.19±0.06	0.10, 0.32	0.18±0.05	0.10, 0.31	0.11±0.06	0.03, 0.27		
Number of colonizers ($\hat{B}_{ij}$)								
Fenced	4.96±5.98	-6.85, 16.57	4.57±2.38	-0.08, 9.24	11.88±5.36	1.39, 22.36	0.09	0.77
Grazed	6.05±2.61	0.98, 11.18	11.39±2.07	7.58, 15.66	7.05±5.23	-3.18, 17.34		
Rate of change in species richness ($\hat{\lambda}$)								
Fenced	1.14±0.29	0.68, 1.80	1.16±0.13	0.92, 1.42	1.43±0.26	1.00, 2.03	0.02	0.89
Grazed	1.20±0.15	0.93, 1.51	1.38±0.08	1.22, 1.54	1.22±0.24	0.82, 1.77		

* parameter estimate

Table 3.6 Mean (±SE) number of individuals detected at all fenced and all grazed replicates per year in southeast Kansas[a,b].

Species		**1996**		**1997**		**2004**		**2005**	
Guild									
Common Name	**Latin Name**	**fenced**	**grazed**	**fenced**	**grazed**	**fenced**	**grazed**	**fenced**	**grazed**
Cavity									
Barred Owl	*Strix varia*	0.67(0.33)	-	0.33(0.33)	-	0.33(0.33)	0.67(0.33)	1.00(0.58)	-
Carolina Chickadee	*Poecile carolinensis*	1.33(0.33)	2.67(0.88)	3.00(0.58)	4.00(0.58)	3.33(0.67)	1.67(0.88)	3.33(1.20)	5.67(1.20)
Carolina Wren	*Thryothorus ludovicianus*	1.33(0.33)	1.33(0.33)	1.33(0.33)	0.67(0.33)	3.33(1.20)	2.33(0.33)	1.33(0.67)	1.00(0.00)
Downy Woodpecker	*Picoides pubescens*	2.33(0.67)	3.00(0.58)	1.33(0.33)	1.33(0.33)	1.00(0.58)	1.67(0.67)	2.67(0.88)	2.00(0.58)
Eastern Bluebird	*Sialia sialis*	0.67(0.67)	1.00(0.00)	-	0.33(0.33)	0.33(0.33)	0.33(0.33)	1.33(0.33)	1.67(0.67)
European Starling	*Sturnus vulgaris*	0.33(0.33)	-	-	-	-	-	-	-
Great Crested Flycatcher	*Myiarchus crinitus*	3.00(0.00)	4.00(0.58)	3.00(0.58)	2.67(0.33)	1.00(0.00)	1.67(0.33)	2.67(0.33)	2.33(0.88)
Hairy Woodpecker	*Picoides villosus*	-	-	0.33(0.33)	-	0.67(0.33)	-	-	-
Northern Flicker	*Colaptes auratus*	-	-	-	-	0.33(0.33)	-	-	-
Pileated Woodpecker	*Dryocopus pileatus*	-	-	-	-	0.33(0.33)	0.67(0.33)	0.33(0.33)	0.33(0.33)
Red-bellied Woodpecker	*Melanerpes carolinus*	4.67(0.33)	3.00(1.00)	3.33(0.33)	4.00(0.00)	2.33(0.33)	2.67(0.33)	3.33(0.67)	4.33(0.67)
Red-headed Woodpecker	*Melanerpes erythrocephalus*	-	-	-	-	-	-	1.00(1.00)	-

Table 3.6 continued

Tufted Titmouse	*Baeolophus bicolor*	3.00(1.00)	4.00(0.58)	3.33(0.33)	3.67(0.67)	5.67(2.19)	1.67(0.33)	2.67(0.67)	3.67(0.88)
White-breasted Nuthatch	*Sitta carolinensis*	0.33(0.33)	-	1.00(0.00)	0.67(0.33)	1.67(0.33)	2.33(0.33)	1.00(0.58)	2.00(1.00)
Ground									
Black-and-white Warbler	*Mniotilta varia*	-	-	-	-	0.33(0.33)	0.67(0.33)	-	1.00(0.58)
Kentucky Warbler	*Oporornis formosus*	-	-	-	-	-	-	0.67(0.33)	0.33(0.33)
Northern Bobwhite	*Colinus virginianus*	-	0.33(0.33)	-	-	0.33(0.33)	0.67(0.67)	0.33(0.33)	-
Wild Turkey	*Meleagris gallopavo*	-	-	0.67(0.33)	0.67(0.67)	1.00(0.58)	0.33(0.33)	0.67(0.33)	0.33(0.33)
Shrub/low-foliage									
Acadian Flycatcher	*Empidonax virescens*	-	-	-	-	0.67(0.67)	-	-	-
American Goldfinch	*Carduelis tristis*	0.67(0.33)	0.67(0.33)	1.00(0.58)	1.00(0.58)	1.67(1.67)	0.67(0.67)	0.33(0.33)	-
American Robin	*Turdus migratorius*	0.33(0.33)	1.33(1.33)	-	-	0.33(0.33)	1.00(1.00)	-	-
Blue-gray Gnatcatcher	*Polioptila caerulea*	0.67(0.33)	0.67(0.33)	0.33(0.33)	0.33(0.33)	0.33(0.33)	0.33(0.33)	1.33(0.33)	0.33(0.33)
Brown Thrasher	*Toxostoma rufum*	1.67(0.88)	2.67(1.33)	1.00(0.58)	2.67(0.33)	-	0.33(0.33)	-	0.67(0.67)
Chipping Sparrow	*Spizella passerina*	-	-	-	-	0.33(0.33)	0.33(0.33)	-	-

Table 3.6 continued

Common Grackle	*Quiscalus quiscula*	0.33(0.33)	0.33(0.33)	1.33(0.88)	-	-	0.33(0.33)	-	-
Dickcissel	*Spiza americana*	-	-	-	-	0.67(0.67)	-	0.33(0.33)	0.33(0.33)
Field Sparrow	*Spizella pusilla*	0.67(0.67)	1.67(0.88)	0.33(0.33)	1.00(0.58)	1.00(0.58)	0.33(0.33)	0.67(0.33)	0.67(0.33)
Gray Catbird	*Dumetella carolinensis*	-	0.33(0.33)	-	-	0.33(0.33)	-	-	-
Indigo Bunting	*Passerina cyanea*	4.33(0.88)	2.33(0.67)	5.00(1.00)	4.33(0.33)	2.67(0.33)	2.00(0.58)	4.33(0.33)	2.00(0.58)
Northern Cardinal	*Cardinalis cardinalis*	7.33(0.33)	4.33(0.33)	5.67(0.33)	7.00(1.00)	6.67(1.76)	5.00(2.00)	7.00(0.58)	6.00(1.73)
Northern Mockingbird	*Mimus polyglottos*	-	0.33(0.33)	0.33(0.33)	0.33(0.33)	0.33(0.33)	0.33(0.33)	0.33(0.33)	-
Wood Thrush	*Hylocichla mustelina*	0.67(0.33)	0.33(0.33)	-	1.00(0.58)	-	-	0.33(0.33)	-
Yellow-billed Cuckoo	*Coccyzus americanus*	3.67(0.33)	3.00(0.58)	3.00(0.58)	3.67(0.33)	4.33(1.45)	2.33(0.33)	3.00(1.15)	4.00(1.15)
Sub-canopy/canopy									
Baltimore Oriole	*Icterus galbula*	-	-	0.33(0.33)	0.67(0.33)	-	-	-	-
Blue Jay	*Cyanocitta cristata*	2.00(1.15)	1.67(0.88)	2.00(1.00)	3.67(0.88)	1.00(0.58)	2.33(1.33)	1.33(0.88)	3.33(0.88)
Eastern Kingbird	*Tyrannus tyrannus*	-	-	0.33(0.33)	0.33(0.33)	-	-	-	-
Eastern Wood Pewee	*Contopus virens*	3.33(0.88)	1.67(0.33)	3.00(0.58)	2.33(0.33)	1.00(0.58)	1.33(0.33)	2.00(0.00)	1.00(0.58)

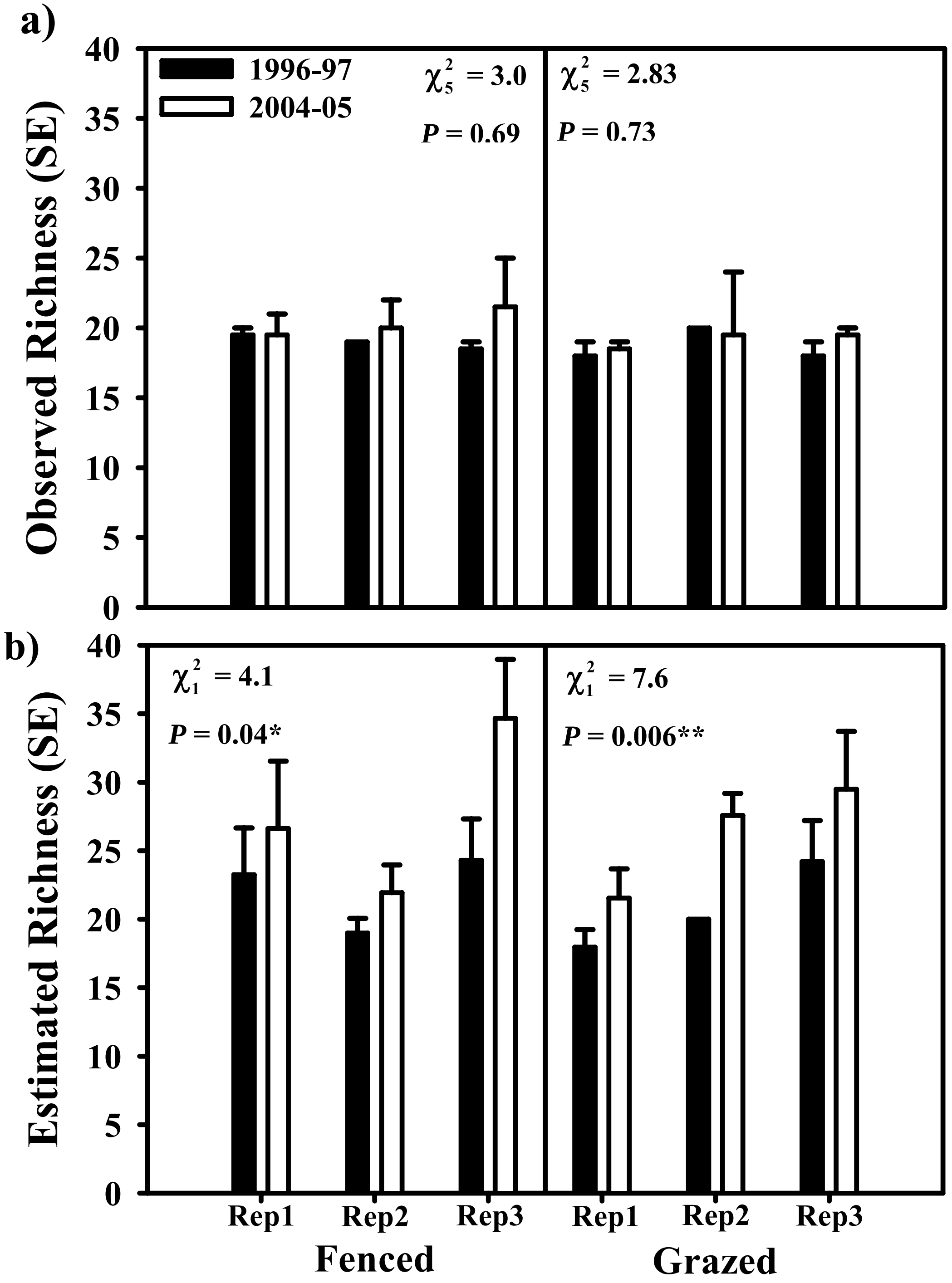
Table 3.6 continued

Fish Crow	*Corvus ossifragus*	-	-	-	-	-	0.33(0.33)	-	-
Great Blue Heron	*Ardea herodias*	-	0.33(0.33)	-	-	-	0.67(0.33)	-	-
Great Horned Owl	*Bubo virginianus*	-	-	-	0.33(0.33)	-	0.33(0.33)	-	-
Green Heron	*Buterides virescens*	0.67(0.33)	-	-	-	0.33(0.33)	0.33(0.33)	-	-
Mourning Dove	*Zenaida macroura*	0.33(0.33)	0.33(0.33)	-	0.33(0.33)	0.33(0.33)	0.67(0.33)	-	-
Red-eyed Vireo	*Vireo olivaceus*	0.33(0.33)	-	1.00(0.58)	0.67(0.67)	-	-	-	1.00(0.58)
Red-shouldered Hawk	*Buteo lineatus*	-	-	-	-	-	-	0.67(0.67)	-
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	-	-	0.33(0.33)	0.67(0.67)	0.33(0.33)	-	-	-
Ruby-throated Hummingbird	*Archilochus colubris*	-	-	-	-	-	-	0.33(0.33)	0.33(0.33)
Summer Tanager	*Piranga rubra*	-	-	-	-	0.67(0.33)	-	0.67(0.33)	0.67(0.67)
Yellow Warbler	*Dendroica petechia*	0.33(0.33)	0.33(0.33)	-	-	0.67(0.67)	-	0.33(0.33)	-
Parasitic									
Brown-headed Cowbird	*Molothrus ater*	1.67(0.67)	2.67(0.33)	0.67(0.33)	1.33(0.88)	0.67(0.33)	1.67(0.88)	3.00(1.15)	0.67(0.33)

[a] For each species, the maximum number of individuals detected on a visit to each replicate was used to obtain the average.

[b] Eastern phoebe (*Sayornis phoebe*) has been omitted from the guild analysis.

Figure 3.1 Riparian bird species richness on the Kansas Army Ammunition Plant: a) average (±SE) observed richness for consecutive years of the study for each replicate site, and b) species richness ($\hat{N}$) estimates generated by robust design mark-recapture models for each replicate site; fenced = cattle excluded, grazed = cattle present.



CHAPTER 4 - Conclusions

Ecological restoration efforts can positively influence plant and animal populations, and have become central to many conservation efforts for rare species or communities. In systems that are often subject to multiple land uses (e.g., agriculture and wildlife management), successful approaches to restoration support both wildlife populations and landowners. Using livestock grazing as a restoration tool may be a useful approach to restoring ecological communities that have an evolutionary history with ungulate grazers, like tallgrass prairie. Livestock grazing can provide a low-cost, relatively low-maintenance alternative restoration method to prescribed burning, intensive chemical treatments, and overseeding with native species. However, understanding community responses to altered rangeland management techniques is imperative to determine whether livestock grazing can be an effective restoration tool.

We found that by allowing cattle to graze tall fescue-dominated pastures during the winter, we decreased the abundance and biomass of the exotic tall fescue, and increased the abundance and biomass of several native, warm-season grasses. Additionally, we observed higher species richness of native plants in winter-grazed pastures compared to the traditional year-round grazed pastures. The grassland breeding bird community showed variable responses to winter-grazing. Winter-grazed pastures had higher species richness of breeding grassland birds than either year-round grazed pastures or native prairie remnants. We observed a higher density of Dickcissels in pastures grazed during the winter only, but Grasshopper Sparrows and Eastern Meadowlarks had higher densities in pastures grazed year-round. Dickcissel density was negatively influenced by the presence of cattle during the breeding season and by a higher abundance of tall fescue in year-round grazed pastures. Grasshopper Sparrow density was negatively influenced by the higher abundance of native grass and associated visual obstruction in some winter-grazed pastures, and was positively influenced by a greater abundance of forbs. We identified no significant habitat variables that influenced the density of Eastern Meadowlarks. Henslow’s Sparrows and Common Yellowthroat were never detected in year-round grazed pastures, but territorial males were detected in low numbers in pastures that had been winter-grazed for five consecutive years. Brown-headed Cowbirds, although detected too infrequently to estimate density, were encountered more frequently in year-round grazed pastures than winter-grazed pastures. Winter-grazing may be more beneficial for some species of grassland breeding birds than for others because of the disparate requirements of nest-site characteristics among species; however, if winter-grazing results in reduced brood parasitism by Brown-headed Cowbirds through decreased cowbird abundance, then grassland-breeding bird communities as a whole could be positively affected.

Quantifying the recovery dynamics in habitats recently subjected to restoration efforts is crucial to evaluating the efficacy of restoration methods. Some ecological communities, especially riparian habitats that are less tolerant to grazing, may show lags in response to changes in rangeland management and passive restoration methods. We evaluated bird community responses in fenced and grazed riparian habitats to see if livestock exclusion positively affected riparian bird species richness and to quantify changes in community vital rates (e.g., local extinction and colonization) since the exclusion of livestock. We used a mark-recapture modeling approach to quantify species richness, which allowed us to account for differences in detectability among species. We found no differences in species richness between fenced riparian sites in which livestock had been excluded for eight years and actively grazed sites. We also observed no differences in local extinction probabilities, species turnover, the number of local colonizers, and the rate of change in species richness between fenced and grazed sites. However, we found substantial differences between unadjusted species counts and species richness estimates from the mark-recapture models, largely due to heterogeneous detection probabilities among species in both fenced and grazed riparian sites. These results suggest that unadjusted counts in community studies can misrepresent the number of species that are present, ultimately resulting in biased estimates of change within the community after restoration.

The accurate quantification of community dynamics after experimental restoration efforts is integral to the development of restoration ecology as a predictive science in which hypotheses about the pace and trajectory of recovery can be made. Approaches that account for differences in detectability among species, such as the mark-recapture approach we used, allow for increased reliability in community surveys and if incorporated into future restoration efforts, should result in a better match between observed and expected restoration timescales.

Appendix A - Riparian encounter histories and geographic coordinates

Appendix Table A.1 Encounter histories[a] for species in fenced and grazed riparian sites on the Kansas Army Ammunition Plant for four primary sampling periods, represented by each year of the study, and 13 secondary sampling periods: 1996:4, 1997:4, 2004:2, 2005:3.

	Fenced/Ungrazed			**Grazed**		
Species	*Replicate 1*	*Replicate 2*	*Replicate 3*	*Replicate 1*	*Replicate 2*	*Replicate 3*
Acadian Flycatcher	0000000010000	-	-	-	-	-
American Goldfinch	0100010100100	-	0011111001000	0100101100000	0000000001000	0010001000000
American Robin	-	-	1000000001000	-	0000000001000	1100000000000
Baltimore Oriole	0000100000000	-	-	0000010000000	-	0000100000000
Barred Owl	1000000000001	0000000001010	1000100000000	0000000010000	0000000010000	-
Black-and-white Warbler	-	-	0000000001000	0000000010001	-	0000000001001
Blue Jay	0000100011000	1110001100001	1111111110111	0000110001110	1111111111111	1111111101110
Blue-gray Gnatcatcher	1100010000100	0000000000010	0010000001010	1000110000000	1000000000000	0000000001001
Brown Thrasher	1000000000000	1110111100000	0000010000000	0000001000000	1110111101010	1101110100000
Brown-headed Cowbird	1001010010100	0100011100100	0010000001100	1101100011000	0011110001000	0100000000111
Carolina Chickadee	0001111101111	1001111011010	1011011111111	1100111110111	1011011010111	1110011100111
Carolina Wren	0111010101111	0110001011000	1101110111111	0100011101100	1101001111011	0010000001111
Chipping Sparrow	-	-	0000000010000	0000000001000	-	-

Table A.1 continued

Common Grackle	-	0000100000000	-	-	-	-
Dickcissel	-	0000000001100	-	-	-	-
Downy Woodpecker	1011000100101	1111100010101	1101111101111	1100101010101	0111001011011	1111010011111
Eastern Bluebird	0000000001011	1010000000100	-	0100001000001	0110000010010	0110000000100
Eastern Kingbird	-	-	0000100000000	-	0000001000000	-
Eastern Phoebe	-	-	-	-	0001000000000	0100000001000
Eastern Wood Peewee	1111101100111	-	1111111111111	1110111010010	1111110001000	1111111101110
European Starling	-	-	0100000000000	-	-	-
Field Sparrow	-	1000011011100	0000000011001	1010011100011	1111111110001	-
Fish Crow	-	-	-	0000000001000	-	-
Gray Catbird	0100000000000	-	0000000010000	-	-	1000000000000
Great Blue Heron	-	-	-	0001000001000	0000000010000	-
Great Crested Flycatcher	1111111110111	1111111111110	1111111110111	1110111110101	1011110110111	1111011110111
Great Horned Owl	-	-	-	-	-	0000100010000
Green Heron	0001000010000	1010000000000	-	-	-	0000000010000
Hairy Woodpecker	0000000001000	0000000100000	0000000010000	-	-	-
Indigo Bunting	1111111111111	0111111110111	1111111111111	1111111111110	1111111111111	0111111111111
Kentucky Warbler	0000000000010	0000000000100	-	0000000000010	-	-
Mourning Dove	-	-	0100000011000	-	0000000001000	0010000110000

Table A.1 continued

Northern Bobwhite	-	-	0000000010100	0100000000000	-	0000000010000
Northern Cardinal	1111111111111	1101111111111	1111111111111	1111111111111	1111111111111	1111111111111
Northern Flicker	0000000001000	-	-	-	-	-
Northern Mockingbird	-	0000001110001	-	-	0001001010000	-
Pileated Woodpecker	-	0000000010010	-	-	0000000010000	0000000010100
Red-bellied Woodpecker	1111111111111	1111111111111	1111111111111	1100111101111	1111111111111	1111111111111
Red-eyed Vireo	1000110000000	-	0000100000000	-	0000101000100	0000000000100
Red-headed Woodpecker	0000000000100	-	-	-	-	-
Red-shouldered Hawk	-	0000000000010	-	-	-	-
Rose-breasted Grosbeak	-	-	0000100010000	-	-	0000100000000
Ruby-throated Hummingbird	-	-	0000000000110	0000000000110	-	-
Summer Tanager	0000000001100	0000000010000	0000000000011	0000000000100	-	-
Tufted Titmouse	1111111111110	1111111111110	1111111011111	1111111111111	1111111111111	1111111111111
White-breasted Nuthatch	0000001011000	0100011001010	0000010011010	0000000001100	0000001010010	0000001011100
Wild Turkey	0000001111011	0000100011010	0000100011010	0000000000110	0000000100000	0000000001000
Wood Thrush	1000000000100	1000000000000	-	0000100000000	1000100000000	-
Yellow Warbler	0000000000100	-	1000000010000	-	1000000000000	-
Yellow-billed Cuckoo	1111111111011	1111111111110	1111111111111	1111111011011	1111111111111	0111111111111

[a] encounter = 1, not encountered = 0

Appendix Table A.2 Geographic coordinates of point count stations in fenced and grazed riparian sites monitored 2004-05 at the Kansas Army Ammunition Plant, Labette Co., Kansas.

Fenced	Grazed
14PC1: N 37° 15.945 W 95° 11.217	*14PC4*: N 37° 15.617 W 95°11.465
14PC2: N 37° 15.856 W 95° 11.306	*14PC5*: N 37° 15.500 W 95° 11.465
14PC3: N 37° 15.750 W 95° 11.343	*14PC6*: N 37° 15.427 W 95° 11.570
24PC5: N 37° 18.128 W 95° 09.177	*24PC1*: N 37° 17.922 W 95° 08.649
24PC6: N 37° 18.119 W 95° 09.286	*24PC2*: N 37° 18.010 W 95° 08.739
24PC7: N 37° 18.133 W 95° 09.416	*24PC3*: N 37° 18.079 W 95° 08.850
30PC1: N 37° 17.715 W 95° 08.374	*24PC4*: N 37° 18.078 W 95° 08.987
30PC2: N 37° 17.606 W 95° 08.397	*30PC5*: N 37° 17.380 W 95° 08.061
30PC3: N 37° 17.542 W 95° 08.286	*30PC6*: N 37° 17.310 W 95° 07.968
30PC4: N 37° 17.439 W 95° 08.249	*30PC7*: N 37° 17.208 W 95° 07.916