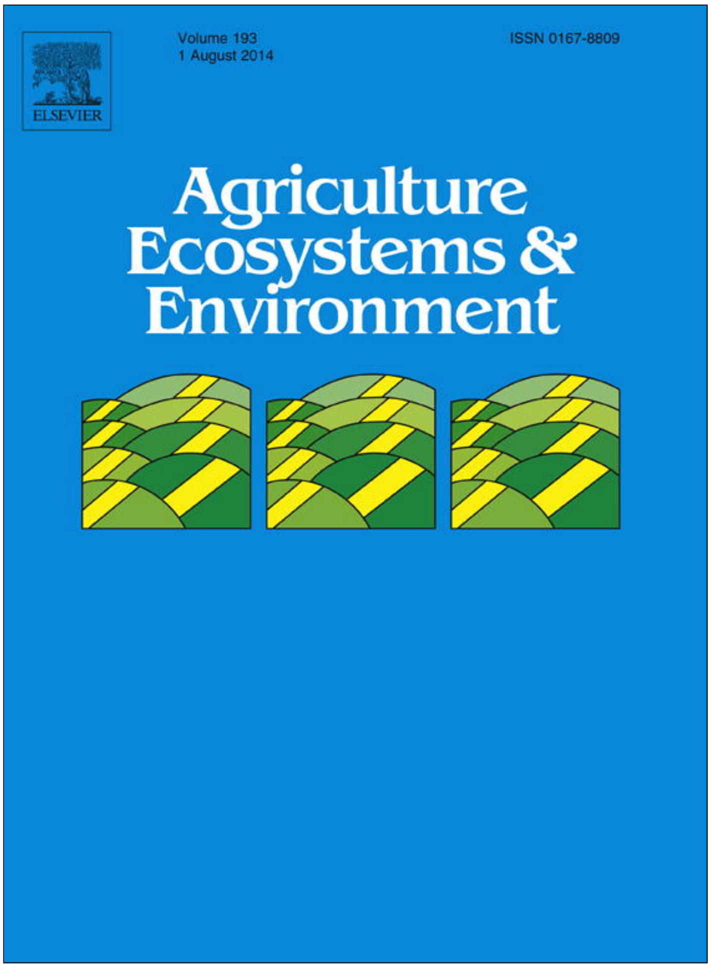
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Grassland bird communities on conservation and marginal grasslands in an agricultural landscape

W. Andrew Cox, L. LaReesa Wolfenbarger*, Lorelle I. Berkeley, Shannon E. Engberg, William M. Janousek, Page E. Klug, Nicholette L. Rider, John P. McCarty

Department of Biology, University of Nebraska Omaha, 103 Allwine Hall, Omaha, NE, 68182, United States

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ABSTRACT

Six years of point count data in eastern Nebraska and western Iowa, USA, were used to investigate how the community structure of grassland birds and the densities of four focal species (common yellowthroat, dickcissel, grasshopper sparrow and sedge wren) varied on conservation lands with differing management strategies (i.e., warm- versus cool-season grasses and low- to high-diversity plantings), and between conservation and unmanaged marginal grasslands (e.g., field borders and terraces). Model-selection results indicated that grasshopper sparrow and dickcissel densities were influenced by grassland type, with higher densities in parcels dominated by warm-season grasses. Species-specific changes in density in response to planting diversity reinforced the value of creating heterogeneous habitat for grassland birds. Densities for all four species were substantially lower in unmanaged marginal grasslands versus conservation parcels and the community structure between the two habitats differed significantly, with generalist species (e.g., American robins, common grackles and grassland species associated with shorter, sparse and patchy vegetation (e.g., horned lark and vesper sparrow)) largely replacing tallgrass specialists in unmanaged marginal grassland parcels.

1. Introduction

Grassland bird species have experienced the steepest long-term population declines of any avian guild in North America (Sauer and Link, 2011) and are the targets of significant conservation planning and management efforts (Rich et al., 2004; Johnson et al., 2004). Remaining grasslands are further threatened by disruption of historical grazing and fire patterns (Brennan and Kuvlesky, 2005), woody encroachment (Briggs et al., 2005), and agricultural intensification (Askins et al., 2007). The cumulative effects of these processes are seen most dramatically in the conversion of >96% of the original tallgrass prairie of the eastern Great Plains to row-crop agriculture and other non-grassland land types (Samson and Knopf, 1994).

Managing remaining grasslands to maximize the abundance and productivity of birds is a conservation priority (Rich et al., 2004). Within landscapes dominated by row crop agriculture, preservation of large grassland fragments is especially important for grassland bird species that may be edge and/or area sensitive (Winter and Faaborg, 1999; Fletcher, 2005; Ribic et al., 2009). Management on large conservation lands can vary, with warm or cool season grasses comprising the dominant plantings, which can influence the abundance of some avian species (Delisle and Savidge, 1997; McCoy et al., 2001; Johnson and Sandercock, 2010). Furthermore, varying the levels of forb and grass diversity can alter the structural characteristics of the vegetation community, which may differentially influence the abundances of grassland species (Johnson and Schwartz, 1993; Delisle and Savidge, 1997). For example, sedge wrens (*Cistothorus platensis*) and grasshopper sparrows (*Ammodramus savannarum*) both occur in tallgrass prairies, but sedge wrens typically prefer tall vegetation with moderate forb cover (Dechant et al., 2002c) whereas grasshopper sparrows (*A. savannarum*) prefer patches with shorter grasses and clumped vegetation (Dechant et al., 2002b).

Efforts to conserve European farmland birds have focused in part on managed and unmanaged field margins in agricultural landscapes (e.g., Robinson and Sutherland, 2002), and the active

* Corresponding author at: Department of Biology, 514C Allwine Hall, Omaha, NE 68182, United States. Tel.: +1 (402) 554 2405; fax: +1 (402) 554 3532.
E-mail addresses: wcox@unomaha.edu (W.A. Cox), lwolfenbarger@unomaha.edu (L.L. Wolfenbarger), lberkeley@mt.gov (L.I. Berkeley), shotch@cox.net (S.E. Engberg), wjanousek@unomaha.edu (W.M. Janousek), pklug@usgs.gov (P.E. Klug), dvmrider@gmail.com (N.L. Rider), jmccarty@unomaha.edu (J.P. McCarty).

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management of field margins has been formerly incorporated into many European Union conservation plans (Vickery et al., 2009). In the United States, by contrast, comparatively little attention has been paid to the use by grassland birds of the unmanaged grassy margins (hereafter, "unmanaged marginal grasslands") associated with the agricultural lands now dominating the landscape. Such habitat can provide food resources for birds (Vickery et al., 2009) and although densities (e.g., Hultquist and Best, 2001), reproductive performance (Best, 2000), and survival (Bro et al., 2004) may be lower than what is found in large block grasslands, unmanaged marginal grasslands often represent the best available habitat in agricultural landscapes where many species infrequently use row-crop fields for foraging or nesting (Best et al., 1995). The recent rise in commodity prices and the associated increase in conversion of grasslands to row-crop fields (Wright and Wimberly, 2013) emphasize the challenges of maintaining, much less increasing, the amount of tallgrass prairie habitat. As such, there is a continued need to manage existing prairies to maximize their conservation potential and to investigate the value of unmanaged marginal grasslands for bird conservation. This is especially urgent given recent suggestions that marginal agricultural land be devoted to the production of biomass for a feedstock for biofuels (Gelfand et al., 2013).

We investigated how land-use and conservation practices influenced the abundance and community structure of grassland birds using 6 years of point count data recorded in tallgrass prairies and unmanaged marginal grasslands associated with agricultural fields in eastern Nebraska and western Iowa, USA. Intensification of land use in this region since the 1920's has produced a landscape with larger fields dominated by corn and soybeans, and less land devoted to grains, pasture, and other crops (Brown and Schulte, 2011). Loss of marginal grasslands from these changes has been at least partially offset by farm programs intended to reduce soil erosion through the planting of grassed terraces, waterways, and bufferstrips (Brady, 2007). Within this landscape, we assessed how abundances of four focal species and the overall avian community structure differed (1) between block conservation grasslands and unmanaged marginal grasslands, (2) between warm- and cool-season dominated conservation grasslands, and (3) as a function of planting diversity within warm-season conservation grasslands. The four focal species were grasshopper sparrow, dickcissel, sedge wren, and common yellowthroat (*Geothylpis trichas*). U.S. Breeding Bird Survey (BBS) data suggest that the sedge wren has exhibited a modest long-term population increase in the United States during 1966–2011 whereas the other three species have exhibited significant declines (Sauer et al., 2012). For conservation efforts within the Western Hemisphere, Rich et al. (2004) categorized the dickcissel as a "Watch Species" because of its declining trends and the grasshopper sparrow as a "Stewardship Species" because its populations are concentrated in a single biome.

2. Methods

2.1. Data collection

Point counts were performed on 131 parcels in eastern Nebraska and western Iowa during 2002–2007. Field and parcel boundaries were defined by landowners and managers and represented distinct management practices and histories (Klug et al., 2009). Most counts in conservation parcels ($n = 109$) occurred in management units within the Boyer Chute and DeSoto National Wildlife Refuges on the eastern border of Nebraska along the Missouri River. Conservation parcels at both refuges were managed under varying seeding and fire regimes and were interspersed among forested and agricultural management units. Other conservation sites included three Conservation Reserve Program (CRP) parcels, three parcels at the Allwine Prairie Preserve (a restored prairie owned and managed by the University of Nebraska at Omaha), one parcel at Cuming City Cemetery (a remnant prairie managed by Dana College), and three privately-owned parcels converted to grasslands for conservation purposes. The mean size of conservation parcels was 16.1 ha (SE 1.8 ha, range 1.6–128 ha). Some conservation parcels ($n = 31$) had originally been planted with seed mixes consisting predominantly of cool season grasses, eight of which were converted from cool- to warm-season planting during the course of the study. Conservation parcels planted with warm season grasses were categorized based on the diversity of plant species included in the seed mix. Low diversity grasslands had been planted with seed mixes containing fewer than 35 species, medium diversity sites were planted with mixes that included 35–65 species, and high diversity sites had been planted with more than 65 species. Some of the older restorations had been over seeded with additional species and their categories were adjusted to reflect the added plant diversity at the time of the surveys. Both Allwine Prairie Preserve and Cuming City Cemetery were classified as high diversity because data provided by land managers indicated >65 species occurred on both preserves. Marginal parcels ($n = 22$) were surveyed within corn and soybean row-crop fields typical of the study region. Marginal grasslands consisted of small linear grassy terraces and/or waterways within the row crops fields and grassy habitat along the field margins. In contrast to the CRP land, which was planted with native species, no marginal grasslands were managed for wildlife conservation. The mean size of the row crop fields surveyed was 62.1 ha (SE 11.3 ha, range 9.9–224.7 ha), with a mean marginal grassland area of 2.5 ha (SE 0.3 ha, range 0.5–5.9 ha). The linear nature of the marginal grasslands resulted in edge to interior ratios (0.308 ± 0.124 m/m^2 SE) that were substantially greater than those for conservation parcels (0.022 ± 0.001 m/m^2 SE).

Between 18 May and 15 June each year parcels were visited 1–11 times (mean = 2.1 ± 0.1 SE) on the basis of parcel area for fields and length for linear buffers, terraces, and roadsides. Each year points were selected within parcels in a manner that maximized coverage of the parcel while minimizing overlap, with points placed $\geq$100 m from parcel edges except for nine parcels for which the edge was <100 m. Some overlap between points occurred, but distance sampling is robust to violations of the assumption of geographic independence among points even when overlap is severe (Buckland, 2006). Point locations within parcels were not fixed across years. Five-minute, unlimited radius counts (except on seven small conservation parcels, where the radii were truncated to avoid counting birds in adjacent parcels) were performed by 16 total observers from 06:00 to 10:00 or 18:00–20:45 when weather conditions were appropriate (i.e., no fog, no or very light precipitation, wind speeds $\leq$25 kmph). Observers were experienced in identifying birds by sight and sound and their competence was confirmed prior to any data collections. Observers generally used range finders to estimate the distance to each observed bird, but they were also trained to estimate the distance when a bird could not be reliably sampled with a range-finder. Nine parcels (7%) were sampled in all 5 years of the study. An addition 39 parcels (29%) were surveyed in 3–4 years of the study and the remaining parcels (64%) were either sampled once or twice.

2.2. Analysis

Subsets of the overall dataset were used for each of three habitat comparisons (unmanaged marginal versus conservation parcels, warm- versus cool-season parcels, and planting diversity within warm-season parcels) because the sampling effort within management regimes changed across years depending upon study objectives. A hierarchical, distance-based model was used to estimate densities of the four focal species that was originally developed by Royle et al. (2004) and is formally implemented in package *unmarked* (Fiske and Chandler, 2011) in program R (R Core Team, 2013). The experimental unit for these analyses was each surveyed point.

Three detection covariates were incorporated into the analyses (observer, wind speed, and time of day). Model fitting was problematic when observers that accounted for relatively few points were included, so observers responsible for <25 points were placed into either an "expert" or "non-expert" group based on the similarities of their experience with bird surveys. Time was included as a categorical rather than continuous variable because of the bimodal distribution of the survey effort (i.e., surveys were performed in mornings and evenings). As such, the morning was divided into two categories (dawn–07:29, 07:30–09:45) and a third category was created for evening counts, which occurred during 18:00–20:45. In addition to the inclusion of habitat type as an abundance covariate, a covariate for year was included to allow for annual variation in abundances. Points that were missing covariate values were excluded from the analyses. Burn histories were not available for many parcels, so the time since the most recent burn was excluded from the analyses. Points from parcels that were burned in the same year of the count were also excluded because we did not consider those counts to be indicative of bird use of the parcels in most years.

Unmarked requires that distances be binned into intervals. To choose appropriate bins, the farthest ~10% of detections were excluded, choosing slightly less or more if a natural break in the distribution of distances suggested a better break point (Buckland et al., 2001). 10–20 m bins were then created for each species, based on the distribution of detections (breaks and maximum distances are reported in Table S1 and Tables 1 and 2). All possible combinations of the detection covariates were considered using both half-normal and hazard key functions, and relative support of the models were evaluated within an information-theoretic approach (Burnham and Anderson, 2002). Models were ranked using Akaike's Information Criterion (AIC), and model support was assessed by examining the difference between the top model and other candidate models (ΔAIC) and the overall weight of evidence supporting each model (w_i). The best detectability model was then incorporated into a set of models with abundance covariates. A null abundance model, a model with year, a model with habitat type (e.g., unmanaged versus conservation parcels), and a model with year and habitat type were considered. Substantial weight for the null model (i.e., if it was top-ranked) was interpreted to indicate a lack of support for an effect of habitat on densities. Model-based predictions of bird densities in each habitat type were generated to present biologically relevant habitat effects. This was accomplished by using the *predict* function of *unmarked* to generate back-transformed estimates of densities for each habitat type, using predictions averaged across the suite of candidate models when model-selection uncertainty was present (Burnham and Anderson, 2002). The *predict* function does not allow a user to average predictions across categorical variables, so model-based predictions were generated for a single representative year for the comparison of densities between planting diversities within warm-season grasslands, which was based on data collected over 4 years. However, if there are just two levels of a categorical variable, it can be treated as a continuous variable and produce correct parameter estimates. This approach was used for the remaining two habitat comparisons (marginal versus conservation parcels, warm- versus cool-season parcels), and model-based predictions were averaged across the 2 years each analysis was based on.

We used the parametric bootstrap option of *unmarked* to assess how well the top ranked model fit the data. Briefly, we simulated 100 datasets from the model and each time we refit the model to the new dataset and generated a Freeman–Tukey fit statistic (sensu Sillett et al., 2012). We then compared the fit statistic from the actual data to the distribution of fit statistics from the simulated data. With this approach, *P*-values >0.975 or <0.025 indicate that the actual data do not fit the model as well as the simulated (i.e., well-fit) data.

Non-metric dimensional scaling (NMDS), typically the most effective ordination technique for ecological community data (McCune and Grace, 2002), was used to evaluate community dissimilarity for each of the three habitat comparisons. Each parcel was the experimental unit for these analyses. NMDS requires equal survey effort within parcels so a subset of the overall dataset was used to meet this requirement (i.e., most parcels lacked identical sampling histories across years). Points from 2003 only were used in the comparison of communities on warm- versus cool-season conservation parcels, and for the comparison across planting diversities within warm season parcels. Data from 2004 and 2005 were used for the comparison of marginal versus conservation parcels. For all analyses, the raw abundance of each species was averaged across the first two point counts at each parcel (four values were averaged for the marginal versus conservation parcels because we used two years of data), and any parcel with <2 counts was excluded. The removal of rare species can reduce statistical noise, improve the clarity of results, and reduce the risk of obtaining spurious results for rare species (i.e., species may appear to prefer a particular

Table 1
Model selection results for four songbird species surveyed in unmanaged marginal parcels (n = 105 points) and conservation parcels (n = 254) during 2004–2005 in eastern Nebraska and western Iowa, USA.

	K[1]	AIC[2]	ΔAIC[3]	w_i[4]
COYE (n = 473; 15 m, 135 m)[5]				
Detection				
Observer (hazard)	10	2396.00	0.00	0.44
Observer + wind (hazard)	11	2397.98	1.98	0.16
Observer + time (hazard)	11	2397.99	1.99	0.16
Observer (half-normal)	9	2399.01	3.01	0.10
Abundance				
Habitat (P = 0.52)[6]	11	2260.21	0.00	0.73
Habitat + year	12	2262.15	1.94	0.27
DICK (n = 704; 15 m, 150 m)				
Detection				
Null (hazard)	4	3058.63	0.00	0.21
Wind (hazard)	5	3058.73	0.10	0.20
Observer (hazard)	10	3059.98	1.35	0.11
Abundance				
Habitat + year (P = 0.48)	6	3017.22	0.00	1.00
GRSP (n = 382; 20 m, 100 m)				
Detection				
Observer (hazard)	10	1706.56	0.00	0.46
Observer + wind (hazard)	11	1708.37	1.81	0.19
Observer + time (hazard)	11	1708.56	2.00	0.17
Abundance				
Habitat (P = 0.55)	11	1639.21	0.00	0.70
Habitat + year	12	1640.91	1.70	0.30
SEWR (n = 181; 15 m, 90 m)				
Detection				
Observer (half-normal)	9	1055.21	0.00	0.21
Observer + wind (half-normal)	10	1055.35	0.14	0.19
Observer + time (half-normal)	10	1055.73	0.52	0.16
Global (half-normal)	11	1056.20	0.99	0.13
Global (hazard)	12	1056.47	1.26	0.11
Abundance				
Habitat (P = 0.50)	10	985.11	0.00	0.71
Habitat + year	11	986.86	1.75	0.29

Detectability models with w_i < 0.10 are not presented.
[1] Number of parameters.
[2] Akaike's information criterion.
[3] The difference between the current and top-ranked model's AIC value.
[4] Weight of evidence supporting the model.
[5] Number of birds detected, distance bin size, and the maximum distance included in each analysis.
[6] *P*-value is from Tukey–Freeman goodness of fit test, with values >0.05 indicating adequate fit.

Table 2
Model selection results for four songbird species surveyed during 2004–2007 in warm season grassland parcels planted with low (n = 161 points), medium (n = 56), and high (133) diversity seed-mix in eastern Nebraska and western Iowa, USA.

	K[1]	AIC[2]	ΔAIC[3]	w_i[4]
COYE (n = 586; 10 m, 110 m)[5]				
Detection				
Observer (half-normal)	9	3159.99	0.00	0.52
Observer + time (half-normal)	10	3161.81	1.82	0.21
Observer + wind (half-normal)	10	3161.96	1.97	0.19
Abundance				
Habitat (P = 0.38)[6]	11	3144.68	0.00	0.66
Habitat + year	14	3145.98	1.30	0.34
DICK (n = 943; 15 m, 135 m)				
Detection				
Observer (hazard)	10	3676.87	0.00	0.23
Null (hazard)	4	3677.61	0.74	0.16
Observer + time (hazard)	11	3678.53	1.66	0.10
Abundance				
Habitat + year (P = 0.30)	15	3653.19	0.00	0.98
Year	13	3660.74	7.55	0.02
GRSP (n = 444; 20 m, 100 m)				
Detection				
Observer (hazard)	10	1825.32	0	0.48
Observer + wind (hazard)	11	1826.87	1.55	0.22
Observer + time (hazard)	11	1827.28	1.95	0.18
Abundance				
Null (P = 0.47)	10	1825.32	0.00	0.29
Habitat	12	1825.49	0.17	0.27
Habitat + year	15	1825.62	0.30	0.25
Year	13	1826.11	0.79	0.19
SEWR (n = 272; 15 m, 105 m)				
Detection				
Observer (half-normal)	9	1518.94	0.00	0.19
Observer + wind (half-normal)	10	1520.00	1.06	0.11
Abundance				
Habitat (P = 0.38)	11	1503.74	0.00	0.78
Habitat + year	14	1506.29	2.55	0.22

Detectability models with w_i < 0.10 are not presented.
[1] Number of parameters.
[2] Akaike's information criterion.
[3] The difference between the current and top-ranked model's AIC value.
[4] Weight of evidence supporting the model.
[5] Number of birds detected, distance bin size, and the maximum distance included in each analysis.
[6] P-value is from Tukey–Freeman goodness of fit test, with values >0.05 indicating adequate fit.

habitat even though the sample size is too small to draw robust conclusions) without substantially influencing output, so we made an *a priori* decision to exclude species that occurred on <5% of parcels (McCune and Grace, 2002).

The *metaMDS* function within package *vegan* in Program R (Oksanen et al., 2013) was used to perform the NMDS analyses. For each analysis, stress as a function of dimensionality was plotted to determine the optimal number of dimensions to use (McCune and Grace, 2002). The Bray–Curtis dissimilarity index was used, and output was considered to be reliable if (1) stress values were <20 and (2) multiple random starts converged on a similar solution, which indicated that a global rather than local solution was reached. The automated scaling and reorientation provided by the *postMDS* function was used to improve the visual clarity of the output. Finally, goodness-of-fit values (R^2) and associated P-values were generated to provide an indication of the strength of the correlation of habitat type with the ordination.

3. Results

3.1. Density estimates

86 points in 59 warm-season conservation parcels and 40 points in 28 cool-season conservation parcels were surveyed during 2002–2003. 22 of the surveys occurred during the evening and 104 in the morning. Detection models with observer as the covariate were top-ranked for all three species (Table S1). Model-selection results did not support an effect of habitat on densities for common yellowthroats, whereas a model with habitat and year was top-ranked for dickcissels and grasshopper sparrows (Table S1). The latter two species were more abundant in parcels with predominantly warm-season grasses, but confidence intervals were wide, reflecting the considerable variability in the results (Fig. S1a–c).

105 points in 22 marginal parcels and 254 points in 60 conservation parcels were surveyed during 2004–2005. All counts were performed in the morning. The best detectability model included a covariate for observer for all species except the dickcissel, the top model for which did not include detection covariates (Table 1). On unmanaged marginal sites 106 dickcissels were detected, whereas just eight common yellowthroats, 20 grasshopper sparrows (13 of which were on a single parcel), and zero sedge wrens were detected. As such, habitat effects were well supported for all species, with the year and null models accounting for 0% of the cumulative model-weight for any species. All four species occurred in substantially higher densities in conservation versus marginal parcels (Fig. 1a–d).

161 points in 27 low-diversity warm-season conservation parcels, 56 points in eight medium-diversity conservation parcels, and 133 points in 23 high-diversity conservation parcels were surveyed during 2004–2007. All counts were performed in the morning. Top-ranked detectability models varied by species (Table 2). Models with a habitat covariate accounted for 100% of the cumulative weight for common yellowthroats, dickcissels, and sedge wrens. In contrast, the null model was most supported for grasshopper sparrows. Densities of common yellowthroats and sedge wrens were lowest and dickcissel densities highest in high-diversity parcels (Fig. 2).

3.2. Community dissimilarity

50 species were detected during the counts on conservation and marginal lands, 33 of which were excluded because they occurred on fewer than 5% of parcels. There was a substantial difference in community composition between marginal (n = 15) and conservation (n = 26) parcels (Fig. 3; R^2 = 0.44, P < 0.01). 43 species were detected during the counts on warm- and cool-season parcels, 30 of which were excluded because of small sample sizes. Most cool-season parcels (n = 17) were closely grouped with one another, but warm-season parcels (n = 45) were spread across the ordination and habitat was only modestly correlated with the ordination (Fig. S2; R^2 = 0.11, P < 0.01). 37 species were detected during the counts for the comparison of low-, medium-, and high-diversity plantings within warm-season parcels, 21 of which were excluded because of small sample sizes. NMDS converged on a solution but the ordination stress was 0.22 so we chose not to interpret the results of the analysis (McCune and Grace, 2002).

4. Discussion

Densities of dickcissels, common yellowthroats, and grasshopper sparrows were substantially higher in conservation versus unmanaged marginal grasslands, and sedge wrens were detected exclusively on conservation parcels. Furthermore, although there was some overlap among the species found in these two habitat types, the ordination analyses showed significant separation between unmanaged marginal grasslands and conservation grasslands (Fig. 3). In particular, unmanaged marginal grasslands were more strongly associated with habitat generalist species such as the American robin, common grackle, and brown-headed cowbird,

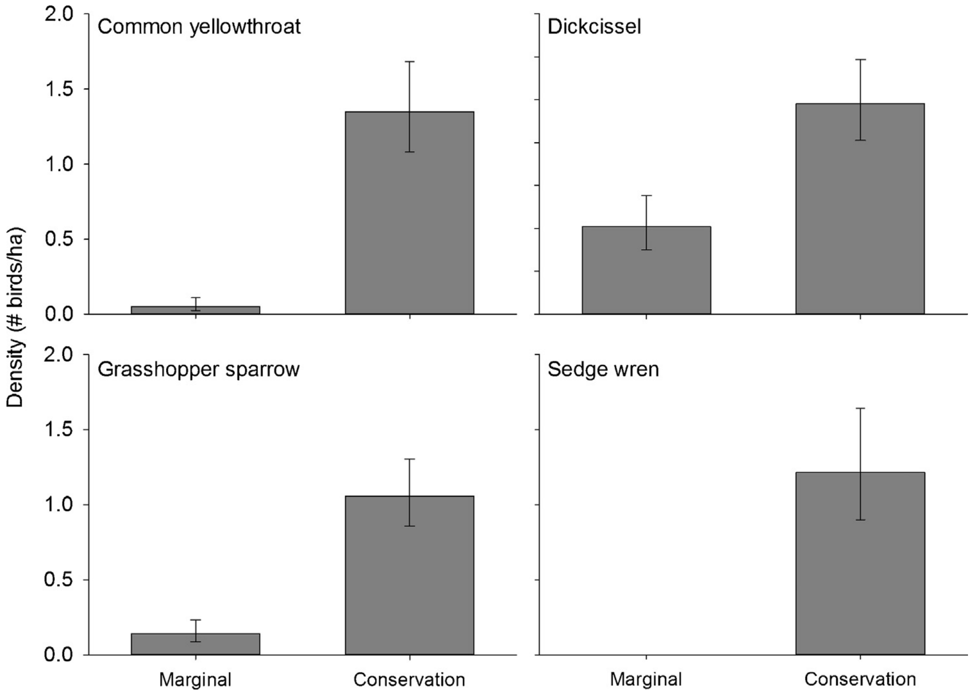

Fig. 1. Predicted densities of four grassland songbird species in unmanaged marginal parcels and conservation grassland parcels in eastern Nebraska during 2004–2005. Error bars represent 95% confidence intervals.

as well as with grassland species associated with shorter, sparser and patchier vegetation than is typically found in tallgrass prairie sites (e.g., horned lark and vesper sparrow; Dechant et al., 2002a; Dinkins et al., 2002).

The estimated densities of dickcissels and grasshopper sparrows reported here on conservation parcels are statistically robust and are similar to those estimated in western and northern Missouri (Jacobs et al., 2012) and southeastern Nebraska (Delisle and Savidge, 1997). However, estimates from unmanaged marginal parcels are not derived from a point composed only of grassland habitat, but rather from grassland-centered farmland habitat because the narrow, linear nature of the unmanaged marginal parcels resulted in points at which a substantial portion of the habitat surveyed was row-crop fields. These estimates are thus lower than they would be if we were able to calculate them for grassland habitat, per se. Although the limitations of the survey design

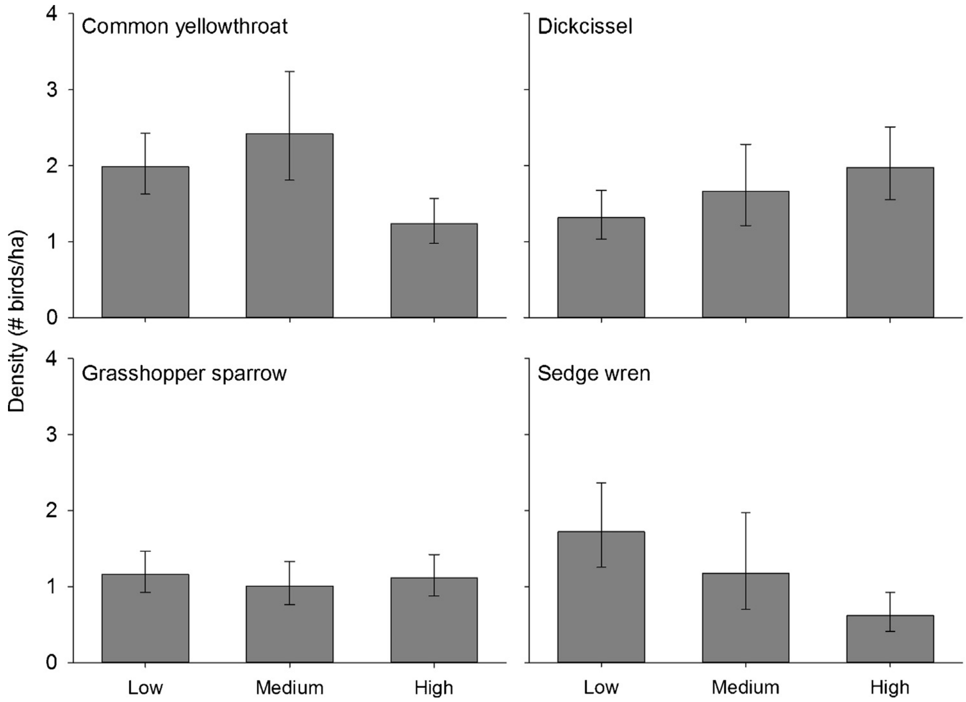

Fig. 2. Predicted densities of four grassland songbird species in warm-season grassland parcels planted with low, medium, and high diversity seed-mix in eastern Nebraska and western Iowa, USA. Data are from 2004 to 2007 but estimates are for 2005 only (see Section 2). Error bars represent 95% confidence intervals.

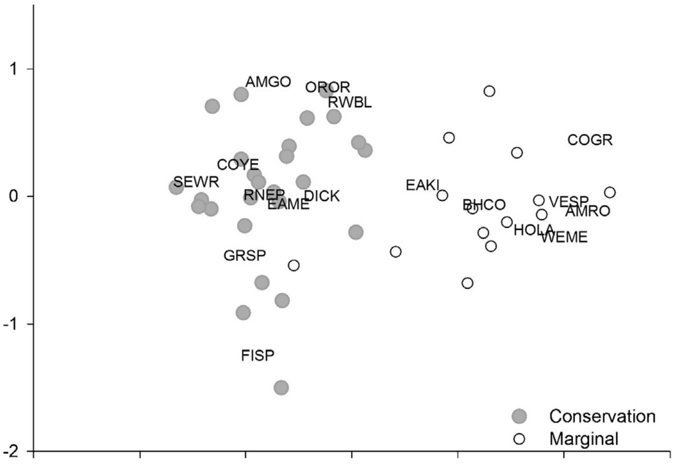

Fig. 3. Non-metric dimensional scaling (NMDS) ordination of grassland songbird community composition in marginal and conservation parcels during 2004–2005. Species' locations indicate points they are most closely associated with (i.e., species overlapping "conservation" points are more closely associated with conservation parcels). AMGO = American goldfinch (*Spinus tristis*), OROR = orchard oriole (*Icterus spurius*), RWBL = red-winged blackbird (*Agelaius phoeniceus*), COYE = common yellowthroat, RNEP = ring-necked pheasant (*Phasianus colchicus*), EAME = eastern meadowlark (*Sturnella magna*), DICK = dickcissel, GRSP = grasshopper sparrow, FISP = field sparrow (*Spizella pusilla*), SEWR = sedge wren, EAKI = eastern kingbird (*Tyrannus tyrannus*), BHCO = brown-headed cowbird (*Molothrus ater*), HOLA = horned lark (*Eremophila alpestris*), WEME = western meadowlark (*Sturnella neglecta*), AMRO = American robin (*Turdus migratorius*), VESP = vesper sparrow (*Pooecetes gramineus*), COGR = common grackle (*Quiscalus quiscula*).

make an apples-to-apples comparison between conservation and unmanaged marginal parcels somewhat problematic, we believe the estimates from marginal lands are still useful for two reasons. First, sedge wrens were absent from unmanaged marginal lands and raw detection rates were extremely low on marginal versus conservation parcels for grasshopper sparrows (1.42 birds/point on conservation parcels, 0.19 birds/point on unmanaged marginal parcels with no birds detected on 93 of 105 marginal parcel points) and common yellowthroats (1.83 birds/point on conservation parcels, 0.08 birds/point on unmanaged marginal parcels with no birds detected on 97 of 105 marginal parcel points). Thus, the density estimates accurately portray the fact that these species rarely use unmanaged marginal lands in the study area. Second, even if the dickcissel density estimates are biased low, they indicate substantial use of unmanaged marginal habitat by the species.

Within conservation lands, warm-season parcels supported greater numbers of dickcissels and, to a lesser extent, grasshopper sparrows compared to cool-season parcels. In addition, planting diversity had pronounced species-specific effects on the densities of dickcissels, sedge wrens, and common yellowthroats but not grasshopper sparrows. These findings might suggest that unmanaged marginal grasslands could be improved by planting warm season grasses and a greater diversity of native forbs. Similar management practices have been used on field margins in Europe and have been shown to increase the abundance and availability of food for birds (reviewed by Vickery et al., 2009), and although there is potential for improved vegetation quality to increase grassland songbird densities in unmanaged marginal lands (see Best, 2000), the success of such efforts may be limited for two reasons. First, previous research demonstrates variable responses by grassland birds to management practices. For example, compared to cool-season grasses, warm-season grasslands may exhibit higher (e.g., Walk and Warner, 2000; Henningsen and Best, 2005) or similar dickcissel densities (Delisle and Savidge, 1997; McCoy et al., 2001), higher (Giuliano and Daves, 2002) or lower (Walk and Warner, 2000; McCoy et al., 2001) grasshopper sparrow densities, and higher (McCoy et al., 2001; Giuliano and Daves, 2002) or lower (Johnson and Schwartz, 1993) common yellowthroat densities. Second, and more critically, unmanaged marginal grasslands such as waterways, roadside ditches, and terraces are by definition narrow and linear in shape. Grasshopper sparrows and sedge wrens are area-sensitive in tallgrass prairies (e.g., Herkert, 1994a,b; Bakker et al., 2002) with grasshopper sparrows also negatively affected by high perimeter-area ratios (Helzer and Jelinski, 1999). Dickcissels are relatively insensitive to patch area (e.g., Winter and Faaborg, 1999; Herkert, 1994b), which may in part explain their comparatively high densities in unmanaged marginal grassland parcels, but nevertheless are often absent in patches with high perimeter-area ratios (Helzer and Jelinski, 1999). As with dickcissels, common yellowthroats are not generally area sensitive (e.g., Johnson and Igl, 2001), but the parcels studied here were substantially smaller than those considered in previous area sensitivity studies.

Our results demonstrate the importance of protected lands to many obligate grassland bird species and the ability of managers to manipulate habitat on conservation grasslands to maximize the densities and diversity of grassland songbirds. However, the results also show that some birds use small, linear grassland patches embedded within a landscape dominated by row-crop agriculture. Given the interspersion of different types of potential grassland bird habitat within the landscape, the dichotomy we draw between marginal and conservation lands reflects a distinct departure from the focus on "farmland birds" by conservation and agricultural programs in much of Europe (Donald et al., 2002; Butler et al., 2007; Hiron et al., 2013). Whether conservation efforts for grassland birds in North America would benefit from shifting its primary focus on remaining parcels of prairie to one with a broader emphasis on landscapes composed of a variety of habitats of different quality might depend on whether cropland itself has the potential to provide resources for native birds. Such a shift in perspective is most likely to occur if production of grass as a feedstock for biofuel production becomes widespread in the Great Plains. To date, increased demand for biofuels has shifted market demands and commodities pricing and has a negative impact on grassland habitats (Wright and Wimberly, 2013). Native grasslands may in fact represent the most economical crop for biofuels when balancing energy production against greenhouse gas emissions and required inputs (Gelfand et al., 2013). If the conversion of grasses into biofuels becomes commonplace, there will surely be impacts on grassland birds, though whether this will help or hinder conservation efforts remains unclear (Robertson et al., 2012). Unmanaged marginal grasslands and other unprotected grassland patches may increase in economic importance, and any consequent growth in the size or frequency of unmanaged marginal grassland patches will alter their value to the conservation of grassland birds.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2014.04.026.

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