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Short-term Effects of Burning and Disking on Songbird Use of Floodplain Conservation Easements

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ABSTRACT.—Extensive conversion of Midwestern riparian areas for agricultural production has had many consequences including reduced habitat for nesting birds. However, more than 120,000 ha of riparian habitat have been restored in this region through USDA conservation programs. In 2001 and 2002, we assessed songbird responses to burning and disking for management of conservation easements in east-central Iowa. We randomly assigned herbaceous riparian fields to burning and disking treatments and collected data on density and species richness of songbirds in these habitats. Total density of grassland and wetland species and red-winged blackbirds (*Agelaius phoeniceus*) were reduced by burning in the first and second breeding seasons after burning; common yellowthroat (*Geothlypis trichas*) density decreased with burning only in the first season. Disking led to increased density of grassland and wetland birds and greater overall avian conservation value on treated relative to untreated fields in the year after treatment. Changes associated with burning and disking treatments were likely related to changes in both vegetation structure and abundance of arthropod food resources. Despite decreased bird densities with burning, fire is a necessary management tool to control woody vegetation. Overall, both burning and disking appear to be effective management practices for maintaining herbaceous riparian habitats for grassland birds.

INTRODUCTION

Since European settlement, the Midwestern United States has undergone major land-cover changes. Prairies and wetlands have been reduced to a fraction of their former extent and rowcrop agriculture has become the dominant land use in many areas (Dahl, 1990; Samson and Knopf, 1994). In Iowa, this conversion of presettlement landscapes to agriculture has resulted in the loss of >99% of native prairie and >95% of wetlands (Bishop, 1981; Smith, 1998). As interfaces between terrestrial and aquatic ecosystems, riparian areas are important landscape components that, like other natural communities, have been greatly altered. The loss of native floodplain functions, including flood-water storage, nutrient and sediment retention, and wildlife habitat, has been more widespread in the Midwest than in any other region of the United States (Brinson *et al.*, 1981; National Research Council, 2002).

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Historically, the structure and composition of Midwestern riparian plant communities were shaped by fine-scale elevation differences and the timing, duration and extent of disturbances such as flooding and fire (Brinson *et al.*, 1981; Gregory *et al.*, 1991; Nelson *et al.*, 1998). Although some riparian areas were forested before European settlement, many of these areas were dominated by extensive grasslands and herbaceous wetlands (Weaver, 1968; Nelson *et al.*, 1998; Benson *et al.*, 2006). This diversity of habitat types and hydrologic conditions historically made riparian areas important for a diversity of plant and animal species (Fredrickson and Reid, 1986; Iowa Department of Agriculture and Land Stewardship, 1999).

In the past two decades, the implementation of Farm Bill programs such as the Conservation Reserve Program (CRP) and Wetlands Reserve Program (WRP) has led to restoration of millions of hectares of grasslands and wetlands. Authorized in 1990, WRP is a voluntary program in which landowners are compensated for taking their land out of agricultural production and restoring natural vegetation (Gray, 2005). Combined with emergency flood-mitigation programs such as the Emergency Wetlands Reserve Program (EWRP), CRP and WRP have restored thousands of hectares of riparian habitat throughout the Midwest. These programs have a variety of goals, including providing habitat for wildlife populations, especially migratory birds (Heard *et al.*, 2000). Through these programs, restoration of riparian grasslands and wetlands may benefit many bird species, including those that have experienced widespread and consistent population declines (Peterjohn and Sauer, 1999; Benson *et al.*, 2006).

To establish and maintain herbaceous habitats in the Midwest, managers use a range of practices including burning and disking. These practices potentially affect the habitat use of numerous bird species by altering vegetation structure or composition and availability of food resources. As a historically important source of disturbance and common management tool for grass-dominated systems in the Midwest, more is known about burning than disking. Burning alters grasslands by removing woody and residual vegetation, encouraging growth of grass species and decreasing arthropod populations (Kucera and Ehrenreich, 1962; Warren *et al.*, 1987; Hulbert, 1988). Disking, typically used to cultivate agricultural fields, is receiving increasing attention by wildlife managers as a method to increase vegetative diversity in largely monotypic grass stands. By mechanically removing existing vegetation, decreasing grass and litter cover, and increasing bare-ground and forb cover, disking has been primarily viewed as a method for providing food-rich brood-rearing habitat for game birds (*e.g.*, Webb and Guthery, 1983; Manley *et al.*, 1994; Madison *et al.*, 1995). The effects of burning on grassland birds vary regionally and among species (Reinking, 2005), and research on bird responses to disking is limited. However, disking relatively small portions of fields may, by creating concentrated patches of arthropod food resources (Benson *et al.*, 2007), increase the density or species richness of birds in these areas.

To provide guidance for future restoration and management activities, we evaluated the effects of burning and disking on the density and species richness of birds breeding in grass-dominated riparian habitats of east-central Iowa. Because burning and disking alter vegetation structure and composition and the abundance and biomass of arthropods (Benson *et al.*, 2007), we expected these management practices to affect bird communities. Specifically, we expected burning and disking to make habitats more suitable for species that are associated with open herbaceous habitats and less suitable for species that prefer densely vegetated herbaceous areas. Additionally, because burning causes decreases in arthropod abundance, while disking increases arthropod abundance (Benson *et al.*, 2007), we expected decreases in bird response variables with burning but increases with disking.

MATERIALS AND METHODS

STUDY AREA

We studied grass-dominated herbaceous habitats in Tama, Benton and Iowa counties of east-central Iowa (Benson *et al.*, 2006). Our study sites were within the >20,000 ha Iowa River Corridor Project (IRC), a cooperative project among the Natural Resources Conservation Service (NRCS), United States Fish and Wildlife Service (USFWS), and Iowa Department of Natural Resources (IDNR). This area was historically dominated by herbaceous vegetation (Benson *et al.*, 2006). Currently, >100 riparian WRP and EWRP easements totaling about 5000 ha of former agricultural land are enrolled in NRCS conservation programs in the IRC. The watershed drains about 1.25 million ha, about 1 million ha of which were cropland prior to restoration (United States Department of Agriculture, 1976).

The plant species composition in IRC fields was dependent on hydrology, previous land use and extent of restoration efforts. We placed fields into two groups based on flood frequency; fields that flood rarely (mesic grasslands) and those that flood frequently (hydric grasslands). Mesic grasslands were generally planted to native grass species, usually switchgrass (*Panicum virgatum*) or big bluestem (*Andropogon gerardii*), and are sufficiently dry in many years to permit burning. Because of a lack of suitable seed mixes for hydric soils, managers allowed these areas to naturally revegetate and they are generally dominated by reed canarygrass (*Phalaris arundinacea*), an invasive perennial (Hoffman and Kearns, 1997; Merigliano and Lesica, 1998). These hydric fields are too wet to permit burning in most years and, consequently, these areas were not included in our current evaluation of management practices although results of different management evaluations for these sites are presented elsewhere (Benson, 2003; Benson *et al.*, 2007). We collected data on 20 mesic fields in 2001 and 2002. We defined fields as separate easements or separate management units within an easement; fields ranged in size from 8.6 to 52.5 ha [\bar{x} = 18.5 ha \pm 2.5 (SE)], were dominated by herbaceous vegetation, generally grasses (Benson *et al.*, 2007), and were recognized as candidates for management by participating agencies. Fields were distributed among 15 easements, with 10 easements containing one and five easements containing two fields (these pairs of fields were an average of 558 m apart).

TREATMENTS

We randomly assigned 10 of 20 mesic fields to a spring burning and disking treatment. Burning of the 10 fields took place between 19 Apr. and 11 May 2001 and was originally planned to facilitate disking. Although disking was scheduled to be completed soon after burning in spring, wet conditions delayed disking until between 27 Jun. and 20 Jul. 2001. Seven fields received a strip-disking treatment, and disking was done with a single pass of a tandem disk to a depth of about 23 cm. Strips were disked about 20 m wide spanning the length of each treated field, typically close to the center of each field ($n = 5$) or on the field's edge ($n = 2$). Although entire fields were burned, only about 1 ha of each field was disked [\bar{x} = 1.12 ha \pm 0.16 (SE), range = 0.67–1.68 ha]; this corresponded to a mean of 7% of each field (SE = 2%, range = 3–15%). Only a small strip was disked to minimize disturbance to the warm-season grass plantings but provide a forb- and arthropod-rich area that could serve as a concentrated resource patch for birds throughout the field. Four burned fields were not disked because of logistical difficulties, and one unburned field was inadvertently disked. Because burned and unburned fields were similar in vegetation density and height of live vegetation at the time of disking (and therefore disking appeared equally effective in both; Benson *et al.*, 2007), the field that was unburned and disked was included

in analyses with those that had been burned and disked. Treated and untreated fields were distributed throughout the length of the IRC, and no two fields within the same easement received the same treatment.

BIRD SURVEYS

We surveyed birds four times between 17 May and 25 Jul. 2001 and again between 23 May and 30 Jul. 2002. We randomly placed four non-overlapping, 50-m-radius point-counts in each field and completed 5 min counts at these points between sunrise and 1000 h (Ralph *et al.*, 1995). Observers were trained before initiation of field work, rotated among sites to minimize bias and did not conduct surveys on mornings with high winds (>24 km/h) or rain. We recorded all birds identified visually or by song within the survey area, the method of identification (visual, song, call) and sex of each individual when possible (based on sexual dimorphism or singing). Birds identified visually were placed into 1 of 5 distance classes, 0–10 m, 11–20 m, 21–30 m, 31–40 m or 41–50 m, and birds flying over count circles were not included in analyses. Most birds (94%) were visually detected during surveys, and we did not estimate distance for individuals identified only by sound because we considered these estimates to be less reliable. We used the same point-count locations in both years, and counts from adjacent fields were >250 m apart. Given the relatively small area disked in treated fields, point-counts were generally located outside of the treated area; however, we expected these disked areas to function as concentrated resource patches where increased arthropod abundance and biomass would lead to increased bird use of the surrounding areas and, thus, have effects at the field level. Based on published territory sizes of bird species in the study area [*e.g.*, 7123 m² for dickcissel (latin names for all species are listed in Appendix), and 5261 m² for common yellowthroat; Schoener, 1968; Schartz and Zimmerman, 1971], we expected a response to occur within 100 m of disked areas and point-counts within treated fields were generally within this radius.

DATA ANALYSES

We calculated density of birds (males/ha) for common species (≥ 40 total observations), and all grassland and wetland species combined (males/ha; Poole *et al.*, 1992–2002; Vickery *et al.*, 1999) using program DISTANCE (Buckland *et al.*, 1993). For grassland and wetland species with ≥ 40 observations, we compared detectability models that incorporated a uniform distribution with cosine or simple polynomial adjustments, a half-normal model with cosine or hermite polynomial adjustments, and a hazard-rate model with cosine adjustments (Buckland *et al.*, 1993). Although our results focus specifically on mesic fields within our study area, we pooled observations from mesic and hydric fields in our study area to increase our sample size and generate detection functions for both field types. Where sample sizes permitted (*i.e.*, 40 observations per group), we examined differences in detectability functions among habitat types (mesic vs. hydric), treatments (disked, burned, untreated), years (2001 vs. 2002) and interactions of these variables. Models were ranked according to Akaike's Information Criterion for small sample sizes (AIC_c; Burnham and Anderson, 1998). We used density estimates produced by DISTANCE for subsequent analyses. For species with < 40 observations, we calculated density as average number of birds at a point divided by the area of the fixed-radius point count (males/ha; equivalent to a uniform detection function). Because we conservatively chose to use only visual detections in DISTANCE to ensure accuracy of distance estimates, for those species that required distance-related adjustments for detection probability (*see* RESULTS), we used the maximum of the DISTANCE and unadjusted (*i.e.*, including sound-only detections) estimates for each sampling round.

In addition to density, we were interested in the effect of treatments on the number of species using fields as well as the overall conservation priority of the species assemblage. Therefore, we used three metrics in addition to density: total number of species observed per field, total number of grassland and wetland species observed per field, and conservation value. Grassland and wetland birds included obligate and facultative species (Poole *et al.*, 1992–2002; Vickery *et al.*, 1999). Conservation value was determined using the density of each grassland- or wetland-associated species and its Partners in Flight (PIF) prioritization score for the Dissected Till Plains (Carter *et al.*, 2000; Fitzgerald and Pashley, 2000; Nuttle *et al.*, 2003; for PIF scores *see* acknowledgments):

$$\text{Conservation value} = \sum_{i=1}^S \text{density}_i \times \text{PIFscore}_i.$$

We used mixed-model Analysis of Variance (ANOVA) to test for effects of burning, disking, year, burn \times year, disk \times year, burn \times disk, and burn \times disk \times year on density, species richness and conservation value (SAS PROC MIXED; Littell *et al.*, 2006). In addition to these fixed effects, we treated field within treatment and field \times year within treatment as random effects to account for the 2 y of data collection and four sample rounds within years. We used the Kenward-Roger approximation for denominator degrees of freedom and, to account for potential heterogeneity of variance, we modeled residual variation among treatment \times year combinations (Littell *et al.*, 2006). We used plots of the residuals from these analyses to confirm that the normal distribution was appropriate. When data did not conform to a normal distribution, usually because of a large number of occasions when zero individuals were observed, we tested for the above treatment, year and interaction effects on presence of these rare species using generalized linear mixed models using a binomial distribution and logit link function, and with the same random effects listed above (SAS PROC GLIMMIX; Littell *et al.*, 2006).

Since burn treatments were applied to fields before the first season of data collection, we expected burn effects to appear as either a main effect (*i.e.*, burn effect in both years) or as a burn \times year interaction (*i.e.*, burn effect in only one year). However, disking was completed either late in the first field season (*i.e.*, after 3 or 4 survey rounds) or between the two field seasons. Consequently, vegetative responses of this treatment were not observed until the second year of our study and we expected bird responses to this treatment to appear as changes between years in disked relative to control fields (*i.e.*, disk \times year interaction). For those few cases where disking occurred before the fourth round of bird surveys, our observations and exploratory analyses confirmed that this treatment had little or no effect on the birds during that season. When significant interactions were present, we examined differences among burn \times year or disk \times year combinations with contrasts. The significance level was set at $P \leq 0.05$ for all analyses. To protect against Type II error, we did not adjust for multiple comparisons (Quinn and Keough, 2002; Moran, 2003). Likewise, because the cost of making Type II errors is high, particularly when dealing with management practices that may adversely affect populations, results with $0.10 \geq P \geq 0.05$ were considered marginally significant (Cousens and Marshall, 1987; Mapstone, 1995; Quinn and Keough, 2002).

RESULTS

DENSITY ESTIMATION

In 2001 and 2002, we observed 25 bird species in mesic fields (Appendix). Five grassland or wetland species were sufficiently abundant to calculate density using DISTANCE (Appendix, Table 1). Uniform functions were best for modeling detectability of all bird

species, and no adjustments to the detection function were made for red-winged blackbirds or dickcissels. The best detection model for common yellowthroats incorporated a habitat-type effect (differences in detection between mesic and hydric fields), with detection functions in both habitats incorporating one cosine adjustment. For sedge wrens, the best detection model incorporated two cosine adjustments. For American goldfinches, the best detection model incorporated a year effect with no adjustments to the uniform function in 2001 data, and one adjustment in 2002 data. For red-winged blackbirds and dickcissels, 100% of the variance associated with density estimates was related to encounter rate, whereas this value varied from 29 to 49% for the other species. Because of a low sample size for grasshopper sparrows, we did not adjust density for detectability and present density as mean number of individuals observed per ha which is equivalent to a model with a uniform detection function. Likewise, because the best detectability model for combined grassland and wetland species was uniform with no adjustments, we calculated density of this group as the sum of species-specific density estimates (both adjusted and unadjusted).

DENSITY, SPECIES RICHNESS AND CONSERVATION VALUE

The burn \times disk and burn \times disk \times year interactions were not significant for any of the variables we evaluated (all $P > 0.05$). The burn \times year interaction was only significant for density of common yellowthroats, and the disk \times year interaction was only significant for combined density of all grassland and wetland species and conservation value (Tables 1 and 2; Figs. 1 and 2). Density of common yellowthroats was lower in burned than unburned fields in 2001 but not 2002 [difference = 0.38 males/ha \pm 0.12 (SE), $F_{1,38} = 10.27$, $P = 0.003$ and 0.07 ± 0.10 (SE), $F_{1,19} = 0.49$, $P = 0.491$, respectively; Fig. 1]. Additionally, there was a significant decrease in density between years in unburned but not burned fields [difference = 0.28 ± 0.11 (SE), $F_{1,48} = 6.73$, $P = 0.013$ and -0.04 ± 0.04 (SE), $F_{1,9} = 0.74$, $P = 0.409$, respectively].

Density of all grassland and wetland species did not differ between fields that did or did not receive a disking treatment in 2001, but the difference was marginally significant in 2002 [difference = 0.30 ± 0.38 (SE), $F_{1,19} = 0.63$, $P = 0.439$ and -0.70 ± 0.39 (SE), $F_{1,20} = 3.14$, $P = 0.091$, respectively; Fig. 2A]. The density of this group decreased between years in fields with no disking treatment but displayed a non-significant increase in those that did receive disking [difference = 0.57 ± 0.26 (SE), $F_{1,132} = 4.99$, $P = 0.027$ and -0.42 ± 0.34 (SE), $F_{1,126} = 1.50$, $P = 0.224$, respectively]. Conservation value did not differ between disked and undisked fields in 2001, but differed marginally in 2002 [difference = 8.18 ± 9.04 (SE), $F_{1,11} = 0.83$, $P = 0.385$ and -14.62 ± 7.57 , $F_{1,9} = 3.73$, $P = 0.083$, respectively; Fig. 2B]. Between years, there was a marginally significant decrease in conservation value in undisked fields, but a marginal increase in disked fields [difference = 10.15 ± 5.45 (SE), $F_{1,17} = 3.46$, $P = 0.079$ and -12.65 ± 7.16 (SE), $F_{1,16} = 3.13$, $P = 0.096$, respectively]. There was a significant effect of burning on density of red-winged blackbirds and all grassland and wetland species combined; density was lower in burned than unburned fields [difference = 0.57 males/ha \pm 0.26 (SE), and 0.72 males/ha \pm 0.33 (SE), respectively; Tables 1 and 2, Fig. 3]. There were no differences between fields with and without a disking treatment for density of individual species, species richness or conservation value, nor were there differences in species richness or conservation value between burned and unburned fields. However, there were significant differences between years in density of red-winged blackbirds, occurrence of American goldfinches, and richness of grassland and wetland species (Tables 1 and 2). There was also a marginally significant difference between years in occurrence of grasshopper sparrows.

TABLE 1.—Density (mean and standard error; males/ha) of common bird species, species richness (species/field) and conservation value in mesic riparian grasslands in east-central Iowa, USA, 2001–2002

Variable	2002													
	2001					Disked					Undisked			
	Unburned (n = 10)	Burned (n = 10)	\bar{x}	SE ^a		Unburned (n = 1)	Burned (n = 6)	\bar{x}	SE ^a		Unburned (n = 9)	Burned (n = 4)	\bar{x}	SE ^a
Density														
Red-winged blackbird	1.19	0.24	0.64	0.13	0.87	—	0.62	0.24	0.24	0.92	0.21	0.24	0.06	
Dickcissel	0.70	0.15	0.68	0.17	0.56	—	0.93	0.22	0.22	0.75	0.17	0.58	0.12	
Common														
yellowthroat ^b	0.43	0.11	0.06	0.02	0.00	—	0.15	0.09	0.09	0.15	0.07	0.03	0.03	
Sedge wren ^b	0.28	0.16	0.25	0.11	0.26	—	0.09	0.09	0.09	0.43	0.23	0.45	0.30	
American goldfinch ^b	0.10	0.03	0.10	0.03	0.00	—	0.07	0.04	0.04	0.07	0.04	0.03	0.03	
Grasshopper sparrow	0.00	0.00	0.06	0.06	0.00	—	0.32	0.21	0.21	0.02	0.01	0.06	0.04	
Grassland/wetland														
species ^{c, d}	2.76	0.31	1.90	0.18	1.77	—	2.25	0.13	0.13	2.36	0.36	1.41	0.40	
Species richness														
Grassland/wetland														
species ^c	5.00	0.39	5.00	0.33	4.00	—	4.50	0.43	0.43	4.11	0.51	4.00	0.82	
All species	5.80	0.39	5.70	0.33	4.00	—	5.67	0.76	0.76	5.00	0.67	4.00	0.41	
Conservation value ^e	52.33	5.96	38.07	4.60	34.25	—	46.95	2.89	2.89	46.85	7.33	30.76	8.43	

^a Standard error estimates are based on variation within point estimates and do not explicitly account for detection probability

^b Density corrected for detectability using program DISTANCE (Buckland *et al.*, 1993)

^c Obligate and facultative grassland and wetland bird species (Poole *et al.*, 1992–2002; Vickery *et al.*, 1999)

^d Includes estimates corrected for detectability and uncorrected estimates

^e Conservation value = $\sum_{i=1}^S \text{density}_i \times \text{PIFScore}_i$

TABLE 2.—Statistics for effects of burning, disking, and year on bird density (males/ha) or presence, species richness (species/field) and conservation value in mesic riparian grasslands in east-central Iowa, USA, 2001–2002

Variable	Burn			Disk			Year			Burn × Year			Disk × Year			
	F	P		F	P		F	P		F	P		F	P		
Density/presence																
Red-winged blackbird ^a	4.66	0.047	0.37	0.555	0.033	4.68	0.02	0.890	0.198	0.02	0.890	1.69	0.103	0.568	0.908	0.688
Dickcissel ^a	0.01	0.909	0.01	0.927	0.220	1.62	0.32	0.579	0.103	0.32	0.579	2.99	0.35	0.568	0.908	0.688
Common yellowthroat ^a	6.17	0.024	0.32	0.582	0.040	4.42	7.10	0.011	0.568	7.10	0.011	0.35	0.01	0.908	0.688	0.688
Sedge wren ^b	0.01	0.920	0.69	0.416	0.813	0.06	1.32	0.253	0.908	1.32	0.253	0.01	0.01	0.908	0.688	0.688
American goldfinch ^b	0.54	0.472	0.34	0.567	0.007	7.51	0.34	0.562	0.688	0.34	0.562	0.16	0.16	0.688	0.688	0.688
Grasshopper sparrow ^b	1.59	0.224	0.00	0.962	0.063	3.89	0.00	0.995	0.496	0.00	0.995	0.48	0.48	0.496	0.496	0.496
Grassland/wetland species ^{a,c}	4.85	0.043	0.38	0.120	0.729	0.12	0.00	0.959	0.019	0.00	0.959	5.67	5.67	0.019	0.019	0.019
Species richness ^a																
Grassland/wetland species ^{a,c}	0.17	0.681	0.17	0.684	0.049	4.18	0.31	0.586	0.131	0.31	0.586	2.69	2.69	0.131	0.131	0.131
All species ^a	0.10	0.760	0.18	0.679	0.112	2.79	0.13	0.727	0.327	0.13	0.727	1.03	1.03	0.327	0.327	0.327
Conservation value ^{a,d}	2.44	0.136	0.20	0.662	0.795	0.07	0.05	0.825	0.023	0.05	0.825	7.25	7.25	0.023	0.023	0.023

^a Effects of burn, disk, year, burn × year, and disk × year were tested with repeated-measures ANOVA (SAS PROC MIXED; Littell *et al.*, 2006; 1, 10–124 df)

^b Effects of burn, disk, year, burn × year and disk × year were tested with generalized linear mixed models using a binomial distribution and logit link function (SAS PROC GLIMMIX; Littell *et al.*, 2006; 1, 16–154 df)

^c Obligate and facultative grassland and wetland bird species (Poole *et al.*, 1992–2002; Vickery *et al.*, 1999)

^d Conservation value = $\sum_{j=1}^5 \text{density}_j \times \text{PIFscore}_j$

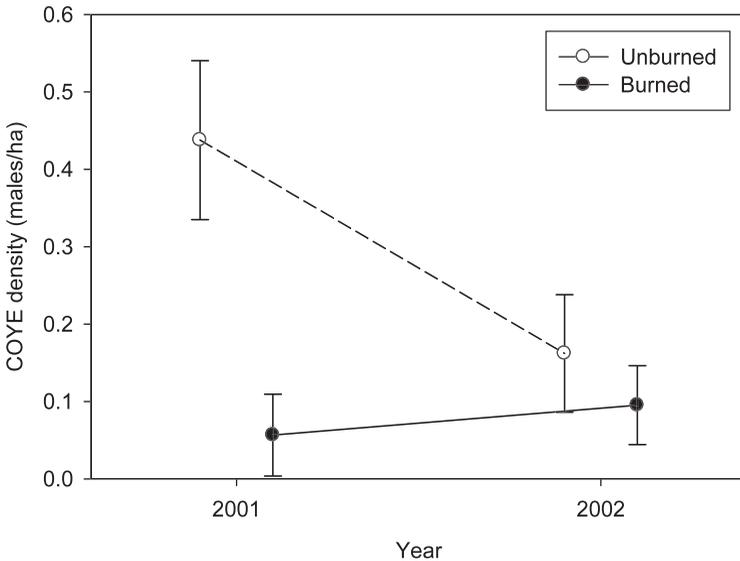


FIG. 1.—Estimated density (males/ha; \pm standard error) of common yellowthroats (COYE) in burned and unburned mesic riparian grasslands in east-central Iowa, USA, 2001–2002

DISCUSSION

The bird species and densities found in restored grasslands of the Iowa River Corridor are similar to those found in other studies of grassland birds in Iowa and throughout the Midwest (*e.g.*, Cink and Lowther, 1989; Johnson and Schwartz, 1993; Patterson and Best, 1996; Fletcher and Koford, 2002; Murray and Best, 2003). However, some grassland obligate species commonly observed in this region, such as bobolinks and meadowlarks, were rare in this study for unknown reasons (Appendix). Nonetheless, these restored grasslands were occupied by a number of grassland or wetland species, including eight that are considered moderate or high conservation priorities (Fitzgerald and Pashley, 2000; Nuttle *et al.*, 2003; Benson *et al.*, 2006).

DENSITY, SPECIES RICHNESS AND CONSERVATION VALUE

Previous studies of the effects of burning on grassland birds have had variable results with species- and region-specific differences in responses to this management practice (Reinking, 2005). In this study, we found decreases in density of grassland and wetland species, and density of two species, common yellowthroats and red-winged blackbirds following burning. Past studies found little effect of burning on density of all species combined (Madden *et al.*, 1999), or decreases in density (Zimmerman, 1992; Robel *et al.*, 1998). Most of the observed decrease in density of all grassland and wetland species combined in our study was driven by changes in red-winged blackbird density, the most abundant species on our study area. Indeed, the estimated difference in blackbird density between unburned and burned fields accounted for 78% of the difference for combined density of all grassland and wetland species. In contrast, some past studies have found no effect (Herkert, 1994) or increases in red-winged blackbirds after burning (Zimmerman, 1992); however, others have found decreases (Robel *et al.*, 1998). Consistent with our results from the year fields were burned (2001), past research suggests that common yellowthroats are negatively impacted by

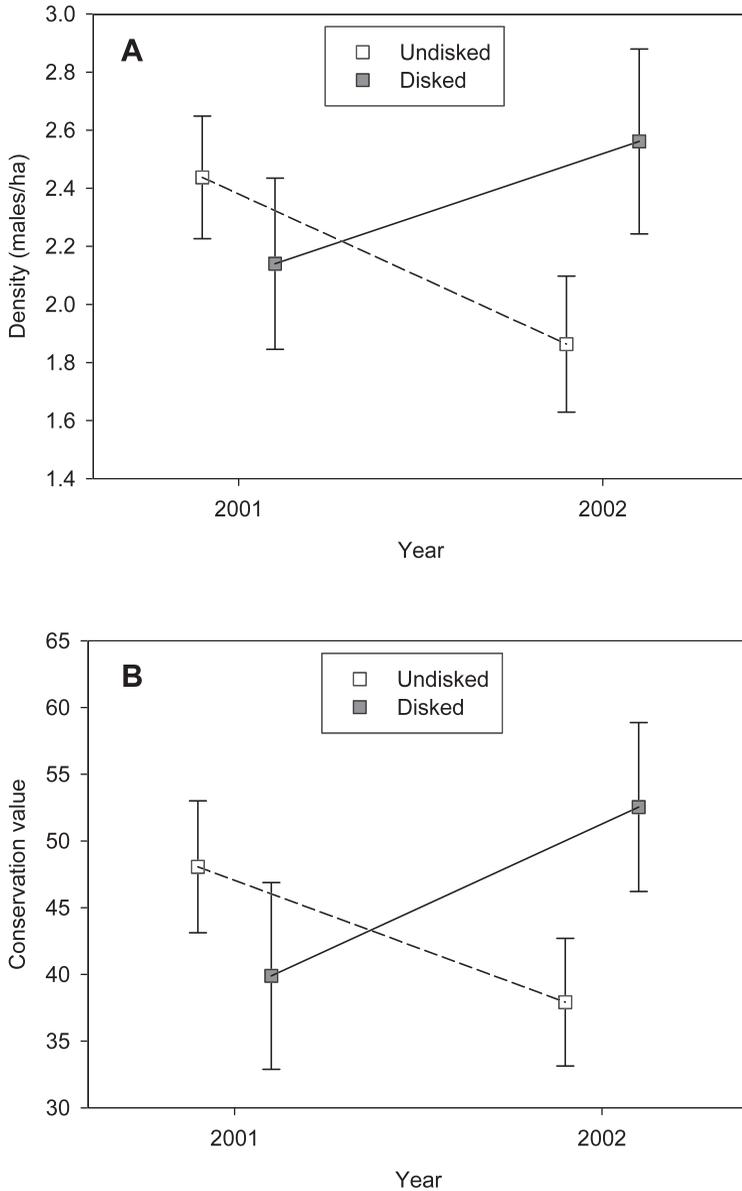


FIG. 2.—Estimated (A) density (males/ha; \pm standard error) of all grassland- and wetland-dependent species combined, and (B) avian conservation value (density weighted by Partners in Flight prioritization scores) in mesic riparian grasslands that did or did not receive a diking treatment in east-central Iowa, USA, 2001–2002

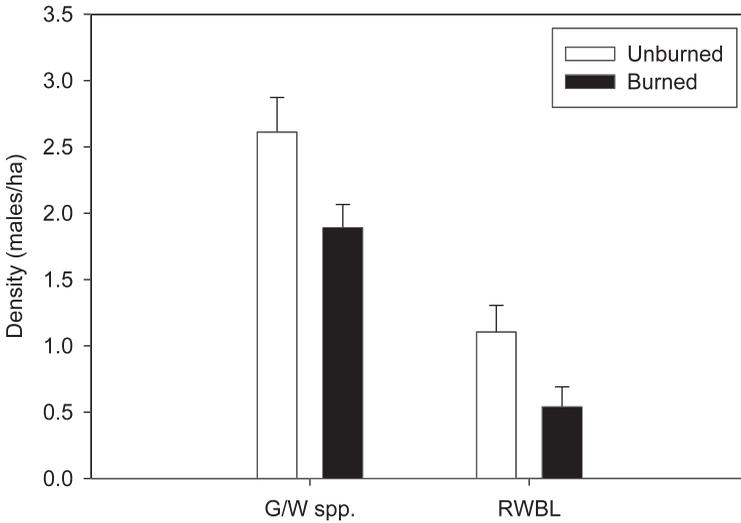


FIG. 3.—Estimated densities (males/ha; \pm standard error) of all grassland- and wetland-dependent species combined (G/W spp.) and red-winged blackbirds (RWBL) in burned and unburned mesic riparian grasslands in east-central Iowa, USA, 2001–2002

burning of their habitat (Zimmerman, 1992; Herkert, 1994; Madden *et al.*, 1999). These differences between burned and unburned areas may have persisted into the second year, but weather-related changes in density between years, with 2002 drier than 2001 (National Oceanic and Atmospheric Administration, 2001, 2002), may have diluted any effect of burn treatments in 2002.

Density or occurrence of other species and species richness were unaffected by burning in this study. This is consistent with previous research on dickcissels (Zimmerman, 1992; Swengel, 1996; Applegate *et al.*, 2002), and sedge wrens (Schramm *et al.*, 1984; Zimmerman, 1992; Herkert, 1994). However, non-significant decreases in density of sedge wrens with burning have been observed in past studies (Herkert, 1994; Robel *et al.*, 1998). Likewise, decreases in density of dickcissels and American goldfinches have been observed (Zimmerman, 1992; Robel *et al.*, 1998; Fuhlendorf *et al.*, 2006). Grasshopper sparrows were unaffected by burning in this study, but have been negatively affected (Huber and Steuter, 1984; Johnson, 1997) and positively affected by burning in other studies (Herkert, 1994; Swengel, 1996; Madden *et al.*, 1999; Fuhlendorf *et al.*, 2006). Similarly, previous studies have found both increases (*e.g.*, Madden *et al.*, 1999) and decreases in species richness with burning (both total and grassland-core species; Zimmerman, 1992). However, these studies evaluated well-established grasslands in more grass-dominated landscapes, in regions with more grassland species, and likely had greater vegetative diversity including cover of woody vegetation. Consequently, the total number of grassland species observed in these studies relative to ours was relatively high and there were fewer species present to respond to fire in our study.

Although two species declined with burning, the unchanged conservation value suggests there was not a shift from high- to low-conservation-priority species, or a measurable impact on density of the highest-priority species. Again, this is because burning most impacted red-winged blackbirds, a low-priority species. The mechanism of the burn-related impacts on blackbirds and common yellowthroats was likely through modification of vegetation

structure rather than depression of food resources. Burning decreased the cover and height of standing dead vegetation and cover and depth of litter but had few effects on potential arthropod food resources in the burn year, and there were no significant differences in arthropod abundance or vegetation structure during the second breeding season (Benson *et al.*, 2007). Red-winged blackbirds nest early and are dependent on residual vegetation relative to other species, and density of both blackbirds and yellowthroats was positively related to litter depth (Benson, 2003). However, the effects of burning on height and cover of residual vegetation and litter depth in 2001 were not observed in 2002 (Benson *et al.*, 2007), so the persistence of burn effects on red-winged blackbirds and combined grassland and wetland species in 2002 is surprising but may relate to site fidelity or real but undetected differences in habitat or food resources in 2002.

Previous research on breeding bird responses to disking is limited. In Texas, abundance and diversity of nongame birds were greater on an area managed for northern bobwhites (including disking, tree cutting, grazing exclosures, brush piles and forb plantings) than a control area (Webb and Guthery, 1983). On our study area, disking led to a change in vegetation structure and composition (including decreased residual vegetation and grass cover and increased forb cover) and an increase in the abundance and biomass of arthropods (Benson *et al.*, 2007). Associated with this change, fields with a disking treatment held a greater density of grassland and wetland birds between 2001 and 2002, although there were no significant differences in density or occurrence of individual species or species richness. Although not significant, these fields had greater numbers of dickcissels and grasshopper sparrows than untreated fields. Because these two species are among the highest priority species for conservation, this led to increased conservation value on disked relative to undisked fields. The results of disking treatments were likely less pronounced than for burning because of the small size of treatments relative to the field size, and because of the strip configuration. Hydric fields with block-shaped disking treatments showed more pronounced increases in density, particularly for dickcissels (Benson, 2003). For both the mesic fields included in this study and the hydric fields, strip-shaped plots distributed the affected area throughout a field and may have diffused rather than concentrated any potential effects, although disking in strips likely better preserves habitat for dense-habitat specialists like common yellowthroats and sedge wrens (Benson, 2003).

The reason that disking led to increased post-treatment density was likely related to changes in habitat and food availability; disking decreased litter, shifted treated areas from grass- to forb-dominated, and increased the abundance and dry biomass of potential arthropod food resources (Benson *et al.*, 2007). For dickcissels, the apparent positive response in mesic fields (and hydric fields; Benson, 2003) is probably related to the increase in forb cover, a habitat variable of known importance to this species (Zimmerman, 1966; Temple, 2002), and the increase in food resources (Benson, 2003). The decreased vegetation density created by disking, if done at a larger scale, would possibly also benefit other species that prefer relatively sparse herbaceous vegetation such as grasshopper sparrows, although species dependent on dense herbaceous vegetation such as common yellowthroats and sedge wrens would likely decline (Fletcher and Koford, 2002; Benson, 2003; Murray and Best, 2003).

Irrespective of treatment effects, there were also between-year changes in density or occurrence of red-winged blackbirds, common yellowthroats, American goldfinches, grasshopper sparrows and grassland-and-wetland-bird species richness. These changes were likely caused by weather differences between the 2 y, with 2001 being relatively wet and 2002 relatively dry (National Oceanic and Atmospheric Administration, 2001, 2002). These types

of inter-annual changes in weather and associated changes in birds are fairly common in grassland systems (e.g., Ahlering *et al.*, 2009). At our study area, these weather differences led to decreases in vegetation height, vegetation cover and arthropod abundance (irrespective of treatment; Benson *et al.*, 2007). Thus, in 2002 fields were less suitable for species dependent on dense vegetation, such as red-winged blackbirds and common yellowthroats but were more suitable for grasshopper sparrows.

MANAGEMENT IMPLICATIONS

Although necessary for controlling invasion of woody plants, frequent burning may have negative effects on bird and arthropod communities. To minimize these effects, burning in most cases should be done at ≥ 3 y intervals. However, as there is great spatial and temporal variation in susceptibility of fields to invasion of woody vegetation, managers may need to adjust burn frequencies in some situations to address these concerns. Although we did not find positive responses to burning for any species, past research suggests that fire can be used to effectively manage habitat for some species, including grasshopper sparrows (e.g., Fuhlendorf *et al.*, 2006). Disking had short-term effects on density of all grassland and wetland bird species combined and overall conservation value but no significant effect on density of individual species or species richness. Disking may be effective for increasing habitat quality for species dependent on relatively open or forb-dominated grasslands. However, because of the effort often taken to establish native-grass plantings on mesic fields, extensive disking should be avoided and alternative methods for increasing forb and arthropod abundance should be used in conjunction with any disking treatments.

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APPENDIX.—Species observed, number of detections of males of each species and total number of observations used to model detection probability and generate density estimates for point counts in mesic riparian grasslands in east-central Iowa, 2001–2002. Species are listed in order of total abundance (both sexes combined)

Species	Males	Total observations used for DISTANCE ^a
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	404	1061
Dickcissel ^b (<i>Spiza americana</i>)	364	618
Common yellowthroat ^b (<i>Geothlypis trichas</i>)	71	96
Sedge wren ^b (<i>Cistothorus platensis</i>)	56	64
American goldfinch (<i>Spinus tristis</i>)	36	107
Grasshopper sparrow ^b (<i>Ammodramus savannarum</i>)	37	
Savannah sparrow (<i>Passerculus sandwichensis</i>)	18	
Song sparrow (<i>Melospiza melodia</i>)	12	
Mourning dove (<i>Zenaida macroura</i>)		
Ring-necked pheasant (<i>Phasianus colchicus</i>)	7	
Indigo bunting (<i>Passerina cyanea</i>)	10	
Baltimore oriole (<i>Icterus galbula</i>)	4	
Bobolink ^b (<i>Dolichonyx oryzivorus</i>)	3	
Common grackle (<i>Quiscalus quiscula</i>)		
Eastern kingbird ^b (<i>Tyrannus tyrannus</i>)		
House sparrow (<i>Passer domesticus</i>)	2	
Killdeer (<i>Charadrius vociferous</i>)		
Brown-headed cowbird (<i>Molothrus ater</i>)	1	
Red-headed woodpecker (<i>Melanerpes erythrocephalus</i>)		
American robin (<i>Turdus migratorius</i>)		
Blue-winged teal (<i>Anas discors</i>)		
Eastern meadowlark ^b (<i>Sturnella magna</i>)	1	
Henslow's sparrow ^b (<i>Ammodramus henslowii</i>)	1	
Northern cardinal (<i>Cardinalis cardinalis</i>)		
Vesper sparrow (<i>Poocetes gramineus</i>)		

^a The observations used to model detection probability as a function of distance (Buckland *et al.*, 1993) included visually detected males in both the mesic fields used for this study and hydric fields used in other research (Benson *et al.*, 2006, 2007)

^b Species considered to be of moderate or high conservation priority (Fitzgerald and Pashley, 2000; Nuttle *et al.*, 2003; Benson *et al.*, 2006)