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DENSITY AND DIVERSITY OF OVERWINTERING BIRDS IN MANAGED FIELD BORDERS IN MISSISSIPPI

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ABSTRACT.—Grassland bird populations are sharply declining in North America. Changes in agricultural practices during the past 50 years have been suggested as one of the major causes of this decline. Field-border conservation practices encouraged by the U.S. Department of Agriculture's National Conservation Buffer Initiative meet many of the needs of sustainable agriculture and offer excellent opportunities to enhance local grassland bird populations within intensive agricultural production systems. Despite the abundant information on avian use of, and reproductive success in, strip habitats during the breeding season, few studies have examined the potential value of field borders for wintering birds. We planted 89.0 km of field borders (6.1 m wide) along agricultural field edges on one-half of each of three row crop and forage production farms in northeastern Mississippi. We sampled bird communities along these field edges during February–March 2002 and 2003 using line-transect distance sampling and strip transects to estimate density and community structure, respectively. We used Program DISTANCE to estimate densities of Song (*Melospiza melodia*), Savannah (*Passerculus sandwichensis*), and other sparrows along bordered and non-bordered transects while controlling for adjacent plant community. Greater densities of several sparrow species were observed along most bordered transects. However, effects of field borders differed by species and adjacent plant community types. Diversity, species richness, and relative conservation value (a weighted index derived by multiplying species-specific abundances by their respective Partners in Flight conservation priority scores) were similar between bordered and non-bordered edges. Field borders are practical conservation tools that can be used to accrue multiple environmental benefits and enhance wintering farmland bird populations. Provision of wintering habitat at southern latitudes may influence population trajectories of short-distance migrants of regional conservation concern. *Received 4 October 2004, accepted 13 June 2005.*

Grassland birds are one of the most sharply declining groups of birds in North America (Knopf 1994, Herkert 1995, Peterjohn and Sauer 1999). Grassland birds experienced a 1.1% per year decline from 1966 to 2002 in the U.S. and a 2.3% per year decline in the southeastern (U.S. Fish and Wildlife Service Region 4) U.S. (Sauer et al. 2003). Many grassland species are now associated closely with agricultural production systems because most (>80%) of the native grasslands in North America have been converted to other uses (Samson and Knopf 1994, Noss et al. 1995, Hunter et al. 2001), principally agricultural production. Although agriculture facilitated range expansions for several grassland species through clearing of forested land (Askins 1999, Arcese et al. 2002), several correlative studies now suggest agricultural intensification as a leading cause of decline for most grassland birds (Vickery et al. 1999, Blackwell and Dolbeer 2001, Murphy 2003).

Numerous changes in production agriculture have occurred within the past 50 years, hastening the decline of grassland birds. Most notable has been the shift from diversified, small-scale farms to large-scale, highly specialized, chemical- and capital-intensive monoculture farming systems. This shift has resulted in the loss of field edge, fencerow, and other non-crop herbaceous communities (Rodenhouse et al. 1993, Warner 1994, Koford and Best 1996). Recent changes in Conservation Reserve Program (CRP) enrollment options (continuous sign-up) now permit partial field enrollments, thus encouraging conservation-oriented production practices (e.g., conservation buffers) without removing an entire field from production. Conservation buffer practices, available in several Farm Bill conservation programs, offer valuable opportunities to create habitat for grassland birds within intensively farmed landscapes. Grassed waterways, contour grass strips, filter strips, riparian buffers, crosswind trap strips, windbreaks, and shelterbelts are conservation buffer practices used to reduce soil erosion (Dillaha et al. 1989), diminish herbicide and nutrient runoff

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into wetlands (Daniels and Williams 1996, Webster and Shaw 1996), and provide wildlife habitat (Bryan and Best 1991, Puckett et al. 1995, Marcus et al. 2000).

Within intensively farmed landscapes, conservation buffers are increasingly the only available semi-permanent grasslands for nesting birds (Warner 1994, Koford and Best 1996). Field borders, defined as intentionally managed herbaceous plant communities along crop field edges to provide environmental and wildlife habitat benefits, are another type of conservation buffer practice. However, unlike conservation buffer practices specifically designed to filter sediments, field borders may be more broadly applied than simply along downslope edges of fields. Field borders may be established where other conservation buffer practices do not meet eligibility criteria, are not cost effective or practical, or are not desired by the producer.

Although herbaceous strip habitats may have limited value as nesting cover because reproductive success is low (Basore et al. 1986, Bryan and Best 1994, Camp and Best 1994), field borders may provide important wintering habitat for numerous short-distance migrants that winter in the southern U.S. Several studies have documented grassland bird use and reproductive success within other agricultural edge habitats (Best 1983, Johnson and Beck 1988, Best et al. 1990, Sparks et al. 1996); however, no studies have addressed explicitly the importance of field borders. Furthermore, most studies of grassland birds have been conducted during the breeding season (Rodenhouse et al. 1993, Herkert et al. 1996, Ryan et al. 1998, Peterjohn 2003). Only Marcus et al. (2000), in North Carolina, addressed the benefits of field borders to wintering birds. Ryan et al. (1998) noted the lack of data detailing winter bird use of CRP fields, and wintering habitat requirements and ecology of most grassland birds are poorly known (Vickery et al. 1999). Herkert et al. (1996) and Peterjohn (2003) contend that the paucity of information on wintering grassland birds limits our ability to develop effective conservation strategies for them.

Our objectives were to estimate the effects of field borders on grassland bird density and diversity during the winter in northeastern Mississippi. We also characterized avian community structure in bordered and non-bordered fields, relative to adjacent plant communities.

METHODS

Study area.—Our study was conducted on three privately owned farms in Clay and Lowndes counties (88° 32′ W, 33° 34′ N), located within the Black Prairie physiographic region of northeastern Mississippi. All farms in the region have a history of agricultural use, most having actively produced crops for >50 years. Primary agricultural production included soybeans (*Glycine max*), corn (*Zea mays*), forage, and livestock. Most row-crop fields on all three study farms were tilled in late fall in preparation for spring planting. The farms were selected based on similarities in cropping practices, landscape composition (approximately 60–80% row crop), soil associations, and landowner cooperation.

Grasslands on each farm consisted predominantly of perennial, exotic, cool-season forage grasses (tall fescue, *Festuca arundinacea*) and warm-season exotics (Bermudagrass, *Cynodon dactylon*; and Bahia grass, *Paspalum notatum*; Smith 2004). Small remnant and reintroduced stands of native grasses (big bluestem, *Andropogon gerardii*; little bluestem, *Schizachyrium scoparium*; and broomsedge, *A. virginicus*) were scattered throughout each farm. Fencerows, drainage ditches, and contour filter strips were dominated by tall fescue and Johnson grass (*Sorghum halepense*). Periodically disturbed areas contained early seral-stage grasses and forbs (paspalum, *Paspalum* spp.; panicgrass, *Panicum* spp.; giant ragweed, *Ambrosia trifida*; annual marshelder/sumpweed, *Iva annua*; Johnson grass; and goldenrod, *Solidago* spp.). Wooded areas were predominantly oak (*Quercus* spp.), green ash (*Fraxinus pennsylvanica*), maple (*Acer* spp.), hickory (*Carya* spp.), sugarberry (*Celtis laevigata*), and eastern redcedar (*Juniperus virginiana*; Smith 2004).

During early spring 2000, we established experimental field borders (6.1 m wide) along row-crop field margins (fencerows, drainage ditches, access roads, and contour filter strips) on one-half of each farm. Mean field size was 26.9 ha (n = 37, range = 2.9–146.9) and mean percentage of the field area given over to field borders was 6.0% (range = 0.5–15.3).

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Overall, field borders (54.3 ha) composed 0.8–1.3% of the land area of bordered sections of each farm. In return, producers were paid an initial $247.10/ha sign-up bonus with a $123.55/ha/year rental rate for land dedicated to field borders. Producers were required not to mow, treat with herbicide, or disk field borders during the duration of the study. Initially, field borders were seeded with a Kobe lespedeza (*Lespedeza striata*) and partridge pea (*Chamaecrista fasciculata*) mix at rates of 11.2 and 3.4 kg/ha, respectively. Severe drought during the 2000 growing season resulted in poor plant growth; therefore, field borders were re-seeded in early 2001. Despite these two attempts to establish field borders, most re-seeded naturally from seed present within the seed bank. During the 2001 growing season, the most common species occurring in field borders were morning-glory (*Ipomoea* spp.), crabgrass (*Digitaria ciliaris*), Johnson grass, hemp sesbania (*Sesbania exaltata*), yellow nutsedge (*Cyperus esculentus*), and ragweed (*Ambrosia* spp.; PJB unpubl. data).

Data collection.—We used line-transect distance sampling and strip-transect sampling to estimate density (birds/ha) and diversity, respectively, of wintering grassland birds. Georeferenced aerial photos and Geographic Information System (GIS) land cover maps were used to delineate field edges. Field edges were divided into 200-m-long sampling units (transects), with the beginning point of each transect situated so that the vegetation type on the non-agricultural side of the transect was homogenous for the length of the transect. The centerline of each transect was situated along the interface of the original (before field borders implemented) row-crop field and adjacent plant community interface. Transects located adjacent to roadways or those that contained field borders that were disturbed inadvertently (e.g., disked, mowed, sprayed) by producers were not included within this sampling frame. Our sampling frame consisted of 110 bordered and 82 non-bordered transects. We then classified each transect based upon combinations of (1) bordered (T) and non-bordered (C) practices on the agricultural side, and by (2) vegetation type (woody [W], herbaceous [G]) and (3) width (strip [S], ≤30 m of continuous vegetation type; block [B], >30 m of continuous vegetation type) on the non-agricultural side. This classification scheme produced eight treatment combinations: TGB, CGB, TGS, CGS, TWB, CWB, TWS, CWS.

During winter 2002, we conducted a pilot study to estimate encounter rates along potential transects within each treatment combination. We concluded that ≥10 200-m transects/treatment combination would provide sufficient numbers of encounters to estimate detection functions for several common species and most guilds. Because the amount and structure of grassland and woodland habitats differed dramatically among farms, we were not able to sample all eight treatment combinations within any one farm. Therefore, we randomly selected 10–11 transects for each treatment combination from the population of transects available across all three farms except for the CGB treatment combination. Only seven transects were available for the CGB treatment combination and all were used. We sampled the same transects in both years of study (2002, 2003), with the exception of two TWB transects with field borders that were accidentally disked by the producer after year 1. These two transects were replaced by two other randomly selected TWB transects on the same farm.

The field border treatment was assigned randomly to one-half of each of the three farms. Field borders were not assigned randomly to individual transects, but rather bordered transects were selected randomly from the population of all bordered transects across all farms. Thus, our study was observational with replication. Additionally, distance sampling assumes implicitly that transects are placed randomly relative to the distribution of objects (birds) within a study area for justifiable extrapolation of sample statistics to the population (Buckland et al. 2001). Our objectives were not to estimate study area density, but rather densities of birds inhabiting or using designated portions (field borders and adjacent communities) of a study area.

We marked transects with flagging at the beginning, end, and at 20-m intervals along each transect to allow observers to monitor their rate of speed and location during the surveys. Sampling was conducted by two observers each year. Within each sampling interval, we randomly assigned transects to an observ-

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er; within each farm, however, we sampled transects in a systematic order to reduce travel time between transects. Each observer sampled 3–8 transects/morning/farm. Transect order within each farm was alternated among repetitions (i.e., transects were sampled in reverse order during the second repetition). Moreover, following completion of the first repetition, observers switched transect schedules. We sampled all transects three times in 2002 and twice in 2003 during February–March, with approximately 3–4 weeks between visits to the same transect (Freemark and Rogers 1995).

We walked at approximately 20 m/min along each transect and made intermittent stops to record the number of individuals and species seen or heard on each side of the transect line. Transects were sampled between 07:00 and 10:00 (CST) with wind speeds <16 km/hr. We assigned observations into one of four perpendicular distance bands (0.0–9.9, 10.0–19.9, 20.0–29.9, and >30.0 m) on each side of the transect line. The first distance band in bordered transects contained the field border, whereas the first distance band for non-bordered transects was the first 9.9 m of crop field. To reduce observer bias, additional observers ($n = 2$) were trained by PJB and SJD prior to sampling (Kepler and Scott 1981, Smith 1984). Each observer was trained in sampling protocol, bird identification (by sight and sound), and distance estimation (Scott et al. 1981). Furthermore, we assumed that observers were able to detect all birds on the transect line, detect birds at their initial location, and assign observations to correct distance categories (Buckland et al. 2001). PJB collected data during both years of the study, whereas each of the additional observers collected data for only 1 year.

Density estimation.—Because avian detection probabilities (Bibby and Buckland 1987, Buckland et al. 2001) and assemblages (Best 1983, Shalaway 1985, Best et al. 1990, Sparks et al. 1996) differ with plant community structure and composition, we decided *a priori* to develop independent detection functions for the agricultural and non-agricultural sides of transects. On the agricultural side, we developed detection functions for bordered (T) and non-bordered (C) transects. We also stratified the non-agricultural side of transects based on vegetation type (W, G) and width (S, B). Thus, we developed six detection functions (T, C, GS, GB, WS, WB) for each species or guild. We tested pooling robustness (Burnham et al. 1980, Buckland et al. 2001) of the six functions by comparing Akaike's Information Criterion (AIC; Akaike 1974) values between distance data fitted to pooled (e.g., using all observations on the non-agricultural side of wooded transects) and unpooled functions (e.g., wood strip and wood block observations on the non-agricultural side of wooded transects). When comparing functions from the same set of data, a greater AIC value of a pooled model—relative to the sum of AIC values of the unpooled models—indicates that individual models fit the data better than a pooled model (Buckland et al. 2001). Testing of model robustness was conducted only between models on the agriculture sides of transects (T, C) and between models within vegetation types on the non-agricultural sides of each transect (i.e., wood block and wood strip within woods). We assumed that detection functions did not differ between or within years; thus, we pooled observations across years and repetitions. Although some species occasionally occurred in loose aggregations, we treated all individuals as unique, independent observations.

We used Program DISTANCE (Thomas et al. 1998) to model detection functions for species and guilds for which we recorded >60 observations within each of the six habitat types (Buckland et al. 2001). Only Song (*Melospiza melodia*) and Savannah (*Passerculus sandwichensis*) sparrows were detected often enough within these six habitat types to develop species-specific detection functions. We also developed detection functions for an ''other sparrows'' group (hereafter other sparrow) by pooling observations of Swamp Sparrow (*Melospiza georgiana*; $n = 364$), Northern Cardinal (*Cardinalis cardinalis*; $n = 306$), White-throated Sparrow (*Zonotrichia albicollis*; $n = 147$), unidentified sparrow ($n = 106$), Eastern Towhee (*Pipilo erythrophthalmus*; $n = 104$), Field Sparrow (*Spizella pusilla*; $n = 27$), Vesper Sparrow (*Pooecetes gramineus*; $n = 16$), Fox Sparrow (*Passerella iliaca*; $n = 14$), White-crowned Sparrow (*Zonotrichia leucophrys*; $n = 12$), Chipping Sparrow (*Spizella passerina*; $n = 6$), and Lark Sparrow

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(*Chondestes grammacus*; $n = 2$). Because these species have somewhat similar foraging strategies during winter (i.e., granivorous ground-feeding birds that forage close to cover; Bent 1968), we assumed that detection probabilities were similar among species and could be modeled with a common detection function.

Prior to analyses, we visually inspected the data by plotting observations by distance band for each detection function. The half-normal base function, with cosine or hermite polynomial adjustment terms, and the hazard-rate base function, with either cosine or polynomial adjustment terms, were selected as likely base-function, adjustment-term combinations that would best model the data. Base functions and series expansion terms, increasing in complexity (number of estimatable parameters), were sequentially evaluated by comparing AIC values among competing models (Burnham and Anderson 1998, Anderson et al. 2000). When a more complex model failed to adequately fit the data relative to the number of parameters within the model (greater AIC), the previous model was selected as the best approximating model. Right truncation was set to 65 m, equal to the midpoint between the beginning of the last distance band (>30 m) and 100 m.

We estimated bird density independently for the agricultural and non-agricultural sides of the transect. We used the T and C detection probabilities (value of probability density function $f_{(x)}$ evaluated at 0) to compute densities for the agricultural side of all bordered and non-bordered transects, respectively. These two density estimates (bordered and non-bordered on the agricultural side) represented the effect of field borders without accounting for birds inhabiting the adjacent plant community. On the non-agricultural side, we used the GB, GS, WB, and WS detection probabilities to estimate densities using only respective transects that had either a field border or no field border on the agricultural side. We then combined these class-specific density estimates to estimate the joint density for a field edge with a given combination of adjacent plant community and border type. For example, we combined the density estimates for the herbaceous block (GB), non-agricultural side of bordered transects with the density estimate of bordered transects (T) on the agricultural side to produce the density estimate for the TGB treatment combination. We believe this approach best accommodates instances where detection functions differ between sides of the same transect line. All reported densities and variances were generated using 1,000 bootstrap samples (with replacement) incorporating detection probabilities and numbers of observations/transect/treatment combination (Buckland et al. 2001). We used a *Z*-test to evaluate border effects between like pairs (e.g., CGB versus TGB) (Buckland et al. 2001). All results were considered significant at $\alpha = 0.05$.

Community structure.—To characterize community structure and relative conservation value of bordered and non-bordered field edges, we calculated species richness, the Shannon-Weaver diversity index (Shannon and Weaver 1949), and total avian conservation value (TACV; Nuttle et al. 2003) using only observations within the first distance interval on each side of the transect centerline. TACV is a weighted index of community conservation value calculated by multiplying species-specific abundances by their respective Partners in Flight (PIF) conservation priority scores (Carter et al. 2000). Species-specific scores were summed across all species within a given transect to produce a transect TACV score. PIF priority scores reflect different degrees of need for conservation attention based on breeding and wintering distributions, relative abundance, potential threats to breeding and wintering habitats, population trend, and a physiographically specific value of area importance (Carter et al. 2000). We used PIF priority scores for species that winter in the East Gulf Coastal Plain physiographic region. ''Fly-overs'' were not included. We used *t*-tests to determine differences in mean species richness, Shannon Diversity, and TACV between bordered and non-bordered transects by adjacent plant community. Where unequal variances occurred, we used Satterthwaite's adjusted degrees of freedom (Milliken and Johnson 1992). All results were considered significant at $\alpha = 0.05$.

RESULTS

We recorded 71 species and 17,562 individual birds while sampling 155.2 km of tran-

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TABLE 1. Sampling effort and model selection of detection functions of wintering Song, Savannah, and other sparrows along bordered and non-bordered agricultural field edges in Clay and Lowndes counties, Mississippi, 2002–2003.

Species/Class	*n*[a]	L[b]	*n*[c]	Model selected	$f_{(o)}$	m[d]	AIC
Song Sparrow							
Border	317	40,000	44	HN[e] + cosine	0.2353	1	15.52
Non-border	76	37,600	38	HN + cosine	0.0940	1	95.63
Pooled Ag.[f]	393	77,600	82	HR + cosine	0.1353	2	163.86
Grass block	187	17,000	17	HR[g] + cosine	0.0890	2	236.66
Grass strip	240	20,600	22	HR + polynomial	0.6189	3	178.82
Pooled grass[f]	427	37,600	39	HN + cosine	0.1332	3	421.99
Wood block	250	20,000	22	HN + cosine	0.1316	1	165.40
Wood strip	137	20,000	21	HR + cosine	0.2025	2	100.31
Pooled wood[f]	387	40,000	43	HR + cosine	0.1453	2	269.64
Savannah Sparrow							
Border	151	40,000	44	HN + cosine	0.1435	1	80.91
Non-border	210	37,600	38	HR + polynomial	0.0350	2	528.49
Pooled Ag.[f]	361	77,600	82	HR + polynomial	0.1045	3	744.35
Grass block	82	17,000	17	HR + cosine	0.1396	2	67.37
Grass strip	463	20,600	22	HN + cosine	0.1331	1	298.10
Pooled grass[f]	545	37,600	39	HN + cosine	0.1300	1	369.13
Wood block	5	20,000	22	HN + cosine	0.0813	1	9.50
Wood strip	98	20,000	21	HN + cosine	0.1632	1	35.43
Pooled wood[f]	103	40,000	43	HN + cosine	0.1513	1	47.79
Other sparrows							
Border	186	40,000	44	HR + cosine	0.8448	2	73.49
Non-border	49	37,600	38	HR + cosine	0.0251	2	139.89
Pooled Ag.[f]	235	77,600	82	HR + cosine	0.6150	3	323.82
Grass block	145	17,000	17	HR + cosine	0.3594	2	282.15
Grass strip	215	20,600	22	HR + cosine	0.6039	3	346.38
Pooled grass[f]	360	37,600	39	HR + cosine	0.5463	3	629.09
Wood block	276	20,000	22	HN + cosine	0.0885	3	537.16
Wood strip	287	20,000	21	HR + cosine	0.6893	3	380.04
Pooled wood[f]	563	40,000	43	HN + cosine	0.0981	3	937.29

[a] Number of observations.
[b] Sampling effort (total length of transects in meters).
[c] Number of transects.
[d] Number of parameters in detection function.
[e] Half-normal base function.
[f] Pooled detection functions were not used to compute density.
[g] Hazard-rate base function.

sects during 2002–2003. The five most abundant species were Red-winged Blackbird (*Agelaius phoeniceus*; 44.7%), American Pipit (*Anthus rubescens*; 11.2%), Song Sparrow (6.9%), Savannah Sparrow (5.7%), and American Robin (*Turdus migratorius*; 4.9%). Detection functions for Song, Savannah, and other sparrows were not robust to pooling across adjacent plant communities (Table 1); therefore, we used detection functions specific to the adjacent plant community to compute density estimates for each species.

Density.—Song Sparrow densities (birds/ha) differed between bordered and non-bordered field edges adjacent to grass block (border = 30.86, SE = 4.19; non-border = 8.29, SE = 2.58; $Z = 4.59$, $P < 0.001$) and wooded strip ($P < 0.001$) plant communities (Table 2). However, no difference in Song Sparrow density was observed between bordered and non-bordered transects adjacent to herbaceous strips ($P = 0.24$) and wooded blocks ($P = 0.35$; Table 2). Savannah Sparrow densities did not differ between bordered and non-bordered transects adjacent to herbaceous blocks (border = 14.95, SE = 6.14; non-border = 4.74, SE = 1.45; $Z = 1.62$, $P = 0.053$), herbaceous strips ($P = 0.13$), wooded blocks (P

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TABLE 2. Mean wintering densities (birds/ha) of Song, Savannah, and other sparrows along bordered and non-bordered agricultural field edges by adjacent plant community in Clay and Lowndes counties, Mississippi, 2002–2003.

Species/ Adjacent plant community[a]	Bordered[b]		Non-bordered				
	Mean	SE	Mean	SE	RES[c]	*Z*-test	*P*-value
Song Sparrow							
Grass block	30.86	4.19	8.29	2.58	272.25	4.59	<0.001
Grass strip	95.87	30.00	70.03	20.90	36.90	0.71	0.24
Wood block	25.34	3.68	28.20	6.38	−10.14	−0.39	0.35
Wood strip	38.17	4.92	10.24	2.18	272.75	5.20	<0.001
Savannah Sparrow							
Grass block	14.95	6.14	4.74	1.45	215.40	1.62	0.053
Grass strip	18.05	9.93	47.51	24.27	−62.01	−1.12	0.13
Wood block	5.40	2.44	2.35	1.23	129.78	1.12	0.13
Wood strip	21.47	15.08	2.28	1.24	841.67	1.27	0.10
Other sparrows							
Grass block	78.20	12.99	19.36	7.96	303.93	3.86	<0.001
Grass strip	138.98	18.05	30.01	7.39	363.11	5.59	<0.001
Wood block	51.44	11.81	12.55	2.70	309.88	3.21	<0.001
Wood strip	128.94	16.98	107.69	26.80	19.73	0.67	0.25

[a] Adjacent plant community on the non-agriculture side of the transect.
[b] Mean is the sum of densities of agricultural and non-agricultural sides of transects.
[c] Relative effect size: ([border − non-border]/non-border) × 100.

= 0.13), and wooded strips (P = 0.10; Table 2).

Other sparrow densities differed with respect to bordered and non-bordered transects along herbaceous block (border = 78.20, SE = 12.99; non-border = 19.36, SE = 7.96; Z = 3.86, $P < 0.001$), herbaceous strip ($P < 0.001$), and wooded block ($P < 0.001$) communities (Table 2). However, densities did not differ along wooded strips (P = 0.25; Table 2). Most Field Sparrow (92.6%) and Swamp Sparrow (91.8%) observations occurred along bordered transects. We recorded similar numbers of Northern Cardinal (border = 145; non-border = 161), Eastern Towhee (border = 44; non-border = 60), Chipping Sparrow (border = 3; non-border = 3), White-throated Sparrow (border = 62; non-border = 85), Vesper Sparrow (border = 7; non-border = 9), and unidentified sparrows (border = 103; non-border = 56) along bordered and non-bordered transects; however, few Fox Sparrows (border = 1; non-border = 13) and no White-crowned or Lark sparrows were recorded along bordered transects.

Community structure.—We recorded 59 species (6,108 individuals) within one distance band on each side of transects. The most abundant species were Song Sparrow (22.7%), American Robin (7.5%), Savannah Sparrow (6.9%), Swamp Sparrow (6.8%), and Northern Cardinal (6.8%). Species richness, diversity, and TACV did not differ between bordered and non-bordered transects, regardless of the adjacent plant community type (Table 3).

DISCUSSION

Brennan (1991), Rodenhouse et al. (1993), and Warner (1994) suggested that the elimination of grassy edge communities around agricultural field edges and fencerow habitats contributed to the decline of Northern Bobwhite (*Colinus virginianus*) and many other grassland species inhabiting farmlands. Most sparrows are ground foragers (Wheelwright and Rising 1993, Arcese et al. 2002) and their use of strip-cover habitats often depends upon vegetation structure (Bryan and Best 1991, Rodenhouse et al. 1993, Camp and Best 1994). We observed greater densities of several sparrow species where field borders were established. However, this effect varied by species and type of adjacent plant community. Song Sparrow and other sparrow densities were greatest where field borders were established along existing grasslands. The habitat

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TABLE 3. Species richness, Shannon-Weaver diversity index, and total avian conservation value using observations within one distance-band on each side of transects along bordered and non-bordered agricultural field edges by adjacent plant community type in Clay and Lowndes counties, Mississippi, 2002–2003.

Measure/ Adjacent plant community[a]	Bordered		Non-bordered				
	Mean	SE	Mean	SE	RES[b]	*t*-test	*P*-value
Species richness							
Grass block	6.60	0.22	5.29	0.78	24.76	1.62	0.15
Grass strip	6.54	0.79	8.00	0.96	−18.25	−1.17	0.26
Wood block	7.00	1.16	9.30	1.42	−24.73	−1.27	0.22
Wood strip	10.18	1.16	11.50	1.12	−11.48	−0.82	0.43
Shannon-Weaver							
Grass block	1.37	0.08	1.28	0.14	7.03	0.58	0.57
Grass strip	1.21	0.09	1.42	0.15	−14.79	−1.20	0.24
Wood block	1.29	0.17	1.35	0.15	−4.44	−0.24	0.81
Wood strip	1.60	0.19	1.92	0.20	−16.67	−1.19	0.25
Total avian conservation value							
Grass block	970.60	145.05	566.00	157.00	71.48	1.86	0.08
Grass strip	1,199.55	272.22	1,403.00	409.61	−14.50	−0.41	0.68
Wood block	677.67	191.01	996.70	273.72	−29.00	−0.98	0.34
Wood strip	2,421.18	1,392.05	694.30	96.97	248.72	1.24	0.24

[a] Adjacent plant community on the non-agricultural side of the transect.
[b] Relative effect size: ([border − non border]/non-border) × 100.

value of herbaceous field borders adjacent to grasslands may seem paradoxical, but most grasslands on our study farms were monotypic stands of cool-season, exotic forage grasses that provided little vertical structure and little seed-production. Only Song Sparrow densities were greater along wooded strip habitats with a field border. Once crops were harvested, field border habitats provided the structural vegetation characteristics commensurate with the foraging ecology of most sparrows. Field borders were recently established (<3 years old) and consisted primarily of seed-producing grasses and forbs coupled with a relatively open understory that facilitated ground-based foraging. Additionally, field borders provided escape cover in close proximity to other foraging sites, mainly row-crop fields containing waste grain. Therefore, we speculate that field borders may enhance the value of existing grasslands and cropland by producing additional foraging habitat and escape cover in close proximity to waste-grain food sources. The net effect of field borders may be to increase usable space and carrying capacity for sparrows in agricultural landscapes.

Given that most sparrow species observed in our study had somewhat similar foraging strategies, we had expected field borders to elicit similar responses across most sparrow species. With the exception of Song, Field, and Swamp sparrows, Savannah Sparrows and five of the other sparrow species were equally abundant along bordered and non-bordered transects, regardless of adjacent plant community. Whereas our estimates for other sparrows were markedly different between bordered and non-bordered transects across all of the adjacent plant communities (except for wooded strips), this effect was heavily weighted by observations of Swamp Sparrows. Swamp Sparrows were most strongly associated with bordered transects and composed a large proportion (31.5%) of other sparrow observations. Thus, our observed border effects for other sparrows were attributable mainly to greater densities of Swamp Sparrows along bordered transects. Collectively, across most adjacent plant communities, we observed greater densities of Song, Field, and Swamp sparrows along bordered transects. Responses of other sparrow species were either equivocal or negative. Overall, field borders apparently elicited greater use from only a few selected species in our study. The effect of field borders on other species or communities in other physiographic regions remains unknown.

Conservation implications.—Field borders

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may provide important habitat in southern agricultural systems where many eastern grassland species of short-distance migrants overwinter. Murphy (2003) reported strong associations between changes in farmland structure and population trends of short-distance migrant grassland birds and suggested that this association existed because short-distance migrants were affected by changes in agricultural landscapes during both the breeding and wintering seasons. The value of strip habitats has been a source of debate regarding their ability to serve as population sources during the breeding season; however, their roles during the wintering period are unknown. The availability of food resources during winter has been shown to enhance survival and body condition of birds (Porter et al. 1980, Brittingham and Temple 1988, Desrochers et al. 1988, Egan and Brittingham 1994). Although the survivorship of birds wintering in strip habitats is not known, we contend that the annual grasses characteristic of these idle communities might provide important thermal and energetic resources (Klute et al. 1997, Best et al. 1998).

Weed seeds are the primary energy source for most wintering sparrows (Wheelwright and Rising 1993, Mowbray 1997, Arcese et al. 2002). We recommend that field borders be maintained in early seral stages through periodic disturbance (e.g., fire or disking) to provide greater quantities of, and accessibility to, seeds of annual plants during the winter (Burger et al. 1990, Millenbah et al. 1996, Best et al. 1998, Greenfield et al. 2002). Seral species, such as giant ragweed, provide comparatively high levels of metabolizable energy relative to other non-agriculture plant seeds (Robel et al. 1979). Additionally, field borders may provide safe access to other highly metabolizable food sources, such as waste grain. Collectively, we suggest that field borders provide important winter habitat for many grassland birds due to their greater abundance of food (weed seed) and more complex vegetation structure for roosting, loafing, thermal, and escape cover than that found in adjacent row crops and grasslands.

Identifying resource management systems that support both birds and farm operators is important for maintaining a diverse farmland avifauna (Rodenhouse et al. 1993, Musters et al. 2001, Murphy 2003). Environmental benefits (e.g., decreased runoff of herbicides and nutrients, reduced soil erosion and sedimentation) of field-border conservation practices are well documented; the wildlife habitat value of field borders, especially during winter, is not as well understood. Our results suggest that field borders support greater densities of certain sparrow species along agricultural field edges during the winter, but they may not necessarily support greater species richness and diversity. These results, combined with our current understanding of environmental and economic benefits of field borders, suggest that field-border conservation practices are compatible with the needs of farm operators while diversifying farmland vegetation structure to enhance local avifauna.

The U.S. Department of Agriculture's National Conservation Buffer Initiative practices, such as field borders, offer potential opportunities for enhancing wintering habitat for numerous grassland birds on southeastern farmlands. Widespread implementation of field-border conservation practices is currently feasible (through Farm Bill programs) and likely to occur given the growing public concern regarding sustainable agriculture. However, as noted by Peterjohn (2003), simple, all-encompassing solutions will not reverse significant declines of farmland birds; field-border conservation practices may only benefit some species in some physiographic regions. We agree with Herkert et al. (1996) and Peterjohn (2003) in their assertion that the greatest gap in our knowledge of farmland bird ecology is winter ecology. We recommend that a greater emphasis be placed on research addressing overwinter benefits of farmland conservation practices to wildlife.

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