Grassland bird associations in a managed agricultural matrix

by

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ABSTRACT

Continued declines of grassland bird populations in the U.S. are largely attributed to grassland loss from conversion to agriculture. Habitat establishment efforts by the U.S. Department of Agriculture (USDA) may stem these declines given adequate ecological insight to guide management protocols. My research evaluated bird responses to four USDA conservation practices in the Mississippi Alluvial Valley, Mississippi, including 1) early-succession afforestation blocks, 2) early-succession, 60-m wide riparian forest buffers, 3) diverse-planted, 30-m wide filter strips, and 4) monotypic-planted, 30-m wide filter strips. I collected data from May to August, 2005-2007, to investigate the influence of conservation practice design, landscape context, and microhabitat on avian community structure, grassland bird nest ecology, Dickcissel fledgling ecology, and Northern Bobwhite landscape-level associations. Strip-transect surveys showed that Red-winged Blackbird (43%) and Dickcissel (42%) dominated bird communities, but other abundant species included Eastern Meadowlark, Indigo Bunting, Mourning Dove, and Northern Bobwhite. Bird diversity was greater in blocks than buffers, and diverse filter strips had the greatest bird densities. I found and monitored large samples of Dickcissel (n = 733) and Red-winged Blackbird (n = 409) nests in all practices, with large blocks attracting the greatest nest densities. Dickcissel nest success (22.9%) positively correlated with nest height, but negatively correlated with nearby (30 m) rowcrop area, grass cover, and horizontal vegetation cover. Red-winged Blackbird nest success was highest in diverse filter strips (23.4%) than on average (8.6%) and positively correlated with nest height, but negatively correlated with litter cover. I attached radio transmitters to 416 Dickcissel nestlings to track fledgling survival. Red imported fire ants and snakes were the predominant predators of fledglings and caused ≥65% of all mortality, which was greatest (83%) ≤2 days postfledging. Fledgling survival positively correlated with age, mass, and perch height, but negatively correlated with time in season, grassland area and edge density in the landscape. I counted 302 Northern Bobwhites at 100 locations in a 7,818 ha region and found 3.3 times more birds in a managed than unmanaged habitats. Their abundances were negatively associated with rowcrop area, wooded area, developed area, and landscape contagion, but positively influenced by all conservation practices, especially diverse filter strips. I recommend integrating 30-m wide conservation buffers within a block-based conservation management system to maximize grassland bird benefits. I also suggest that buffers be managed to maximize vegetative structural complexity in concert with moderate cover.

CHAPTER 1.
GENERAL INTRODUCTION

Large-scale conversions of native U.S. grasslands to agriculture have resulted in severe grassland bird population declines (Herkert 1991, Peterjohn and Sauer 1999, Brennan and Kuvlesky 2005). Current trends in agricultural intensification (e.g., larger farm field sizes, hedgerow removal, farm consolidation, crop monocultures, and transgenic crops) predict a dismal future for grassland birds without bold efforts to replace a significant amount of lost grassland habitat (Chamberlain et al. 2000, Donald et al. 2001). A wildlife-friendly farming effort is being led by the U.S. Department of Agriculture (USDA), but necessitates scientific information to implement management protocols that optimize benefits for grassland birds (Green et al. 2005). Conservation buffer practices are a special focus of this effort because of their enhanced compatibility in agricultural matrices compared to block-shaped habitat patches. Conservation buffers (e.g., riparian buffers, field borders, and filter strips) are vegetative strips intended to improve water quality by mitigating nonpoint source pollution and, more recently, provide wildlife habitat and preserve biodiversity (McKenzie 1997, Best 2000). In Mississippi, information about grassland bird response to these habitats is crucial because most understanding of grassland bird ecology comes from research conducted in the Midwest (Best 2000).

Conservation buffers provide habitat for a diverse suite of breeding (Henningsen and Best 2005, Conover 2005), and wintering (Smith et al. 2005, Conover et al. 2007) birds. Herbaceous buffers in the Mississippi Alluvial Valley (MAV) are attractive to Dickcissels (Conover et al. *in press*) and as such, managing for vertical cover and forbs may enhance such species-specific benefits (Zimmerman 1982, Hughes et al. 1999). A potential drawback to conservation buffers is their high perimeter-to-area ratio, which may enhance edge effects and confound wildlife benefits by rendering them population sinks or ecological traps (Gates and Gysel 1978, Helzer and Jelinski 1999, Heske et al. 1999, Woodward et al. 2001). Buffers of greater width may increase bird use through greater overall area (Vickery et al. 1994, Winter and Faaborg 1999), reduced nest predation (Gates and Gysel 1978), increased nesting opportunities (Shalaway 1985, Conover 2005), and vegetative heterogeneity (Wiens 1974). Local factors such as vegetation can also determine patch quality for grassland nesting birds (Wiens 1969, Renken and Dinsmore 1987, McCoy et al. 2001). Vegetative characteristics can influence bird use through resource availability, floral diversity, and structural heterogeneity (Cody 1968, Willson 1974, Benton et al. 2003). Hence, an understanding of local vegetation, landscape characteristics, and patch design may enhance our ability to maximize grassland bird benefits from these habitat practices. My

study focused on four habitat practices that are implemented by the USDA through the Conservation Reserve Program. These practices are 1) early-succession, afforestation blocks, and conservation buffers that include 2) diverse-planted filter strips, 3) monotypic-planted filter strips, and 4) early-succession riparian forest buffers.

My dissertation research examined the overall effects of these managed habitats within the intensive agricultural matrix of the MAV. I documented avian community structure to assist understanding of overall bird use of these habitats. Grassland bird nesting activity and success were studied to provide an indication of the demographic consequences associated with habitat management (Fletcher et al. 2006). I further focused on Dickcissel (*Spiza americana*) fledglings to offer insight into a particularly vulnerable grassland bird life-history stage (Berkeley et al. 2007, Suedkamp Wells et al. 2007). Finally, I examined habitat use based on Northern Bobwhite (*Colinus virginianus*) abundances, which is often perceived as a flagship species for many North American grassland ecosystems (Riffell et al. 2008). The combined investigation of these four aspects of grassland bird ecology will provide information relevant to improving the management protocols associated with these established grassland habitats.

THESIS ORGANIZATION

This dissertation comprises four manuscripts written for publication in multiple scientific journals. Chapter 1 entails a synthesized background to my overall research objectives. Chapter 2 is an evaluation of avian community structure in established habitat practices within an intensive, rowcrop agricultural landscape in the MAV. Chapter 3 focuses on the nesting ecology of Dickcissels and Red-winged Blackbird (*Agelaius phoeniceus*) relative to habitat practice and influences of local scale vegetation, landscape characteristics, and other biological processes. Chapter 4 investigates the effects of microhabitat and landscape characteristics, amongst other biological processes, on the survival of Dickcissel fledglings and their associated causes of mortality. Chapter 5 is a spatially-explicit evaluation of the association between Northern Bobwhite abundance and landscape characteristics. Chapter 6 concludes with a synthesis of key results from my dissertation research.

The candidate was responsible for data collection, statistical analyses, and preparation of the text; Dr. Stephen J. Dinsmore provided guidance and editorial advice on all chapters, and Dr. L. Wes Burger, Jr. (Mississippi State University) also provided substantial guidance.

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CHAPTER 2.
AVIAN COMMUNITIES IN A FRAGMENTED AGRICULTURAL LANDSCAPE

ABSTRACT

Declines in populations of many grassland birds have been linked to agricultural land use and increasing intensification. Establishing patches of non-crop, native herbaceous vegetation in agricultural landscapes may help reverse these declines by providing suitable breeding habitat. To optimize the balance between habitat benefits and agronomics, we need to understand bird community responses to a variety of patch designs. I used strip transects to evaluate bird richness, abundance, Shannon diversity, and Total Avian Conservation Value in 20 replicates of four conservation practices in an intensive, rowcrop agricultural landscape in the Lower Mississippi Alluvial Valley from May-July, 2005-2007. Conservation practices included: 1) large blocks of structurally-diverse, early-succession vegetation, 2) 30-m wide filter strips established in a monoculture of tall, dense switchgrass, 3) 30-m wide filter strips with a diverse forb/grass planting, and 4) 60-m wide, early-succession riparian forest buffers. The breeding bird community was dominated by Red-winged Blackbird (43% of total) and Dickcissel (42% of total); Eastern Meadowlark, Indigo Bunting, Mourning Dove, and Northern Bobwhite were the next most abundant species. Dickcissels were at least 1.8 times more abundant in large block and diverse filter strips than other buffers. I observed significantly greater Shannon diversity in large blocks than any strip habitats ($P < 0.05$), but the blocks and strips did not significantly differ in abundance or overall conservation value. Diverse filter strips had 1.6 times greater bird density (7.2 birds/0.6 ha), on average, than all other practices. Based on these data, I recommend managing for grassland birds in an intensive agricultural landscape by integrating a complement of large, early-succession blocks and buffers that are at least 30-m wide and planted with a diversity of native forbs and grasses.

INTRODUCTION

The conversion of native habitat to agriculture in the U.S. resulted in grassland bird range shifts (Hurley and Franks 1976) and severe population declines (Herkert 1991, Peterjohn and Sauer 1999, Brennan and Kuvlesky 2005). By the time cropland development stabilized in the 1950s (Lubowski et al. 2006), avian communities amongst farmlands were composed primarily of species that could exploit the remnant habitats that persisted largely on field margins (Warner 1994). Agricultural advances have since facilitated intensification (e.g., larger field size, hedgerow removal, farm consolidation, crop monocultures, transgenic crops) to maximize crop yield and based on European studies, may represent the primary future threat to bird conservation

on U.S. farmlands (Chamberlain et al. 2000, Donald et al. 2001, Benton et al. 2002), particularly from inadequate food availability for adults and juveniles (Benton et al. 2002, Newton 2004). In the U.S., intensification practices that are documented to affect bird populations include field size enlargement and field margin removal (Best et al. 1990, Herkert 1991, 1994, Warner 1994). For example, "clean farming" is a modern practice that eliminates non-crop, field margin vegetation in favor of maximum utilization of arable acres, thus eradicating the already sparse, wildlife habitat from intensive, agricultural landscapes (Best 1983).

The combination of native grassland degradation throughout the Midwest (Noss et al. 1995) and grassland bird range expansions from agriculture (Hurley and Franks 1976) creates a need and an opportunity to support these populations by establishing early-succession habitat on agricultural landscapes outside the Midwest. The Mississippi Alluvial Valley (MAV) is an ideal location as flood control in the early 1900s facilitated conversion from the largest (10 million ha) contiguous, forested wetland system in North America to predominantly agricultural production, with remaining (20%) extant forest distributed in isolated fragments (Brown et al. 1999). Furthermore, the widespread practice of "clean farming" in the MAV (R. Conover, personal observation) and attraction of birds to herbaceous borders on field margins during breeding and non-breeding seasons (Conover et al. 2007, Conover et al. *in press*) enhances its year-round conservation potential.

United States croplands are 99% privately owned (Lubowski et al. 2006), so the U.S. Department of Agriculture initiated conservation efforts in 1985 through economic incentives under the Conservation Reserve Program (CRP). The CRP has effectively established considerable amounts of grassland habitat on private lands to enhance farmland sustainability, although its goals did not incorporate wildlife conservation until 2004. However, the continued declines (Ryan et al. 1998) and limited reproductive potential (e.g., Dickcissel, *Spiza americana*; McCoy et al. 1999) for some species in existing CRP habitats provoked the enhancement of program objectives to incorporate wildlife benefits using conservation buffers. Conservation buffers (e.g., riparian buffers, field borders, filter strips) are vegetative strips intended to reduce soil erosion, improve water quality by mitigating nonpoint source pollution, and more recently, provide wildlife habitat and preserve biodiversity (McKenzie 1997, Best 2000). Herbaceous strip habitats attract high bird densities relative to crop fields (Bryan and Best 1991, Camp and Best 1993) and are used by wintering and breeding birds of conservation concern (Marcus et al. 2000, Puckett et al. 2000, Smith et al. 2005a, Smith et al. 2005b, Conover et al. 2007). Furthermore, herbaceous buffers in the MAV are attractive to Dickcissels (Conover et al. *in press*) and as such,

managing for vertical cover and forbs may enhance such species-specific benefits (Zimmerman 1982, Hughes et al. 1999). The potential for conservation buffers to balance wildlife benefits with landowner economics (Barbour 2006) may amplify their future role in agricultural sustainability.

A potential drawback to conservation buffers is their high perimeter-to-area ratio, which may elicit edge effects and confound wildlife benefits by rendering them population sinks or ecological traps (Gates and Gysel 1978, Helzer and Jelinski 1999, Heske et al. 1999, Woodward et al. 2001). Management protocols that minimize edge effects depend on understanding ecological interactions based on buffer placement, structural design, disturbance regimes, and landscape context. Increased width reduces edge abruptness by decreasing perimeter-to-area ratio and provides more habitat farther from the edge. This alteration may increase bird use through greater overall area (Vickery et al. 1994, Winter and Faaborg 1999), reduced nest predation (Gates and Gysel 1978), increased nesting opportunities (Shalaway 1985, Warner 1992, Conover 2005), arthropod abundance (DiGiulio et al. 2001), and vegetative heterogeneity (Wiens 1974). In addition to width, vegetative composition influences bird use through resource availability, floral diversity, and structural heterogeneity (Cody 1968, Willson 1974, Benton et al. 2003). Herbaceous planting strategies that promote diverse, native flora will enhance structural diversity and may increase bird abundance and diversity (Warner 1992, Bryan and Best 1994, McCoy et al. 2001, Burger 2000). Woody substrates also increase vertical structure and heterogeneity, which can result in increased bird use of strip habitats (Best 1983). However, tradeoffs exist between sown grass strips that promote rapid vegetative growth but suppress plant invasion and non-sown grass plots, which may have reduced cover post-establishment but increased floral diversity (Kleijn et al. 1998). Grass-dominated habitats are also likely to have reduced overall vegetative heterogeneity, which may reduce bird diversity (Wiens 1974).

This study focused on bird community structure and species-specific use in four CRP practices: 1) monotypic filter strip (FSM; planted 30-m wide with switchgrass (*Panicum virgatum*)), 2) diverse filter strip (FSD; planted 30-m wide with grass/forb mixture), 3) riparian forest buffers (RFB; 60-m wide, hardwood planting in early-succession stage), and 4) afforestation blocks (BLK; hardwood planting in early-succession stage). My objectives were to define avian community structure related to type of conservation practice, patch shape (strip vs. block), and local physiognomic features as determined by repeated, randomized vegetation surveys. I hypothesized (1) that greater area and woody composition of block habitats would enhance bird-use over buffer practices, (2) enhanced width of RFB would attract higher bird

abundance and richness than filter strips, (3) increased vegetative diversity and heterogeneity of FSD would attract increased bird-use over FSM, and (4) Dickcissel abundance would positively correspond to forb and vertical cover.

METHODS

Study site.—I studied bird responses to conservation practices on a 2,630 ha farm in Coahoma County, Mississippi located in the lower MAV in Bird Conservation Region (BCR) 26. The study farm was selected for its enrollment in a range of conservation practices and otherwise resembled the current MAV landscape, which is dominated by large fields of ditch-to-ditch, rowcrop agriculture (primarily cotton and soybean) and negligible topographic relief. The farm had recently established various semi-natural vegetation patches to promote wildlife-friendly farming and consisted of 48% rowcrop, 30% early-succession hardwood afforestation plantings, 14% forested or herbaceous wetlands, 4% conservation buffers, 2% wooded, and 2% herbaceous drains. Farm crops were soybean in 2005 and 2007, whereas 2006 was wheat and a late soybean planting. Though crop type varied temporally, consistent spatial and crop emergence patterns permitted relative comparisons without crop influence.

Conservation practices.—I monitored bird communities on 80 transects (20 per conservation practice) that were randomly selected from the farm-wide population. Practices included: (1) filter strips planted with a forb/grass mixture (FSD), (2) filter strips planted with only switchgrass (FSM), (3) early-succession, riparian forest buffers (RFB), and (4) early-succession, hardwood blocks (BLK). Sample sizes were lower for FSD (n = 14) and FSM (n = 6) during 2005, the first year of study implementation.

Filter strips (30 m width) were established in spring 2004 and FSD was planted with partridge pea (*Chamaecrista fasciculata*, 4.5 kg/ha seeding rate), Indian grass (*Sorghastrum nutans*, 1.7 kg/ha seeding rate), little bluestem (*Schizachyrium scoparium*, 5.6 kg/ha seeding rate), and big bluestem (*Andropogon gerardii*, 1.7 kg/ha seeding rate), whereas FSM was seeded at a high density (9.0 kg/ha) of only switchgrass. Some filter strips had artificial perch sites from a nearby center-pivot irrigation system, which was assumed to have negligible effect on bird habitat use (Vickery and Hunter 1995). Riparian forest buffers (60 m wide) were planted in fall 2004. Conservation buffers were established in similar landscape context, located on field margins between a riparian zone (e.g., drainage ditch, stream) and rowcrop field. Landscape context differed for some RFB and FSD plots, which were juxtaposed to both herbaceous and wooded riparian zones. BLKs were established in fall 1999 and thus had enhanced woody growth, but retained primarily herbaceous vegetation. BLKs and RFBs were planted with Nuttall's oak

(*Quercus nutallii*), water oak (*Quercus nigra*), and willow oak (*Quercus phellos*). All habitats were colonized at differing levels with local, non-planted vegetation that predominantly included horsetail (*Conyza canadensis*), redvine (*Brunnichia cirrhosa*), vetch (*Vicia* sp.), goldenrod (*Solidago* spp.), giant ragweed (*Ambrosia trifida*), curly dock (*Rumex crispus*), dewberry (*Rubus trivialis*)*,* blackberry (*Rubus oklahomus*), johnsongrass (*Sorghum halepense*), erect poison ivy (*Toxicodendron radicans*), broomsedge (*Andropogon virginicus*), honey locust (*Gleditsia triacanthos*), American elm (*Ulmus americana*), and sugarberry (*Celtis laevigata*).

Conservation practice assessment.—I evaluated the local physiognomy of conservation practices by surveying vegetative structure and cover during a standardized time frame (14 June to 02 July in 2007) to permit relative comparisons amongst conservation practices. Trained field technicians conducted three habitat surveys on each of 20 plots per practice, totaling 60 surveys per practice. Individual survey locations were spatially randomized by length and width in meter intervals within plots and consisted of a 4 m radius circle divided into quadrants to estimate horizontal cover, heterogeneity, and vegetative composition. The limited filter strip population sample on the farm resulted in some adjacent surveyed buffers (n = 6), although I do not expect this small amount of spatial autocorrelation to substantively bias these data. Measured variables included proportions of live vegetation cover types (i.e., forb, grass, and woody) to quantify structure, vertical cover, horizontal cover, and standing dead vegetation cover. Horizontal cover was visually estimated using the mean proportion of ground cover across 4 quadrants per survey. Vertical cover was estimated as effective vegetative height using a modified Robel pole, which records the height of total visual obscurity from all 4 cardinal directions (Robel et al. 1970).

Bird community assessment.—I conducted strip-transect surveys monthly (May, June, and July) in the breeding seasons of 2005 to 2007 to account for seasonal and annual changes in bird-use patterns. Transects were evenly paced for 10 minutes and were 200 m long, with bird detections recorded within a 30 m band perpendicular to the observer. Walking pace and distance estimates of bird detections were assisted by marked plot edges and systematically placed field tape. The same observer conducted all surveys three hours post-sunrise (Central Standard Time) on days with no precipitation and wind <12 km/hr. I randomly sampled transects per farm region, although sampling within regions was non-random to reduce travel time. Buffer transects were on the buffer-ditch edge and counts were unidirectional 30 m in the buffer; whereas BLK transects bisected the plot with 30 m bi-directional observations; mean values for both sides were used for subsequent BLK analyses. Bird detections were recorded in three 10 m distance bands that were subsequently clumped for analyses and flyovers were excluded. I did not calculate

detection functions for conservation buffers as their ecotonal context violates the assumption of uniformly distributed vegetation cover types, and the transect locations were not randomized, but intentionally situated on the buffer-ditch edge (Buckland et al. 2001). I therefore constrained bird observations to ≤30 m of the transect line and assumed a 1.0 detection probability. This assumption was supported by a Program DISTANCE (Buckland et al. 2001) analysis that reported a 1.0 detection probability for BLK observations ≤30 m (R. Conover, personal observation), which was similar to observations by Diefenbach et al. (2003). BLK vertical cover estimates were representative of all practices except FSM (Figure 1). I assumed constant species-specific detection probabilities across conservation practices, as unequal probabilities could not be accounted for in community metric estimation (Rotella et al. 1999).

Bird community structure was analyzed using abundance (birds/0.6 ha), species richness (total species/0.6 ha), Total Avian Conservation Value (TACV; TACV/0.6 ha; Nuttle et al. 2003), and Shannon diversity index (H', H'/0.6 ha, Shannon and Weaver 1949). TACV indicates relative conservation value by incorporating Partners in Flight (PIF) bird conservation priority ranks (Nuttle et al. 2003), which are based on regional population trends, global population size, regional area importance value, global breeding and wintering distributions, regional threats to breeding habitat, and global threats to wintering habitat, where regional indicates BCR 26 (Carter et al. 2000, Panjabi et al. 2005). This is calculated by multiplying species' abundances by their breeding status PIF rank for the MAV (http://www.rmbo.org/pif/scores/scores.html, accessed 10 May 2008) and summing species-specific TACV scores within study plots. The PIF estimation protocol for area importance had been altered since Nuttle et al. (2003); my ranks are based on the updated version, which indexes area importance from relative density scores that reflect mean density of a species in the MAV relative to the BCR with the highest breeding season density (Panjabi et al. 2005). Migratory, non-breeding species and unidentified birds were excluded from TACV calculations.

I compared community structure and species-specific abundances among conservation practices using a mixed-model, repeated-measures analysis of variance with the MIXED procedure in SAS® software, Version 9.1.3 (SAS Institute Inc., 2002). This analysis incorporated seasonal and annual variation using month and year as repeated time effects, with conservation practice as a fixed main effect and transects as random subject effects. I applied an unstructured covariance structure, as selected by model fit using lowest Akaike Information Criterion scores (Akaike 1973). Denominator degrees of freedom were computed using the Kenward-Roger adjustment, which incorporates variance of the F-value to minimize bias when using an

unstructured covariance matrix (Kenward and Roger 1997). Pair-wise comparisons were evaluated with a Student's t-test and the Tukey-Kramer *P*-value adjustment to conservatively account for unequal sample sizes in 2005. Estimates are reported as least squares means, which are predicted margins for a balanced population. Results were considered statistically significant at $\alpha \leq 0.05$ for purposes of hypothesis testing. I accounted for annual differences in community metrics by presenting year-specific means of metrics to clarify habitat-based effects (Figures 2-5).

As these buffers were from a randomly selected sample, some FSD and RFB transects were adjacent to wooded (n = 6, n = 8) and non-wooded (n = 14, n = 12) field margins, respectively. Wooded edges were defined as adjacent habitats that had predominantly woody (>50%) ground cover and, though some FSM had woody edges, their sample sizes were insufficient for statistical testing. I accounted for this effect separately by testing the relationship between abundance and adjacent habitat types using t-tests, and in our primary analyses, this effect represents natural variation that is typical of the MAV landscape.

RESULTS

Conservation practices.—I documented differences among practices in vegetative structure and cover (Figure 1). Filter strips were grass-dominated with minimal colonization by forbs or woody vegetation. FSM had the most horizontal and vertical cover of all practices, but the least amount of horizontal or structural heterogeneity. RFB had the lowest overall cover, being largely composed of forbs and woody substrates. BLK had moderate cover and heterogeneity, but were primarily defined by relatively balanced proportions of forb, grass, and woody components.

Bird response.—I detected 34 bird species using habitats within 30 m of the transect line throughout the breeding seasons of 2005 to 2007. BLK attracted the most species (n = 25) overall, then FSD (n = 20), with RFB and FSM equal at 17 species. Red-winged Blackbird (*Agelaius phoeniceus*; 43%) and Dickcissel (42%) dominated bird communities across all conservation practices. Eastern Meadowlark (*Sturnella magna*; 4%), Indigo Bunting (*Passerina cyanea*; 2%), Mourning Dove (*Zenaida macroura*; 2%), and Northern Bobwhite (*Colinus virginianus*; 2%) were also frequently observed (Table 1). Dickcissel was most dominant in BLK (55%) and least dominant in FSM (30%), whereas Red-winged Blackbird exhibited the opposite trend, being most dominant in FSM (63%) and least in BLK (25%, Table 1).

Conservation practices differed significantly in bird abundance ($F_{3,72} = 12.08$; $P < 0.01$), richness ($F_{3,79.9} = 6.36$; $P < 0.01$), TACV ($F_{3,67.7} = 10.48$; $P < 0.01$), Shannon diversity ($F_{3,71.5} = 13.50$; $P < 0.01$), and Dickcissel abundance ($F_{3,78.9} = 10.77$; $P < 0.01$). Bird abundance was

similar across conservation practices except FSD, which supported 2.4 to 3.3 more birds/transect than other habitats across years (Table 2), with similar abundances to BLK in 2006 (Figure 2). BLK had consistently more species/transect than other practices in all years (Figure 3). BLK and FSD practices had significantly higher richness than RFB, but did not differ from FSM (Table 2). FSD had higher TACV than other habitats within years (Figure 4) and 4.0 to 5.8 more TACV/0.6 ha than other practices across years (Table 2), whereas other practices had similar TACV within and across years (Figure 4). BLK had significantly higher Shannon diversity compared to all other practices across years (Table 2), including 1.4 and 1.3 times more than the second most diverse habitat in 2005 and 2006, respectively (Figure 5). Although Dickcissels were the most dominant community member in BLK, their highest abundances occurred in FSD, with approximately 1.4 and 1.1 individuals/0.6 ha more in FSD and BLK, respectively, than FSM or RFB (Table 2). My evaluation of adjacent habitat type (wooded vs. non-wooded) indicated greater mean bird abundance for FSD (1.63 ± 1.01 birds/0.6 ha; $P = 0.11$) and RFB (2.43 ± 0.93 birds/0.6 ha; $P = 0.01$) adjacent to non-wooded margins.

DISCUSSION

Bird communities were more similar among early-succession conservation practices than expected. This may be attributed to the relative dominance of all habitats by just two species (Dickcissel and Red-winged Blackbird) or the scarcity of non-crop habitat in the surrounding landscape. The prompt and extensive bird use of these habitats is promising and variation in bird response among practices elucidates patterns of habitat use that may assist future establishment protocols.

Conservation practices.—BLK was the most structurally diverse habitat with moderate proportions of forbs, grasses, and woody substrates. Relatively low horizontal cover with moderate vertical cover and heterogeneity further defined them. Conservation buffer vegetation differed from BLK in that either grasses or forbs dominated individual buffer practices. FSD and FSM were largely composed of grasses, whereas RFB was primarily colonized by forbs. The absence of herbaceous plantings in RFB resulted in an early-succession stage characterized by low vegetative cover and pioneering species. Given the disproportionately large amount of non-crop habitat (hence, seed availability) on this farm relative to the MAV landscape, I would expect even lower floral invasion and vegetative cover of early-succession RFB habitats elsewhere in the MAV (Twedt 2006). This filter strip evaluation represents post-establishment vegetation, but prior to any disturbance regime.

Adjacent habitat types (wooded vs. non-wooded) influenced the abundance of grassland birds and I observed Dickcissel and Red-winged Blackbird abundance to be greater in FSD (1.8 and 2.2 times greater, respectively) and RFB (2.7 and 4.2 times greater, respectively) that were adjacent to non-wooded compared to wooded field margins. These results support previous observations in similar strip habitats (Henningsen and Best 2005).

Bird response.—I detected seasonal and year effects on bird use within habitats, which was expected from the dynamic nature of early-succession flora and annual variation in rowcrop plantings. BLK habitat attracted the greatest species-specific abundances for common but non-dominant birds, including Grasshopper Sparrow (*Ammodramus savannarum*), Northern Bobwhite, Mourning Dove, and Eastern Meadowlark (Table 1). Greater vegetative structure (Willson 1974) and area (Warner 1992, Winter and Faaborg 1999) likely contributed to the elevated bird species richness and diversity in BLK than buffer habitats, as predicted. BLK habitat also supported greater overall bird densities and conservation values than all buffers except FSD (Table 2). Higher bird density in FSD than BLK may have resulted from their diverse vegetative cover and ecotonal context (Bryan and Best 1991). Hence, whereas BLK habitats attracted a larger suite of birds than buffers, they did not consistently support greater densities. The comparison of linear and block habitat is not an attempt to compare wildlife benefits, but to improve understanding of the benefits conservation buffers provide relative to a more ideal habitat option. FSD had higher bird densities and TACV than all other buffers, thus supporting my prediction of elevated bird use over FSM from increased structural diversity and heterogeneity, a byproduct of planting regimes. Conversely, FSM and RFB were on opposite ends of the vegetative cover spectrum, yet both received low relative bird-use, indicating a reduction of buffer quality with extremely high or low cover. Additionally, although this study was designed to test effects of conservation practices and not specific effects of buffer width (see Conover et al. *in press*), similarities in bird abundance, richness, and TACV, contradict the presumption that block or wider buffer habitats will necessarily support more birds, species, and conservation value per unit area. The effects of width alone may have been masked by the greater relative effect of vegetation composition and landscape context. The lack of a block vs. buffer effect may be explained by the reduced placement of buffers on wooded edges, where wider buffers increase grassland bird attraction (Conover et al. *in press*), or by increased relative vegetative cover in the more narrow, filter strips than wider RFB. As such, the differential bird benefits between 60-m and 30-m wide early-succession, conservation buffers remain ambiguous. Future research should explore the practicality of incorporating herbaceous plantings in riparian

forest buffers to expedite wildlife benefits by enhancing vegetative cover prior to sufficient woody emergence.

Conservation practices that supported the greatest bird use had greater vegetation structural complexity in concert with moderate heterogeneity and cover. Dickcissel was the most dominant community member of BLK, but second to Red-winged Blackbird in all buffers. Reduced abundances of Dickcissel in RFB relative to FSD contradicted my prediction and may be explained by the paucity of cover in RFB (Patterson and Best 1996, Hughes et al. 1999). The benefits of conservation buffers for Dickcissels should be reproductively evaluated as this species may exhibit demographic, area-sensitive responses regardless of distribution (Winter and Faaborg 1999) that may facilitate continued population declines in CRP habitats (McCoy et al. 1999).

The performance of FSD highlights the potential wildlife benefits provided by buffer habitats with a mixed forb/grass planting strategy. Contradicting patterns of reduced bird diversity but increased TACV in FSD compared to BLK suggest the possibility of an abundance-driven TACV. As such, management decisions should not rely solely on TACV, but incorporate multiple community metrics. Furthermore, an understanding of bird reproductive ecology in these conservation practices will elucidate potential drawbacks related to edge effects and permit a more informed comparison of relative benefits between conservation buffer and block habitats (Van Horne 1983, Vickery 1992). Increased bird use of habitats with enhanced vegetative structural diversity may be explained by a correlative response in the arthropod community (DiGiulio et al. 2001) and hence, increased food resources. However, Davros (2005) reported a lack of arthropod response to filter strip width, vegetative cover, or forb cover; although this was performed using biomass by order, a relatively crude measurement.

Conservation implications.—My study highlights the importance of local vegetation features to enhance wildlife benefits from conservation buffers. Filter strip plantings should incorporate diverse forbs and grasses or at minimum, moderate seeding rates to promote colonization of local pioneer species to enhance vegetative heterogeneity. High seeding rates of grasses, particularly monotypes, should be avoided to prevent excessively dense vegetative cover and/or homogeneous structure. Differences in bird communities between buffer types demonstrated the value of incorporating multiple buffer practices to manage for whole-farm biodiversity (Renken and Dinsmore 1987, Marshall and Moonen 2002). Additionally, I suggest establishing buffers on a non-wooded edge if management objectives prioritize supporting grassland bird populations.

Whereas the abundance of field margin habitat has been decreasing in agricultural landscapes, the environmental role is increasingly important (Marshall 2002). Strip-shaped habitat patches effectively balance landowner and wildlife needs to overcome social challenges associated with biodiversity amongst agriculture (Firbank 2005). With the increasing shift of conservation attention from nature reserve designs to managing habitat in production-based landscapes, the potential for conservation buffers to support bird populations amongst intensive, rowcrop agriculture is of keen importance. Given previously confirmed benefits and limitations of CRP habitats for grassland birds (Ryan et al. 1998, McCoy et al. 1999), this study enumerates the potential for conservation buffers to expand the breadth of wildlife benefits provided by the program. I recommend the employment of multiple conservation buffer practices using diversified planting regimes as a secondary alternative to block habitats for mitigating farmland wildlife population declines from agricultural intensification (Donald et al. 2001, Newton 2004).

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Table 1. Bird species are listed in descending order of overall number of detections in 200x30 m strips (0.6 ha) in conservation practices in the intensive, rowcrop landscape of the Mississippi Alluvial Valley, MS, 2005-2007.

Species	All practices		FSD[b]			FSM			RFB			BLK		
	No.	RA[a]	No.	RA	Δ%[c]	No.	RA	Δ%	No.	RA	Δ%	No.	RA	Δ%
Red-winged Blackbird	1331	0.427	452	0.444	1.6	393	0.628	20.0	284	0.438	1.1	202	0.246	-18.2
Dickcissel	1310	0.421	427	0.419	-0.2	189	0.302	-11.9	241	0.372	-4.9	453	0.551	13.2
Eastern Meadowlark	131	0.042	25	0.025	-1.8	7	0.011	-3.1	39	0.060	1.8	60	0.073	3.1
Indigo Bunting	75	0.024	39	0.038	1.4	1	0.002	-2.2	32	0.049	2.5	3	0.004	-2.1
Mourning Dove	61	0.020	13	0.013	-0.7	10	0.016	-0.3	12	0.019	-0.1	26	0.032	1.2
Northern Bobwhite	55	0.018	21	0.021	0.3	4	0.006	-1.1	6	0.009	-0.8	24	0.029	1.2
Common Yellowthroat	27	0.009	10	0.010	0.1	4	0.006	-0.2	6	0.009	0.1	7	0.009	0.0
Grasshopper Sparrow	26	0.008	0	0.000	-0.8	1	0.002	-0.7	0	0.000	-0.8	25	0.030	2.2
Northern Cardinal	19	0.006	6	0.006	0.0	1	0.002	-0.5	12	0.019	1.2	0	0.000	-0.6
Other species	79	0.025	25	0.025		16	0.026		16	0.025		22	0.027	
All birds	3114		1018			626			648			822		

[a]RA = relative abundance of a bird species within a conservation practice

[b]Conservation practices: FSD (filter strip with diverse planting regime), FSM (filter strip with monotypic planting regime), RFB (early-succession, riparian forest buffer), BLK (early-succession, hardwood afforestation block)

[c]Proportional difference between specific habitat RA and total number of observations for species across habitats

Table 2. Pair-wise Student's T-test results that compare avian abundance, species richness, Shannon diversity, Dickcissel (*Spiza americana*) abundance, and total avian conservation value (TACV) amongst four conservation practices in the intensive, rowcrop landscape of the Mississippi Alluvial Valley, MS, 2005-2007.

Main effects		Abundance			Richness			TACV			Shannon diversity			Dickcissel abundance		
		Δ mean ± SE[a]	T	P[c]	Δ mean ± SE	T	P	Δ mean ± SE	T	P	Δ mean ± SE	T	P	Δ mean ± SE	T	P
2005	2006	2.38 ± 0.51	4.65	***	0.70 ± 0.10	7.39	***	5.04 ± 1.07	4.72	***	-0.076 ± 0.038	-1.99	0.12	0.67 ± 0.22	3.04	**
2005	2007	1.43 ± 0.53	2.37	0.05	0.58 ± 0.10	5.59	***	3.97 ± 1.16	3.43	**	0.063 ± 0.036	1.77	0.19	0.04 ± 0.25	0.17	0.867
2006	2007	-0.9 ± 0.36	-2.65	*	-0.1 ± 0.07	-1.87	0.16	-1.06 ± 0.54	-1.95	0.13	0.139 ± 0.038	3.61	**	-0.6 ± 0.14	-4.58	***
FSD[b]	FSM	2.79 ± 0.71	3.96	**	0.22 ± 0.17	1.30	0.57	5.02 ± 1.34	3.75	**	0.091 ± 0.040	2.25	0.12	1.45 ± 0.34	4.18	***
FSD	RFB	3.49 ± 0.61	5.72	***	0.47 ± 0.15	3.12	*	5.97 ± 1.11	5.37	***	0.103 ± 0.035	2.93	*	1.43 ± 0.31	4.59	***
FSD	BLK	2.64 ± 0.61	4.32	***	-0.14 ± 0.15	-0.90	0.80	4.16 ± 1.11	3.74	**	-0.096 ± 0.035	-2.72	*	0.31 ± 0.31	0.99	0.325
FSM	RFB	0.70 ± 0.69	1.02	0.74	0.25 ± 0.16	1.55	0.42	0.95 ± 1.30	0.73	0.89	0.012 ± 0.040	0.30	0.99	0 ± 0.34	-0.05	0.962
FSM	BLK	-0.16 ± 0.69	-0.23	1.00	-0.4 ± 0.16	-2.15	0.15	-0.86 ± 1.30	-0.66	0.91	-0.187 ± 0.040	-4.73	***	-1.1 ± 0.34	-3.35	**
RFB	BLK	-0.86 ± 0.59	-1.45	0.47	-0.60 ± 0.15	-4.11	***	-1.81 ± 1.06	-1.70	0.33	-0.200 ± 0.034	-5.82	***	-1.1 ± 0.31	-3.68	***

[a]Least squares mean difference ± standard error, with sign denoting positive or negative trend

[b]Conservation practices: FSD (diverse-planted filter strip), FSM (monotypic-planted filter strip), RFB (early-succession, riparian forest buffer), BLK (early-succession, hardwood block)

[c]*P*-values represent a Tukey-Kramer adjustment

*P<0.05, **P<0.01, ***P<0.001

Figure 1. Vegetation variables were measured for filter strips with diverse (FSD) and monotypic
(FSM) planting regimes, early-succession riparian forest buffers (RFB) and early-succession,
hardwood blocks (BLK) during late-June 2007 in the Mississippi Alluvial Valley, MS.

Figure 2. Bird abundance estimates for filter strips with diverse (FSD) and monotypic (FSM) planting regimes, early-succession riparian forest buffers (RFB) and hardwood blocks (BLK) during the summers (May-July) of 2005 (filled circles), 2006 (open circles), and 2007 (filled triangles) in the Mississippi Alluvial Valley, MS.
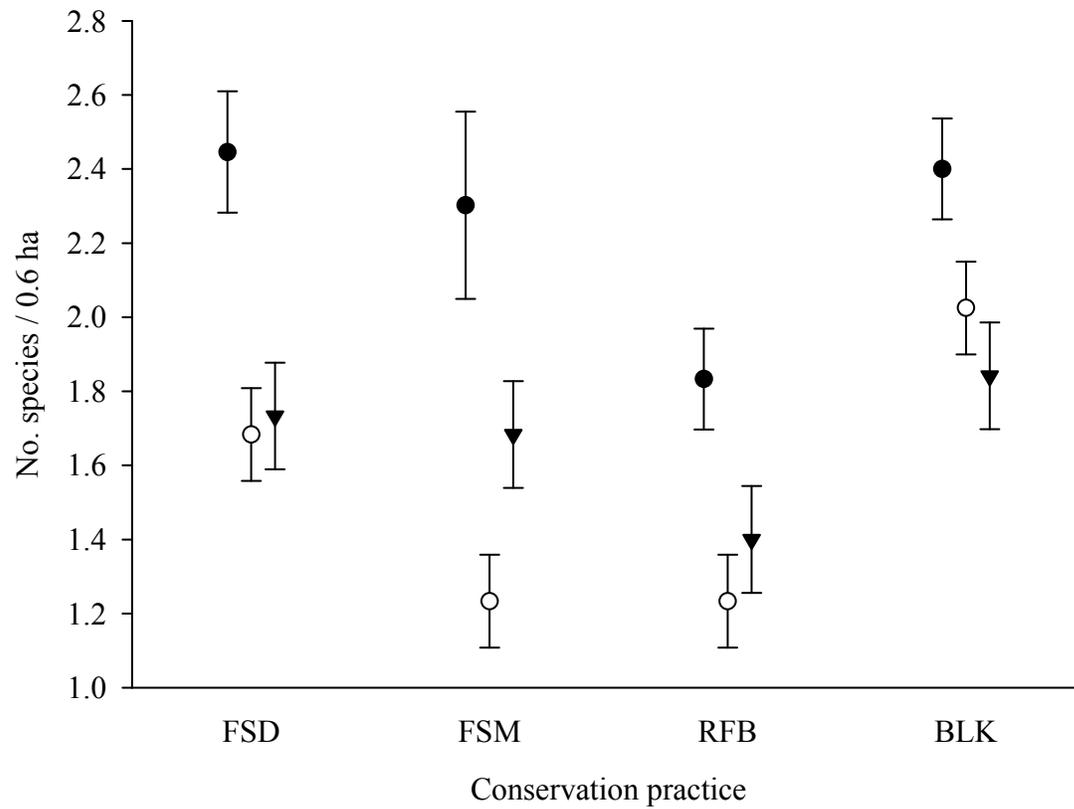
Figure 3. Bird species richness estimates for filter strips with diverse (FSD) and monotypic (FSM) planting regimes, early-succession riparian forest buffers (RFB) and hardwood blocks (BLK) during the summers (May-July) of 2005 (filled circles), 2006 (open circles), and 2007 (filled triangles) in the Mississippi Alluvial Valley, MS.
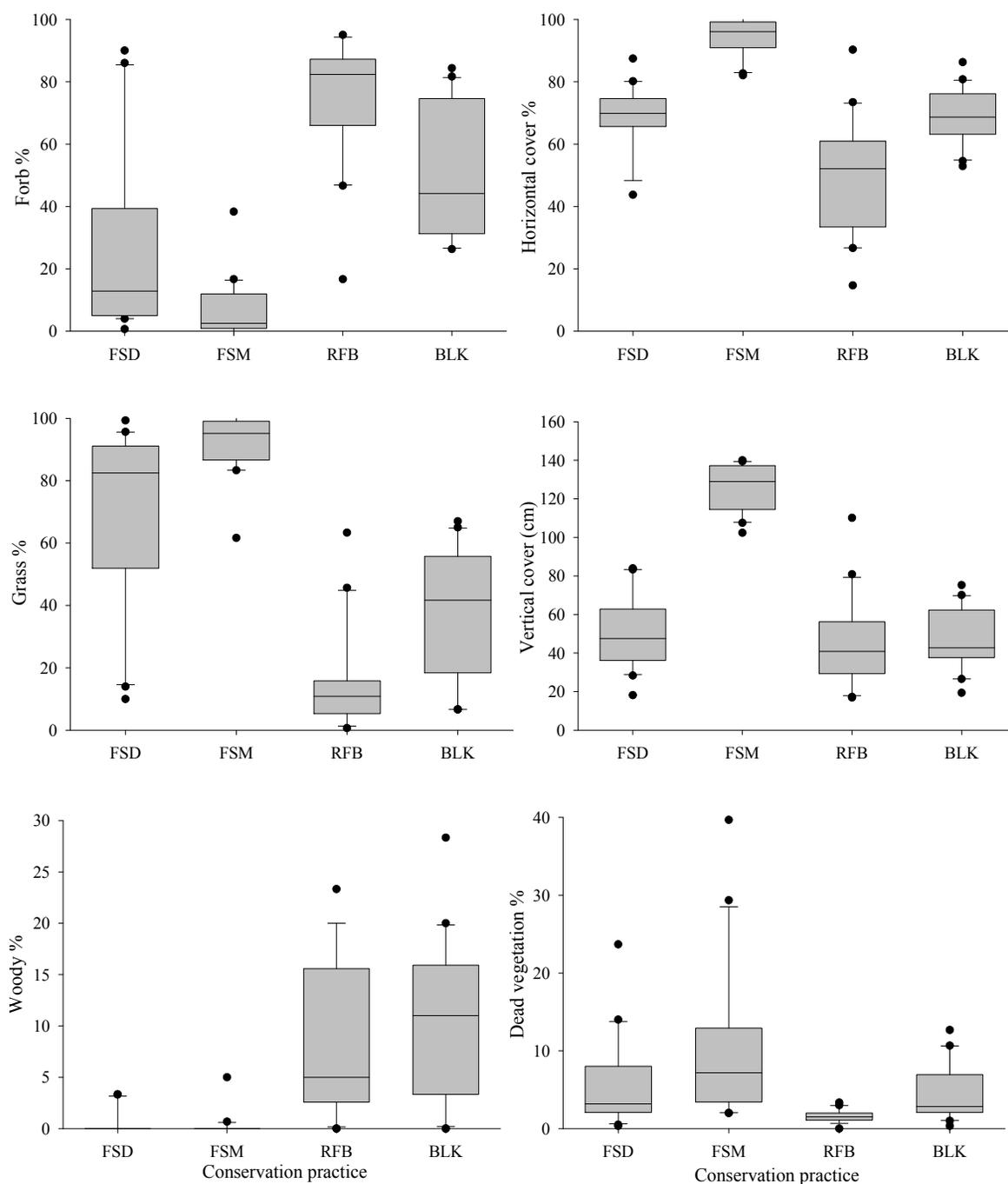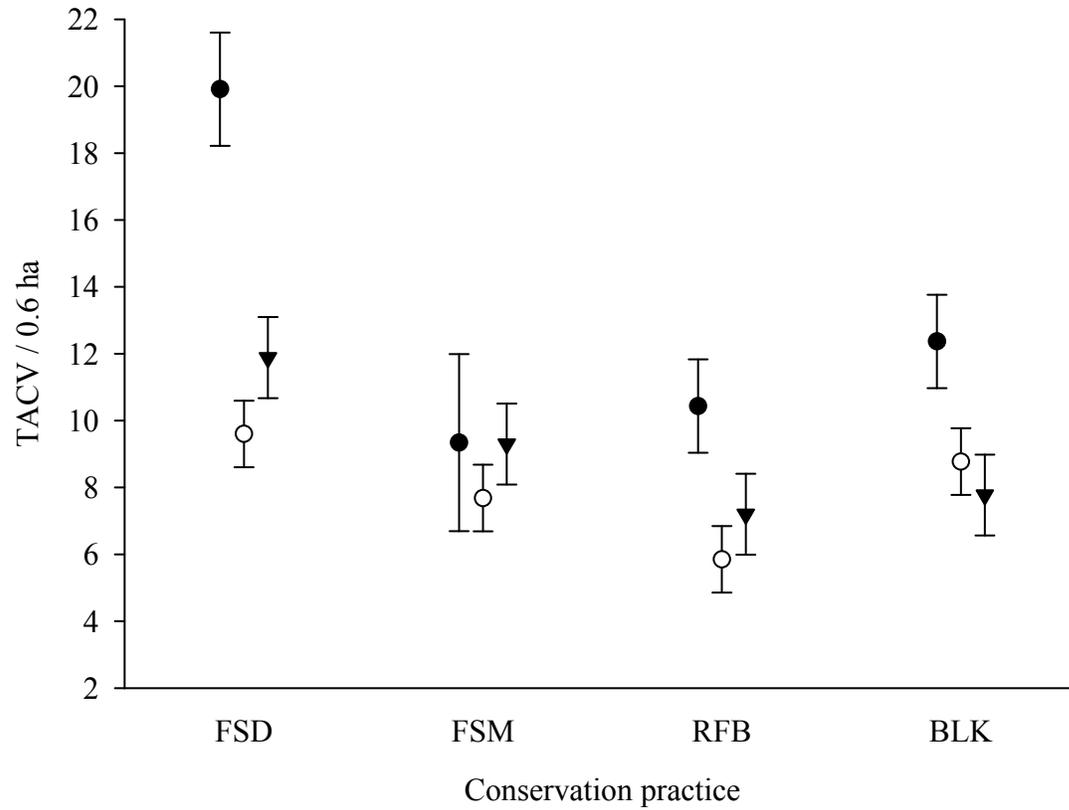
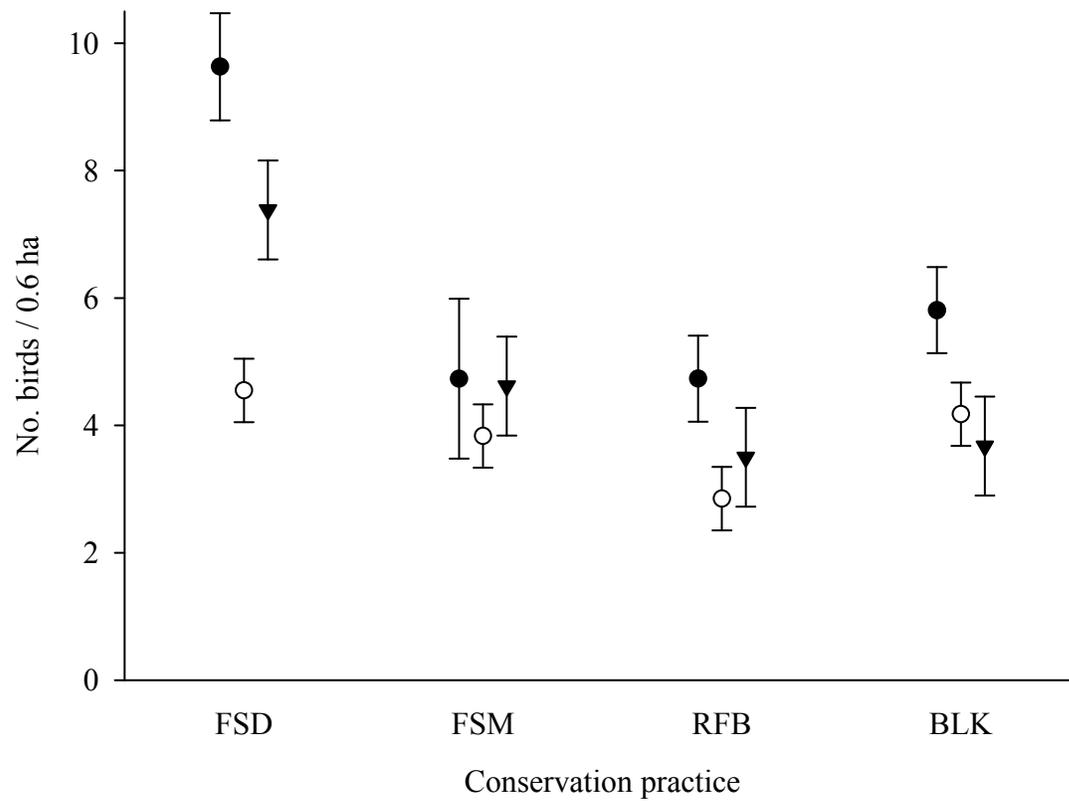Figure 4. Total avian conservation value (TACV) estimates for filter strips with diverse (FSD) and monotypic (FSM) planting regimes, early-succession riparian forest buffers (RFB) and hardwood blocks (BLK) during the summers (May-July) of 2005 (filled circles), 2006 (open circles), and 2007 (filled triangles) in the Mississippi Alluvial Valley, MS.

Figure 5. Shannon diversity index (*H'*) for filter strips with diverse (FSD) and monotypic (FSM) planting regimes, early-succession riparian forest buffers (RFB) and hardwood blocks (BLK) during the summers (May-July) of 2005 (filled circles), 2006 (open circles), and 2007 (filled triangles) in the Mississippi Alluvial Valley, MS.
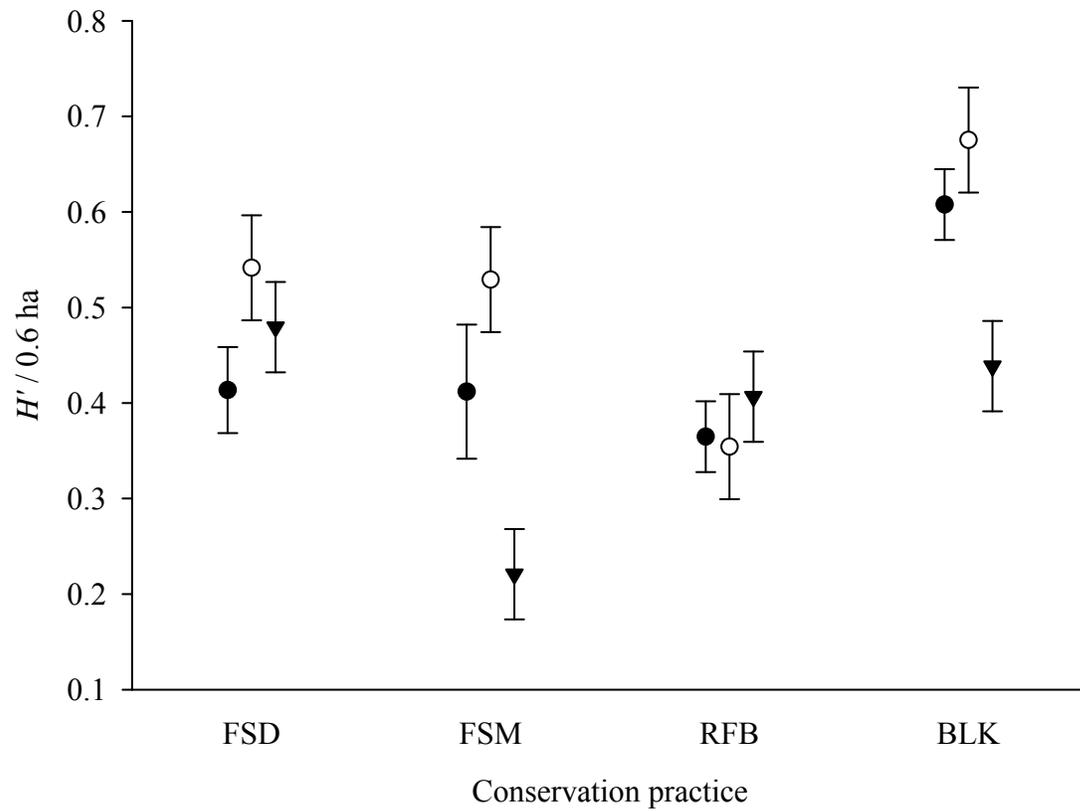
CHAPTER 3.
GRASSLAND BIRD NEST ECOLOGY AMONGST INTENSIVE AGRICULTURE: IMPORTANCE OF PATCH DESIGN

ABSTRACT

Agricultural intensification and continued population declines in grassland birds have stimulated interest to establish suitable nesting habitat amongst intensive, rowcrop agriculture. To maximize compatibility in a production-based landscape, patch design may be constrained to narrow, strip habitats. An understanding of patch shape and area, vegetation structure, and landscape context is necessary to optimize management decisions for nesting grassland birds. Nest searching and monitoring occurred in four conservation practices with accompanying vegetation surveys at random locations and nest sites in northwest Mississippi. Conservation practices included: 1) large blocks of structurally diverse, early-succession vegetation, 30-m wide filter strips established as either 2) a monotype of tall, dense switchgrass or 3) a more diverse mixture of one forb and three grass species, and 4) 60-m wide, early-succession riparian forest buffers. I modeled Dickcissel and Red-winged Blackbird nest survival relative to local vegetation, landscape context, and conservation practice in Program MARK. During three breeding seasons (2005-2007), I monitored Dickcissel (n = 733) and Red-winged Blackbird (n = 409) nests in all four practices. Dickcissel nested in greater densities in the large blocks compared to buffers, whereas Red-winged Blackbird densities were less consistent and annually variable in all habitats. Dickcissel nests were detected more frequently in riparian forest buffers and diverse filter strips compared to monotypic filter strips, but survival was similar among conservation practices. I observed slightly higher Red-winged Blackbird nest densities in monotypic filter strips compared to other buffers. Dickcissel nest survival was positively related to nest height, but negatively related to rowcrop within 30 m, grass cover, horizontal vegetation density, and location in riparian forest buffers. Red-winged Blackbird nest success was higher in diverse filter strips (23.4%) than on average (8.6%) and was positively related to nest height and negatively influenced by percent cover of dead vegetation. Grassland bird recovery will require substantively increasing the amount of diverse native grasslands in agricultural landscapes. To maximize nesting grassland-bird benefits amongst intensive rowcrop agriculture, larger blocks of diverse native grasslands are necessary as the foundation for wildlife-friendly farming. Blocks of habitat can be complemented with conservation buffer practices (e.g., filter strips, upland habitat buffers) that increase connectivity and usable space. Buffers should be ≥30 m wide and planted with a diversity of forbs and grasses to maximize vegetative structural complexity. I further recommend the incorporation of perennial grasses in riparian forest buffer design to improve nest success.

INTRODUCTION

Continued and projected declines of bird populations are globally associated with agricultural land-use practices (Teyssèdre and Couvet 2007). This is evident in North America, where large-scale conversions of native grassland ecosystems to agriculture contributed to dramatic population declines of grassland birds (Askins 1993, Knopf 1994, Peterjohn and Sauer 1999, Askins et al. 2007). After land-use conversions stabilized in the 1950s (Noss et al. 1995, Lubowski et al. 2006), many grassland birds persisted in agriculture-dominated landscapes by exploiting remnant patches of non-crop vegetation (Warner 1994, Best 2000). Growing worldwide food demands facilitated intensified agricultural practices by stimulating the need for greater crop yield efficiency (Tilman et al. 2002, Robertson and Swinton 2005, Green et al. 2005), including the removal or degradation of many of these remnant patches (Warner 1994). The negative effects of agricultural intensification to bird communities on farmlands highlight our need to understand alternative options that mitigate these impacts (Donald et al. 2001, Wilson et al. 2005). Wildlife-friendly farming may reverse these effects by integrating patches of natural vegetation amongst agriculture to enhance biodiversity with minimal impact on agricultural yield (Green et al. 2005). The U.S. Department of Agriculture (USDA) has pioneered this replacement of lost habitat in agricultural landscapes, primarily with economic incentives through the Conservation Reserve Program (CRP). The CRP has helped stabilize many grassland bird populations, although continued declines of some species highlight opportunities for program improvement (Ryan et al. 1998, Koford 1999, Herkert 2007).

An obstacle to these conservation efforts is the need to balance wildlife benefits and landowner economics. This balance often constrains the area, design, and context of established patches to a high perimeter-to-area ratio (Helzer and Jelinski 1999), small overall area (Vickery et al. 1994, Winter and Faaborg 1999), and close proximity to habitat edges (Gates and Gysel 1978, Suarez et al. 1997, Heske et al. 1999). These small, strip-shaped patches on private, agricultural lands may be vulnerable to negative edge or area effects that can confound intended benefits for grassland birds (Gates and Gysel 1978, Vickery et al. 1994, Winter and Faaborg 1999, Best 2000). Edge effects typically occur within 50 m of a land-cover discontinuity and are defined by elevated rates of brood parasitism and unnaturally high nest predator densities that may reduce nest success (Paton 1994). Habitat edges are classified as abrupt (e.g., woodland-grassland) or gradual (e.g., agriculture-grassland) and amongst agriculture, nest predation near abrupt edges can be twice as high compared to gradual edges (Suarez et al. 1997, Chalfoun et al. 2002). Some grassland birds may also exhibit stronger avoidance behavior of grasslands near abrupt edges than gradual edges (Jensen and Finck 2004). These severe edge effects are a conservation concern when occurring in attractive habitat

because they may cause habitat patches to function as ecological traps (Gates and Gysel 1978, Suarez et al. 1997, Woodward et al. 2001). Area effects denote the relationship between a patch's colonization potential and its total area; smaller patches may have lower nesting bird species richness from reduced colonization rates of area-sensitive species (Vickery et al. 1994, Winter and Faaborg 1999); although for grassland birds, colonization may also be mediated by conspecific attraction (Ahlering and Faaborg 2006) or total available habitat in the landscape (Ribic and Sample 2001). Grassland birds may also experience lower nest success in smaller patches from increased nest predation (Johnson and Temple 1990). However, most research on bird nesting success in strip habitats has been conducted in narrow fencerows, waterways, and field borders typically less than 10 m wide. Little research has examined population performance of birds using wider conservation buffer practices as one component of comprehensive conservation management system that includes both block and buffers habitats.

Despite the potential drawbacks, conservation buffers (i.e., strips of non-crop vegetation) can benefit grassland birds by providing foraging habitat, movement corridors, nesting habitat, and escape cover (Best 2000, Marcus et al. 2000, Smith et al. 2005, Puckett et al. 2000, Conover 2005). Furthermore, their contribution to total grassland area may enhance landscape suitability for grassland birds (Ribic and Sample 2001). The potential of conservation buffers to attract high bird densities can translate to significant avian benefits in intensive agricultural landscapes (Camp and Best 1994, Patterson and Best 1996, Conover et al. *in press*). However, most buffer studies have been limited to game species or avian community structure, and typically focus on just one conservation practice (Puckett et al. 2000, Smith et al. 2005, Riddle et al. 2008, Conover et al. *in press*). Hence, there remains a paucity of insight on non-game wildlife responses and productivity parameters in conservation practices that would enhance the reliability and scope of conservation decisions (Van Horne 1983, Vickery et al. 1992). Grassland bird nesting activity and nest success provide an indication of the demographic consequences associated with habitat establishment and management (Fletcher et al. 2006). Furthermore, conservation buffer management decisions are largely reliant on research conducted in the Midwest (Warner 1994, Best 2000, Henningsen and Best 2005), although substantial amounts of buffers exist elsewhere that provide nesting habitat for grassland birds (Conover 2005, Clark and Reeder 2007). Conservation buffer practices in Mississippi are predominantly riparian forest buffers (i.e., CP22; 53,638 ha) and filter strips (i.e., CP21; 3,235 ha) (Clark and Reeder 2007), yet no research has been conducted on nesting birds in filter strips and a minimal amount exists for birds using riparian forest buffers in Mississippi (Twedt et al. 2002).

Local factors such as vegetation can also determine patch quality for grassland-nesting birds (Wiens 1969, Renken and Dinsmore 1987, McCoy et al. 2001). Nest-site vegetation can influence nest success by visual concealment from dense cover or abundant potential nest sites from vegetative complexity (Chalfoun and Martin 2009). Dickcissel (*Spiza americana*) nest success has been linked with local vegetation, including dead and live vegetative cover, vegetation height, grass cover, and overall ground cover (Hughes et al. 1999, Winter 1999). Other nest-site characteristics also influence survival; for example, both Dickcissel and Red-winged Blackbird (*Agelaius phoeniceus*) nests may survive better in nests that are higher off the ground (Harmeson 1974, Dinsmore and Dinsmore 2007). Spatial processes (e.g., habitat fragmentation, amount of grassland in the landscape) can strongly influence nest success and in some landscapes, may supersede local-scale effects (Donovan et al. 1997, Heske et al. 1999). Spatial characteristics can predict habitat use by nest predators in agricultural landscapes (Chalfoun et al. 2002, Larivière 2003, Thompson et al. 2002) and are therefore relevant to understanding grassland bird nest ecology in strip habitats (Warner 1994). Habitat fragmentation and proximity to wooded edges are two primary factors that have negatively impacted Dickcissel nest success (Winter and Faaborg 1999, Dixon et al. 2008).

The Dickcissel and Red-winged Blackbird are grassland-nesting birds that may exhibit area sensitivity (Bakker et al. 2002, Kammin 2003), yet are also habitat generalists that will exploit grassland patches amongst intensive rowcrop agriculture in Mississippi (Herkert 1994, Yasukawa and Searcy 1995, Temple 2002, Conover et al. *in press*). Both species are polygynous; have altricial young; construct open-cup nests interwoven among branches in the center of grass, forb, or woody substrates; prefer thick compared to sparse vegetation; and experience high rates of nest predation (Yasukawa and Searcy 1995, Temple 2002). Contrasts in their nest ecology include that Dickcissel nests are typically closer to the ground, have greater surrounding vegetation density and overhead concealment, and they silently depart their nest when a predator approaches, whereas Red-winged Blackbirds respond with alarm calls and mobbing (Yasukawa and Searcy 1995, Temple 2002). Given their similarities in breeding habitats yet contrasting reproductive strategies, they provide a functional multi-species approach to evaluate the quality of established grasslands based on nest density and survival. These species are also of conservation interest as the Dickcissel is a Partners in Flight species of continental concern (Panjabi et al. 2005) and has undergone slight population declines (-0.3%/year from 1966 to 2007; Sauer et al. 2008), whereas the Red-winged Blackbird has undergone significant population declines (-0.9%/year) throughout the U.S. from 1966 to 2007 (Sauer et al. 2008). However, even large CRP fields and extant remaining grasslands may be population sinks for both species (McCoy et al. 1999, With et al. 2008).

My study evaluated grassland-bird nesting use of strip and block early-succession habitats in an intensive, rowcrop agricultural landscape. I chose two abundant grassland birds (Dickcissel and Red-winged Blackbird) with similar, but contrasting life-history strategies to examine the relationship between nesting activity and nest success in four established conservation practices. Biological variables of interest included year, seasonal variation, nest stage, precipitation, local vegetation, nest-site characteristics, and spatial information at fine (30 m) and coarse (400 m) scales. These variables were evaluated relative to conservation practice to determine overall importance of patch vegetation and landscape context for nesting grassland birds in an intensive agricultural landscape.

METHODS

Study site.—My study took place on a 2,630 ha farm located in Coahoma County, Mississippi, USA (34° 18' N, 90° 34' W), which is in the MAV (Mississippi Alluvial Valley; Bird Conservation Region 26). This farm exemplifies the current MAV landscape, which is dominated by large fields of ditch-to-ditch, rowcrop agriculture of primarily cotton (*Gossypium* sp.) and soybean (*Glycine* sp.) that are fragmented by a network of drainage ditches, streams, wood lines, and fencerows, with negligible topographic relief. However, the farm differed from the surrounding landscape in that it recently established various semi-natural vegetation patches to promote wildlife-friendly farming and consisted of 48% rowcrop, 30% early-succession hardwood afforestation plantings, 14% forested or herbaceous wetlands, 4% conservation buffers, 2% wooded, and 2% herbaceous drains. Soybean was the only farm crop in 2005 and 2007, whereas 2006 had early (May-early June) wheat and a late (mid-June) soybean planting. All rowcrop fields on the farm endured similar tillage regimes. Soil associations on the farm included Sharkey clay (41%), Dowling clay (15%), and Tunica silty clay (11%) (Soil Survey Staff 2008). Rainfall primarily occurred late in the breeding season of 2005 (May = 2.72 cm, June = 1.83 cm, July = 17.20 cm), early in 2006 (May = 13.21 cm, June = 2.72 cm, July = 3.53 cm), and was more consistent and greater overall in 2007 (May = 14.99 cm, June = 5.00 cm, July = 14.96 cm).

Conservation practices.—I monitored bird nesting activity in four distinct early-succession, conservation practices that varied by vegetation structure and patch shape. Practices included (1) diverse filter strips (FSD) planted with a forb-grass mixture, (2) monotypic filter strips (FSM) planted with only switchgrass (*Panicum virgatum*), (3) early-succession, riparian forest buffers (RFB), and (4) early-succession, hardwood afforestation blocks (BLK). I randomly selected 20 study plots for each conservation practice from a pre-determined, farm-wide sample. Sample sizes were less for FSD (n = 14) and FSM (n = 6) in 2005, the first year of study implementation.

Both filter strip practices were 30 m wide and planted in spring 2004. FSD was planted with partridge pea (*Chamaecrista fasciculata*, 4.5 kg/ha seeding rate), Indian grass (*Sorghastrum nutans*, 1.7 kg/ha seeding rate), little bluestem (*Schizachyrium scoparium*, 5.6 kg/ha seeding rate), and big bluestem (*Andropogon gerardii*, 1.7 kg/ha seeding rate), whereas FSM was seeded at a high density (9.0 kg/ha) of only switchgrass. Riparian forest buffers were planted 60 m wide in the fall of 2004. Conservation buffers were established in similar landscape contexts, located on a field margin between a riparian zone (e.g., drainage ditch, stream) and a rowcrop field. Landscape context differed for some RFB and FSD plots, as they were juxtaposed to either herbaceous or wooded riparian zones. BLKs were established in the fall of 1999 and thus had enhanced woody growth, but retained primarily herbaceous vegetation. BLK and RFB were predominantly planted with Nuttall's oak (*Quercus nutallii*), water oak (*Quercus nigra*), and willow oak (*Quercus phellos*). All habitats were invaded at differing levels with local, non-planted vegetation that predominantly included horsetail (*Conyza canadensis*), redvine (*Brunnichia cirrhosa*), vetch (*Vicia* sp.), goldenrod (*Solidago* spp.), giant ragweed (*Ambrosia trifida*), curly dock (*Rumex crispus*), dewberry (*Rubus trivialis*), blackberry (*Rubus oklahomus*), johnsongrass (*Sorghum halepense*), poison ivy (*Toxicodendron radicans*), broomsedge (*Andropogon virginicus*), honey locust (*Gleditsia triacanthos*), American elm (*Ulmus americana*), and sugarberry (*Celtis laevigata*). Although filter strips will be maintained with a disturbance regime, no management had yet been implemented.

Nest searching and monitoring.—An intensive nest search effort ensued from 15 May to 20 July during three years (2005-2007). Two to four people searched for nests by systematically walking through each plot with 2 m between each person and mildly disturbing the vegetation with 1 m sticks to flush brooding birds. All plots were searched with equal effort in proportion to their area to permit relative nest density comparisons. I found nests by visual detection, flushing parents, or behavioral cues (e.g., carrying food or nesting material, alarm calls). Search protocol changed for FSM in 2007 to a rope-drag technique because the vegetation was too thick and tall to effectively search from within the buffer. This should not bias nest density results because high vegetative cover during the previous two years caused nest searching to occur primarily based on flushing bird observations, and rope dragging uses the same approach (Wiens 1969). To ensure prompt relocation and minimize disturbance to nests and surrounding vegetation, I recorded their location with a handheld Global Positioning System (GPS, GeoExplorer II; Trimble Navigation, Sunnyvale, California). I supplemented re-location efforts with maps of nearby vegetative landmarks (e.g., patch of goldenrod) and by tying 10-cm pieces of orange flags to the base (below the herbaceous canopy) of

vegetative substrates at 5 m to the north and south. Flags were mostly concealed and served as a backup locator in the event of GPS failure or nest destruction.

I monitored nests every 2-4 days, primarily in the afternoon, after dew had evaporated to minimize scent trails, and used care to minimize trampling vegetation or creating dead-end trails that may cue nest predators (Martin 1993). Monitoring data included nest stage (laying, incubation, and nestling), clutch/brood size, nest age (in days), parent behavior, and nest fate. I estimated age during incubation by candling all eggs in the clutch and recording the age of the most developed embryo (Lokemoen and Koford 1996), and during the nestling stage from species-specific voucher images (R. Conover, unpublished data). Egg-candling procedures included a 9-cm piece of ¾-cm pipe insulation directed towards the sun to observe embryo development. I classified failed nests as depredated, abandoned, weather-caused, human-caused, or unknown. Nest fate determinations accounted for nest condition (i.e., disturbance), nest-site clues (e.g., feces, feathers, eggshells, etc.), parent behavior (e.g., alarm calls or food carrying), fledgling presence, and nestling age at previous visit. I considered nests successful in the incubation and nestling stages if ≥1 egg hatched or nestling fledged, respectively. Nest visitation occurred on or after the expected fledging date to increase the accuracy of nest fate conclusions. Observed nest depredation events are rare and nest condition can be useful to provide a crude classification on nest predator types (Peterson et al. 2004). I classified nests as intact (i.e., no nest damage) or torn (i.e., nest visibly damaged) to offer insight on the cause of depredation by snakes or mammals, respectively (Thompson and Burhans 2003, Peterson et al. 2004). I report minimal relative nest density as the number of nests detected per 100 ha. Nest density was compared between buffers adjacent to wooded and non-wooded vegetation to determine the effect of nearby mature wooded habitat on nest distributions. These estimates assume all active nests had equal detectability across habitats and hence; caution should be used in interpretation.

Nest-site characteristics.—I measured a suite of variables associated with the immediate nest-site because previous studies have demonstrated their importance to nest success (Martin 1993). Nest-site variables included nest height (Harmeson 1974), nest concealment (Blankespoor 1970), and surrounding vegetation structure (Hughes et al. 1999, Winter 1999). Nest height is the distance from the ground to the nest bowl rim, and nest concealment is the percentage of total cover (live or dead) directly above the nest bowl. I quantified nest-site vegetation in an 8-m diameter, circular plot centered on the nest. Vegetation measurements included vertical cover (visual obstruction reading, VOR), horizontal vegetation density, vegetation structure (percentage of live cover types), and percentage of standing dead vegetation cover. VOR indicated effective vegetation height (where visibility was obstructed by vegetation), which was recorded to the nearest centimeter using a

modified-Robel pole placed at the nest site, with the metric calculated as the average of four
measurements (one at each cardinal direction) at a 4 m distance and 1 m height (Robel et al. 1970,
Renken and Dinsmore 1987). Horizontal density is the average percentage of total cover from each
of four quadrants within the 8-m diameter circular plot and was visually estimated after walking
within each quadrant. Live cover percentages are visually estimated proportions of total live cover
that include forb, grass (i.e., grass-like structures), and woody substrates, whereas standing dead
vegetation is an independent estimate of the proportion of total dead cover. I recorded nest-site
vegetation measurements 25 days (±5 days) after nest initiation to standardize potential bias from
vegetative growth. I also recorded the number of active red imported fire ant (RIFA, *Solenopsis
invicta*) mounds within 10 m and report these estimates as mound density (mounds/ha). Mound
activity was determined by poking a stick through the mound surface and observing whether or not
RIFA subsequently emerged. Vegetation from nest sites in 2007 was compared with surveys from
random locations within each conservation practice during 2007. Random vegetation surveys were
conducted during the peak of nest initiation (mid-late June) to capture vegetation characteristics that
are maximally comparable to nest-site vegetation.

   Landscape variables.—I characterized nests at a larger spatial scale because studies have
shown that landscape context is important for nest success (Donovan et al. 1997, Winter and Faaborg
1999, Thompson et al. 2002). I downloaded nest locations using GPS Pathfinder Office© (version
2.90), and exported into ArcGIS© (version 9.3, ESRI, Redlands, California). Nest locations were
spatially referenced using the North American Datum 1983 (zone 15-North) and Universal
Transverse Mercator Zone 15N coordinate system. GPS locations were differentially corrected using
local (<100 km) base station data downloaded through GPS Pathfinder Office to maximize nest
location accuracy. I calculated nest distances to wooded and nearest edges using the point distance
function in ET GeoWizards 9.8 in ArcMAP©. I classified the wooded edge as mid-late succession
(≥15 years old) woody patches and the nearest edge as any land-cover discontinuity. Three years
(2005-2007) of digital orthophoto quadrangles from the National Agriculture Imagery Program and
updated information on USDA-Farm Services Agency conservation practice enrollment provided data
to permit land-cover digitization. I used ArcINFO to buffer and clip spatially explicit, land-cover
information from Geographic Information System (GIS) vector data at fine (30 m) and coarse (400 m)
landscape scales surrounding each nest. Fine scale buffer distance is based on approximate radius of
Dickcissel and Red-winged Blackbird breeding territories (Yasukawa and Searcy 1995, Temple
2002). I selected 400 m as the landscape scale buffer radius because this distance is correlated with
predation risk for an open-cup songbird nest relative to landscape-level characteristics (Kus et al.

2008). I calculated land-cover composition, total patch area, and total edge using FRAGSTATS (McGarigal et al. 2002). Land-cover composition is important to define landscape context and included rowcrop, herbaceous, early-, mid-, and late-succession woody, and developed (i.e., roads and homesteads) classes. Other land-cover types existed (e.g., wetland), but were not within 400 m of any nests. Total edge measures the degree of spatial heterogeneity (i.e., fragmentation) in the landscape, with greater edge indicating increased habitat fragmentation (McGarigal et al. 2002).

   Nest survival analyses.—I used maximum likelihood estimation with a logit link function in Program MARK to model nest survival (i.e., daily nest survival probability) of Dickcissel and Red-winged Blackbird nests (White and Burnham 1999, Dinsmore et al. 2002). I evaluated daily nest survival across the nesting seasons of each species, with the earliest discovered nest standardized as day one. Nest survival data were modeled from 13 May to 16 August (96 d) for Dickcissel and 10 May to 12 August (95 d) for Red-winged Blackbird across the three study years combined. Modeling procedures and covariates were identical for both species. I modeled effects of year, nest stage (egg-laying, incubation, and nestling), and patch treatment (BLK, RFB, FSD, and FSM) in MARK as groups. Model construction applied *a priori* determined biologically important variables that included nest-site vegetation structure, nest height, overhead concealment, clutch size, and fine and coarse scale spatial characteristics. I imputed missing covariate values using means from nests of the same species, year, and vegetation treatment. Imputed values for Dickcissel included nest concealment (n = 24; 3%), nest height (n = 4; <1%), and nest vegetation (n = 42; 6%). Imputed values for Red-winged Blackbird included nest concealment (n = 13; 3%), nest height (n = 2; <1%), and nest vegetation (n = 8; 2%). Nest survival data of late-season nests were right-censored, which caused most missing covariate measurements as vegetation surveys were conducted 25 days after clutch initiation.

   I chose a hierarchical modeling approach because my analyses included multiple sources of spatial and temporal variation in nest survival and I wanted to keep the model set small to minimize the risk of over-fitting the data (Burnham and Anderson 2002). Furthermore, predation effects can operate on multiple scales, with larger scale processes constraining effects on bird productivity that occur at smaller scales (Thompson et al. 2002). I constructed models in four hierarchical steps that included 1) additive and interactive effects from year, nest stage, and seasonal time trends (linear and quadratic), 2) patch treatments, 3) landscape variables (fine and coarse scales) and edge proximities, and 4) local vegetation and nest-site characteristics, with daily precipitation added to the best model. In step one I evaluated daily nest survival rate (DSR) as an interactive and additive function of year and nest stage, and seasonal variation within years as a constant (no time trend), logit-linear time

trend, or logit-quadratic time trend. I used information theoretic approaches for model selection with the adjusted Akaike Information Criterion (AIC_c) as the selection tool (Akaike 1973, Burnham and Anderson 2002). In the above hierarchical procedure, the best and all competitive ($\Delta AIC_c \leq 2.0$) models were advanced to subsequent steps as reference models. Covariates were added singly to reference models within steps to test for independent effects; combined effects were subsequently evaluated for covariates that competed ($\Delta AIC_c \leq 2.0$) with reference models. I computed the ratio of differences in log-likelihood values as an estimate of the proportion of deviance explained by the best model as compared with the global model (Skalski et al. 1993, *but see* Dinsmore et al. 2003). This calculation compared the best model, the global model, and the constant (no covariates) model as an approximate measure of model fit. The constant model represents the Mayfield estimate with only one parameter (Mayfield 1961, 1975), whereas the global model represents the full amount of variation documented by all measured variables, which included full year, nest stage, and seasonal effects in addition to all biological covariates for Red-winged Blackbird (128 parameters) and Dickcissel (171 parameters). I represented seasonal nest survival patterns for each species by plotting time-specific nest survival relative to the most influential environmental variable and conservation practice, using values derived from the incubation stage during 2007. I chose incubation and 2007 because they represent the middle values of nest survival for stages and years, respectively.

Nest success estimates represent the probability of a nest surviving across the entire nesting period and thus, are distinct from DSR. I calculated nest success estimates using the best approximating models for each species to predict the influence of individual covariates and their combinations. These estimates were calculated based on the range of observed covariate values and with other covariate values held constant (Shaffer and Thompson 2007). Nest success estimates were standardized at a start date of 1 June 2007, which was peak nest initiation. Nest success was calculated using empirically-based nesting periods for both species. These stage-specific (i.e., incubation and nestling) time periods were calculated using only nests found before stage initiation and monitored through stage success, except egg-laying. Egg-laying time periods were derived from average clutch sizes, as both species lay one egg/day and begin incubation with the penultimate egg (Yasukawa and Searcy 1995, Temple 2002). Dickcissel mean clutch size was 3.83 ± 0.62 (n = 135). Hence, Dickcissel nest success was based on a 19 day nest period (2 d egg-laying, 10 d incubation, 7 d nestling). Red-winged Blackbird mean clutch size was 3.01 ± 0.76. Hence, Red-winged Blackbird nest success was based on a 21 day nest period (1 d egg-laying, 11 d incubation, 9 d nestling).

I made predictions on the outcome of this study based on current understanding of grassland bird ecology amongst agriculture. My hypotheses included that (1) nest density will be greater in

block-shaped patches, (2) nest survival will positively correlate with patch area, vegetative cover, and structural complexity, and (3) nest survival will inversely correlate with distance to an abrupt edge.

RESULTS

Nesting activity.—I found a total of 1,314 nests of 14 species during the 3-year study, including 376 nests of 8 species in 2005, 554 nests of 9 species in 2006, and 384 nests of 12 species in 2007. Dickcissel (55.8%, n = 733) and Red-winged Blackbird (31.1%, n = 409) were the two most prevalent nesting species across all conservation practices, and Mourning Dove (5.1%), Eastern Meadowlark (3.8%), and Northern Bobwhite (1.8%) nests were also frequently detected (Appendix A). Dickcissel nests were common in all years (n = 149, 2005; n = 340, 2006; n = 244, 2007), whereas Red-winged Blackbird nesting activity declined annually (n= 179, 2005; n = 147, 2006; n = 88, 2007). Dickcissel relative nest density was at least 3.6 times greater in BLK than any conservation buffers during any year, although nest densities increased annually in all buffers (Table 1, Figure 1). Dickcissel regularly nested in RFB and FSD but almost completely avoided FSM until 2007. Red-winged Blackbirds nested frequently in all treatments during at least one year of the study, with relative nest densities varying amongst treatments and years (Table 1, Figure 2). They nested most frequently in BLK and FSD in 2005, but densities in those treatments declined annually and their highest nest densities occurred in FSM during 2006 and in RFB during 2007 (Table 1, Figure 2). Red-winged Blackbirds had similar nest densities adjacent to non-wooded and wooded vegetation, respectively, in RFBs (nests/100 ha ± 95% CI; 103.17 ± 60.14, n = 23; 64.81 ± 41.20, n = 27) and filter strips (nests/100 ha ± 95% CI; 93.33 ± 35.21, n = 75; 86.67 ± 56.17, n = 25) across years. Dickcissel nest densities were also slightly greater in RFBs adjacent to non-wooded than wooded field margins, respectively, (nests/100 ha ± 95% CI; 65.22 ± 32.40, n = 23; 37.04 ± 34.14, n = 27), but exhibited the opposite pattern in filter strips (nests/100 ha ± 95% CI; 3.11 ± 13.07, n = 75; 63.33 ± 30.23, n = 25).

Seventy-eight percent of all Dickcissel nests in conservation buffers were initiated before 31 May, with only 5% initiated after 15 June. Their nesting phenology in buffers differed from blocks, in which nest initiation occurred consistently from early May to 15 July (Figure 3a). Red-winged Blackbird nest initiation occurred primarily from 15 May to 15 June in buffer (54% of nests initiated) and block (70% of nests initiated) treatments (Figure 3b). Dickcissel continued to nest in blocks later in the breeding season, whereas Red-winged Blackbirds late-season nesting activity was mostly in buffers (Figure 3).

Nest-site characteristics.—Grass cover dominated FSM and FSD although FSD had moderate (26%) forb cover. Forbs dominated vegetation in RFB (74%), but BLK had relatively

similar amounts of grass and forb cover (Table 3). Woody cover was at least 20 times greater in BLK and RFB than filter strips, although woody substrates were considerably taller in BLK from advanced maturation. Nest-site substrates varied greatly between conservation practices. In BLK, Dickcissel nested mostly in woody vegetation (90.4%) and Red-winged Blackbirds commonly used both woody (66.4%) and forb (30.9%) substrates. Dickcissel nests in BLK were mostly in small oaks (n = 306, 50%) and sugarberry (n = 193, 32%), whereas Red-winged Blackbird nests were mostly in small oaks (n = 61, 27%), goldenrod (n = 52, 23%), and American elm (n = 42, 19%). In RFB, Dickcissel and Red-winged Blackbirds, respectively, nested frequently in forb (59.7%, 47.4%), woody (23.4%, 31.6%), and vine (14.3%, 17.1%) substrates, and Dickcissel's preferred forb was goldenrod (43%). Dickcissel nested mostly in grasses in monotypic (75%) and diverse (69.4%) filter strips, and Red-winged Blackbirds were similar in FSM (77.9%), but nested more frequently in forbs (54.5%) than grasses (34.1%) in FSD. Overhead nest concealment was consistently greater for Dickcissel nests and ranged from 1.4 (RFB) to 3.7 (FSM) times greater than that of Red-winged Blackbirds (Table 2). Red-winged Blackbirds nested higher than Dickcissels in all habitats and were approximately 2 times higher in all buffers, but similar in BLK (Table 2). Dickcissel clutch sizes were similar in FSD and RFB, slightly smaller in BLK, and undetermined in FSM from lack of data. Red-winged Blackbird clutches were slightly larger in RFB and FSD than BLK or FSM (Table 2). Nest-site vegetation for both species generally had greater vertical cover than at random (Table 3). In RFB, both species' nests had greater horizontal vegetation density than expected at random and Red-winged Blackbirds selected in favor of dead vegetation cover, but slightly against woody cover (Table 3). Overall, both species selected sites with greater vertical and woody cover and less grass cover (Table 3).

Nest survival.—Depredation was the leading cause of nest failure for Dickcissel (87.2%, 387 of 444 losses) and Red-winged Blackbird (90.2%, 323 of 358 losses). Observed Dickcissel nest depredation events included RIFA (n = 19), speckled kingsnake (*Lampropeltis getula holbrooki*, n = 3), and nine-banded armadillo (*Dasypus novemcinctus*, n = 1), and nest condition after unobserved events was torn (n = 53) and intact (n = 169). I observed Red-winged Blackbird nest depredation by RIFA (n = 9) and southern black racer (*Coluber constrictor priapus*, n = 1), and nest condition after unobserved events was torn (n = 50) or intact (n = 82). Dickcissel nest outcomes included depredation (n = 387), abandoned (n = 45), right censored (n = 23), failed from Brown-headed Cowbird (*Molothrus ater*) parasitism (n = 7), human-caused failure (n = 3), weather-caused failure (n = 2), and other failure (n = 1). Only 2.6% (n = 17) of Dickcissel nests were parasitized by cowbirds in all conservation practices across years; 10 parasitism events did not cause failure. Red-winged

Blackbird nest outcomes included depredation (n = 323), abandoned (n = 30), human-caused failure (n = 3), weather-caused failure (n = 1), and other failure (n = 1).

My nest survival data included an effective sample size of 6,454 nest days for Dickcissel and 2,872 for Red-winged Blackbird. In the first step of my hierarchical model selection, nest survival was best explained for both species by the interaction between year and nest stage as well as a quadratic seasonal time trend (Tables 4, 5), which indicated higher nest survival early and late in the season, but lower during peak nest initiation from late-May to mid-June (Figures 4, 5). Dickcissel models were improved in step 2 by adding RFB and BLK conservation practices, largely uninfluenced by land-cover types, but slightly improved by rowcrop within a 30 m radius and total patch area for models with RFB (Table 4). Horizontal vegetation density, % grass cover, and nest height improved Dickcissel nest survival models in the final step with nest-site characteristics (Table 4). Overall, Dickcissel nest survival was negatively related to RFB (β_{RFB} = -0.399 on a logit scale, SE = 0.182, 95% CI = -0.756, -0.041), rowcrop within 30 m (β_{30Row} = -1.879, SE = 1.365, 95% CI = -4.554, 0.797), horizontal vegetation density (β_{HD} = -0.005, SE = 0.002, 95% CI = -0.010, -0.001), and grass cover (β_{Grass} = -0.004, SE = 0.002, 95% CI = -0.007, -0.0003), but positively with nest height (β_{NH} = 0.007, SE = 0.002, 95% CI = 0.003, 0.012). The best model for Dickcissel explained 15.2% of the proportion of deviances.

Red-winged Blackbird nest survival models exposed a strong, positive relationship to FSD. Many land-cover covariates were included in competitive models, but most of the effects were non-significant (Table 5). Models were improved in the final step with the addition of dead vegetation cover and nest height. Nest survival was negatively related to dead vegetation cover ($\beta_{DeadVeg}$ = -0.008, SE = 0.004, 95% CI = -0.016, -0.001), but positively related to FSD (β_{FSD} = 0.549, SE = 0.337, 95% CI = -0.111, 1.209) and nest height (β_{NH} = 0.003, SE = 0.002, 95% CI = -0.0004, 0.007). Red-winged Blackbird models were only marginally improved with addition of conservation practice, land-cover, or nest-site covariates and the best model explained 7.9% of the proportion of deviances (Table 5).

Dickcissel nest success was 22.94% (95% CI = 10.82%, 36.40%) in all conservation practices, lowest in RFB (success = 10.60%, 95% CI = 2.62%, 24.78%) (Table 6), and most improved with elevated nest height (Table 6). Red-winged Blackbird nest success in all conservation practices was 8.61% (95% CI = 2.27%, 19.75%) and was best in FSD (Table 7).

DISCUSSION

Dickcissel and Red-winged Blackbird nested in all buffer and block conservation practices established amongst intensive, rowcrop agriculture; albeit with varied nesting densities, temporal use

patterns, and success. Their attraction to these vegetative patches provides reason for promise and caution regarding grassland bird conservation and associated management practices. Below, I discuss the relevance of my results to nesting activity, nest-site characteristics, and nest survival as a basis for recommendations about changes in conservation practices to benefit these and other grassland birds.

Nesting activity.—Information on nesting activity provides wildlife managers with an understanding of species-specific nesting habitat use. BLKs had enhanced vegetation structural diversity and attracted greater overall nesting activity than any conservation buffers, particularly for Dickcissel. This increased nesting activity in BLKs supports my first hypothesis and may be attributed to area-sensitive behavior (Vickery et al. 1994, Winter and Faaborg 1999) and local vegetation features (Wiens 1969, Renken and Dinsmore 1987). Enhanced nesting use of BLKs by Dickcissel is probably related to their preference to place elevated nests in woody plants (Overmire 1962), which were scarce in buffers. However, Dickcissel nest sites are also defined by tall, dense vegetative cover (Zimmerman 1971, Renken and Dinsmore 1987), which was comparable or less in BLKs than buffers (Table 3), especially during the early nesting season (R. Conover, personal observation), which may explain their initial attraction to conservation buffers. Red-winged Blackbirds nested abundantly in blocks during 2005 and 2006, yet their nest densities decreased annually, which suggests diminished nesting suitability in BLKs for this species at approximately eight years post-establishment.

Lower overall nest densities in buffers likely relates to some buffers being adjacent to wooded edges, as Dickcissels (Hughes et al. 1999, O'Leary and Nyberg 2000, Jensen and Finck 2004) and to a lesser extent, Red-winged Blackbirds (Kammin 2003), are reported to exhibit lower nest densities near wooded vegetation. Although I observed similar results, the nest density differences between wooded and non-wooded buffers were weak and Dickcissel actually had higher nest densities in filter strips adjacent to wooded habitat. Regardless, my estimates of nest density in buffers (0-183 nests/100 ha and 67-375 nests/100 ha) are lower than previously reported for Dickcissel and Red-winged Blackbird, respectively, in similar strips (i.e., grassed waterways) with no adjacent wooded edges in the Midwest (276 nests/100 ha and 556 nests/100 ha; Bryan and Best 1994). Elevated nest density estimates for Dickcissel in the Midwest may correspond with being in closer proximity to the core of their range (Husak and Linder 2004). Henningsen and Best (2005) also reported higher Red-winged Blackbird nest density in Iowa warm-season grass filter strips adjacent to non-wooded edges (1,141.9 ± 248.9 nests/100 ha) and slightly greater adjacent to wooded edges (181.6 ± 22.6 nests/100 ha ± SE). Their Dickcissel nest density estimates are comparable (153.8 ± 38.1 nests/100 ha) to this study although I observed considerably greater Dickcissel nesting

activity in filter strips adjacent to wooded edges than in the Midwest (Henningsen and Best 2005). This observation may be attributed to my study having on average, slightly wider filter strips and greater vegetative vertical density (Conover 2005). Furthermore, I documented greater Dickcissel nest densities in buffers during at least one year than was recorded in cool-season grass filter strips (68.5 ± 23.0; Henningsen and Best 2005). Hence, I propose that filter strip practices in the MAV and Midwest may have similar conservation potential for these species. Although conservation practices were all primarily composed of early-succession, herbaceous vegetation, I suggest using caution when interpreting effects of patch shape from differences in patch age and planting regimes.

In comparison to blocks, filter strips on the study farm had limited vegetation structural complexity, which corresponds to their planting regime of high grass density intended for erosion control. This may have caused the increased nesting activity early in the breeding season, as Dickcissels require standing dead vegetation for territorial male perches and female nesting sites early in the nesting season (Zimmerman 1971, 1982) and the absence of sufficient nesting cover can delay nesting activity (Robel et al. 1998). Red-winged Blackbirds had variable year effects in their buffer nest use and did not exhibit a preference toward any particular buffer practice. This pattern may relate to their generalist habitat selection behavior, as they were previously reported to be impartial to strip habitat type or management regime in southern Québec (Deschênes et al. 2003). Although untested, I speculate their apparently neutral preference for strip habitats may be partially explained by drainage ditch characteristics, as they regularly loitered in these adjacent ditches (R. Conover, personal observation). Red-winged Blackbird nest densities declined after the first year in FSD and the second year in FSM, whereas Dickcissel nest densities increased in both filter strip treatments during the third year. Whether postponing filter strip disturbance until after the third year is beneficial to Dickcissels remains untested. Nesting opportunities in RFB may increase concomitantly with woody growth and increased vegetation diversity associated with maturation, as well as increased vegetative cover and height for Dickcissel (Zimmerman 1971). Riparian forest buffers were only two to four years old during this study and based on previous studies, they received more nesting use than expected (Twedt et al. 2002).

The increased early season use of buffers and diminished use thereafter by Dickcissels is potentially explained by greater vegetative growth and/or residual standing vegetation than BLKs, habitat copying (i.e., birds select patches based on previous inhabitant's reproductive success, Doligez et al. 1999), geographic dispersal of buffer-nesting females after nest failure (Walk et al. 2004), or seasonally enhanced crop growth. Habitat copying and geographic dispersal hypotheses are unlikely in filter strips, where Dickcissels experienced similar nest success to BLKs. I speculate this

pattern is best explained by a combination of local vegetation features and greater structural heterogeneity of buffer vegetation may result in more consistent seasonal nesting activity. Birds may also be redistributing across crop fields, although this was not tested. Further insight on the causal mechanism of this trend could be determined by monitoring Dickcissel nesting activity during the early breeding season in buffers post-management (Robel et al. 1998).

Nest-site characteristics.—Dickcissel nests had greater overhead concealment than Red-winged Blackbird nests in all conservation practices, which supports previous observations of their preference for complete overhead cover (Temple 2002). Notwithstanding, my estimates of overhead concealment in RFB and FSD were only 50-55% and with concomitantly lower relative vegetative cover and height this may suggest the local vegetation was not ideal for nesting Dickcissels (Gross 1921, Harmeson 1974, *but see* Dechant et al. 2003). Dickcissel also typically prefer habitats with moderate to high amounts of forb cover, which existed in RFB and BLK, but was less in filter strips (Dechant et al. 2003). Given the moderate to high amounts of overall vegetation and forb cover in concert with more complex vegetative structure, advanced woody growth, and larger overall area, it is not surprising that the greatest densities of Dickcissel nests occurred in BLKs (Delisle and Savidge 1997, Winter and Faaborg 1999). As vegetative maturation continues on RFBs, I anticipate an increasing attractiveness to nesting Dickcissels from greater vegetative height and cover and taller woody substrates.

Red-winged Blackbirds generally placed their nests higher off the ground, in taller vegetative substrates, and in thicker vertical cover than Dickcissels. Similar to Dickcissels, their selection for enhanced vertical cover and moderate amounts of grass and forb cover than at random suggests that continued maturation in RFB and disturbance management to increase vegetative heterogeneity in filter strips may improve the suitability of these strips for nesting Red-winged Blackbirds. Red-winged Blackbird and Dickcissel nest sites had similar dead vegetation cover as random sites in buffers, but less than random sites in BLK. Overall, both species selected for similar vegetative structure and had comparable forb, grass, and woody cover surrounding their nests. These data provide no indication that either species exhibited avoidance behavior regarding RIFA mound densities.

Nest survival.—An understanding of nesting habitat use helps wildlife managers perceive land-cover and local vegetation features from the species' perspective, but may be misleading without a correlation to reproductive performance (Van Horne 1983, Vickery et al. 1992). This study documented 22.9% Dickcissel nest success across all conservation practices on the farm, which is higher than the 13.2% and 14.9% reported in Kansas CRP fields (Hughes et al. 1999), comparable to

tallgrass prairie fragments in Missouri (19.7% and 39.8%, Winter 1999), and lower than CRP fields in Missouri (29.7%) that were considered sink habitat (McCoy et al. 1999). In contrast to these studies, I estimated nest survival using modeling techniques that incorporated information on local vegetation and land-cover features to permit a deeper inspection of environmental variables that influenced nest survival. My data indicates a significant negative relationship between Dickcissel DSR and RFB, although this effect is not significant when interpolated to an overall nest success estimate. Nest success estimates highlight the potential for habitat management to improve local vegetation features based on Dickcissel reproductive benefits. Overall, these estimates demonstrate that Dickcissel nest success may be as low as 3% in RFBs of poor vegetative quality, but up to 38% in other conservation practices and sites that are not immediately adjacent to rowcrop fields, have reasonably tall (102.9 cm) vegetative substrates, moderate (36%) horizontal vegetative cover, and moderately low (12.4%) grass cover. Dead vegetation cover (Hughes et al. 1999) and total vegetative cover (Harmeson 1974, Zimmerman 1982) were previously shown to positively relate to Dickcissel nest success. I did not find support for either of these effects, in contrast to my predictions. I also observed a quadratic seasonal trend that indicated highest nest survival early and late, but lower survival during the middle of the season at peak nest initiation (Roos 2002).

Low Dickcissel nesting success in RFB (10.6%) provides reason for concern, as it was their preferred buffer for nesting (Table 1) and is the most abundant conservation buffer type in Mississippi (Clark and Reeder 2007). Although it warrants deeper inspection, this pattern indicates that 60 m wide, early-succession (<4 years old), riparian forest buffers may be ecological traps for nesting Dickcissels (Gates and Gysel 1978) on this farm. However, Dickcissel nest success could increase to 23% in RFBs given ideal vegetation conditions. This is still considerably less than previously reported for Dickcissels in 4-10 year old, oak-planted RFBs in the MAV, which was 36.3% success in a 19-day nest period (Twedt et al. 2002) and indicates the potential for increased nest success in RFB practices over the next six years. Whereas this trend appears contrary to previous reports of a positive relationship between strip width and nest success (Major et al. 1999), I propose an explanation based on local vegetation and predator composition. Riparian forest buffers in this study had comparably low vertical and horizontal vegetation cover and Dickcissel nests were often easily visible, which can increase nest predation rates (Angelstam 1986). Although these local vegetative features were not significant in my models, they are well known as important components for Dickcissel nest success (Harmeson 1974, Zimmerman 1982, Hughes et al. 1999, Winter 1999). Furthermore, I suspect the relatively low vegetative cover in RFBs facilitated higher invasibility by RIFA, resulting in significantly higher mound densities in this conservation practice than elsewhere (Allen 2004, Menzel

and Nebeker 2008). These mound densities may have corresponded to Dickcissel nests in RFB being more frequently depredated by RIFA (20%) than any other habitats on the farm, which may have a partially additive effect on nest depredation rates based on the incidental manner in which most predators find nests (Angelstam 1986). Based on the absence of visible damage to the majority of depredated Dickcissel (70%) and Red-winged Blackbird (61%) nests (Peterson et al. 2004) and four direct observations of snake predation, I speculate that snakes were an abundant nest predator on this farm. These observations reflect previous studies that report snakes as primary predators of bird nests in Missouri (Thompson et al. 1999, Thompson and Burhans 2003), Texas (Sperry and Weatherhead 2009), and specifically for Dickcissels (Facemire and Fretwell 1980, Zimmerman 1984). Elevated rates of snake depredation may also explain the positive effect of nest height on nest success, which was previously suggested for Dickcissels (Harmeson 1974).

Overall Red-winged Blackbird nest success was 8.61% in this study. This estimate is considerably lower than was reported in CRP fields in Missouri that were considered sink habitat (27.6%; McCoy et al. 1999). This is also lower than reported for nests in marsh habitat (26.7%), particularly when nests were at least 12 inches above the water surface (38.2%; Goddard and Board 1967). Brown and Goertz (1978) reported similarly high estimates (32.5-33.3%) for Red-winged Blackbird nests between 1.2 and 2.4 m above the ground, but their estimates for lower nests (<0.6 m) were more similar (12.5%) to my estimates. I observed that Red-winged Blackbird nest success was considerably greater in diverse filter strips (23.4%) than other conservation practices. This estimate is considerably greater than Red-winged Blackbird nest success documented in Midwest filter strips (9.6-15.6%; Henningsen and Best 2005). Twedt et al. (2002) reported 23.8% success in a 21-day nest period in 4-10 year old RFBs in the MAV that, similar to Dickcissel, suggests the potential for increased nest success over the next six years. My models predicted that nest success was higher in FSDs with degraded vegetation than other habitats with improved vegetation. Given that nest survival was negatively related to dead vegetation cover, this species may benefit from the use of a fire disturbance regime, provided that sufficient cover and height of nesting substrates remains or grows back rapidly. This pattern of reduced dead vegetation has been previously reported, albeit with no effect on nest success (Warren and Anderson 2005). Red-winged Blackbird nest depredation events were slightly more often observed in torn condition after being depredated than Dickcissels, although most had no obvious signs of depredation, suggesting snakes were also a common predator of their nests.

The only land-cover type that explained variation in nest survival was rowcrop cover within 30 m of Dickcissel nests, although I also report a weak positive effect of patch area for Dickcissel

(Winter and Faaborg 1999). I did not detect an effect of edge proximity as predicted (Suarez et al. 1997), although this pattern is not surprising (Jensen and Finck 2004). Red-winged Blackbirds had competitive models that included total amount of edge within 30 m, total edge in 400 m, developed cover in 400 m, rowcrop cover in 400 m, and late-succession forest cover in 400 m, but none were strong effects. The relatively weak influence of landscape-level factors has been previously documented for Dickcissel (Hughes et al. 1999) and other grassland birds (Davis et al. 2006), further providing support for the hypothesis that for Dickcissel, and possibly Red-winged Blackbird, nest success is more strongly related to local vegetation features (Hughes et al. 1999).

Conservation implications.—My study highlights the importance of implementing multiple conservation practices in an intensive rowcrop, agricultural landscape to benefit grassland-nesting birds. The BLK practice had the highest nest density for both species and was particularly attractive to Dickcissels. I recommend its establishment as the foundation for a whole-farm conservation management strategy in the MAV that initially targets grassland bird conservation. Riparian forest buffers provided attractive nesting habitat and I expect nesting use by both species to continue increasing with woody maturation. However, with high RIFA mound densities and low Dickcissel nest success, RFB provided suboptimal benefits on this farm. I suggest investigating effects of integrating perennial grasses and forbs to complement the woody component and provide nesting habitat for grassland birds in an area of lower RIFA colonization potential. This approach may also provide additional water quality benefits through reduced sediment runoff (Lee et al. 2003). Although slow-growing oaks had not grown tall enough in three years to provide nesting substrates with much height from the ground, this planting strategy is preferable for grassland birds over fast growing trees, in which case maturation occurs quickly and reduces suitability of RFBs for grassland nesting species (Twedt et al. 2002). Monotypic and diverse filter strips provided dense, grass-dominated strips that contrasted with the heterogeneous vegetation structure of early-succession blocks and RFBs. Based on their attractiveness as nesting sites early in the season, I recommend implementing disturbance regimes for filter strip management on a rotational basis to maintain an undisturbed portion of the filter strip population each year. These practices were less attractive to Dickcissel than Red-winged Blackbird, which may be compensated by increasing vegetative heterogeneity, particularly in the monotypic filter strips. Increasing vegetation structure in filter strips may also expand their seasonal use by nesting Dickcissels. Regardless, I observed no evidence to support my initial prediction of low nest success in these strip habitats relative to blocks and in contrast, found that Red-winged Blackbird nest success was highest in diverse-planted filter strips. This study was geographically limited and hence, supplemental investigations across a larger

geographic range would improve local management decisions to optimize avian benefits from these conservation practices.

Based on the reproductive benefits of filter strips for grassland birds in concert with the incompatibility of large block habitat in intensive, agricultural landscapes, I recommend their implementation to complement the ecological functions of a block-based management strategy. My study highlights the challenges facing habitat management for grassland birds in intensive, agricultural landscapes and illustrates the importance of local vegetation and hence, planting regimes to properly manage conservation buffers for nesting birds.

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Table 1. Relative nest densities for Dickcissel (*Spiza americana*) and Red-winged Blackbird (*Agelaius phoeniceus*) in four conservation practices on an intensive rowcrop, agriculture farm in the Mississippi Alluvial Valley, MS, 2005-2007.

Species	Year	BLK Nests / 100 ha ± 95% CI	*n*	RFB Nests / 100 ha ± 95% CI	*n*	FSD Nests / 100 ha ± 95% CI	*n*	FSM Nests / 100 ha ± 95% CI	*n*
Dickcissel	2005	558.33 ± 155.90	20	41.67 ± 22.17	20	35.71 ± 50.54	14	0.00 ± 0.00	6
	2006	1241.67 ± 352.88	20	100.00 ± 58.77	20	100.00 ± 64.47	20	8.33 ± 16.33	20
	2007	662.50 ± 212.12	20	183.33 ± 71.69	20	166.67 ± 71.10	20	91.67 ± 50.13	20
Red-winged Blackbird	2005	504.17 ± 156.52	20	70.83 ± 41.51	20	345.24 ± 281.26	14	166.67 ± 223.15	6
	2006	312.50 ± 96.90	20	66.67 ± 49.71	20	83.33 ± 64.90	20	375.00 ± 145.85	20
	2007	108.33 ± 71.19	20	179.17 ± 75.20	20	66.67 ± 64.47	20	91.67 ± 50.13	20

These estimates only include nests found in predetermined study plots for each practice, thus excluding opportunistically found nests.

Table 2. Red-winged Blackbird (*Agelaius phoeniceus*) and Dickcissel (*Spiza americana*) nest-site characteristics in four conservation practices amongst rowcrop agriculture in the Mississippi Alluvial Valley, MS, 2005-2007.

	Nest-site characteristic	Conservation practice[a]							
		BLK		RFB		FSD		FSM	
		Mean ± 95% CI	*n*	Mean ± 95% CI	*n*	Mean ± 95% CI	*n*	Mean ± 95% CI	*n*
Dickcissel	Nest concealment (%)	70.55 ± 2.21	593	56.24 ± 7.57	72	50.83 ± 10.80	35	79.67 ± 20.34	9
	Nest rim height (cm)	71.63 ± 2.86	606	25.53 ± 3.68	77	33.50 ± 6.57	34	46.67 ± 9.47	12
	Nest plant height (cm)	168.11 ± 13.25	170	86.61 ± 9.07	44	80.45 ± 9.13	20	115.73 ± 22.40	11
	Clutch size	3.78 ± 1.20	121	4.25 ± 0.74	8	4.33 ± 1.02	6	no data	
Red-winged Blackbird	Nest concealment (%)	37.70 ± 4.17	219	41.61 ± 7.21	74	21.44 ± 7.88	46	21.44 ± 6.49	62
	Nest rim height (cm)	81.15 ± 4.29	223	65.86 ± 6.17	72	74.38 ± 10.09	45	89.28 ± 6.43	68
	Nest plant height (cm)	139.57 ± 14.37	28	126.77 ± 9.55	43	158.63 ± 42.47	8	161.82 ± 17.76	11
	Clutch size	2.98 ± 1.31	56	3.27 ± 1.23	22	3.30 ± 0.94	10	3.00 ± 1.10	14

[a]BLK = large, early-succession afforestation blocks; RFB = early-succession riparian forest buffers; FSD = filter strips with diverse, forb and grass, planting; FSM = filter strips planted with monotype of switchgrass

Table 3. Mean estimates ±95% confidence intervals for nest-site characteristics measured at random and nest-site locations for Dickcissel (*Spiza americana*) and Red-winged Blackbird (*Agelaius phoeniceus*) in four conservation practices on an intensive, rowcrop agriculture farm in the Mississippi Alluvial Valley, MS, 2007.

| Survey type | Conservation practice | *n* | Nest-site characteristic | | | | | | |
			VOR[a]	% HD[b]	% Grass	% Forb	% Woody	% Dead veg	RIFA[c] mounds / ha
Random	FSD	60	49.60 ± 4.75	67.72 ± 2.93	70.70 ± 7.19	25.97 ± 7.26	0.42 ± 0.27	24.82 ± 5.26	82.80 ± 28.35
	FSM	60	126.34 ± 3.07	94.02 ± 1.48	92.12 ± 2.38	6.55 ± 2.32	0.28 ± 0.28	15.72 ± 3.04	27.60 ± 15.80
	RFB	60	44.81 ± 5.74	49.53 ± 4.63	15.30 ± 4.15	74.18 ± 5.03	8.68 ± 1.92	15.75 ± 2.90	288.74 ± 56.26
	BLK	60	46.87 ± 3.95	68.62 ± 2.23	37.42 ± 5.27	50.33 ± 5.22	10.67 ± 1.92	27.78 ± 4.78	53.08 ± 22.44
Dickcissel	FSD	20	64.28 ± 5.41	69.21 ± 8.60	71.70 ± 15.04	23.75 ± 13.87	4.55 ± 8.82	21.20 ± 8.64	101.91 ± 42.86
	FSM	11	115.70 ± 19.99	86.45 ± 12.20	87.64 ± 13.47	12.36 ± 13.47	0.00 ± 0.00	10.36 ± 5.03	34.74 ± 48.68
	RFB	44	61.38 ± 5.17	61.56 ± 5.00	12.59 ± 7.22	81.27 ± 7.57	6.07 ± 2.81	19.43 ± 4.66	257.67 ± 41.78
	BLK	167	70.43 ± 4.18	70.46 ± 3.77	25.83 ± 3.12	51.65 ± 3.38	21.91 ± 2.43	20.41 ± 3.09	72.79 ± 16.41
Red-winged Blackbird	FSD	8	95.88 ± 23.46	63.59 ± 11.74	38.75 ± 22.05	50.00 ± 23.57	0.00 ± 0.00	26.88 ± 14.10	95.54 ± 62.42
	FSM	11	134.00 ± 8.02	90.32 ± 3.54	81.27 ± 18.58	12.91 ± 12.07	5.09 ± 8.02	13.55 ± 6.79	23.16 ± 30.45
	RFB	43	77.42 ± 7.35	58.02 ± 4.06	15.86 ± 8.70	77.16 ± 9.00	4.81 ± 1.96	29.00 ± 7.80	239.97 ± 51.36
	BLK	26	57.85 ± 11.54	67.52 ± 8.60	31.04 ± 8.76	49.27 ± 8.68	17.96 ± 4.94	19.65 ± 7.41	73.49 ± 34.41

[a] Visual obstruction reading (VOR) is a measurement of effective vertical vegetative cover

[b] Horizontal vegetation density, or ground cover

[c] Red-imported fire ant (RIFA)

Table 4. Model selection results for Dickcissel (*Spiza americana*) nest survival in conservation practices amongst row-crop agriculture in the Mississippi Alluvial Valley, MS, 2005-2007.

Model[a]	$\Delta AICc$[b]	w_i[c]	K[d]	Deviance
Step 4				
Year*Stage+TT+RFB+30Row+HD+Grass+NH	0.00	0.27	16	2245.80
Year*Stage+Year*TT+RFB+30Row+HD+Grass+NH	0.39	0.23	20	2238.14
Year*Stage+Year*TT+RFB+30Row+Grass+NH	0.99	0.17	19	2240.76
Year*Stage+TT+RFB+30Row+HD+Grass+NH+Precip	1.65	0.12	17	2245.44
Year*Stage+TT+RFB+30Row+HD+Grass+woody+NH	1.84	0.11	17	2245.63
Year*Stage+Year*TT+RFB+Patcharea+30Row+HD+Grass+NH	1.97	0.10	21	2237.71
Step 3				
Year*Stage+Year*TT+RFB+30Row	14.91	0.00	17	2258.70
Year*Stage+Year*TT+RFB+Patcharea+30Row	16.06	0.00	18	2257.84
Year*Stage+TT+RFB+30Row	16.66	0.00	13	2268.49
Step 2				
Year*Stage+Year*TT+BLK	17.78	0.00	16	2263.58
Year*Stage+Year*TT+RFB	18.02	0.00	16	2263.82
Year*Stage+Year*TT+RFB+BLK	18.58	0.00	17	2262.37
Year*Stage+TT+RFB	19.16	0.00	12	2273.00
Year*Stage+TT+BLK	19.55	0.00	12	2273.39
Step 1				
Year*Stage+Year*TT	23.61	0.00	15	2271.42
Year*Stage+TT	24.53	0.00	11	2280.37

[a] Models are ranked by ascending $\Delta AICc$. TT = quadratic seasonal trend; RFB = riparian forest buffer; 30Row = rowcrop cover within 30 m; HD = horizontal vegetation density; NH = nest height; BLK = early-succession, hardwood block

[b] Denotes the difference of AICc (Akaike's information criterion adjusted for small sample size) between candidate models. AICc of the best model is 2277.88.

[c] AIC model weight

[d] Number of parameters.

Table 5. Model selection results from Red-winged Blackbird (*Agelaius phoeniceus*) nest survival in block and buffer conservation practices amongst rowcrop agriculture in the Mississippi Alluvial Valley, MS, 2005-2007.

Model[a]	ΔAICc[b]	w_i[c]	K[d]	Deviance
Step 4				
Year*Stage+TT+FSD+DV+NH	0.00	0.12	14	1381.86
Year*Stage+TT+FSD+30TE+DV+NH	0.83	0.08	15	1380.66
Year*Stage+TT+DV+NH	0.95	0.07	13	1384.83
Year*Stage+TT+FSD+DV	1.27	0.06	13	1385.15
Year*Stage+TT+FSD+400Late+DV+NH	1.49	0.06	15	1381.33
Year*Stage+TT+FSD+DV+NH+Precip	1.58	0.05	15	1381.41
Year*Stage+TT+FSD+DNE+DV+NH	1.78	0.05	15	1381.62
Year*Stage+TT+FSD+30TE+DV	1.81	0.05	14	1383.66
Year*Stage+TT+FSD+400Developed+DV+NH	1.85	0.05	15	1381.69
Year*Stage+TT+FSD+DWE+DV+NH	1.88	0.05	15	1381.72
Year*Stage+TT+FSD+DWE+30TE+DV+NH	1.88	0.05	15	1381.72
Year*Stage+TT+FSD+patcharea+DV+NH	1.89	0.05	15	1381.73
Year*Stage+TT+DV	1.90	0.05	12	1387.80
Year*Stage+TT+FSD+400TE+DV+NH	1.90	0.04	15	1381.74
Year*Stage+TT+FSD+400Row+DV+NH	2.00	0.04	15	1381.84
Step 3				
Year*Stage+TT+FSD	4.07	0.02	12	1389.97
Year*Stage+TT	4.72	0.02	11	1392.63
Year*Stage+TT+FSD+30TE	5.06	0.02	13	1388.94
Year*Stage+TT+FSD+DWE	5.40	0.01	13	1389.27
Year*Stage+TT+FSD+Patcharea	5.53	0.01	13	1389.41
Year*Stage+TT+FSD+400Late	5.54	0.01	13	1389.42
Year*Stage+TT+FSD+DNE	5.66	0.01	13	1389.54
Year*Stage+TT+FSD+400Mid	5.83	0.01	13	1389.70
Year*Stage+TT+FSD+400TE	5.90	0.01	13	1389.78
Year*Stage+TT+FSD+DWE+30TE	5.90	0.01	14	1387.76
Year*Stage+TT+FSD+400Row	6.00	0.01	13	1389.88
Year*Stage+TT+FSD+400Developed	6.01	0.01	13	1389.89
Year*Stage+TT+FSD+30Row	6.05	0.01	13	1389.93
Year*Stage+TT+FSD+400Early	6.05	0.01	13	1389.93
Year*Stage+TT+400Late	6.06	0.01	12	1391.96
Step 2				
Year*Stage+TT+FSD	4.07	0.02	12	1389.97
Year*Stage+TT	4.72	0.02	11	1392.63
Step 1				
Year*Stage+TT	4.72	0.02	11	1392.63
Year*Stage	6.52	0.01	9	1398.47

[a] Models are ranked by ascending ΔAICc. TT = quadratic seasonal trend; FSD = diverse filter strip; DV = dead vegetation cover; NH = nest height, 30 = amount of spatial metric within 30 m; TE = total edge; 400 = amount of spatial metric within 400 m; Late = late-succession forest; DNE = distance to nearest edge; DWE = distance to wooded edge; Row = rowcrop cover; Mid = mid-succession forest; Early = early-succession habitat

[b] Denotes the difference of AICc (Akaike's information criterion adjusted for small sample size) between candidate models. AICc of the best model is 1410.01.

[c] AIC model weight

[d] Number of parameters.

Table 6. Nest success estimates and 95% CIs for covariates from the best approximating model of Dickcissel (*Spiza americana*) nest survival. These estimates are specific to positive and negative values of coefficients and best and worst case management scenarios for nest survival in all habitats or only in RFB (riparian forest buffer) on an agricultural farm in the Mississippi Alluvial Valley, MS, 2005-2007.

Scenario[a]	RFB	NH (cm)	% Grass cover	30Row (ha)	% HD	Nest success	95% CL lower	95% CL upper	Sig.[b]
All habitats	0	65.89	27.71	0.01	57.99	0.229	0.108	0.364	
RFB	1	65.89	27.71	0.01	57.99	0.106	0.026	0.248	
NH, −1SD	0	28.93	27.71	0.01	57.99	0.148	0.053	0.278	
NH, +1SD	0	102.85	27.71	0.01	57.99	0.323	0.171	0.472	B+
Grass, −1/2SD	0	65.89	12.39	0.01	57.99	0.250	0.121	0.389	B+
Grass, +1/2SD	0	65.89	43.04	0.01	57.99	0.210	0.095	0.342	
30Row, zero cover	0	65.89	27.71	0.00	57.99	0.236	0.112	0.372	B+
30Row, +1SD	0	65.89	27.71	0.05	57.99	0.208	0.092	0.345	
HD, −1SD	0	65.89	27.71	0.01	35.97	0.259	0.122	0.407	B+
HD, +1SD	0	65.89	27.71	0.01	80.01	0.202	0.090	0.332	
Best, all habitats	0	102.85	12.39	0.00	35.97	0.384	0.213	0.541	A+, B+
Best, RFB	1	102.85	12.39	0.00	35.97	0.230	0.079	0.422	
Worst, all habitats	0	28.93	43.04	0.05	80.01	0.096	0.028	0.207	A−
Worst, RFB	1	28.93	43.04	0.05	80.01	0.029	0.004	0.110	A−, B−

[a] NH = nest height; 30Row = cover of rowcrop within 30m radius (ha); HD = % Horizontal cover; Grass = % grass cover; SD = standard deviation

[b] Letters, A and B, represent a significance difference (Sig.; as determined by 95% CLs) and direction of trend for scenario-specific comparisons. Hence, A+ represents the highest possible nest survival and B- is the lowest, and blanks have no significant comparisons

Table 7. Nest success estimates and 95% CIs for covariates from the best approximating model of Red-winged Blackbird (*Agelaius phoeniceus*) nest survival. These estimates are specific to positive and negative values of coefficients and best and worst case management scenarios for nest survival in all habitats or only in FSD (diverse filter strip) on an agricultural farm in the Mississippi Alluvial Valley, MS, 2005-2007.

Scenario[a]	FSD	NH (cm)	% Litter cover	Nest success	95% CI lower	95% CI upper
All habitats	0	79.06	17.80	0.086	0.023	0.197
FSD	1	79.06	17.80	0.234	0.043	0.512
NH, −1SD	0	47.23	17.80	0.067	0.014	0.175
NH, +1SD	0	110.89	17.80	0.109	0.032	0.232
Litter, −1SD	0	79.06	0.57	0.117	0.033	0.254
Litter, +1SD	0	79.06	35.04	0.061	0.013	0.159
Best, all habitats	0	110.89	0.57	0.144	0.044	0.290
Best, FSD	1	110.89	0.57	0.319	0.078	0.599
Worst, all habitats	0	47.23	35.04	0.045	0.008	0.139
Worst, FSD	1	47.23	35.04	0.158	0.018	0.432

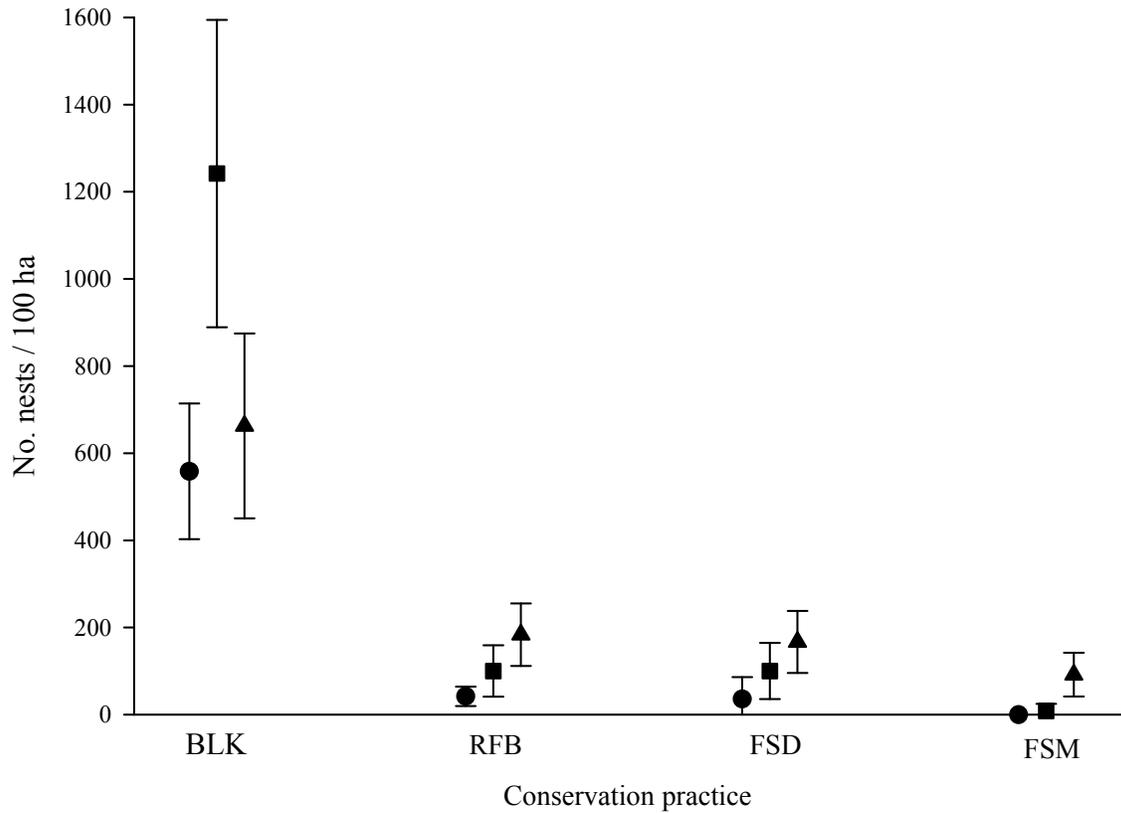[a] NH = nest height; Litter = % litter cover; SD = standard deviation

Figure 1. Relative nest densities (mean ± 95% CI) for Dickcissel in early-succession, afforestation blocks (BLK), early-succession, riparian forest buffers (RFB), diverse filter strips (FSD), and monotypic filter strips (FSM) during breeding seasons of 2005 (filled circles), 2006 (filled squares), and 2007 (filled triangles) on an agricultural farm in the Mississippi Alluvial Valley, MS.
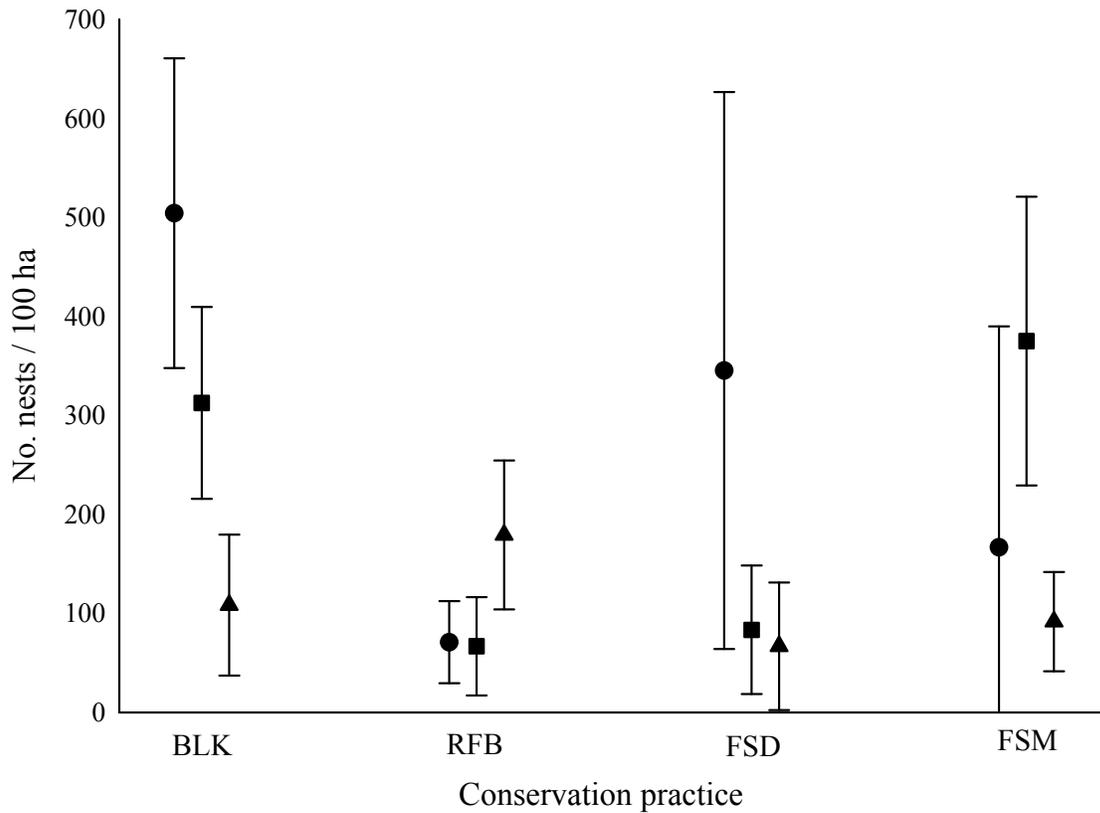
Figure 2. Relative nest densities (mean ± 95% CI) for Red-winged Blackbird in early-succession, afforestation blocks (BLK), early-succession, riparian forest buffers (RFB), diverse filter strips (FSD), and monotypic filter strips (FSM) during breeding seasons of 2005 (filled circles), 2006 (filled squares), and 2007 (filled triangles) on an agricultural farm in the Mississippi Alluvial Valley, MS.
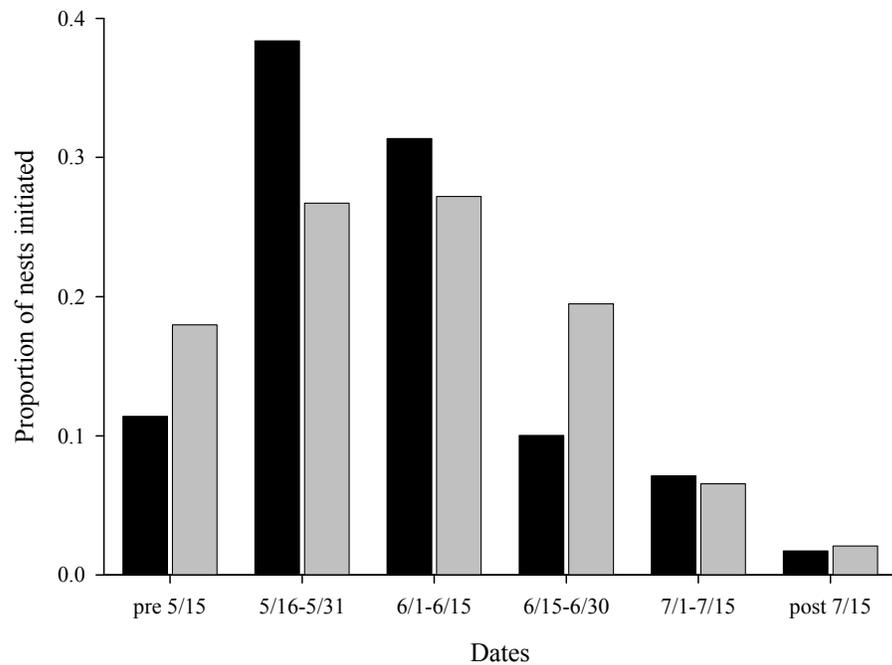
a.
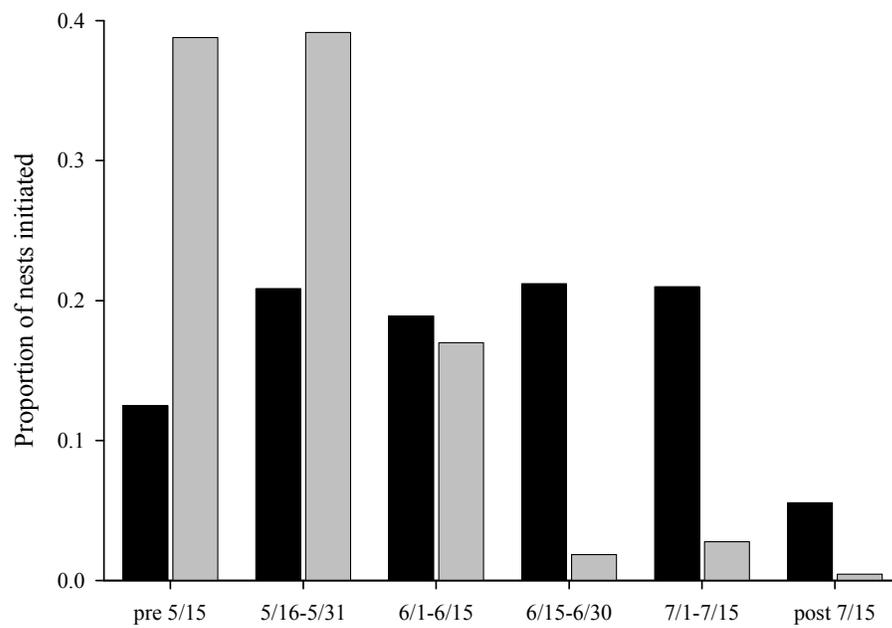
b.

Figure 3. Total proportions of nests initiated relative to nesting activity in early-succession block (black bars) and all conservation buffer (gray bars) practices for Dickcissel (a) and Red-winged Blackbird (b) on an agricultural farm in the Mississippi Alluvial Valley, MS, 2005-2007.
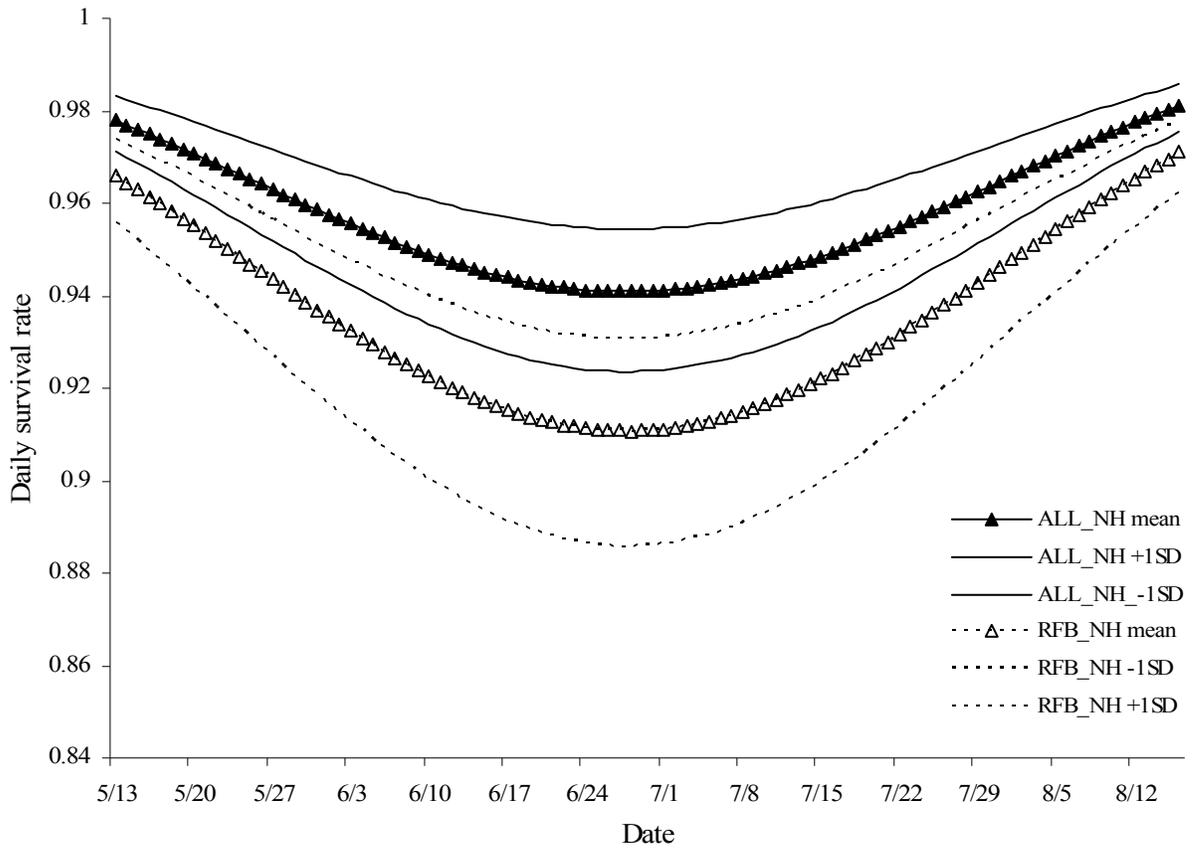
Figure 4. Dickcissel daily nest survival estimates across the breeding season for all conservation practices combined (filled triangles) ±1 SD in nest height (solid lines) and for riparian forest buffers (open triangles) ±1 SD in nest height (dotted lines) amongst intensive agriculture in the Mississippi Alluvial Valley, MS, 2005-2007.
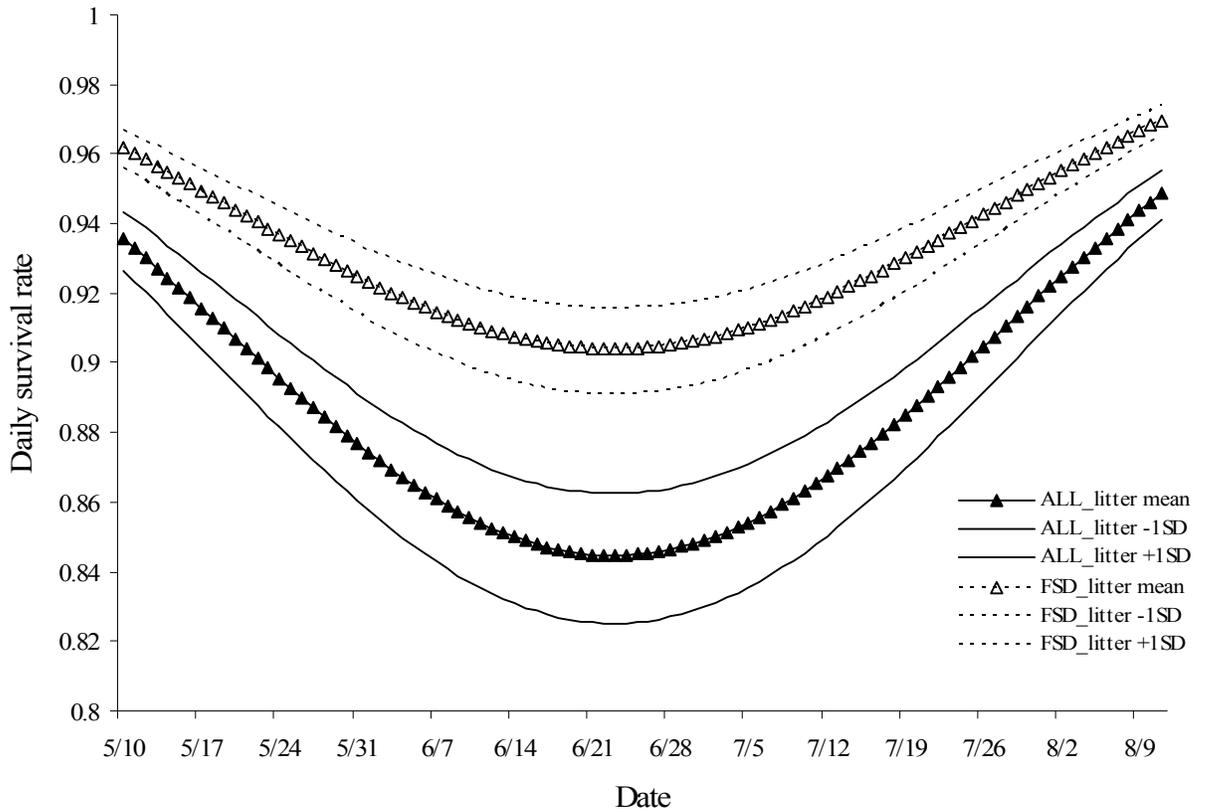
Figure 5. Red-winged Blackbird daily nest survival estimates across the breeding season for all conservation practices combined (filled triangles) ±1 SD in dead vegetation cover (solid lines) and for diverse-planted filter strips (open triangles) ±1 SD in dead vegetation cover (dotted lines) amongst intensive agriculture in the Mississippi Alluvial Valley, MS, 2005-2007.

Appendix A. Summary of breeding bird nesting activity (listed in descending order) and apparent nest success in four conservation practices on a rowcrop agricultural farm in the Mississippi Alluvial Valley, MS, 2005–2007.

Species	Total nests	% nest success	FSD[a]			FSM			RFB			BLK		
			2005	2006	2007	2005	2006	2007	2005	2006	2007	2005	2006	2007
Dickcissel	733	36.3	3	12	20		1	11	9	24	44	137	303	169
Red-winged Blackbird	409	13.5	30	10	8	6	45	11	16	16	43	122	76	26
Mourning Dove (*Zenaida macroura*)	67	31.3							2	5	1	14	25	20
Eastern Meadowlark (*Sturnella magna*)	50	32.0	1		1	1			4	3	5	12	16	7
Northern Bobwhite (*Colinus virginianus*)	23	30.4	3		1				7	4	1	3	2	2
Grasshopper Sparrow (*Ammodramus savannarum*)	12	41.7										4	7	1
Yellow-billed Cuckoo (*Coccyzus americanus*)	5	0												5
Northern Mockingbird (*Mimus polyglottus*)	2	0										1		1
Brown Thrasher (*Toxostoma rufum*)	6	33.3											3	3
Eastern Towhee (*Pipilo erythropthalmus*)	1	100												1
Northern Cardinal (*Cardinalis cardinalis*)	1	100							1					
Blue Grosbeak (*Guiraca caerulea*)	1	100											1	
Indigo Bunting (*Passerina cyanea*)	3	0								1	2			
Orchard Oriole (*Icterus spurius*)	1	100												1
Total nests	1314		37	22	30	7	46	22	39	53	96	293	433	236

[a]FSD = diverse-planted filter strip, FSM = monotypic-planted filter strip, RFB = early-succession riparian forest buffer, BLK = early-succession, hardwood afforestation block

CHAPTER 4.
DICKCISSEL (*SPIZA AMERICANA*) FLEDGLING SURVIVAL AMONGST ROWCROP AGRICULTURE IN MISSISSIPPI

ABSTRACT

Continued declines of grassland bird populations throughout North America are associated with agricultural intensification and warrant the replacement of lost grassland habitat in agricultural landscapes. Incorporating bird priorities in the management protocols of these habitat establishment efforts will enhance subsequent benefits for grassland birds. Considerable amounts of early-succession habitat have been established amongst rowcrop matrices in the Mississippi Alluvial Valley, yet we lack an understanding of the response by fledgling grassland passerines to these habitats. Dickcissel fledgling ecology can provide important insight into habitat management as this is the most vulnerable life stage for this grassland bird of conservation concern. I attached radio transmitters to 416 Dickcissel fledglings in two years (2006 and 2007) and tracked them twice daily for 15 days postfledging to document cause-specific mortality and examine survival relationships with microhabitat, landscape context, and fledgling biology. Red imported fire ants (RIFA) and snakes were the predominant predators and caused ≥65% of all mortality. Mortality was greatest (83%) ≤2 days postfledging, and survival had a strong positive relationship with fledgling age. Fledgling survival declined substantially over the season and was negatively influenced by the amount of grassland area and edge density in the surrounding landscape. Body mass and perch height positively correlated with survival rates. Conservation practices had similar rates and causes of mortality, except in riparian forest buffers, where no fledglings survived past day two and depredation events were dominated by RIFA. Elevated mortality rates and low overall survival highlights the importance of considering fledgling ecology in management protocols for established vegetation patches amongst intensive agriculture. I recommend that conservation practices are managed to maintain adequate vegetative cover early in the breeding season (May to early June) when fledgling survival is highest. Furthermore, riparian forest buffers may function as ecological traps in areas with RIFA and I suggest additional research be conducted to determine effects of alternative establishment strategies, such as inclusion of perennial grasses or forbs, which may reduce intensity of RIFA depredation on grassland bird fledglings.

INTRODUCTION

The historically vast North American grasslands have been largely replaced by agriculture (Noss et al. 1995) and many grassland bird populations have declined precipitously as a result (Knopf 1994, Peterjohn and Sauer 1999). Increasing food demands from growing human populations facilitate

agricultural practice intensification (e.g., fencerow removal, chemical applications, monotypic crop fields), which reduce opportunities for wildlife conservation on U.S. farmlands. Wildlife management approaches have concomitantly shifted towards wildlife-friendly farming, which aims to enhance farmland biodiversity by integrating natural vegetation patches that minimally impact crop yield, thereby reducing the need to replace lost cropland elsewhere (Green et al. 2005). On private, agricultural landscapes, land managers are challenged to optimize wildlife benefits from established habitat patches while maintaining compatibility with producer economics.

To achieve this balance between crop yield and grassland bird benefits, wildlife professionals must understand the impact of established, non-crop vegetation on grassland-bird population trajectories. In Mississippi, the U.S. Department of Agriculture (USDA) has established afforested blocks (BLK) and conservation buffers that include diverse-planted filter strips (FSD), monotypic-planted filter strips (FSM), and riparian forest buffers (RFB). Current management decisions for these and similar established patches are largely reliant on studies of bird community structure and nesting ecology (Marcus et al. 2000, Smith et al. 2005, Conover 2005, Henningsen and Best 2005). Whereas these studies provide meaningful insight on bird habitat use and productivity, researchers have raised concerns about the potential of similar studies to provide misleading indications of habitat quality (Van Horne 1983, Vickery et al. 1992) or failure to represent the full range of habitat types used during critical life stages (Cohen and Lindell 2004, White et al. 2005). Notwithstanding recent improvements in reproductive performance evaluation (Jones and Geupel 2007), circumstances potentially exist when nest success alone may inaccurately reflect fledgling success (Keedwell 2003). Birds are particularly vulnerable during the fledgling life stage, which is the time period between nest departure and parental independence for a young bird (Kershner et al. 2004, Berkeley et al. 2007). Obstacles in research logistics often result in wildlife management decisions that are based on an incomplete understanding of a species' life history. Because of reduced radio transmitter size and weight, grassland passerines have recently been the focus of several studies on fledgling ecology. These studies have reported higher mortality rates than during any other life stage, especially immediately postfledging (Yackel Adams et al. 2001, Wheelwright and Templeton 2003, Kershner et al. 2004, Yackel Adams et al. 2006, Berkeley et al. 2007, Suedkamp Wells et al. 2007). Based on these studies, further investigation of this life-history stage in the Dickcissel is warranted.

Despite efforts to establish early-succession habitat for grassland birds in the southeastern U.S., our understanding of grassland bird response to these established patches comes primarily from the Midwest (Best 2000, Henningsen and Best 2005). The Dickcissel (*Spiza americana*) is a Partners in Flight species of conservation concern that is of particular interest for its affinity to nest in

established grassland habitats in the Southeast, notably in the Mississippi Alluvial Valley (MAV; Conover 2005; see also Chapter 3). Recent radio-tracking studies of Dickcissel fledglings have provided information on their habitat use, although these studies have been geographically focused on the Midwest (Berkeley et al. 2007, Suedkamp Wells et al. 2007). As with other grassland birds, Dickcissel fledglings have high mortality rates immediately postfledging (Berkeley et al. 2007, Suedkamp Wells et al. 2007). As established habitats in the Southeast target grassland bird conservation, it is crucial that associated management protocols incorporate fledgling ecology of at least one member of the grassland bird community. Of particular interest are early-succession RFBs, the most abundant conservation buffer in Mississippi, although wildlife responses to this practice in this region remain largely unknown (Clark and Reeder 2007).

This study evaluated Dickcissel postfledging survival in response to specific habitat management practices in Mississippi. My primary objectives were to investigate causes of Dickcissel fledgling mortality and to evaluate the relationship between fledgling survival and microhabitat features, landscape characteristics, and USDA conservation practices within the intensive, rowcrop agricultural landscape of the MAV. I hypothesized that survival would positively correlate with fledgling age, mass, vegetative cover, and grassland area. Insight on Dickcissel postfledging ecology will mediate a more holistic habitat management approach that accounts for a broader realm of life-history strategies for a grassland bird.

METHODS

Study site.—The study farm is 2,630 ha and located in Coahoma County, Mississippi, USA (34° 18' N, 90° 34' W), which is in the MAV (Mississippi Alluvial Valley; Bird Conservation Region 26). This farm exemplified the current MAV landscape, which is dominated by large fields of ditch-to-ditch, rowcrop agriculture of primarily cotton (*Gossypium* sp.) and soybean (*Glycine* sp.) that are fragmented by a network of drainage ditches, streams, wood lines, and fencerows, with negligible topographic relief. However, the farm differed from the surrounding landscape from recent establishment of various semi-natural vegetation patches to promote wildlife-friendly farming and consisted of 48% rowcrop, 30% early-succession hardwood afforestation blocks, 14% forested or herbaceous wetlands, 4% conservation buffers, 2% wooded, and 2% herbaceous drains. Farm crops included early wheat (May-early June) and late soybean (mid-June planting) in 2006 and only soybean in 2007, and all fields endured similar tillage regimes. Soil associations on the farm included Sharkey clay (41%), Dowling clay (15%), and Tunica silty clay (11%) (Soil Survey Staff 2008). Rainfall occurred earlier in 2006 (May = 13.21 cm, June = 2.72 cm, July = 3.53 cm) and more consistently and in greater abundance in 2007 (May = 14.99 cm, June = 5.00 cm, July = 14.96 cm).

Nest location and radio-transmitter attachment.—Intensive Dickcissel nest search efforts took place from 15 May to 20 July during 2006 and 2007 by two to four people systematically walking through vegetation with 2 m between each person while mildly disturbing the vegetation with 1 m sticks to flush brooding birds. Nest monitoring occurred every 2-4 days using extreme care to avoid damaging vegetation that surrounded the nest (Martin 1993). Dickcissels fledge in 7-10 days post-hatching with a mass of approximately 18-20 g (Temple 2002, Berkeley et al. 2007). At 1-3 days prior to fledging (6-8 days of age), I attached a U.S. Geological Survey metal band and radio-transmitter to nestlings. I also recorded nestling body mass (g) using a 50 g Pesola® spring scale, as earlier studies have found a relationship between mass and postfledging survival (Naef-Daenzer et al. 2001, Suedkamp Wells et al. 2007). This time period allowed birds to grow sufficiently so that I could properly size their harness for a good fit with slight slack to allow room for growth. I aged nestlings based on reference images from Dickcissel nestlings of known ages (R. Conover, unpublished data). Radio transmitters were attached to all nestlings in the brood except individuals that were significantly underdeveloped because they were neither ready to fledge nor of sufficient physical stature to bear a transmitter package. I tagged all individuals in the brood to ensure sufficient sample sizes and based on previous evidence that reported survival of individual Dickcissels was independent from brood mates (Suedkamp Wells 2007). Dependency of brood size was tested using a chi-squared goodness-of-fit test for all possible outcomes of 5 brood sizes (n = 20).

I attached transmitters with a modified Rappole and Tipton (1991) technique that entailed gluing the transmitter to the bird above the synsacrum with the antenna oriented down the tail and attaching a cotton-elastic leg harness that allowed room for fledgling growth (Kershner et al. 2004). I later observed parents pecking at transmitters in nests and subsequently documented several that were removed, so I altered the attachment technique. Hence, I constructed custom-sized body harnesses that ran behind the leg and over the wing and remained attached even if the glue detached from the body (R. Conover, personal observation). Handling time was <5 minutes per bird. Harnesses were visually inspected daily for fit and bird wing movement. Body harnesses were removed from birds within approximately 4 days postfledging (4.08 ± 0.23, days ± SE), which is 3 days before flight ability significantly increases (Berkeley et al. 2007), and replaced with the standard Rappole and Tipton (1991) leg harness technique. Because there are no previous data to determine the effect of this technique, I provide descriptive statistics to compare movement distances for fledglings of equal ages with leg versus body harnesses. After transmitter attachment, nestlings were returned to the nest head-first and monitored for several minutes or until they were pacified. When re-harnessing birds, I banded them with a unique color combination of UV-resistant Darvic bands (Avinet, Dryden, New

York) to facilitate visual confirmation of older, more mobile birds. Radio-transmitters weighed 0.65 g, 0.05 g less than previously confirmed suitable for Dickcissel fledglings (Suedkamp Wells et al. 2003). Transmitter batteries (Ag 379) had an estimated life of 29-38 days at 24 pulses per minute and transmit via a 10-cm whip antenna (Advanced Telemetry Systems, Inc., Isanti, MN, USA). Total transmitter package weight (including harness) was 0.75-0.8 g, which has a mass load of 4% based on mean Dickcissel fledging mass (Temple 2002). I used 35 individual transmitters in 2006 and 50 in 2007 and therefore, often re-used transmitters to exploit their battery life. After transmitter recovery, harnesses were removed, residual glue scraped off, and the entire unit was soaked and scrubbed in 90% isopropyl alcohol prior to re-deployment.

Radio tracking.—Tracking commenced the following morning after transmitter attachment. I tracked fledglings using an R2000 receiver and three-element Yagi antenna (Advanced Telemetry Systems, Inc., Isanti, MN, USA). I visually confirmed locations once per day until fledglings moved $\geq$5 m from the nest, and twice per day thereafter (i.e., tracking period = ½ day) until a) the bird died, b) the signal was lost, or c) the transmitter fell off (Berkeley et al. 2007). A tracking period was considered unsuccessful if there was no visual confirmation of the bird's status and was followed with an intensive tracking effort during the subsequent period. I recorded fledgling locations twice daily in alternating time block cycles that included early morning (sunrise to 0929 hours), late morning (0930 to 1229 hours), afternoon (1230 to 1559 hours), and evening (1600 hours to sunset) to reflect a broad spectrum of diurnal activity (Garton et al. 2001, Suedkamp Wells et al. 2007). In the case of a lost signal, I searched all adjacent fields and continually expanded the search radius until the bird was found or the transmitter was considered undetectable. After locating an individual, I recorded spatial coordinates using a global positioning system, percentage of overhead cover, perch height, parental presence, and distance from the nest on each tracking period. I also marked the perch site of the individual with a small (10 cm) piece of blue flagging at (if the bird moved while tracking) or near the fledgling's location to reduce disturbance but permit a precise vegetation survey at a later date. Movement distances from the previous location were measured using the "distance between points" function in HawthsTools (Version 3.27; Beyer 2004). Tracking was only conducted at times of negligible precipitation to avoid disturbing birds during adverse weather conditions. I quantified mortality causes as red imported fire ant (RIFA, *Solenopsis invicta*), snake, mammal, or unknown. RIFA deaths were visually determined by their presence on fledgling bodies. Snake depredations were determined visually or behaviorally while tracking the transmitter in the snake. Mammal depredations were assigned based on teeth marks in the tag and torn condition of the harness (Suedkamp Wells et al. 2007). I also recorded initial escape behavior whenever possible, including

flutter, freeze (i.e., motionless), run/hop, and fly. Flutter is classified as an ineffective flight attempt that resulted in near zero displacement, whereas fly represents flight attempts with measurable displacement. Upon a flight response, flight distances (m) were paced off by trained individuals to provide an indication of flight abilities. I recorded these behavioral responses because they may offer supplemental insight to assist our understanding of age-specific mortality (Mumme 1992, Grubb et al. 1998, Yackel Adams et al. 2001).

Environmental variables.—I evaluated microhabitat at every fledgling location within an 8-m diameter, circular plot for 15 days postfledging. Surveys typically occurred between 11-20 days after the tracking period, with a mean of 16.9 ± 0.7 (SE) to mitigate fledgling disturbance and standardize potential effects from vegetation growth. I measured a suite of microhabitat variables that are known or suspected to influence fledgling survival in this species (Berkeley et al. 2007, Suedkamp Wells et al. 2007). Microhabitat metrics included percentage of horizontal vegetation density (HD), percentage of total vertical cover (i.e., total cover; Tcov), vegetative structure (proportions of forb, grass, and woody cover), percentage of cover within a 20 cm vertical range of the fledgling perch site (i.e., side cover), and RIFA mound density (mounds/ha) within 5 m. These or similar metrics have been previously shown to be of importance to survival and habitat use during the postfledging period of grassland birds (Kershner et al. 2004, Berkeley et al. 2007, Suedkamp Wells et al. 2007). Horizontal cover was visually estimated by averaging cover percentages from four quadrants in the survey plot. Total cover was determined using 3 readings (at 0/360, 120, and 240 degrees) from a modified Robel pole and recording the mean of proportions of vegetative cover in seven 20 cm vertical subsections within 1.4 m of the ground, at a distance of 4 m, and visual height of 1 m (Robel et al. 1970, Renken and Dinsmore 1987). Side cover was similarly estimated, except only for the 20 cm vertical subsection in which the fledgling was perched to represent actual fledgling concealment, which may reduce predator detection (Overskaug et al. 1999).

I evaluated landscape characteristics at fine and coarse spatial scales to capture fledgling habitat use for comparison with their survival. Spatially explicit information of surrounding land-cover types may be important to fledgling survival because they may prefer habitats that differ from nest sites (Cohen and Lindell 2004). The fine scale was quantified at 30 m for birds <7 days old and at 100 m for birds 7≤15 days old to incorporate environmental variables that reflect age-specific movement potential, as determined by movement distances recorded by Berkeley et al. (2007) and this study (Figure 3). Additionally, 30 m represents the approximate size of an adult Dickcissel territory, which was important to the survival of recent fledglings in a separate study (Berkeley et al. 2007). The coarse scale was evaluated at 400 m, a distance that reflects land-cover metrics relevant

to nest predation risk in passerines (Kus et al. 2008). I developed a geographic information system (GIS) in ArcGIS (ArcMap Version 9.3) from 2007 National Agriculture Imagery Program digital orthophoto quarter quadrangles, complemented by empirical ground observations. Metrics at both scales included percentages of wooded, rowcrop, and grassland area and edge density (total edge in m/ha), which were calculated with FRAGSTATS (McGarigal et al. 2002). These land-cover types were selected for their relative dominance in the study landscape and because Dickcissel fledglings may select habitats different from their natal site (Berkeley et al. 2007). Edge density indicates amount of spatial heterogeneity in the landscape and reflects habitat fragmentation (McGarigal et al. 2002).

   Statistical methods.—I modeled fledgling survival 15 days postfledging, a period when birds require parental care (Zimmerman 1993, Berkeley et al. 2007). I constructed models using maximum likelihood estimation on the logit scale in the nest survival module (Dinsmore et al. 2002) of Program MARK (White and Burnham 1999). My dataset had 151 encounter occasions at half-day intervals from 26 May to 9 August. Mean start tracking date for all birds across both years was 28 June, which was subsequently used as the start date for my predicted survival estimates. I applied hierarchical modeling procedures with 5 steps and performed model selection using adjusted Akaike Information Criterion scores (AIC$_c$; Akaike 1973, Burnham and Anderson 2002). Competitive models ($\leq$2 ΔAIC$_c$) within each step were advanced to the subsequent step as reference models. Within model steps, I evaluated all single covariates and combinations relative to all reference models. Step 1 evaluated linear and quadratic effects of fledgling age, year, linear and quadratic seasonal trends within year, and an interactive effect of year and each seasonal trend. Step 2 evaluated landscape variables at the coarse scale (400 m), and step 3 evaluated fine scale (30 m) variables. Step 2 and 3 covariates included edge density, grassland area, wooded area, and rowcrop area at their respective scales. Step 4 defined the microhabitat and included effects of woody cover, forb cover, grass cover, total vertical cover, horizontal density, and RIFA mound density. Finally, step 5 evaluated fledgling side cover, perch height, overhead concealment, and nestling body mass, which was only added singly at the end to the best approximating model.

   I censored individuals that were not detected for >1 day but subsequently relocated. I otherwise imputed values for missing covariate values associated with ½ day tracking periods, which was primarily for recent fledglings before they moved 5 m from the nest site or from the adverse weather conditions that occasionally postponed radio-tracking efforts. I imputed measurements from the nearest ½ day interval instead of mean values across a time period to maintain spatial relevance of

imputed environmental variables. I imputed values for 15.6% (n = 272) of fledgling locations, which predominantly (80%) included fledglings ≤1.5 days old.

Fledgling survival rates are the probability that a fledgling survives ½ day and were calculated based on coefficients and their respective covariate mean values ± 1 SD (as computed by Program MARK) for 15-day time periods that start at the mean tracking initiation date of 28 June 2007. Predictions based on covariate values were calculated only for coefficients with significant trends (e.g., 95% CI not overlapping zero). I also computed fledgling survival rates relative to seasonal trends for three (early, middle, late) 15-day time periods to illustrate differences in survival probability associated with timing of fledging. Seasonal time periods started ±15 days of the mean seasonal radio-tracking initiation date and thus included 12 June to 26 June (early), 28 June to 11 July (middle) and 12 July to 26 July (late). I also calculated fledgling success as the probability of an individual surviving a complete 15-day time period, which differed from the period-specific fledgling survival rates that capture the probability of surviving ½ day (i.e., one tracking period).

RESULTS

I attached a total of 416 radio transmitters to nestlings during both years (n = 185 in 2006, n = 231 in 2007), although only 200 (2006 = 101 and 2007 = 99) were confirmed to have fledged and provided radio-tracking data. Mean nestling body mass at time of transmitter attachment was 16.20 ± 0.12 (g ± SE) and age (days post hatching) was 6.8 ± 0.03 (days ± SE). Fledglings either died (n = 308) or survived until they could no longer be tracked (n = 105), with 3 fates unknown from lost data. Of individuals that fledged the nest and were subsequently radio tracked, I observed 63.4% mortality in 2006 (64 of 101) and 72.7% mortality in 2007 (72 of 99). These deaths predominantly (83%) occurred for young birds (≤2 days old); only 27% of birds that lived past day two died. Twenty-two birds lived through the 15-day tracking period. Primary causes of mortality were RIFA (34.5%), snakes (30.2%), and mammals (15.9%), but also included an owl (<1%), drowning (<1%), humans (<1%), and unknown causes (16.8%). I captured most of the snakes responsible for killing 93 nestlings and fledglings and identified them (in descending order of fledglings killed) as Southern Black Racer (37.6%, *Coluber constrictor priapus*), Speckled Kingsnake (20.4%, *Lampropeltis getula holbrooki*), Black Ratsnake (6.5%, *Elaphe obsoleta obsoleta*), Eastern Cottonmouth (4.3%, *Agkistrodon piscivorus piscivorus*) and unidentified (31.2%). Mortality rates were high in all USDA conservation practices and RIFA were the principal predator (Table 1). Fledgling behavioral response at my approach varied with age. They primarily froze during their first 4 days beyond the nest and occasionally ran or flutter flew. They began using flight as an escape tactic by day four and from day eight on this was their predominant response (Table 2). An evaluation of movement

distances from the nest and previous locations showed no significant differences (based on 95% CI) between leg and body harness techniques for fledglings up to 5 days old (Figures 1 and 2). I tracked ≥20 individual fledglings at all ages throughout the 15-day period and noted an increase in fledgling movements at day 7 (Figure 3). Fledglings consistently moved farther from the nest site with age, but did not average >100 m distance from the nest until approximately day 10 (Figure 4). Parental attendance continued for at least 35 days postfledging, although vigilance of care declined with increased fledgling age (Figure 5).

> *Fledgling survival.*—Radio-tracking efforts included an effective sample size of 1,741 tracking periods across both years. Seasonally, sample sizes were highest from early to mid June and throughout July (Figure 6). I observed a dependent effect of fledgling survival with original brood size ($\chi^2 = 54.98$, df = 15, $P < 0.01$). However, >92% of the contribution to the test statistic came from only two potential brood outcomes out of 20. Hence, most brood combinations fit the model well and I proceeded with the analysis because I observed that brood mate spatial locations differed and Suedkamp Wells et al. (2007) reported no effect of brood dependence on individual Dickcissel fledgling survival.

> Fledgling survival models exceeded computational limits in Program MARK because most covariates were time-varying, thus necessitating an alteration of my hierarchical model selection procedures. Previously described procedures were followed through step 2, but in step 3 I was unable to advance all competitive (≤ 2 ΔAIC_c) models. Hence, for all remaining model steps I advanced only the best three models from each step. Survival was best explained in step 1 by a quadratic age trend and an interaction between year and a linear seasonal trend, with no other competitive models (Table 3). Step 2 revealed competitive effects for the addition of grassland area as well as grassland area plus edge density at the coarse scale. Both of these models advanced to step 3, where edge density at the fine scale was competitive when combined with coarse scale grassland area and edge density. These three models remained as the best model set through step 4, where I documented a minimal effect of microhabitat variables (Table 3). In the final step, models were improved by the addition of perch height and fledgling body mass at capture, but bird concealment metrics had a negligible effect (Table 3).

> Coefficient trends (95% CLs) of covariates in the best approximating model included coarse scale grassland area (-0.049; -0.076, -0.022), coarse scale edge density (-0.010; -0.023, 0.004), perch height (0.020; 0.008, 0.032), and mass (0.115; -0.016, 0.246), with significant effects (95% CI did not overlap zero) of grassland area and perch height. Dickcissel fledgling success decreased linearly across the breeding season and success in a 15-day period was nearly zero by mid-June (Figure 6).

This trend had a strong influence on the overall probability of fledgling success relative to fledglings that departed a nest during early, middle, or late time periods within a season (Figure 7). Although I did not find a strong effect using predicted covariate values, a best case scenario of decreased grassland area and increased perch height enhanced predicted fledgling survival (Figure 8).

DISCUSSION

Birds are particularly vulnerable during the fledgling life stage because of their limited mobility and dependence on parental care (Anders et al. 1997, Naef-Daenzer et al. 2001, Cohen and Lindell 2004, Kershner et al. 2004) while learning critical life skills (Wheelwright and Templeton 2003). As such, an understanding of fledgling ecology is valuable to complement previous studies of avian community structure and nesting ecology and assist management decisions aimed at bird conservation. Other recent radio-tracking studies in the Midwest have uncovered numerous environmental factors associated with Dickcissel fledgling survival (Berkeley et al. 2007, Suedkamp Wells et al. 2007). However, with an increasing amount of grassland vegetation being established within agricultural landscapes in Mississippi, there is a need to complement these studies with knowledge of local biological effects.

Fledgling ecology.—Previous studies have indicated that application of a leg harness, attachment of transmitter, or repeated radio-tracking does not negatively influence fledglings (Rappole and Tipton 1991, Powell et al. 1998, Naef-Daenzer et al. 2001, Suedkamp Wells et al. 2003). My use of a body harness for secure transmitter attachment on recently fledged Dickcissels is a previously untested technique. Based on evidence of fledgling movements from nest sites and previous locations, there was no effect of the body harness compared with the commonly applied leg harness. Furthermore, I tracked 203 individuals with body harnesses and only documented two incidents of fledglings getting stuck because of the harness, an effect that also occurs with leg harnesses (Suedkamp Wells et al. 2007). Hence, I do not suspect this harness technique contributed to additional mortality, although caution should be used with interpretations until this technique receives further testing.

I recorded similar movement distances from nests relative to postfledging age as did a separate study in Nebraska and western Iowa (Berkeley et al. 2007). Additionally, I observed marked escalation of fledgling movements at seven days postfledging, which is consistent with previous research (Berkeley et al. 2007). These movement distances were in concert with my observations of increased continuous-flight potential, which corresponded with use of flight as the primary escape response in older (8 day) fledglings. After leaving their natal habitat, fledglings were often observed in soybean rowcrop fields and wooded areas in addition to grassland habitats, although

I did not quantify these data. Immediately after fledging, the primary escape responses were to remain motionless or run, which is not surprising given their limited flight potential and corresponds with previous research that demonstrated quieter behavior resulted in reduced predation (Mumme 1992). My observations that male and female Dickcissel parents continued to care for young until they were at least 35 days old contrasts with previous reports in Kansas that parents only provide care for 14 days and that males abandon young to join postbreeding flocks (Zimmerman 1993). However, these data are in agreement with recent research that documented parental care (sex of parent was not reported) for at least four weeks postfledging (Berkeley et al. 2007).

   Fledgling survival.—Most (83%) mortality for Dickcissel fledglings occurred in very young (≤2 days old) birds. This is consistent with previous studies of Dickcissel fledglings. Suedkamp Wells et al. (2007) reported 87% of Dickcissel fledgling mortality occurred in the first week and Berkeley et al. (2007) reported elevated mortality throughout the first eight days of life. High mortality immediately postfledging has also been reported in other grassland songbirds (Yackel Adams et al. 2006), including 64% mortality for Eastern Meadowlarks (*Sturnella magna*) in the first three days postfledging (Kershner et al. 2004). My mortality rates were higher than other studies of Dickcissel fledglings, which may be attributed to the presence of RIFA at the field site (Suedkamp Wells et al. 2007). Age had a strong effect in my survival models, which is commonly reported for fledgling studies and concurs with reports of high predation rates immediately postfledging (Naef-Daenzer et al. 2001, Kershner et al. 2004). My observation of a strong seasonal decline in fledgling survival was not expected given that the same study area had high nest survival late in the season (see Chapter 3). This result contrasts with another grassland bird study that documented higher survival during the mid- and late season (Yackel Adams et al. 2006), but is supported by another study on tits (Family Paridae) that reported a strong decrease in survival with mortality rates five times greater in late June than mid-May (Naef-Daenzer et al. 2001). The positive relationship between survival and fledgling mass has been previously demonstrated for Dickcissel (Suedkamp Wells et al. 2007) and other grassland birds; this result may be associated with increased feeding abilities of larger fledglings during drought years (Yackel Adams et al. 2006) or a greater ability to survive periods of food shortages (Lack 1954, Magrath 1991).

   My survival analysis of landscape features was performed in a "top down" hierarchical procedure because predation effects operate on multiple scales, with larger scale processes providing constraints relative to smaller scale effects on bird productivity (Thompson et al. 2002). The negative relationship between survival and grassland area within 400 m contradicted my prediction and has not been previously reported. However, similar to an explanation by Zimmerman (1982) for lower

Dickcissel nest survival in preferred nesting habitat, I speculate this pattern is related to grassland areas in the study landscape having greater predator abundance and richness than surrounding land-cover types such as large rowcrop fields. This is also consistent with another study that reported predator activity in an Iowa agricultural landscape was greatest near grassland blocks (Kuehl and Clark 2002). The relationship between survival and edge density at both coarse and fine scales indicates potential negative effects caused by increased landscape patch heterogeneity. Similar effects are seldom reported in other studies, although this pattern could be associated with negative edge effects caused from increased predator abundance and activity associated with amount of edge in the landscape, which is commonly reported in studies of nest success (Gates and Gysel 1978, Heske et al. 1999, Chalfoun et al. 2002). I documented negligible effects of microhabitat features on fledgling survival. This may be partially explained by the increased movement of Dickcissels at 7 days, which potentially diluted the effects of local vegetation on survival. This contrasted with my prediction and previous research, which often show a positive relationship between Dickcissel fledgling survival and vertical and horizontal vegetation cover, as well as total grass cover (Berkeley et al. 2007). Fledgling concealment was also not related to survival, which may have been partially confounded because Dickcissels closer to the ground have increased concealment, yet remain vulnerable to the majority of predators in this system (i.e., RIFA and snakes). The positive association between perch height and survival is also likely explained by the predator community, as RIFA and snakes inhabiting fields are more mobile on the ground and find prey opportunistically (Allen 2004, Weatherhead et al. 2003). Another study in the same system (see Chapter 3) reported increased survival of Dickcissel nests placed higher off the ground, which occurred in habitats with greater woody cover. Hence, the presence of woody vegetation was important to fledgling survival on this farm as it allowed them to spend less time on the ground, where they may be more vulnerable to predators.

Mortality causes.—Red imported fire ants were the primary confirmed predator of Dickcissel fledglings in this study. This likely contributed to my observations of low overall survival and high mortality immediately postfledging, as RIFA depredated fledglings at a mean age of 0.51 ± 0.11 days ($\pm$ SE). This result is of conservation importance, as RIFA depredation was largely negligible for birds older than 1 day and the oldest bird confirmed to be killed by RIFA was 4 days old. The estimated mortality caused by RIFA may be high as their presence was easily detected and could have occurred post-mortality; however, I do not believe this was a common occurrence as fledglings were tracked on a frequent basis (twice daily) and RIFA were often observed on fledglings during the tracking period prior to their death. Snakes were also a frequent predator, which concurs with

previous studies of Dickcissel fledglings in Missouri, Nebraska and western Iowa (Berkeley et al. 2007, Suedkamp Wells et al. 2007). Frequent snake depredation is also consistent with reports from grassland songbird nests in old fields in Missouri (Thompson and Burhans 2003, Stake et al. 2005). The most commonly identified snake was Southern Black Racer (*Colubris constrictor priapus*) and I suspect that most of the depredations from unidentified snakes were also this species, as it was the only snake species confirmed at the field site that was difficult to track and capture. I speculate that many unknown causes of mortality resulted from the intense heat at my field site, which has been previously suggested for Dickcissel fledgling deaths in Missouri (Suedkamp Wells et al. 2007). The field site also experienced occasional periods of drought, which likely resulted in some unclassified mortality events although assigning fate to this cause was difficult (Yackel Adams et al. 2006). Mortality in USDA conservation practices was highest in RFB where nearly all deaths were caused by RIFA. This is a noteworthy observation, as RFB had the highest RIFA mound density of all practices (see Chapter 3). I radio-tagged 38 fledglings in RFBs in 2007 and 28 were killed within one day by RIFA. RFBs are the most abundant conservation buffer practice in Mississippi (Clark and Reeder 2007), attract greater Dickcissel nest densities than other conservation buffers, are negatively related with Dickcissel nest survival (see Chapter 3), and had the highest Dickcissel fledgling mortality rate among conservation practices. This highlights a potential conservation concern for RFBs in geographic areas that contain high RIFA densities.

       Conservation implications.—The fledgling period is likely the most vulnerable life stage for Dickcissels (Berkeley et al. 2007, Suedkamp Wells et al. 2007). Our understanding of the relationship between fledgling survival and microhabitat and landscape context of established patches may enhance management protocols for grassland birds. The high early season nesting activity in conservation buffers in concert with substantially greater fledgling success rates during this time period may result in a disproportionate effect on total recruits to the Dickcissel population. Furthermore, the high nest success occurring late in the breeding season (see Chapter 3) may not benefit Dickcissel population trajectories if fledgling survival is low. As such, I recommend management practices that promote increased vegetative cover early in the breeding season. My study highlights the importance of region-specific studies to capture effects of local biological processes, as RIFA had a substantial effect on Dickcissel fledgling survival that had not been previously reported. This effect was most obvious in RFBs, which appeared to function as an ecological trap (Gates and Gysel 1978) on this farm. I suggest that future research investigate the potential of integrating perennial grasses and/or forbs in RFBs to reduce RIFA mound densities, as these buffers may provide suitable breeding and postfledging habitat in the absence of RIFA.

Furthermore, my results elucidate the need to understand fledgling ecology in concert with other life stages to realize the full spectrum of grassland bird ecology and enhance efficacy of management decisions. Management protocols that focus on maximizing fledgling survival within just a few days postfledging may substantially enhance grassland bird benefits in these conservation practices. In summary, my study highlights that the fledgling life stage is a vulnerable period for Dickcissels, particularly during the first two days postfledging, and I urge land managers to incorporate knowledge of fledgling ecology in their management decisions.

ACKNOWLEDGMENTS

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Table 1. Cause-specific mortality for Dickcissel fledglings in four conservation practices amongst intensive agriculture in the Mississippi Alluvial Valley, MS, 2006-2007.

Practice[a]	Year	n	Mortality[b]	RIFA		Snake		Mammal		Other[c]	Unknown
				n	%	n	%	n	%		
BLK	2006	112	62	28	25	25	22	13	12	3	12
	2007	114	66	35	31	14	12	19	17	7	17
FSD	2006	6	83	1	17	1	17	3	50	0	0
	2007	15	67	3	20	1	7	2	13	4	1
FSM	2007	4	100	4	100	0	0	0	0	0	0
RFB	2006	3	67	2	67	0	0	0	0	0	0
	2007	33	97	28	85	2	6	2	6	0	0

[a]BLK = early-succession, afforestation blocks, FSD = diverse-planted filter strips, FSM = monotypic filter strips, RFB = early-succession, riparian forest buffer

[b]This estimate represents the proportion of fledglings confirmed dead relative to all radio-tracked birds

[c]These values include human, weather, drowning, plucked birds, and one owl

Table 2. Initial escape behavior and mean flight distances (m) for Dickcissel fledglings in the Mississippi Alluvial Valley, MS, 2007.

Age (days)	Total dead	Flutter n	Flutter %	Freeze n	Freeze %	Run/Hop n	Run/Hop %	Fly n	Fly %	Flight mean ± SE
1	99	6	6	76	77	17	17	0	0	
2	78	10	13	43	55	21	27	4	5	
3	72	7	10	34	47	18	25	13	18	
4	49	2	4	22	45	10	20	15	31	8.30 ± 3.03
5	48	10	21	12	25	10	21	16	33	8.67 ± 2.04
6	35	1	3	9	26	13	37	12	34	14.35 ± 4.47
7	42	1	2	14	33	8	19	19	45	12.42 ± 1.98
8	38	0	0	9	24	7	18	22	58	12.63 ± 1.89
9	40	4	10	5	13	4	10	27	68	20.57 ± 2.99
10	30	3	10	3	10	3	10	21	70	21.00 ± 5.33
11	31	1	3	4	13	3	10	23	74	20.10 ± 2.96
12	28	1	4	2	7	2	7	23	82	22.00 ± 3.79
13	27	0	0	1	4	1	4	25	93	19.94 ± 2.34
14	26	0	0	1	4	1	4	24	92	17.74 ± 2.33
15	25	0	0	1	4	1	4	23	92	23.67 ± 1.42

[a] Mean flight distances were only calculated for ages when at least 10 flights were measured.

Table 3. Hierarchical model selection results for Dickcissel (*Spiza americana*)
fledgling survival in the Mississippi Alluvial Valley, MS, 2006-2007.

Model[a]	ΔAICc[b]	w_i[c]	K[d]	Deviance
Model step 5				
AgeTT+Year*T+Gr400+ED400+Perch+Mass	0.00	0.36	10	663.19
AgeTT+Year*T+Gr400+ED400+Perch	0.75	0.25	9	665.96
AgeTT+Year*T+Gr400+ED400+ED_fine+Perch	1.27	0.19	10	664.46
AgeTT+Year*T+Gr400+Perch	1.28	0.19	8	668.52
Model step 4				
AgeTT+Year*T+Gr400	13.15 (0.00)	0.00	7	682.41
AgeTT+Year*T+Gr400+ED400	13.33 (0.18)	0.00	8	680.57
AgeTT+Year*T+Gr400+ED400+ED_fine	13.38 (0.23)	0.00	9	678.60
AgeTT+Year*T+Gr400+HD	13.76 (0.60)	0.00	8	680.99
AgeTT+Year*T+Gr400+ED400+HD	14.13 (0.98)	0.00	9	679.34
AgeTT+Year*T+Gr400+ED400+ED3+HD	14.17 (1.02)	0.00	10	677.36
AgeTT+Year*T+Gr400+ED400+ED3+Forb	14.59 (1.44)	0.00	10	677.78
AgeTT+Year*T+Gr400+Forb	14.63 (1.48)	0.00	8	681.87
AgeTT+Year*T+Gr400+ED400+Forb	14.71 (1.56)	0.00	9	679.93
AgeTT+Year*T+Gr400+RIFA	14.84 (1.68)	0.00	8	682.07
AgeTT+Year*T+Gr400+ED400+ED_fine+RIFA	14.95 (1.79)	0.00	10	678.14
AgeTT+Year*T+Gr400+Woody	15.06 (1.90)	0.00	8	682.29
AgeTT+Year*T+Gr400+Tcov	15.09 (1.94)	0.00	8	682.32
AgeTT+Year*T+Gr400+ED400+RIFA	15.09 (1.94)	0.00	9	680.30
AgeTT+Year*T+Gr400+Grass	15.13 (1.97)	0.00	8	682.36
Model step 3				
AgeTT+Year*T+Gr400	13.15 (0.00)	0.00	7	682.41
AgeTT+Year*T+Gr400+ED400	13.33 (0.18)	0.00	8	680.57
AgeTT+Year*T+Gr400+ED400+ED_fine	13.38 (0.23)	0.00	9	678.60
AgeTT+Year*T+Gr400+ED_fine	13.94 (0.79)	0.00	8	681.17
AgeTT+Year*T+Gr400+Row400	14.36 (1.21)	0.00	8	681.59
AgeTT+Year*T+Gr400+Row_fine	14.38 (1.23)	0.00	8	681.61
AgeTT+Year*T+Gr400+ED400+Row_fine	14.49 (1.33)	0.00	9	679.70
AgeTT+Year*T+Gr400+Gr_fine	14.90 (1.75)	0.00	8	682.14
AgeTT+Year*T+Gr400+Row400+ED_fine	14.93 (1.78)	0.00	9	680.15
AgeTT+Year*T+Gr400+ED400+Gr_fine	15.13 (1.98)	0.00	9	680.35
Model step 2				
AgeTT+Year*T+Gr400	13.15 (0.00)	0.00	7	682.41
AgeTT+Year*T+Gr400+ED400	13.33 (0.18)	0.00	8	680.57
AgeTT+Year*T+Gr400+Row400	14.36 (1.21)	0.00	8	681.59
AgeTT+Year*T+Gr400+Wd400	15.16 (2.00)	0.00	8	682.39
Model step 1				
AgeTT+Year*T	23.31 (0.00)	0.00	6	694.58

[a] Models are ranked by ascending ΔAICc. TT = quadratic seasonal trend; T = linear seasonal trend; 400 = spatial metric refers to within 400 m; fine = spatial metric refers to within 30 or 100 m; Gr = grassland cover; Perch = perch height; ED = edge density; HD = horizontal vegetation density; RIFA = red imported fire ant; Tcov = total vertical cover; Row = rowcrop cover; Wd = all woody cover >15 years old.

[b] ΔAICc is based on a best approximating model with an AICc value of 683.32. Numbers in parentheses denote within-stage ΔAICc values
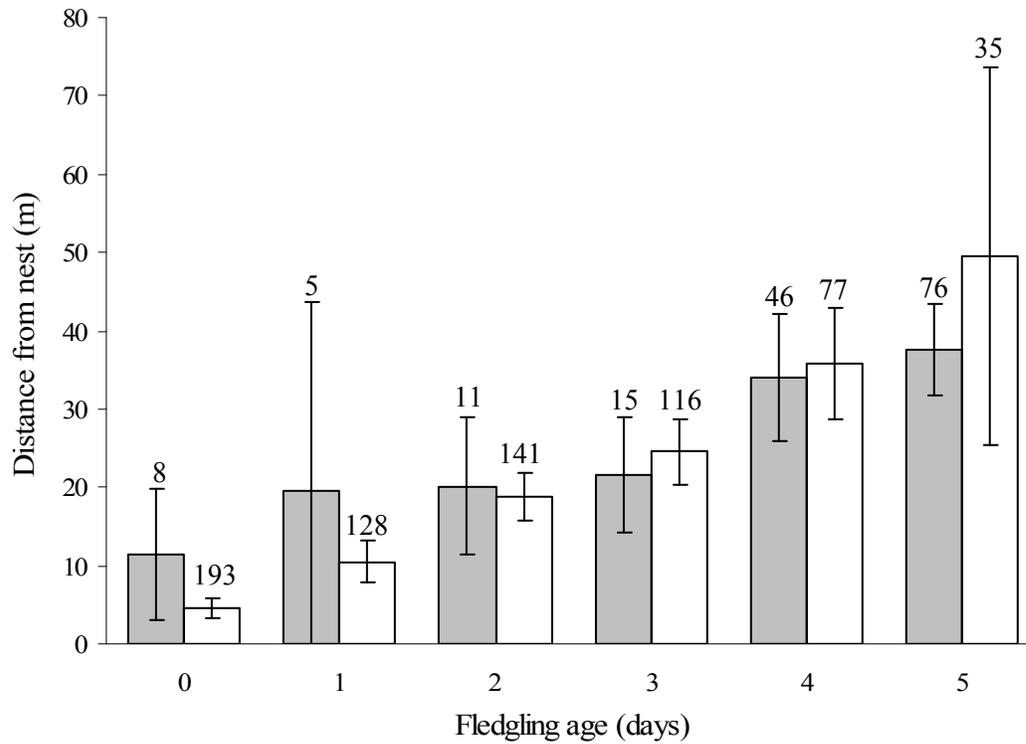
[c] AIC model weight

[d] Number of parameters

Figure 1. Distances moved (±95% CI, sample sizes above error bars) away from nest within 5-days
postfledging by Dickcissel fledglings with leg (shaded bars) and body (open bars) harness styles
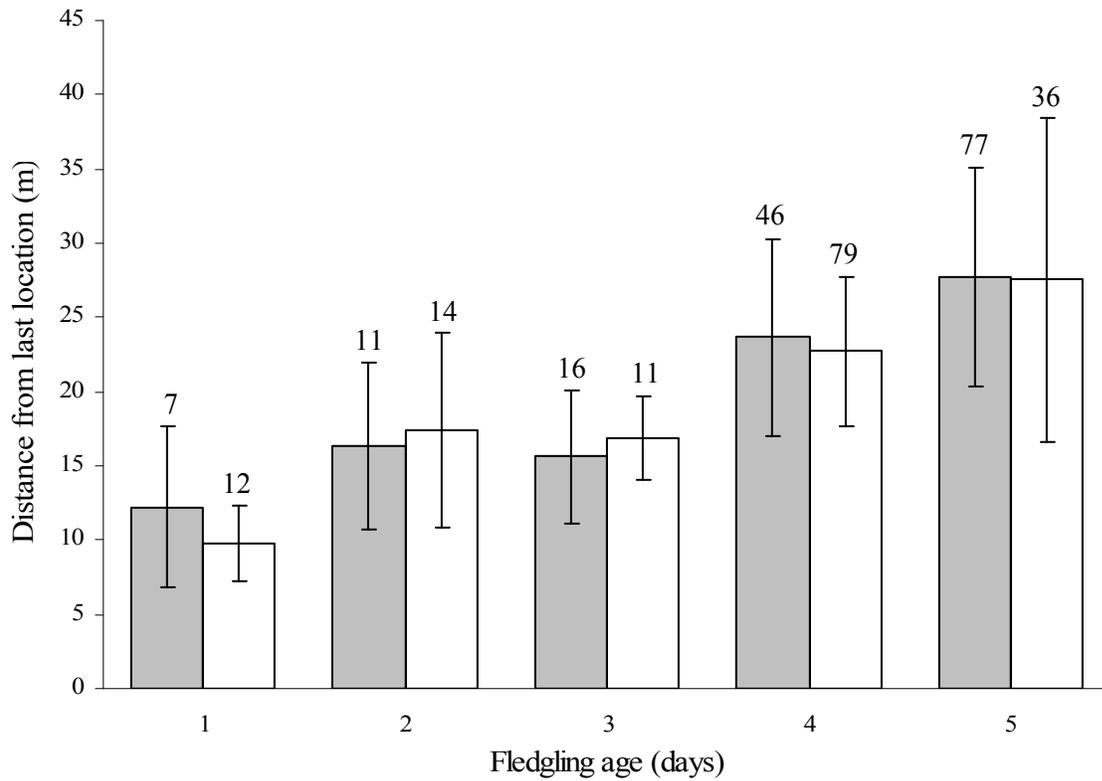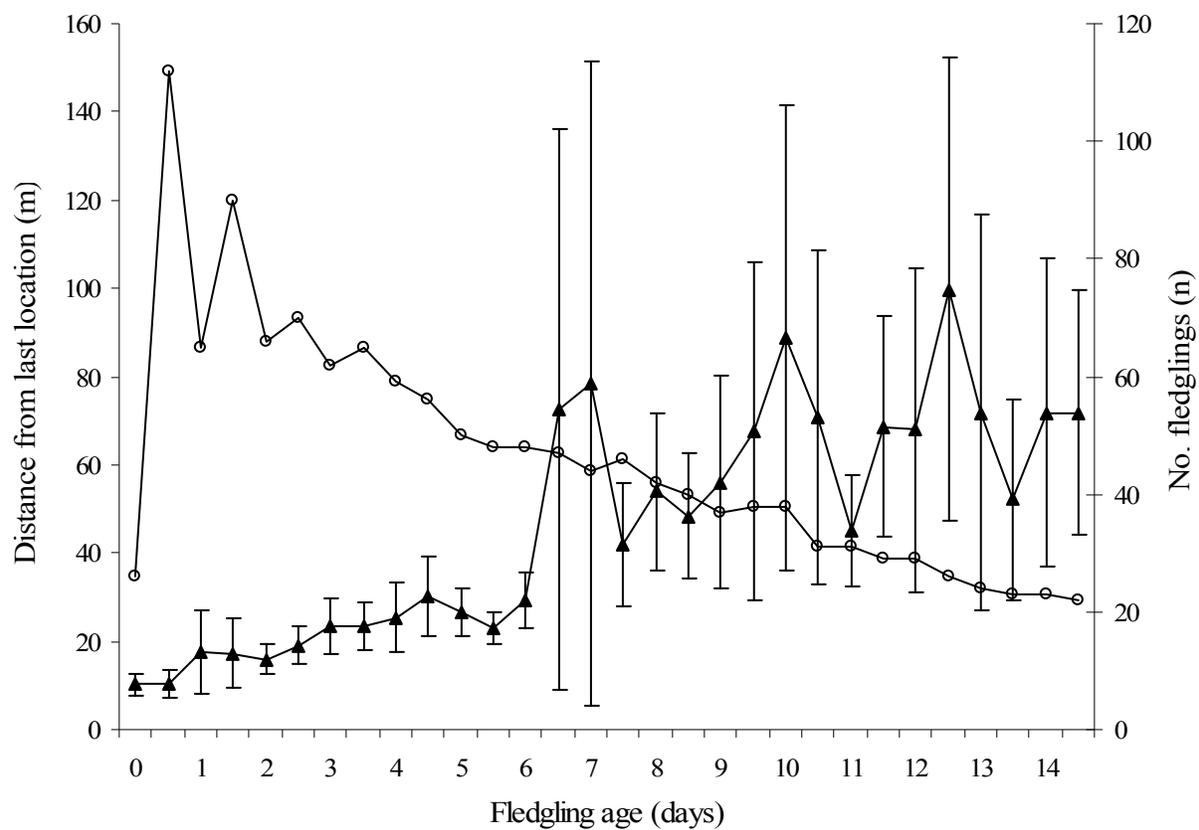in the Mississippi Alluvial Valley, MS, 2006-2007.

Figure 2. Distances moved (±95% CI, sample sizes above error bars) from previous location per day within 5-days postfledging by Dickcissel fledglings with leg (shaded bars) and body (open bars) harness styles in the Mississippi Alluvial Valley, MS, 2006-2007.

Figure 3. Mean distance (m) moved (±95% CI) since previous tracking location relative to postfledging age (days) for Dickcissel fledglings in the Mississippi Alluvial Valley, MS, 2006-2007. The right y-axis refers to fledgling sample size.
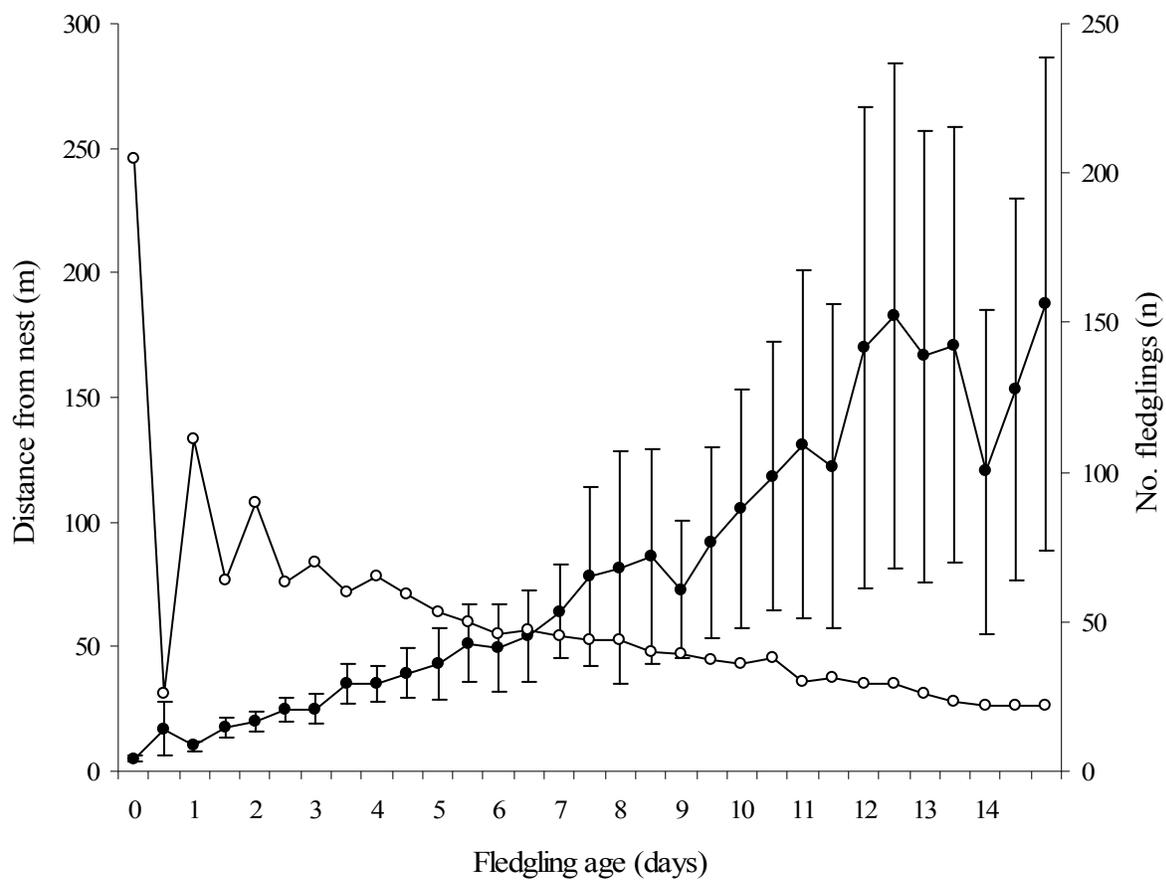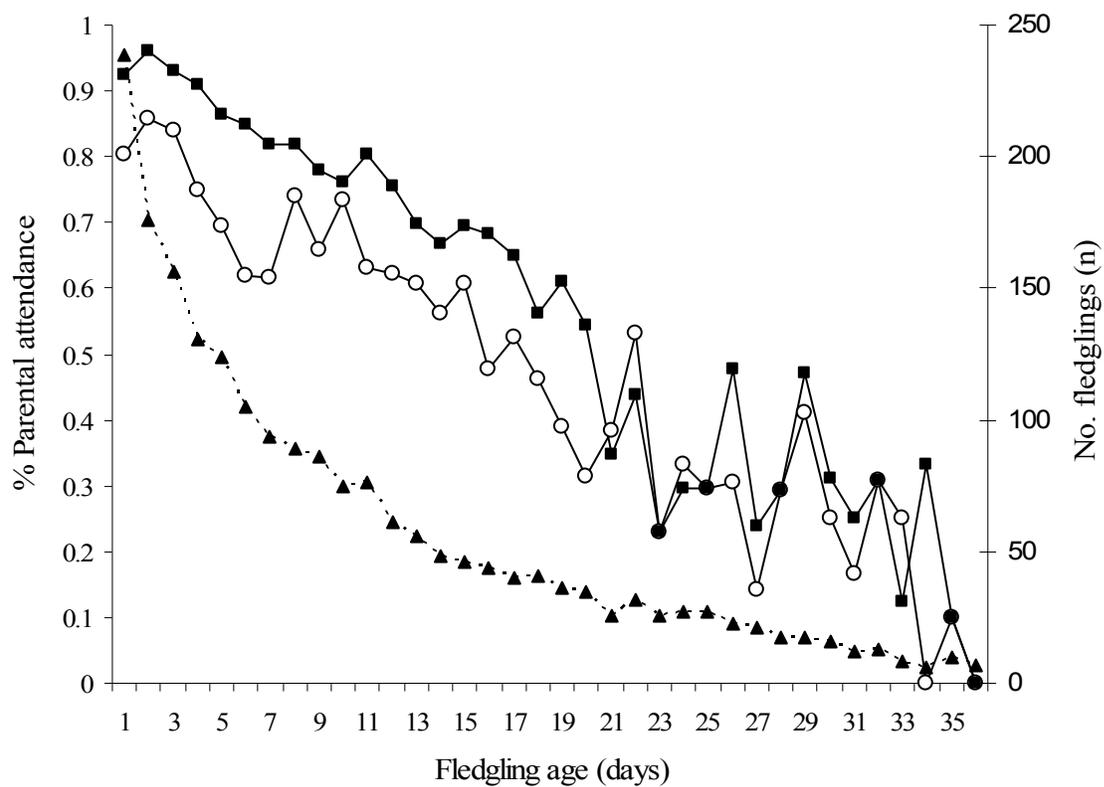
Figure 4. Distance (m) moved (±95% CI; filled circles) away from the nest relative to fledgling age
(days; open circles) for Dickcissel fledglings in the Mississippi Alluvial Valley, MS, 2006-2007.
The right y-axis refers to fledgling sample size.

Figure 5. Observed proportions of parental attendance for male (open circles) and female (closed squares) adult Dickcissels relative to fledgling age (n = filled triangles) in the Mississippi Alluvial Valley, MS, 2006-2007. The right y-axis refers to sample size.
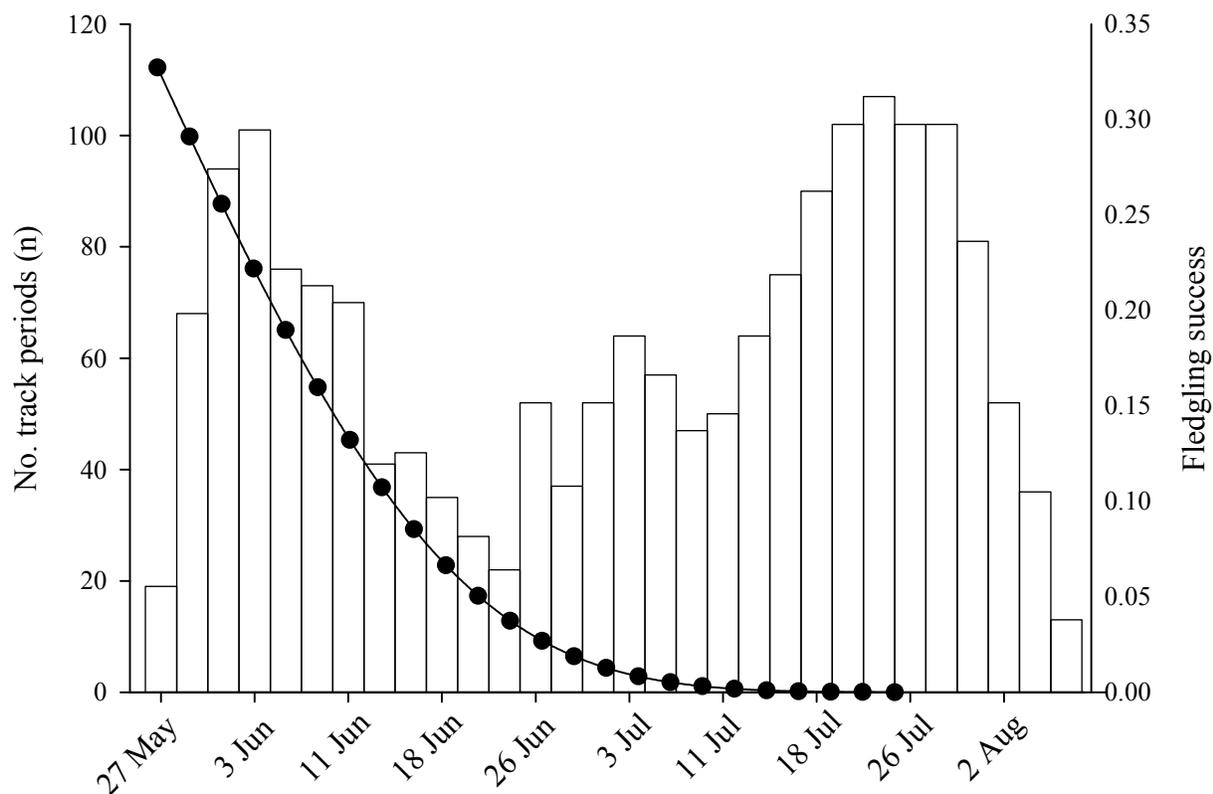
Figure 6. Probability of surviving a 15-day period (filled circles) and effective sample sizes (i.e., radio ½ days) in 5-day intervals (histogram) for Dickcissel fledglings in the Mississippi Alluvial Valley, MS, 2006-2007.
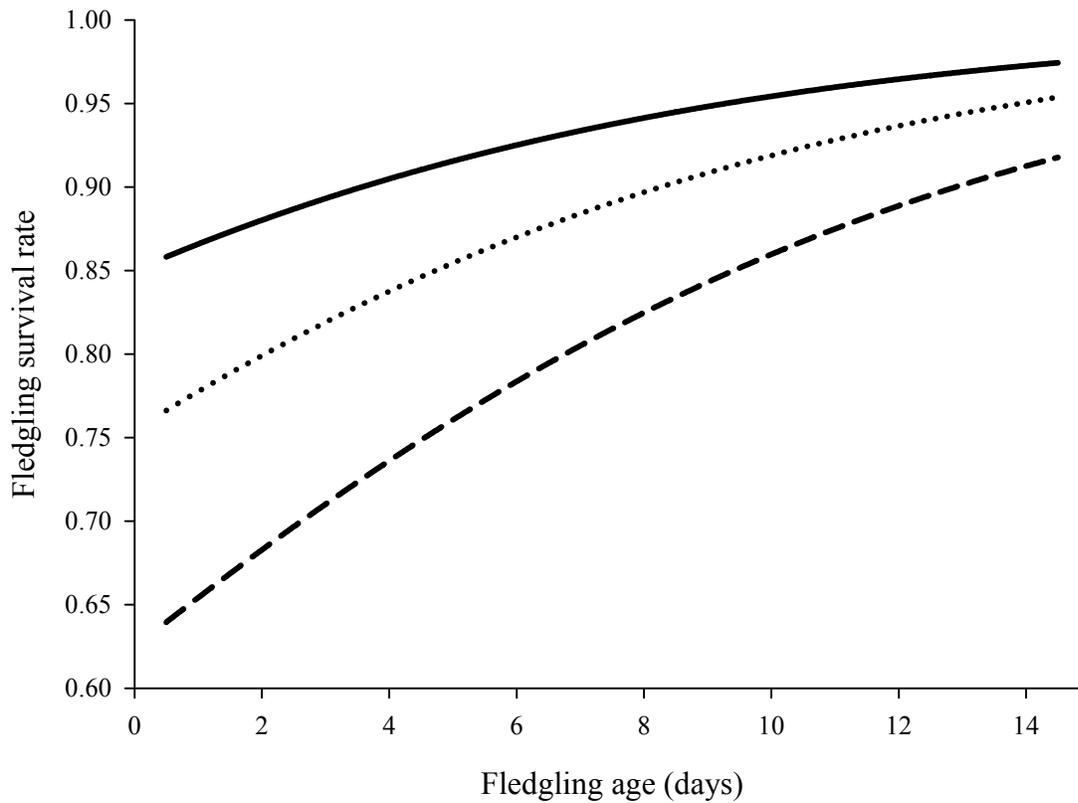
Figure 7. Dickcissel fledgling survival rates (half-day intervals) as a function of postfledging age (days) in the Mississippi Alluvial Valley, MS, 2006-2007. Survival decreased concomitantly from early (12 June to 26 June, solid line), to middle (28 June to 11 July, dotted), and late (12 July to 26 July, dashed) in the breeding season, but increased within each 15-day period as a result of fledgling age.
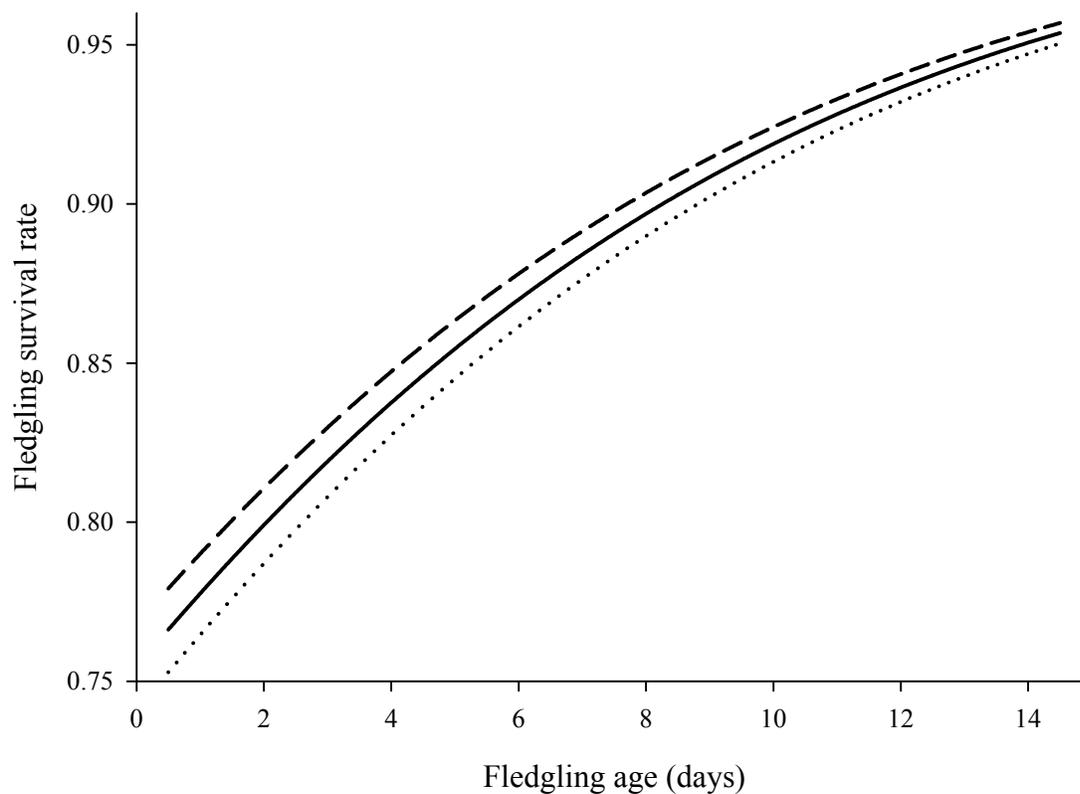
Figure 8. Dickcissel fledgling survival rates (half-day intervals) were estimated in a 15-day time period, with a 28 June start date, in the Mississippi Alluvial Valley, MS, 2007. This graph portrays important covariates (as reported by Program MARK) at mean values (solid line, coarse scale grassland area (Gr400) = 1.76 ha, perch height = 1.66 cm), in a best survival scenario (dashed line, Gr400 (-1SD) = 0.85 ha and perch height (+1 SD) = 3.12 cm), and worst survival scenario (dotted line, Gr400 (+1SD) = 2.66 ha and perch height (-1 SD) = 0.19 cm).

CHAPTER 5.
LANDSCAPE-LEVEL RESPONSE OF NORTHERN BOBWHITE (*COLINUS VIRGINIANUS*) TO A CONSERVATION MANAGEMENT SYSTEM IN THE MISSISSIPPI ALLUVIAL VALLEY

ABSTRACT

Northern Bobwhite populations have been declining for several decades, and these declines have continued post-implementation of the Conservation Reserve Program (CRP). As Northern Bobwhite are strongly associated with agriculture, establishing suitable non-crop habitat in rowcrop landscapes may help sustain or restore their populations. Northern Bobwhites use a diversity of land-cover types in heterogeneous landscapes and respond positively to herbaceous strips. As such, an optimal management strategy is to target establishment of various block and strip-shaped non-crop habitats within a Conservation Management System (CMS). I investigated Northern Bobwhite responses to a CMS in a 7,818 ha region within the intensive agricultural landscape of the Mississippi Alluvial Valley. This region encompasses a CMS that has established four primary conservation practices through the CRP, including early-succession, afforestation blocks, early-succession riparian forest buffers, diverse filter strips, and monotypic filter strips. I counted 302 singing male Northern Bobwhites at 100 fixed point count stations across three years (2005-2007) and modeled their abundance relative to landscape characteristics and conservation practices within a 400m radius of each point. I observed lower abundances at sites with increased rowcrop area, increased wooded area, increased developed area, and greater landscape contagion. All four conservation practices positively related to Northern Bobwhite abundances, which were 3.3 times greater within the CMS than the surrounding unmanaged landscape. Diverse filter strips had the strongest effect, whereas the effect of monotypic filter strips was negligible. My study highlights the important supplemental effects of properly managed filter strips to Northern Bobwhite populations in a CMS. I recommend incorporating 30 m wide strips of forbs and perennial grasses for Northern Bobwhite, but not monotypic plantings.

INTRODUCTION

Range-wide population declines of Northern Bobwhite (*Colinus virginianus*) have been recognized for several decades (Brennan 1991, Veech 2006). These declines have been largely attributed to agricultural practices and other factors realized at broad landscape scales (Hunter et al. 2001, Burger 2002). Continued declines (-4.48 %/year from 1985-2007; Sauer et al. 2008) after implementation of the Conservation Reserve Program (CRP) has heightened conservation concerns (Brennan and Kuvlesky 2005). In the Mississippi Alluvial Valley (MAV) their numbers have declined locally at 2.26 %/year from 1966-1985 and 6.76 %/year from 1985-2007 (Sauer et al. 2008). Land-use changes

have reduced useable space in the landscape for Northern Bobwhite, which locally prefer patches with a diversity of vegetation types and successional stages as well as permanent suitable cover (Ellis et al. 1969, Burger et al. 1990, Guthery et al. 2005). Guthery (1999) suggested that the goal of bobwhite habitat management should be to maximize usable space through time in a given landscape, where usable space is defined at the landscape level as the proportion of point-locations that provide the structural characteristics and resources required for survival and reproduction. However, bobwhite depend on multiple successional stages (e.g., annual weed, perennial grass, shrub) to fulfill seasonal life-history requisites (nesting, brood-rearing, roosting, predator avoidance, etc.). Within managed agricultural landscapes, different seral stages are most often distributed in patches defined by land-use practices. Thus, at the landscape scale, bobwhite are often most abundant in landscapes defined by diverse and heterogeneous patch distribution with moderate amounts of grassland, rowcrop, and woody edge, although their response to land-cover edges remains uncertain (Roseberry and Klimstra 1984, Roseberry and Sudkamp 1998, Guthery et al. 2001, Twedt et al. 2007). Conflicting reports of associations between bobwhite abundance and specific land-use practices or land cover (e.g., woodlands, rowcrop, grassland, etc.) are most likely due to the respective landscape context in which the various studies were conducted and the nonlinear nature of habitat relationships.

The Northern Bobwhite Conservation Initiative (NBCI) is designed to restore populations to their 1980 densities by implementing management practices that provide usable space in multiple land-use classes across large regional areas (Bird Conservation Regions). Within the planning boundaries of the NBCI, agricultural landscapes have been identified as most easily improvable for bobwhite because they can provide usable space given appropriate conservation practices, are amenable to manipulation and thus have potential for increasing local Northern Bobwhite populations (Guthery et al. 2005, Riddle et al. 2008). Rowcrop exists predominantly on private lands and hence, the use of farm policy (e.g., CRP) is the most effective manner to achieve a critical mass of habitat suitable to restore and sustain Northern Bobwhite populations (Burger et al. 2006). Furthermore, it is preferable for these efforts to proceed in a targeted, landscape-level approach rather than managing on diffuse, fine-scales or region-wide scales, as such management strategies may be ineffective to restore Northern Bobwhite populations (Burger 2002, Williams et al. 2004).

Rowcrop agriculture represents 87.7% of all land-use types in the MAV that are considered suitable and manageable habitat for Northern Bobwhite (Dimmick et al. 2002). In production-based, agricultural landscapes where large, grassland blocks are less compatible with land-use objectives, management decisions require further insight on the differential effects of block and strip grassland patches. Conservation buffers (i.e., non-crop strips of vegetation) are targeted for enrollment by the

NBCI and are more conducive than large blocks in production-based landscapes. Multiple buffer practices have been reported to fulfill several Northern Bobwhite life-history requirements, including escape cover, movement corridors, and nesting habitat (Puckett et al. 2000, Palmer et al. 2005). Riddle et al. (2008) demonstrated a positive influence of linear and nonlinear buffers on populations in an agriculture-dominated landscape, although their landscape lacked the moderate levels of grassland cover that is of primary importance to Northern Bobwhite (Roseberry and Sudkamp 1998, Guthery et al. 2001, Twedt et al. 2007). Smith and Burger (*in press*) reported that depending on the amount of buffers in the landscape, buffers alone may not provide sufficient habitat to increase and sustain Northern Bobwhite populations, thus highlighting the need to understand the influence of multiple conservation practices with different shape, area, and vegetation structure on bobwhite populations.

Based on recommendations for a targeted, landscape-level approach, an optimal conservation management system (Burger 2002) may require integrating greater overall grassland cover in addition to conservation buffers. However, because bobwhite are dependent on multiple successional stages to meet different life-history requisites, they are often most abundant in landscapes with moderate, but not indefinitely large, amounts of grassland cover. It is therefore crucial to test for supplemental benefits provided by buffers in landscapes containing larger grassland patches prior to implementation. Spatially explicit habitat models can assist these management decisions by providing insight on farm-level, habitat associations of Northern Bobwhite during the breeding season (Schairer et al. 1999). My study investigated the response of Northern Bobwhite to the application of a comprehensive, objective-driven CMS at the farm level in the intensively rowcropped MAV. I hypothesized that bobwhite abundance would relate 1) positively to overall grassland area and focal conservation practices, but 2) negatively with wooded, developed, and rowcrop area in this landscape.

METHODS

Study site.—The 7,818 ha study area included a targeted 2,630 ha study farm in the center. The target farm was located in Coahoma County, Mississippi, USA (34° 18' N, 90° 34' W), which is in the MAV (Bird Conservation Region 26). This physiographic region is characterized by intensive, rowcrop (e.g., cotton, soybean, and corn) operations. The MAV has nominal topographic relief and the agricultural landscape is sparsely fragmented with non-crop, strip habitats. The landscape matrix surrounding the study farm was intensively rowcropped and composed of 83% rowcrop, 8% forested or herbaceous wetland, 4% wooded, 3% developed, and less than 2% non-crop herbaceous cover.   In contrast, the targeted farm had implemented a myriad of conservation practices as part of an overall CMS. This property was composed of 48% rowcrop, 30% early-succession hardwood afforestation

plantings (CP3a), 14% forested or herbaceous wetlands, 4% strip habitats, 2% forested, and 2% herbaceous drains. Soil associations on the farm included Sharkey clay (41%), Dowling clay (15%), and Tunica silty clay (11%) (Soil Survey Staff 2008). Monthly rainfall was relatively sparse during all years in June (2005 = 1.83 cm, 2006 = 2.72 cm, 2007 = 5.00 cm).

   Northern Bobwhite monitoring.—I conducted breeding season, call count surveys for singing male Northern Bobwhite in a two-week period (7-21 June) across three years (2005-2007). I monitored 100 call count stations systematically distributed in a 10x10 square grid across the study area, with 800 m between each station, extending 1.6 km beyond the CMS boundary. Locations of singing males were recorded to limit counts to unique individual males. Surveys lasted 5 minutes, occurred within 2 hours after sunrise, and were conducted on mornings with no precipitation and wind speed <12 km/hr. Counts were conducted by multiple observers, although this did not likely cause any bias because the Northern Bobwhite song is easily recognizable and all observers received some training prior to surveys. Forty-two points fell within the CMS and 58 fell on unmanaged farm land outside the CMS.

   Landscape characteristics.—The surveyed landscape included four conservation practices of conservation interest, including early-succession, bottomland hardwood, afforestation blocks (BLK), and three conservation buffers; early-succession, riparian forest buffers (RFB), monotypic filter strips (FSM) and diverse filter strips (FSD). Afforestation blocks had a mean area of 25.80 ± 4.14 ha. They were planted primarily with Nuttall's oak (*Quercus nutallii*), water oak (*Quercus nigra*), and willow oak (*Quercus phellos*) in the fall of 1999 and were still in the early-succession seral stage, with considerable herbaceous cover and enhanced growth transitioning to shrub-successional habitat. Riparian forest buffers were planted 60 m wide with hardwood trees in the fall of 2004, but throughout the study were dominated by pioneer species. Pioneer species that naturally invaded were similar for RFBs and BLKs, including horsestail (*Conyza canadensis*), redvine (*Brunnichia cirrhosa*), vetch (*Vicia* sp.), goldenrod (*Solidago* spp.), giant ragweed (*Ambrosia trifida*), curly dock (*Rumex crispus*), dewberry (*Rubus trivialis),* blackberry (*Rubus oklahomus*), johnsongrass (*Sorghum halepense),* poison ivy (*Toxicodendron radicans*), and broomsedge (*Andropogon virginicus*). Both filter strip practices were established in the spring of 2004. Monotypic filter strips were densely planted (9kg/ha) with switchgrass (*Panicum virgatum*), which completely dominated the vegetative community. Diverse filter strips were planted with partridge pea (*Chamaecrista fasciculata*, 4.5 kg/ha seeding rate), Indian grass (*Sorghastrum nutans*, 1.7 kg/ha seeding rate), little bluestem (*Schizachyrium scoparium*, 5.6 kg/ha seeding rate), and big bluestem (*Andropogon gerardii*, 1.7

kg/ha seeding rate). Filter strips will be maintained using planned disturbance regimes, but no management occurred before or during this study.

I developed a geographic information system (GIS) in ArcGIS (ArcMap Version 9.3) from 2007 National Agriculture Imagery Program digital orthophoto quarter quadrangles, complemented by empirical ground observations, to characterize the landscape composition and structure in a 7,818 ha region encompassing the study area. Eighteen identified land-cover types included rowcrop (69%), late-succession forested wetland (10%), early-succession CRP (7%), mid-succession (>15 years old) CRP (4%), late-succession forest (4%), developed (2%), RFB (1%), Wetland Reserve Program (WRP) that was entirely herbaceous (<1%), FSD (<1%), herbaceous drainages (<1%), wetland (<1%), miscellaneous herbaceous (<1%), pasture (<1%), wooded drainage (<1%), food plot (<1%), FSD (<1%), open water (<1%), and FSM (<1%). These were re-classified into 10 land-cover classes that I considered relevant based on previous research on Northern Bobwhite ecology. Reclassified land-cover classes included 1) grassland (i.e., herbaceous, pasture, buffers, early-succession blocks, food plots, herbaceous drainage, WRP), 2) wooded (i.e., late-succession forest, all mid-succession forest, late-succession forested wetland, wooded drainage), 3) rowcrop, 4) developed, 5) non-treatment grassland (i.e., all grassland that was not a CRP practice), 6) RFB, 7) FSD, 8) FSM, 9) BLK, and 10) all conservation buffers. I included five additional spatial metrics previously reported of importance to Northern Bobwhite (Roseberry and Sudkamp 1998, Guthery et al. 2001, Twedt et al. 2007). They included 11) total grassland edge, 12) total wooded edge, 13) total rowcrop edge, 14) edge density (total edge m/ha), and 15) landscape contagion, for a total of 15 independent variables. I characterized landscape variables within a 400 m radius (50.24 ha) surrounding each of the 100 call count stations to represent the median size of Northern Bobwhite home range in Mississippi (Smith 2005). I used ArcINFO software to buffer and clip spatially-explicit, land-cover information from ArcGIS vector data at each call count station. Spatial metrics were calculated using FRAGSTATS software and included area of all land-cover classes, total edge for grassland, wooded, and rowcrop cover, edge density, and landscape contagion (McGarigal et al. 2002). Edge density and contagion are measures of landscape spatial heterogeneity, which has been previously correlated with Northern Bobwhite distributions (Roseberry and Klimstra 1984, Roseberry and Sudkamp 1998, Twedt et al. 2007). Edge density indicates the amount of edge (m) to area (ha) and contagion indicates overall landscape clumpiness (McGarigal et al. 2002).

Statistical evaluation.—I used abundance-based habitat modeling to assess relationships between Northern Bobwhite and landscape context. My count data were Poisson distributed and I therefore applied a Poisson regression using the GLIMMIX procedure in SAS® software, Version

9.1.3 (SAS Institute Inc., 2002). Independent variables did not change annually, so I summed count data across years and modeled all effects as fixed. Model selection was performed with the Akaike's Information Criterion, adjusted for small sample sizes (AICc), to differentiate explanatory ability of several biologically meaningful model sets, while accounting for parsimony (Akaike 1974, Burnham and Anderson 2002). My data were overdispersed ($\hat{c} > 1$) and so I used QAICc values for model selection, which were calculated using the overdispersion parameter for the global model ($\hat{c} = 2.74$; Burnham and Anderson 2002). Model construction was performed in two steps. I first modeled effects of background landscape characteristics, which included all land-cover types and spatial metrics of interest that did not include conservation practices of conservation interest. All competitive models (≤ 2 ΔAICc) were moved to the second step. The second model construction step incorporated single covariates and relevant combinations of conservation practices to determine their influence on Northern Bobwhite abundance relative to the background landscape composition. I used model averaging for models in the second step to account for model selection uncertainty in coefficient estimation (Burnham and Anderson 2002). I also provide a descriptive summary of the land-cover types at stations that detected 0, 1$\leq$3, and >3 Northern Bobwhites.

RESULTS

I counted 104 singing males in 2005, 90 in 2006, and 108 in 2007. On average, I detected 3.3 times more Northern Bobwhites (mean ±95% CI birds/station) within the CMS planning boundary (1.67 ± 0.31) than the surrounding unmanaged landscape (0.50 ± 0.16). Stations with no detections had significantly greater surrounding rowcrop cover and less grassland cover compared to areas with >3 Northern Bobwhites (Figure 1). Northern Bobwhite occurred in areas with greater cover of BLK, RFB, and FSD, but not FSM (Figure 2).

The first model construction step included background landscape characteristics and demonstrated complementary negative effects of rowcrop, wooded, and developed cover on Northern Bobwhite distributions, as well as a strong additive effect of landscape contagion (Table 1). The second model step was strongly influenced by total grassland cover and grassland edge, although the best model included additive effects of BLK and FSD. Competitive models included RFB, but FSM had a negligible impact on Northern Bobwhite abundance. Weighted coefficient estimates (± SE) for landscape characteristics included negative influences of wooded area (-0.019 ± 0.009), developed area (-0.128 ± 0.040), rowcrop area (-0.010 ± 0.005), and contagion (-0.015 ± 0.006). Of the 9 most competitive models, FSD occurred in 4, BLK in 3, RFB in 2, FSM in 0, and merged conservation buffers in 2. All conservation practices were positively related with Northern Bobwhite abundance in

the landscape and weighted coefficient estimates (± SE) in descending order include FSD (0.133 ± 0.048), RFB (0.023 ± 0.029), BLK (0.010 ± 0.007), and FSM (0.003 ± 0.052).

DISCUSSION

The relationship between Northern Bobwhite population declines and agricultural practices provide land managers with the challenge of establishing non-crop, early-succession vegetation that is compatible with production-based landscapes (Brennan 1991, Burger 2002). This is best achieved with a focal approach that applies federal farm policies however, necessitates an understanding of bobwhite response to multiple CRP practices at the landscape scale (Burger et al. 2006).

Background landscape.—Northern Bobwhite occurred throughout the CMS planning boundary in a non-uniform distribution. Their abundances were negatively related to rowcrop, wooded, and developed area, but positive with grassland area, thus supporting our predictions. Though they may positively relate to rowcrop area, this effect occurs mostly in landscapes dominated by wooded cover with sparse open areas (Riddle et al. 2008). My result of bobwhite rowcrop avoidance supports previous research in agricultural landscapes (Parnell et al. 2002). Forested areas can provide suitable habitat for Northern Bobwhite, though they largely avoid wooded areas with a closed canopy (Schairer et al. 1999, Parnell et al. 2002). This trend was supported by my study, as the wooded patches in the CMS were established in 1989-1991 and had a predominantly (>75%) closed canopy. I observed that bobwhite abundances were largely uninfluenced by wooded edge, which contrasts previous research (Roseberry and Sudkamp 1998) and may relate to their preference for open, grassland habitats that were reasonably abundant in this landscape. Twedt et al. (2007) showed Northern Bobwhite to be negatively related with land-cover edge density and grassland edge in the West Gulf Coastal Plain. I documented a poor relationship with total edge density however; the only strong edge response I detected in this CMS was a positive one to grassland edge. This result is biologically sensible, as bobwhite prefer heterogeneous landscapes with moderate amounts of grassland area (Roseberry and Klimstra 1984, Roseberry and Sudkamp 1998, Twedt et al. 2007). The strong negative relationship with landscape contagion most likely reflects the dominance of large rowcrop fields.

Conservation practices.—Mississippi has enrolled more CRP habitat than any southeastern U.S. state (Burger 2005). As such, it is critical that land managers understand relationships between these habitats and wildlife responses. Positive relationships existed between all four conservation practices and Northern Bobwhite abundance, which support my predictions and previous observations related to grassland cover and buffers (Burger et al. 1990, Roseberry and Sudkamp 1998, Guthery et al. 2001, Riddle et al. 2008). Diverse filter strips had the strongest relationship and explained the

most variation as a single covariate, demonstrating the importance of diverse buffers. This substantiates previous claims that buffers improve landscape suitability for Northern Bobwhite (Puckett et al. 2000, Riddle et al. 2008, Smith and Burger *in press*). The positive effect of FSD on bobwhite abundance is also not surprising, as the diverse, thick cover of this buffer practice likely provides suitable nesting cover (Burger et al. 1990, Puckett et al. 1995). In contrast to FSD, the sparse vegetation and increased bare ground cover of early-succession RFBs may benefit Northern Bobwhite by providing brooding and roosting habitat (Klimstra and Ziccardi 1963, Wiseman and Lewis 1981), although this may be compromised from high densities of red imported fire ants (*Solenopsis invicta*) (Allen et al. 1993, see Chapters 3, 4). Monotypic filter strips had a positive, but negligible effect on bobwhite abundance. The weak relationship with this buffer practice likely relates to its extremely high vegetative density and paucity of vegetation structure. This study occurred pre-management and I speculate both filter strip practices will provide improved habitat for Northern Bobwhite post-disturbance, when vegetation is more diverse (Greenfield et al. 2002). The complementary effects between BLK and conservation buffers were evident, suggesting buffers provide supplemental benefits that further enhance the suitability of landscapes with large grassland blocks. My result of supplemental benefits provided by multiple habitat types concurs with previous recommendations that management of surrounding habitats is necessary to achieve sustainable Northern Bobwhite populations (Palmer et al. 2005, Smith and Burger *in press*). Smith and Burger (*in press*) recommend establishing a minimum of 5-10% land-cover with buffers to elicit a Northern Bobwhite population response. My study elucidates the supplemental benefits provided by less buffer area (4%) in landscapes with substantial amounts (30%) of alternative early-succession vegetation. Whereas this study provides support for all focal conservation practices, it is not an effective replacement to a local-scale investigation in determining patch quality related to specific life-history requirements.

   Conservation implications.—Northern Bobwhite are a flagship species for many North American grassland ecosystems, and associated habitat management practices can impact many non-target grassland songbirds (Riffell et al. 2008). Although the Northern Bobwhite is not native to the bottomland hardwood forests that historically dominated the MAV, their current exploitation of this geographic region represents a promising conservation opportunity given the extensive loss of North American grasslands (Noss et al. 1995). I recommend integrating a suite of conservation buffer practices and large block early-succession habitats across a focal area-sized landscape to increase and maintain bobwhite abundances. Particularly, the establishment of 30 m filter strips with a diverse planting regime can elicit a greater response than monotypic filter strips. This study highlights that

even in an intensive agricultural landscape; a focal, comprehensive habitat management approach can restore and maintain Northern Bobwhite populations. I suggest the NBCI goal of restoring bobwhite populations to their 1980 levels is plausible if conservation practices are deployed with adequate intensity over a sufficiently large geographic region.

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Table 1. Results from abundance-based model selection using $QAIC_c$ for Northern Bobwhite (*Colinus virginianus*) relative to landscape characteristics and four conservation practices in the Mississippi Alluvial Valley, MS, 2005-2007.

Model	$\Delta QAIC_c$[a]	$QAIC_c\ w_i$	K[b]	Deviance
Model step 2				
Row+Wood+Developed+Contagion+BLK+FSD	0.00	0.14	8	476.30
Row+Wood+Developed+Contagion+Grassland TE	0.19	0.12	7	480.45
Row+Wood+Developed+Contagion+Grassland	0.26	0.12	7	480.62
Row+Wood+Developed+Contagion+FSD	0.41	0.11	7	481.05
Row+Wood+Developed+Contagion+BLK+FSD+RFB	1.22	0.07	9	475.92
Row+Wood+Developed+Contagion+Buffers	1.39	0.07	7	483.72
Row+Wood+Developed+Contagion+RFB+FSD	1.61	0.06	8	480.70
Row+Wood+Developed+Contagion	1.62	0.06	6	487.87
Row+Wood+Developed+Contagion+BLK+Buffers	1.97	0.05	8	481.69
Row+Wood+Developed+Contagion+RFB	2.29	0.04	7	486.19
Row+Wood+Developed+Contagion+BLK	2.38	0.04	7	486.45
Row+Wood+Developed+Contagion+FSM	2.90	0.03	7	487.87
Model step 1				
Row+Wood+Developed+Contagion	1.62 (0.00)	0.06	6	487.87
Row+Wood+Developed	3.78 (2.16)	0.02	5	497.19
Row+Wood+Developed+ED	4.42 (2.80)	0.01	6	495.54
Row+Developed	4.84 (3.22)	0.01	4	503.38
Row+Wood+Developed+Row TE	4.93 (3.31)	0.01	6	496.92
Row+Wood+Developed+Wood TE	4.93 (3.31)	0.01	6	496.94
Row+Wood+Developed+Grassland NT	4.99 (3.37)	0.01	6	497.09
Developed	8.73 (7.11)	0.00	3	517.22
Row+Wood	9.66 (8.04)	0.00	4	516.58
Developed+Wood	9.70 (8.08)	0.00	4	516.68
Row	10.12 (8.50)	0.00	3	521.01
Wood	17.06 (15.44)	0.00	3	540.02

[a] $\Delta QAICc$ values are based on a best model QAICc of 182.27. Values in parentheses denote within stage $\Delta QAICc$.

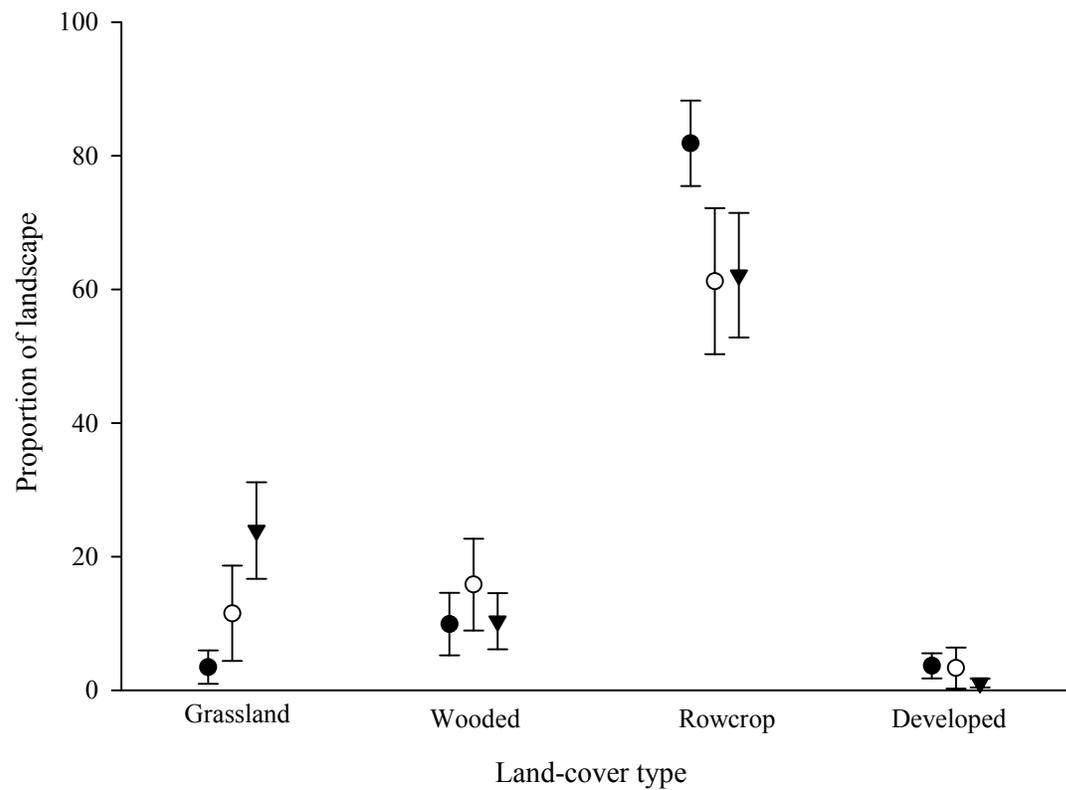[b] This total includes the estimation of the overdispersion parameter, c-hat

Figure 1. Mean landscape proportions (±95% CI) for general land-cover types relative to number of singing male Northern Bobwhites counted (0 = filled circles, 1 to 3 = open circles, >3 = filled triangles) at 100 call-count stations in the Mississippi Alluvial Valley, MS, 2005-2007.

Figure 2. Average proportions ±95% CIs of conservation practices in the landscape relative to number of singing male Northern Bobwhites counted (0 = filled circles, 1≤3 = open circles, >3 = filled triangles) at 100 call-count stations in the Mississippi Alluvial Valley, MS, 2005-2007.
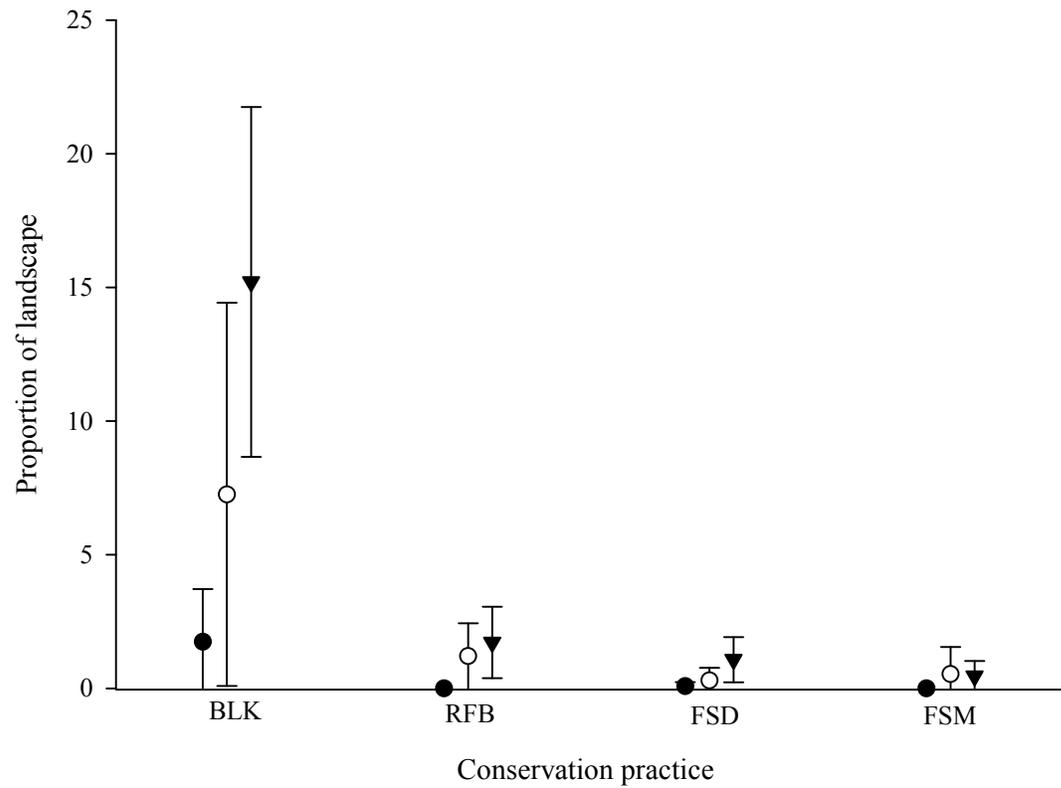
CHAPTER 6.
GENERAL CONCLUSIONS

My research investigated the avian benefits of four conservation practices and their associated establishment procedures, landscape context, and microhabitat features as well as general landscape effects related to wildlife-friendly farming. Conservation practices included early-succession, afforestation blocks (BLK), early-succession riparian forest buffers (RFB), diverse-planted filter strips (FSD), and monotypic-planted filter strips (FSM). Early-succession blocks were the most structurally diverse habitat with moderate proportions of forbs, grasses, and woody substrates. Conservation buffer vegetation differed from BLK in that either grasses or forbs dominated individual buffer practices. My work with birds focused on a general assessment of community structure, a study of nest success as a means of assessing reproductive performance, an in-depth study of the fledgling survival of a focal species (Dickcissel, *Spiza americana*), and a broader landscape-level habitat assessment of a second focal species (Northern Bobwhite, *Colinus virginianus*). My study's results will aid management decisions that may enhance grassland bird benefits in established habitat patches amongst intensive agriculture in the Mississippi Alluvial Valley (MAV).

Avian community structure was more similar amongst early-succession conservation practices than expected, which may be attributed to the relative dominance of all habitats by just two species (Dickcissel and Red-winged Blackbird) and the scarcity of other non-crop habitat in the surrounding landscape. BLKs attracted the greatest abundances of breeding birds and had elevated species richness compared to conservation buffers, which was likely related to vegetative structural complexity (Willson 1974) and greater area (Winter and Faaborg 1999). Bird densities were highest in FSD, perhaps from their enhanced vegetative cover and ecotonal context (Bryan and Best 1991).

Dickcissel and Red-winged Blackbird nested in all buffer and block conservation practices, albeit with varied nesting densities, temporal use patterns, and success. Early-succession blocks attracted greater nesting activity than conservation buffers, particularly for Dickcissel, which is likely related to their preference to place nests in woody plants (Overmire 1962), more complex vegetative structure (Delisle and Savidge 1997), and larger patches of habitat (Winter and Faaborg 1999). Both species selected for similar vegetative structure and had comparable amounts of forb, grass, and woody cover surrounding their nests. I observed no indication that either species exhibited avoidance behavior of high red imported fire ant (RIFA, *Solenopsis invicta*) mound densities. I documented 22.9%

Dickcissel nest success across all conservation practices, which is comparable to previous studies (Hughes et al. 1999, Winter 1999) but still low (McCoy et al. 1999). Low Dickcissel nest success in RFB (10.6%) may indicate a conservation concern because it was their preferred nesting buffer and is the most abundant conservation buffer type in Mississippi (Clark and Reeder 2007). Red-winged Blackbird nest success was 8.61%, which is considerably lower than in Conservation Reserve Program fields that were considered population sinks (McCoy et al. 1999). Their greatest nest success was in FSD (23.4%). I speculate that snakes were an abundant nest predator in this system, which may explain the positive effect of nest height on Dickcissel and Red-winged Blackbird nest success (Harmeson 1974).

Most (83%) Dickcissel fledgling mortality occurred in very young (≤2 days old) birds, which concurs with previous studies (Berkeley et al. 2007, Suedkamp Wells et al. 2007). The elevated mortality rates at my field site relative to others may be attributed to the presence of RIFA (Suedkamp Wells et al. 2007). Age had a strong effect in my survival models and mortality rates decreased for birds surviving past day two. I detected a strong seasonal decline in fledgling survival, which indicates that early season may be the most crucial time for Dickcissel conservation in these practices. The landscape context was important to fledglings, with a negative influence of the combination of greater grassland area with high amount of edges. I observed negligible effects of microhabitat features on fledgling survival, but similar to nest survival, a positive influence of perch height. RIFAs were the primary predator of Dickcissel fledglings and they most commonly depredated birds ≤1 day postfledging. Snakes were also a frequent predator, which concurs with previous studies of Dickcissel fledglings in Missouri, Nebraska, and western Iowa (Berkeley et al. 2007, Suedkamp Wells et al. 2007). Mortality in conservation practices was highest in RFB where nearly all deaths were caused by RIFA.

I documented Northern Bobwhite (*Colinus virginianus*) abundances to negatively relate with greater rowcrop area, wooded area, and landscape contagion, but positively influenced by all four conservation practices. Northern Bobwhites were 3.3 times more abundant within the targeted habitat management area than the surrounding unmanaged landscape. The strongest positive effect was for FSD, whereas FSM had a negligible influence.

Overall, my study highlights the importance of vegetation establishment protocols to enhance wildlife benefits from conservation buffers. Habitats that supported the most diverse

and abundant bird communities as well as greater nesting densities had high structural diversity and moderate vegetative heterogeneity and cover. I recommend that filter strip plantings incorporate diverse forbs and grasses or, at a minimum, moderate seeding rates to promote invasibility of local pioneer species to enhance vegetative heterogeneity. Additionally, I recommend implementing multiple conservation buffer practices to mitigate farmland wildlife population declines from agricultural intensification (Donald et al. 2001, Newton 2004). Overall this research confirmed avian benefits in all conservation practices, although I suggest land managers use caution when establishing RFBs in regions inhabited by RIFAs, as they appear to have a strong negative effect on nest and fledgling survival. I suggest that future research investigate alternative management strategies, such as use of perennial grasses, to reduce RIFA density to enhance wildlife benefits. I also recommend establishing early-succession blocks as the foundation for a whole-farm conservation management strategy in the MAV that initially targets grassland bird conservation.

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