EVALUATION OF PATCH-BURN GRAZING ON SPECIES RICHNESS

AND DENSITY OF GRASSLAND BIRDS

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by

DAVID J. STROPPEL

Dr. Mark R. Ryan, Thesis Co-supervisor

Mike Hubbard, Thesis Co-supervisor

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The undersigned, appointed by the dean of the Graduate School, have examined the thesis entitled

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AND DENSITY OF GRASSLAND BIRDS

presented by David J. Stroppel,

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and hereby certify that, in their opinion, it is worthy of acceptance.

Professor Mark R. Ryan

Michael W. Hubbard

Professor Frank R. Thompson III

Professor Rose-Marie Muzika

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Evaluation of patch-burn grazing on species richness and density of grassland birds

David J. Stroppel

When European settlers arrived in North America they found large, unfragmented, tallgrass prairies. This historic presettlement prairie ecosystem in Missouri has been described as a tallgrass plant community that was subject to frequent fire and grazing with few trees and shrubs (Kurz 2003, Nelson 2005).

Mechanized agriculture converted much of the prairie into crop fields, fragmenting the prairie, reducing the prevalence of fires and shrinking the tallgrass prairie to 5% of its presettlement range (Sampson and Knopf 1994). Today, only about 36,000 hectares (90,000 acres) of Missouri's original 6 million hectares (15 million acres) of prairie still exist (Kurz 2003). Along with the loss of the tallgrass prairie, grassland bird populations have undergone dramatic declines in spite of management efforts. The North American Breeding Bird Survey data show that 70% of the 29 bird species characteristic of North American prairies declined between 1966 and 1993. These grassland bird species are declining faster than any other guild of terrestrial birds on this continent (Knopf 1994). This suggests that techniques currently used to mange rangelands may be insufficient to maintain biological diversity (Holecheck et al. 1998). To reverse this decline, the remaining tracts of tallgrass prairie must be managed in a way to provide the diverse habitat needs of the avian community.

The presence of grazing during the evolution of prairie ecosystems helped promote biodiversity. This suggests that biodiversity could be enhanced on today's grasslands by mimicking the temporal and spatial grazing patterns of presettlement prairies (Fuhlendorf and Engle 2001).

Heterogeneity when defined as variation in vegetation structure, composition, density and biomass, influences species diversity, habitat variety and ecosystem function (Christensen 1997, Wiens 1997, Bailey et al 1998). Many species depend on the interspersion of diverse habitat types scattered throughout a heterogeneous landscape (Fuhlendorf and Engle 2001). Heterogeneity leads to biodiversity and should serve as the framework for ecosystem management (Christensen 1997, Ostfeld 1997, Wiens 1997).

The variation in habitat requirements of grassland birds shows us that heterogeneity is important. The structure of grassland avian communities is influenced by the structural heterogeneity of the plant communities (Wiens 1974). Certain bird species require specific structural characteristics in grasslands (Cody 1985). The variation in habitat requirements of coexisting grassland bird species supports the necessity of heterogeneous grasslands for maintaining diverse avian communities (Fuhlendorf and Engle 2001). For example, Greater Prairie Chickens (*Tympanuchus cupido*) need short, sparsely vegetated ground for courtship displays, sites with residual vegetation for nesting, and, in general, little woody vegetation (Johnson et al. 2004). Upland Sandpipers (*Bartramia longicauda*) use short vegetation for foraging, short to medium height vegetation for brood rearing, and taller vegetation for nesting (Johnson et al. 2004). Grasshopper Sparrows (*Ammodramus savannarum*) prefer short to intermediate height, clumped grasslands interspersed with patches of bare ground for foraging and for nesting they require moderate amounts of litter (Bent 1968, Blankespoor 1980, Vickery 1996). The Eastern Meadowlark (*Sturnella magna*) prefers intermediate height grasslands with a moderate amount of forbs and little woody vegetation. The meadowlark forages for insects on the ground or in the soil and nests on the ground within short grasslands that are moderately to heavily grazed (Skinner et al 1984). These species, and others, prefer different habitats for different stages of their life cycle, and often they need these habitats in close proximity to one another. To achieve this mix of habitats, juxtaposed in such a way as to benefit several different species; a mix of management techniques is needed that mimic the interaction of historic processes that shaped prairies.

The eastern tallgrass prairie community evolved under the interactive effects of climate, topography, soil types and conditions, fire, and grazing (Axelrod 1985). The combined effects resulted in diverse plant communities of grasses and forbs in different successional stages scattered across the landscape. Today, large contiguous tracts of native prairie are rare, and managers are trying to replicate the interaction of fire and grazing on relatively small, isolated patches of native and restored prairie. These smaller prairies may function differently than larger prairies of presettlement times, but the historical processes of fire and grazing are still necessary to maintain functioning prairie systems.

One promising prairie management technique is patch-burn grazing (PBG). PBG mixes annual burning with summer grazing to increase vegetation heterogeneity. This technique may help create the mix of plant communities similar to what occurred historically. PBG created diverse plant structure scattered across the landscape (Fuhlendorf and Engle 2001) and has potential for enhancing grassland bird populations.

The benefits of PBG to grassland bird populations versus burning alone have not been tested in Missouri. It has been investigated in other states with promising results. Our overall objective was to conduct a confirmatory analysis of the effects of PBG on species richness and density of grassland birds. The diversity in vegetative structure created by PBG, should lead to greater species richness, and higher densities for some species (e.g. Grasshopper Sparrow). We expect estimated density will be greater in the control unit for the DICK and HESP and greater in the grazing unit for EAME and GRSP. We will use a student's t-test to check for significant differences in mean estimated density.

It is the goal of this study to determine where avian species richness and density is higher relative to the treatment effects from grazing, years since burning and the interaction of grazing and years since burning. This information could prove valuable to managers who wish to provide habitat conditions for a large suite of grassland birds but are concerned about making the lands they manage unsuitable to the species that are currently present or to species of conservation concern.

Study Areas

The study areas were Taberville, Wah'Kon-Tah, Niawathe and Bethel Prairies. All areas are owned by MDC, The Nature Conservancy or both and were located in either the Osage Plains or Springfield Plateau Region in southwest Missouri. Both the Osage Plains and the Springfield Plateau have soils formed from eroding rock and may have bedrock near the surface (Kurz et al. 2003).

Taberville Prairie located in St. Clair County, is 651 hectares (1,608 acres). Taberville is an Ozark border prairie with silt loam soils from shale and sandstone (Kurz et al. 2003). Taberville was hayed and grazed on a three year rotation prior to the introduction of a PBG system. The surrounding land use was 70% cropland and 30% grassland with 25% of the grassland used as pasture while the other 5% is hayed (Gilmore, Pers, Comm.).

Wah’Kon-Tah prairie, in St. Clair and Cedar Counties, at 1,157 hectares (2,858 acres) is the largest, easternmost tallgrass prairie in North America (Kurz et al. 2003). It also is an upland Ozark border prairie, but unlike Taberville, Wah’Kon-Tah has cherty silt loam soils from cherty limestone. Wah’Kon-Tah was under a hay, burn, and rest rotation prior to being patch burned grazed. Surrounding land use was 95% grassland with 90% in pasture and 5% hayed, the remaining 5% is cropland (Gilmore, Pers, Comm.).

The last two sites are smaller, with the entire conservation area being in the study. At 130 hectares (320 acres), Niawathe prairie in Dade County, is an Ozark border dry mesic upland prairie, with silt loam soils formed from shale and sandstone (Kurz et al. 2003). Prior to the application of PBG, about 30 hectares (80 acres) of Niawathe was hayed annually on rotation. Additionally, Niawathe was burned on a five-year rotation. Both the control and grazing units were under the same management. The surrounding landscape was 50% fescue (*Festuca sp)*, 25% native prairie, and 25% cropland (Hedges, Pers, Comm.).

Bethel Prairie, a small (105 hectare [260 acre]) tract in Barton County is an upland prairie over shallow sandstone (Kurz et al. 2003). Until the late 1990’s Bethel was grazed continuously and rarely burned. The area surrounding Bethel was 15% cropland, 60% native grassland and 25% fescue and Caucasian bluestem (*Bothriochloa ischaemum*). All neighboring grasslands are hayed and continuously grazed. These sites were selected for PBG because they were all native grasslands in similar landscapes making comparisons between sites possible.

Methods

Field Methods

In this study, PBG is a management strategy which was applied by dividing a prairie into three patches one of which was burned each spring. Patch 1 was burned in 2005, patch 2 in 2006 and patch 3 in 2007 (Map 1). Cattle are introduced shortly following the burn (mid-April) and are allowed to graze until mid-August. The only fence is a perimeter fence to contain the cattle, allowing them to graze freely across all three patches of the grazing unit.

To obtain information on species richness and density, transects were laid out in each patch of each unit in a stratified random sampling pattern (Map 2). Each patch typically had two transects and each unit had six. In March 2006 an arson fire burned much of the PBG unit on Wah'Kon-Tah. Surveys were conducted in the portions of Wah'Kon-Tah that were burned on the same rotation as the other study areas. No surveys were conducted in the arson burn area.

We used a distance sampling method where bird observations were recorded from transects. Distance sampling is based on determining a detection function. We used line transect sampling and program DISTANCE (version 5.0; Thomas et al. 2005) to compute the probability of detection (p) for the avian species we detected in this study (Buckland et al. 2001). The detection function compensated for the fact that detectability decreases with increasing distance from the observer (Rosenstock et al. 2002). Distance sampling also allowed us to model detection probability by observer, study site, treatment and other habitat characteristics.

We placed the first transect within a patch a random distance between 50 m and 90 m from the perimeter of the patch. Additional transects were placed at least 100 m from the previous transect and all transects ended at least 50 m from the perimeter of the patch.

Distance sampling requires estimates or measures of the distance to each bird perpendicular to the transect (x). Three assumptions (Buckland et al. 2001) critical to acquiring accurate detection function estimates are: (1) birds on the line are always detected; (2) birds are detected before any movement triggered by the observer and (3) distances are estimated or measured accurately. To meet assumptions one and two, trained observers traveled slowly or stopped occasionally along the line to listen and observe the line ahead of them (Rosenstock et al. 2002) and focused on identifying birds on the line before they were disturbed. We used Bushnell Yardage Pro 800 laser rangefinders to measure perpendicular distance to eliminate distance estimation errors.

Counts were conducted in treatment and control units on all four study sites. On three of the four prairies the control unit was immediately adjacent to the grazing unit. At Taberville the control and treatment units were separated by 325 m (Map 1). Each count was conducted between 30 minutes before sunrise and 1000 hours. Transects were walked when the sky was clear, no precipitation or fog was present and winds were less than 16 kph (<10 mph). On the two larger prairies treatment and controls units were approximately 90 ha (220 acres) each. On Bethel and Niawathe prairies (the two smaller sites), treatment and control units were approximately 53 ha (130 acres) and 65 ha (160 acres) respectively.

A team of four observers sampled two prairies each day. Two observers sampled all transects on one area while the other observers sampled a different prairie. Transects were rotated so that the first transect and the starting point of each transect was different each day.

We conducted 12 surveys of each patch on each site in 2006 and in 2007. Two sites were sampled simultaneously each morning with one observer in the control unit while another observer surveyed the treatment unit. To obtain a precise density estimate with a coefficient of variation of less than 0.15 we needed a sample size (*n*) between 60-100 for each species (Rosenstock et al. 2002, Buckland et al. 2001).

Cover board readings were taken from each site in June. The percent of visual obstruction from 0 – 10 cm and maximum vegetation height in centimeters was recorded at dozens of plots within each patch of each unit. The number of plots varied depending on the size and shape of each patch with a range of 55 – 110 vegetation plots per patch.

The expected relationships between avian density, grazing and years since burning for the species covered in this paper are listed in Appendix A. For both Dickcissel (DICK) and Henslow's Sparrow (HESP) we expected a negative response to grazing and a positive response to years since burning. That is, we expected estimated density of these two species to increase as the number of years since burning increased or to decrease with grazing. For Eastern Meadowlark (EAME) and Grasshopper Sparrow (GRSP) we expected a positive relationship to grazing and for GRSP, a negative response to increasing time since burning. EAMEs prefer mid-height grasslands so we predicted they would vary in their response to fire. Mean estimated densities will be pooled across years, averaged, and then a student's t-test will be used to check for significant differences between means.

We chose to analyze data for DICK and HESP for three reasons. One, they were the two largest datasets respectively, two: both species are associated with dense stands of tallgrass prairies and three; HESP are in decline and DICK are a species of management concern in Missouri (MDC 2009). We expected both species to show preference for the taller, thicker grasslands found in the control units and have lower density estimates in the grazed units and even lower numbers in the grazed and burned patches.

To get a picture of how PBG affects other species, we chose to also evaluate EAME and GRSP densities. These species are associated with short to medium height grasslands and their sample sizes were adequate for evaluating patch level abundances. The EAME is common in Missouri but the GRSP is designated a species of management concern (MDC 2009).

Model Selection and Statistical Analysis

Models for this paper were ranked using an information theoretic approach. We chose to use Akaike's Information Criterion (AIC) to select the best supported model from the given candidate models (Burnham and Anderson 2002). We used AICc (AIC adjusted for small sample size) to rank the candidate models (Buckland et al. 2001). Models were ranked using ΔAICc which is the difference between the best supported model and all other models in the model set. Models that were within two AICc units of the best supported model were considered well supported (Burnham and Anderson 2002).

We estimated detection probability and avian abundance with program DISTANCE. Detection function models were determined for each species by evaluating non-covariate and covariate models. The non-covariate model group included; uniform, hazard rate and half-normal key functions with simple polynomial, hermite polynomial and cosine series expansion combinations. For the covariate analysis only half-normal or hazard rate functions were used as the uniform function does not allow for the evaluation of covariates and the exponential key function has an implausible shape near zero distance (Thomas et al. 2006).

The hazard rate and half-normal models are suggested for Multiple Covariate Distance Sampling (MCDS) by Buckland et al. (1993). MCDS analysis allowed us to determine which variable or combinations of variables produced a better supported detection function. To reduce variance in the density and abundance estimation calculations, we right truncated the data 5% (Buckland et al. 1993).

The first set of covariate models included study site, observer, treatment, sample year and years since burning. The second set of covariate models incorporates vegetation data from cover board readings including mean cover at ground level from 0 – 10 cm (Mean CS 1), mean cover at maximum height (Mean Max) and coefficients of variation from each of these strata, denoted CV CS 1 and CV Max respectively.

All well supported models from the first three groups were combined to form the mixed model group. This allowed us to evaluate if the covariates improved the fit of the detection function beyond the basic non-covariate models. We used AICc values to rank the models in the candidate set and the best supported model from this group was then used to estimate density. The mixed model group is listed on Table 1.Total number of detections for each species, detection probabilities with standard error (SE), and density with SE are listed in Appendix B.

Statistical Analysis

To determine PBG effects on avian species richness and density we referenced the bird survey results to the patch or unit they were located in to show their response to burning, grazing or the combination of burning and grazing. Species richness was the number of avian species observed in each patch. To test for differences in species richness we performed an

ANOVA in SAS using the GLM procedure (SAS Institute 2000). We looked for differences in species richness and abundance by study site, treatment, years since burning, and treatment plus years since burning.

Visual analysis of the data indicated a problem in the 2007 observations with more detections at zero distance than expected. Buckland et al. (2001) report that if >10% of detections are at zero distance this could result in unreliable distance measurements. The data were then left truncated at 0.9 m because of these excess detections. We compared the data with and without left truncation and found it did not change the density estimates but left truncation of the data did reduce the confidence intervals and resulted in lower coefficients of variation. Even though the 2006 data did not have an issue with greater than expected observations at zero distance we truncated both the 2006 and 2007 datasets.

To evaluate density at the patch level, estimated density was evaluated by years since burning with Year 0 denoting the patches that were burned the spring immediately preceding the survey, and Year 1 or 2 being one or two years of growth since they were last burned.

Results

Species richness analysis

Species richness was greater in PBG units ($\bar{x}$ = 20.2 +/- SD 2.77) than in control units ($\bar{x}$ = 17.5 +/- SD 3.09; p = 0.02) (Table 2). Fourteen species occurred uniquely on PBG units (Table 3). The Upland Sandpiper, Horned Lark, Greater Prairie Chicken, Scissor-Tailed Flycatcher and Eastern Bluebird occurred frequently on PBG sites, but only rarely (<3 observations) on control units (Table 4). Seven species occurred uniquely or frequently on control units, but only rarely (< 3 observations) on PBG units (Table 5). Of these, none were true grassland species, none were state endangered and one, the Eastern Wood-Pewee, is a species of management concern.

The Greater Prairie Chicken occurred frequently in the Year 0 and Year 2 grazing patches and once in the Year 0 control patch. Prairie chickens were never detected in the other control patches. The Upland Sandpiper and Horned Lark were detected in all three grazing patches but were never found in the control units. The Scissor-Tailed Flycatcher occurred frequently in the Year 0 and Year 2 grazing patches and once in the Year 1 control patch but was never found in the other control patches.

The ANOVA analysis showed significant differences in species richness between study sites ($P = 0.03$), no significant difference between years since burning ($P = 0.46$) and no significant difference with the years since burning plus treatment interaction ($P = 0.49$; Table 6). The difference in species richness between study sites was expected, all four study sites were native prairies but they varied in size, shape, latitude and terrestrial natural community.

Detection function model selection

For both DICK and EAME the uniform simple polynomial detection function was the best supported (Table 1) with the lowest AICc score as well as having a $w_i = 1.00$. For the GRSP the non-covariate hazard rate key function with a simple polynomial series expansion was the best supported. It too had the lowest AICc score and a $w_i = 1.00$. The observer covariate model was the best fit for the HESP with the lowest AICc score and a $w_i = 1.00$. This means that a large portion of the variance in the abundance estimate can be explained by variables other than distance, in this case the observer (Buckland et al. 2004). This could represent differences in the observer’s vision, hearing or experience identifying HESPs or it could be related to the bird's behavior, coloration or song which makes it a challenge for the observer to detect.

Density

DICK and HESP estimated densities were higher in the control units while EAMEs and GRSPs had higher estimated densities in the grazing units (Fig 1). Mean estimated density for the DICK in the grazing units was $\bar{x}$ = 1.03 birds/ha (+/- SD 0.466) and $\bar{x}$ = 1.48 birds/ha (+/- SD 0.687) in the control units (two-tailed t-test: p = 0.071). HESP mean estimated density was $\bar{x}$ = 0.22 birds/ha (+/- SD 0.176) and $\bar{x}$ = 0.30 birds/ha (+/- SD 0.116) for the grazing and control units respectively (p = 0.1924).

For the EAME grazing unit mean estimated density was $\bar{x}$ = 0.19 birds/ha (+/- SD 0.081) whereas control unit estimated density was only $\bar{x}$ = 0.07 birds/ha (+/- SD 0.045; p = 0.0002). The GRSP was also much more prevalent in the grazing units with an estimated density of $\bar{x}$ = 0.12 birds/ha (+/- SD 0.077) and only $\bar{x}$ = 0.01 birds/ha (+/- SD 0.010) in the control units (p = 0.0001).

When densities were analyzed by years since burning (control patches, Fig. 2a-d), DICK mean estimated density (Fig. 2a) was lowest in the Year 2 patches with $\bar{x}$ = 0.30 birds/ha (+/- SD 0.241), the lowest estimated density observed for this species in any patch, and highest in Year 1 with $\bar{x}$ = 1.74 birds/ha (+/- SD 0.988). A two-tailed t-test revealed no significant difference in estimated density between the Year 0 ($\bar{x}$ = 1.39 +/- 0.743 SD) and Year 1 (p = 0.4399) patches. There were significant differences in DICK estimated density between the Year 0 and Year 2 (p = 0.0014) patches and between the Year 1 and Year 2 (p = 0.0013) patches.

HESP estimated density (Fig. 2b) was lowest in the Year 0 patches with $\bar{x}$ = 0.09 birds/ha (+/- SD 0.074) and highest in the Year 1 patches with $\bar{x}$ = 0.49 birds/ha (+/- SD 0.190; p = 0.0001). HESP densities in Year 2 were intermediate ($\bar{x}$ = 0.33 +/- SD 0.269; Year 0 vs. Year 2: p = 0.0249; Year 1 vs. Year 2: p = 0.2047).

The lowest estimated densities for EAMEs (Fig. 2c) occurred in the Year 0 patches with $\bar{x}$ = 0.07 birds/ha (+/- SD 0.050), the lowest estimated density observed for this species in any patch, and the highest estimated densities occurred in the Year 2 patches with $\bar{x}$ = 0.11 birds/ha (+/- SD 0.088; p = 0.2183; Year 0 vs. Year 1; p = .7504; Year 1 vs. Year 2; p = 0.4913).

The greatest GRSP estimated density (Fig. 2d) occurred in Year 1 at $\bar{x}$ = 0.03 birds/ha (+/-SD 0.056) and lowest in Year 2, the lowest estimated density for this species in any patch, with $\bar{x}$ = 0.02 birds/ha (+/- SD 0.014; p = 0.3512: Year 0 vs. Year 1; p = .6818: Year 0 vs. Year 2; p = 0.4484).

When densities were analyzed for the grazing patches by years since burning, the highest DICK estimated density (Fig. 3a) was in the Year 1 patches ($\bar{x}$ = 1.68 birds/ha +/- SD 0.971; p = 0.9043) followed by Year 0 ($\bar{x}$ = 0.80 birds/ha +/-SD 0.651; p = 0.1133) and the lowest estimated density was found in the Year 2 patches ($\bar{x}$ = 0.72 birds/ha +/- SD 0.270; p = 0.0052).

In the grazing patches, HESP estimated density (Fig. 3b) was lowest in the Year 0 patches ($\bar{x}$ = 0.07 birds/ha +/- SD 0.111; p = 0.7615,), the lowest estimated density we observed for this species in any patch, higher in Year 1 ($\bar{x}$ = 0.23 birds/ha +/-SD 0.254; p = 0.0373) and highest in Year 2 ($\bar{x}$ = 0.31 birds/ha +/- SD 0.138; p = 0.8539).

EAME estimated density (Fig. 3c) in the grazing patches was lowest in Year 0 ($\bar{x}$ = 0.12 birds/ha +/- SD 0.91; p = 0.2131) and equal in Year 1 ($\bar{x}$ = 0.24 birds/ha +/- SD 0.152; p = 0.0341) and Year 2 ($\bar{x}$ = 0.24 birds/ha +/- SD 0.108; p = 0.0303), respectively.

GRSP estimated density (Fig. 3d) in the grazing patches was highest in Year 0 ($\bar{x}$ = 0.21 birds/ha +/- SD 0.160; p = 0.0062), lower in Year 1 ($\bar{x}$ = 0.12 birds/ha +/- SD 0.114; p = 0.0843) and lowest in Year 2 ($\bar{x}$ = 0.06 birds/ha +/- SD 0.042; p = 0.0153).

Discussion

Species Richness

Patch-burned grazed units show increased species richness and increased EAME and GRSP estimated density, the latter identified as a species of management concern in Missouri (MDC 2009). This supports other researchers (Fuhlendorf et al. 2006, Coppedge et al. 2008) who concluded that PBG provides habitat for a wider suite of grassland birds than burning alone.

In this two-year study, only one observation of a Greater Prairie Chicken and one of a Scissor-Tailed Flycatcher were recorded in a control unit. These are important observations as the prairie chicken is a state endangered species and the Scissor-Tailed Flycatcher is a species of management concern in Missouri (MDC 2009). The Greater Prairie Chicken has been shown to be area sensitive and was absent on prairie fragments less than 77 ha (Winter and Faaborg 1998) in southwest Missouri. It appears that to enhance grassland habitats for these species, a combination of grazing and fire provides the range of habitat conditions they utilize during the breeding season. To create the diversity of habitats needed by a wide variety of species, grasslands should include areas that are burned, unburned, grazed, ungrazed, burned and grazed, as well as rested (unburned and ungrazed).

Species richness was not significantly related to years since burning or years since burning plus treatment. This was expected as it's the heterogeneity across the PBG unit that would provide the habitat diversity needed to support a wider suite of grassland birds, not the habitat created within a given patch (burn year) or patch treatment.

Density

EAME and GRSP estimated densities were significantly greater in the PBG units. This result is consistent with the literature for these two species as they prefer short to intermediate height grasslands, which were abundant in the grazing units (Roseberry and Klimstra 1970, Wiens 1973, Vickery 1996). This response was expected given both species' preference for grazed grasslands (Walk et al. 2000, Ryan et al. 2003, Skinner et al. 1984) and is also consistent with others (Powell 2006) who found grazing to have a positive effect on GRSP abundance.

No significant difference in estimated density was detected between grazed and ungrazed units for the DICK or HESP. Given both species reported preference for taller, dense grasslands with deep litter and forbs (Wiens 1973, Skinner et al. 1984, Winter 1998, Winter 1999) it was expected that their densities would be greater in the control units which are characterized by taller, denser grasslands with high litter. However, many studies have reported that the DICK (Overmire 1963, Wiens 1973, Skinner et al. 1984, Bock et al. 1993) and HESP (Skinner et al. 1984, Swengel 1996) use light to moderately grazed grasslands.

Because the patch-burn grazed units were moderately grazed, the grasslands could recover quickly from fire and were therefore very attractive to both species.

DICK estimated density related to years since burning was significantly lower in Year 2. We detected no difference in Year 0 or Year 1 estimated densities, with a peak in estimated density occurring in Year 1. Zimmerman (1992) also reported that spring burning (Year 0) did not affect DICK abundance in Kansas, and Powell (2006) reported DICK abundance in burned sites to be highest in Year 1 and then declined over the next two years. Other studies have shown DICK abundance to be unrelated to years since burning (Zimmerman 1993, Swengel 1996, Winter 1999). It appears that the vigorous growth of the grasslands following spring burning resulted in suitable habitat within the year of burning and for the following year, but this continued growth resulted in less desirable habitat conditions in Year 2, significantly reducing estimated density.

In the grazing patches, DICK estimated density was significantly greater than the control patches in Year 2 and peaked in Year 1. Powell (2006) reported similar results in grazed pastures with DICK estimated abundance highest in Year 1, and declining in Year 2. This would suggest that the addition of grazing animals extends the impact of the prescribed fire for an additional year creating habitat capable of supporting densities in Year 2 greater than that of burning alone.

HESP density was strongly influenced by fire in this study, with estimated density related to years since burning lowest in Year 0, highest in Year 1, and down in Year 2, all three responses were significant. These results are consistent with other studies that show this species preferring taller grasslands with high litter cover (Wiens 1969, Skinner et al. 1984) and with studies (Herkert 1994, Powell 2006) that found HESP abundance to be lowest in recently burned habitats. It is likely that HESP density was lowest in Year 0 due to the lack of litter, lack of standing dead vegetation and/or lack of forbs for song perches.

The peak in HESP estimated density in Year 1 and the decrease in Year 2 differs from other studies that found HESP abundance increased over time, peaking two (Herkert 1994) and three years following burning (Powell 2006), but is consistent with Powell's (2006) findings in Kansas where HESP abundance was greater in the control patches for both Year 1 and Year 2. This decrease in Year 2 could be attributed to the aggressive growth of the grasslands following spring burning resulting in less desirable habitat than in the Year 1 patches.

In the grazing patches, HESP estimated density was lower than the control patches for Year 0, Year 1 and for Year 2. Estimated densities were significantly greater only for Year 1. This is consistent with Powell's (2006) findings in Kansas where HESP abundance was greater in the control patches for both Year 1 and Year 2. Powell (2006) also reported grazing eliminated or nearly eliminated HESP from grazed sites, but this study does not show elimination or near elimination of HESPs from the PBG units.

EAME estimated density was not related to years since burning. Estimated density was lowest in Year 0 and peaked in Year 2 control patches. In the grazing patches, estimated density was greater in all three patches with a peak in Year 1 and Year 2 estimated density equal to Year 1. This is consistent with Powell (2006) who found EAME abundance lowest in recently burned sites (Year 0) and highest in Year 2. Results show significantly greater estimated density in the PBG patches in Year 1 and Year 2. This was expected as the meadowlark usually responds positively to grazing (Bock et al. 1993).

GRSP estimated density was not related to years since burning. Estimated density was very low in all three years with the lowest density in Year 2 and highest in Year 1. In the grazing patches estimated density was greater in all three patches with a peak in Year 0. Estimated densities were significantly greater in Year 0 and Year 2. Powell (2006) reported estimated abundance to be lowest in Year 0, highest in Year 1 with Year 2 being nearly equal to Year 1 for both grazed and ungrazed sites. The results of this study differ from Powell as we detected the greatest density for GRSPs in Year 0 grazing patches and then a decline in each of the following years. This was expected given the GRSP's preference for short, grazed grasslands (Wiens 1973, Ryan et al. 2003, Vickery 1996). It appears that PBG is a viable management technique to boost densities of this and other species associated with short grasslands.

Research and Management Recommendations

PBG increases avian species richness and density of some species during the breeding season when applied to native tallgrass prairies. Further research is needed to assess its effects on avian species richness and density on warm season grass restorations. Winter habitat use of patch-burn grazed prairies by resident species also should be evaluated.

As with any management technique, some species will benefit from PBG and some will not. Spring burning results is grass-dominated prairies that lack the plant diversity needed to support a wider suite of grassland birds, while PBG provides short, heavily grazed grasslands as well as taller less disturbed areas adjacent to each other. This heterogeneity in the prairie landscape is capable of supporting a wider range of bird species than burning alone. This combination of habitats supplies what is needed to maintain acceptable densities of species associated with tall grasslands while increasing densities of uncommon species that rely on shorter grasslands.

Managers who wish to boost densities of GRSPs, Upland Sandpipers, Greater Prairie Chickens, and other short-grass associated species should consider the application of PBG to the prairies they manage but must be aware of the species that already occur on these sites and the potential impact it might have on them. For example, species like the DICK and HESP, which are associated with tall, dense grasslands, may have lower densities on PBG prairies. Attention should also be paid to the size of the grazing units to ensure they are large enough to support populations of area-sensitive species such as the Greater Prairie Chicken who did not occur on Missouri prairies smaller than 77 ha (Winter and Faaborg 1988).

Budget limitations pressure managers to impact as many hectares as possible for the purpose of reporting accomplishments to maximize federal aid reimbursements. The goal of burning, grazing or managing prairies can be impacted by the bottom line. We suggest the goal of prescribed burning and grazing be focused on enhancing biodiversity in the plant and bird communities of the prairie ecosystem rather than on accomplishment reporting.

Table 1. Mixed model group for each species from Program Distance which includes the best supported non-covariate detection function, the best covariate model from Group 1 and the best supported models from the Habitat covariate group.

DICK Mixed Models

Name	ΔAICc	AICc	LogL	K	w_i
Uniform Simple Polynomial	0.00	65028.30	-32509.15	0	1.00
CV CS 1	77.58	65105.88	-32549.94	1	0.00
Study Site	91.28	65119.58	-32554.79	3	0.00
					1.00

HESP Mixed Models

Name	ΔAICc	AICc	LogL	K	w_i
Observer	0.00	10522.92	-5252.39	7	1.00
Hazard Rate Simple Polynomial	34.20	10557.12	-5275.55	0	0.00
CV CS 1	49.94	10572.86	-5283.42	1	0.00
CV Max	51.94	10574.86	-5284.42	1	0.00
Mean Max	51.94	10574.86	-5284.42	1	0.00
Mean CS 1	51.94	10574.86	-5284.42	1	0.00
					1.00

EAME Mixed Models

Name	ΔAICc	AICc	LogL	K	w_i
Uniform Simple Polynomial	0.00	10269.74	-5129.85	0	1.00
Study Site	11.96	10281.70	-5136.83	3	0.00
Observer	12.52	10282.26	-5133.07	7	0.00
Mean Max	16.04	10285.78	-5140.88	1	0.00
					1.00

GRSP Mixed Models

Name	ΔAICc	AICc	LogL	K	w_i
Hazard Rate Simple Polynomial	0.00	4627.16	-2310.56	0	1.00
Observer	18.58	4645.75	-2313.71	7	0.00
Sample Year	18.84	4646.00	-2319.98	1	0.00
CV CS 1	22.92	4650.09	-2322.02	1	0.00
Mean CS 1	23.55	4650.72	-2322.34	1	0.00
CV Max	23.58	4650.75	-2322.35	1	0.00
Mean Max	23.61	4650.77	-2322.36	1	0.00
					1.00

Table 2. Species richness analysis results by treatment.

Treatment		Mean	Std Dev
	Control	17.5	3.09
	PBG	20.2	2.77

Table 3. Species that occurred only in grazing units.

American Crow
American Robin
Black Capped chickadee
Blue Grosbeak
Eastern Bluebird
Great Blue Heron
Green Heron
Horned Lark
Killdeer
Northern Flicker
Orchard Oriole
Turkey Vulture
Upland Sandpiper
Wood Thrush

Table 4. Species that occurred frequently in the grazing units but occurred less than three times in the control unit.

Upland Sandpiper
Horned Lark
Greater Prairie Chicken
Scissor-Tailed Flycatcher
Eastern Bluebird

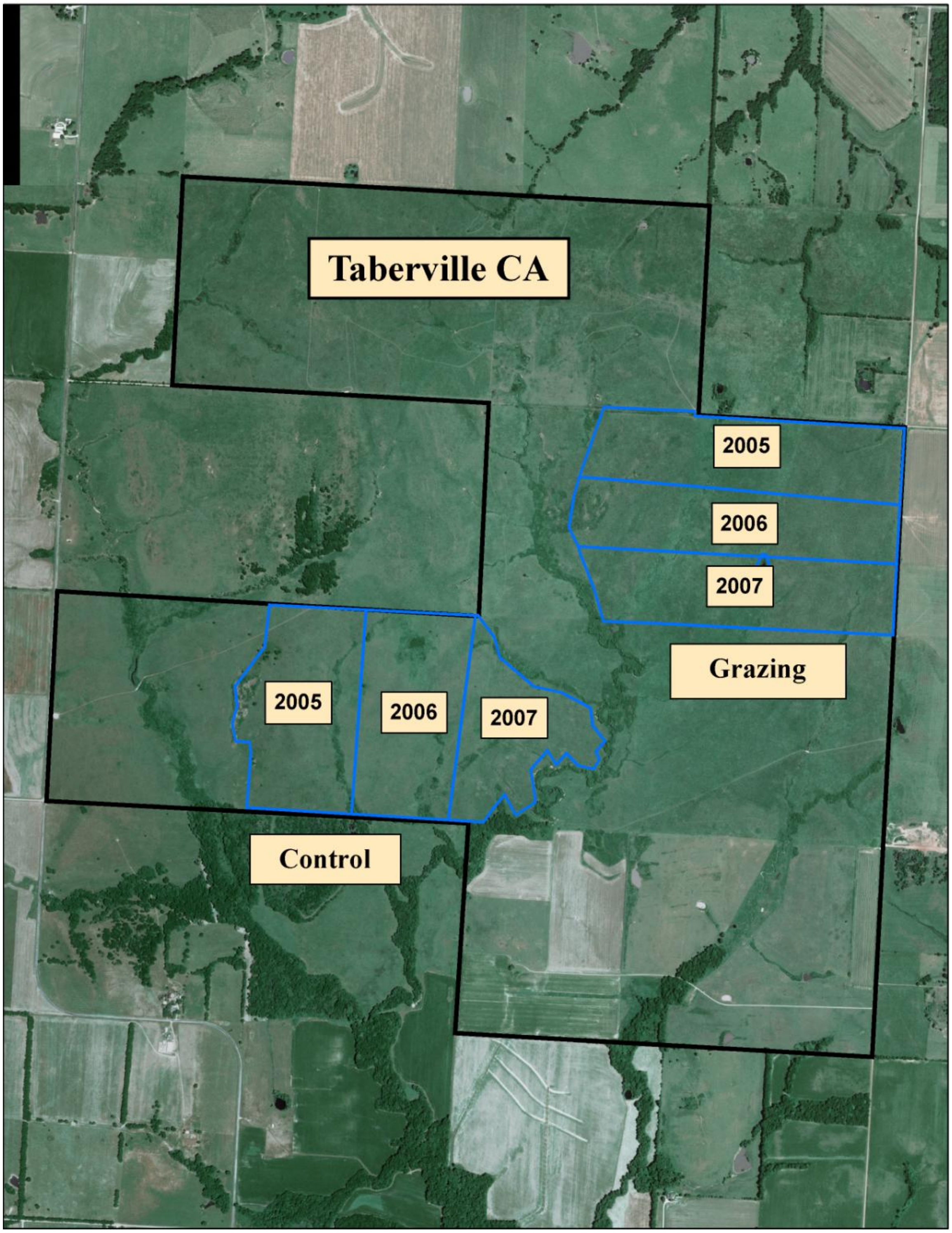
Table 5. Species that occurred only in the control units.

Baltimore Oriole
Carolina Wren
Cedar Waxwing
Downy Woodpecker
Eastern Tufted Titmouse
Eastern Peewee
Red Bellied Woodpecker

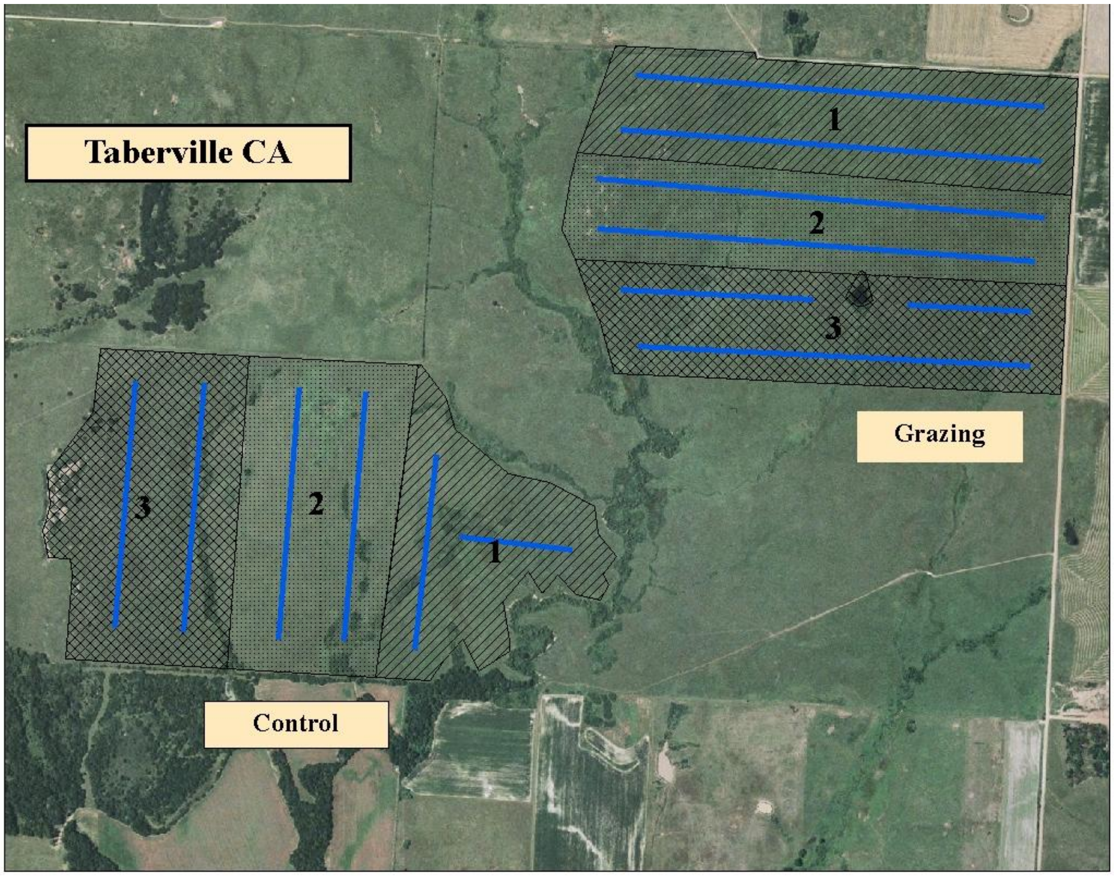
Table 6. Species richness analysis results by variable.

Source	DF	Type 1 SS	Mean Square	F Value	Pr > F
Study Site	3	72.365	24.122	3.74	0.03
Treatment	1	44.010	44.010	6.82	0.02
Years Since Burning	2	10.688	5.344	0.83	0.46
Years Since Burning + Treatment	2	9.521	4.760	0.74	0.49

Map 1. Spatial location of treatment units and patch burn year at Taberville CA.



Map 2. Transect locations, patch number and units at Taberville CA.



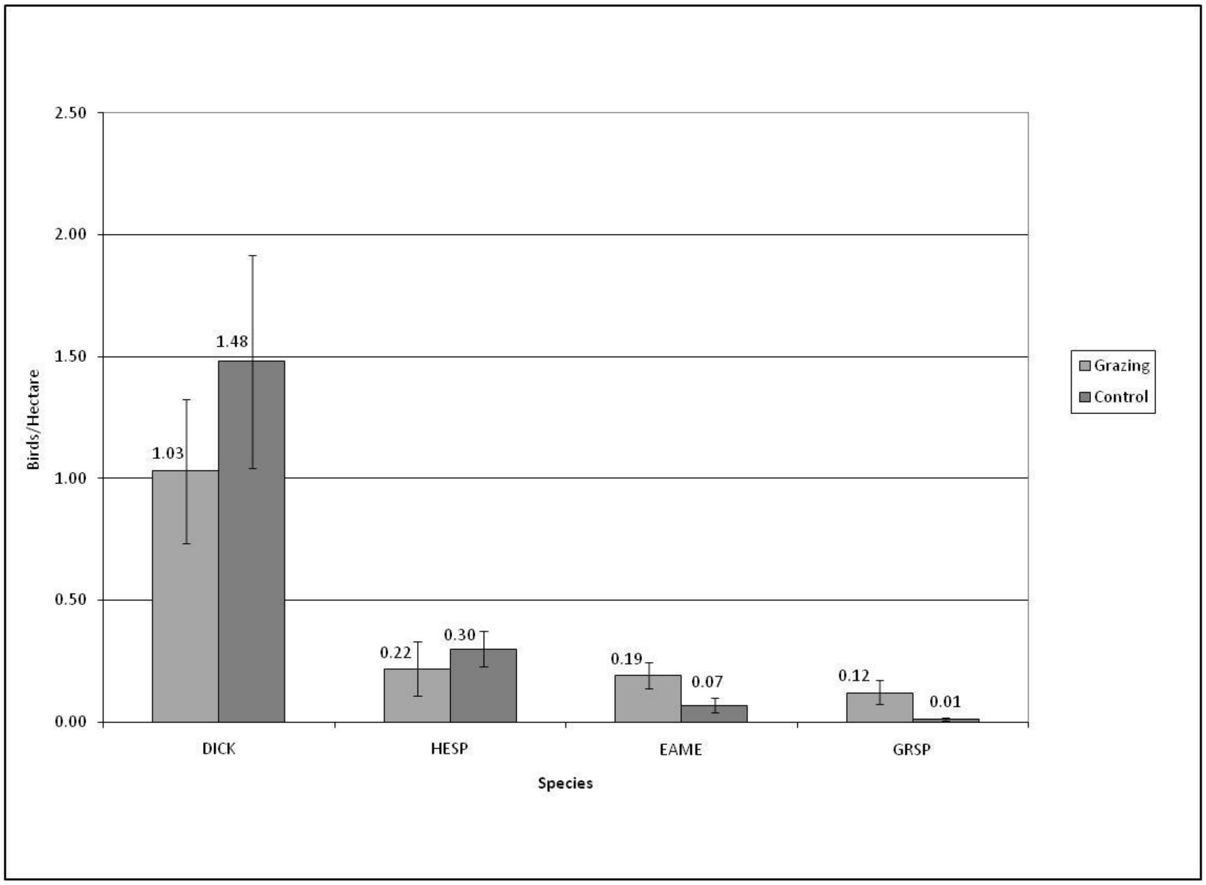


Figure 1. Estimated density for each species by treatment.

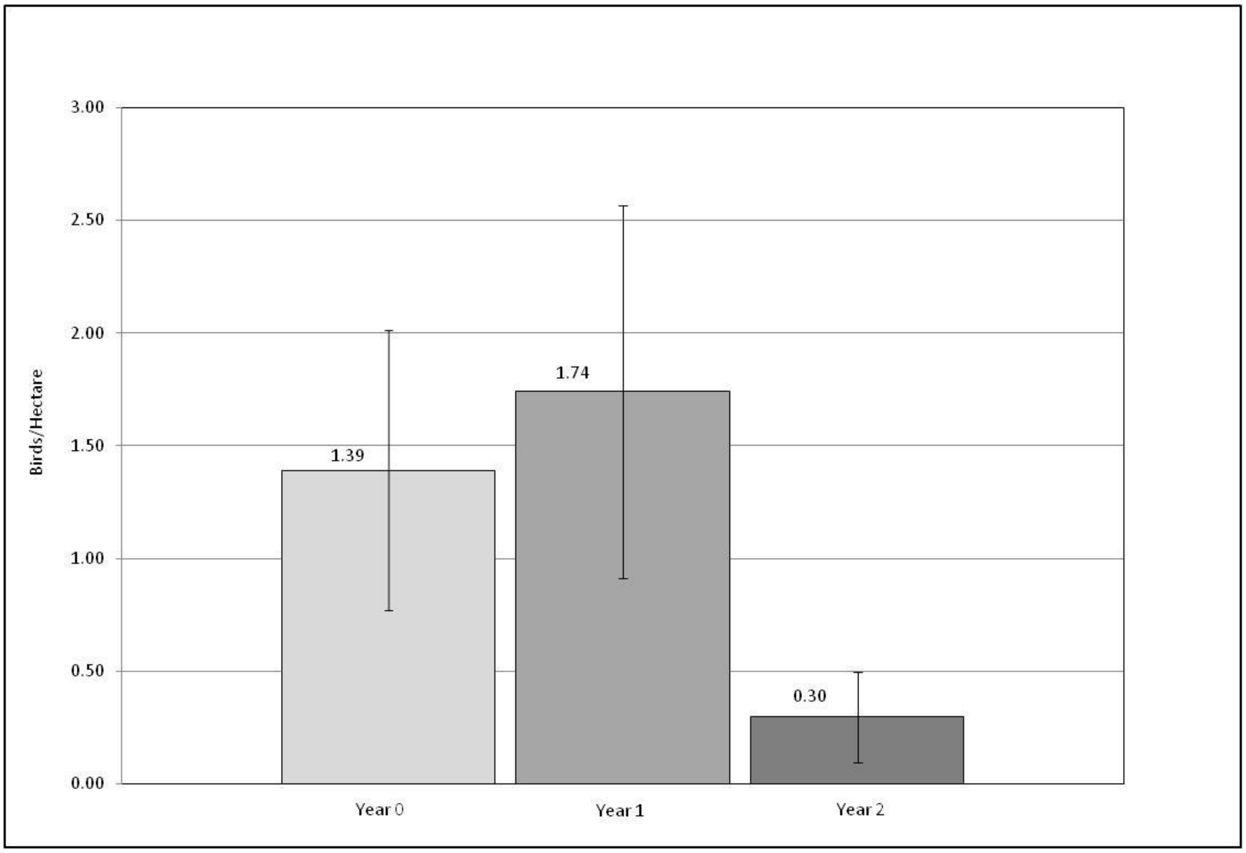


Figure 2a. Dickcissel estimated density by years since burning with 95% confidence intervals.

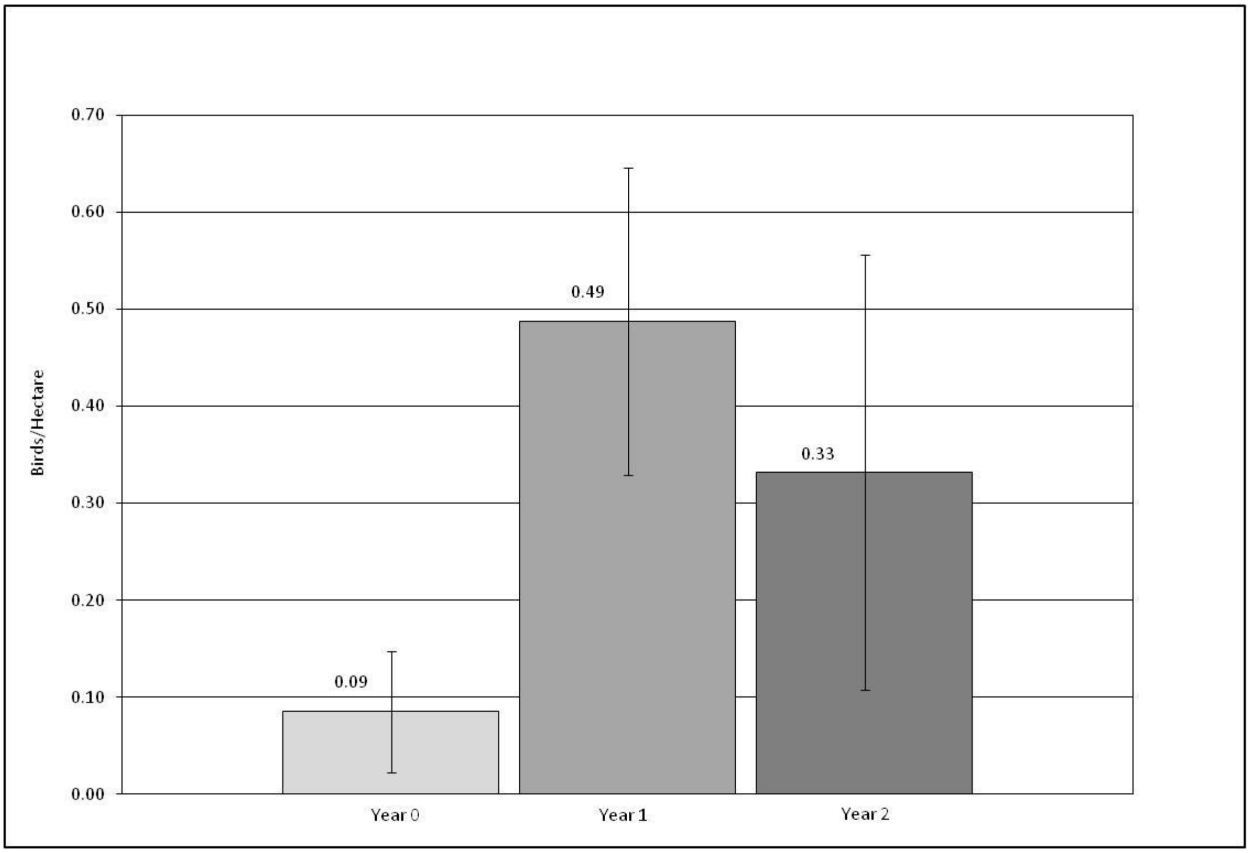


Figure 2b. Henslow's Sparrow estimated density by years since burning with 95% confidence intervals.

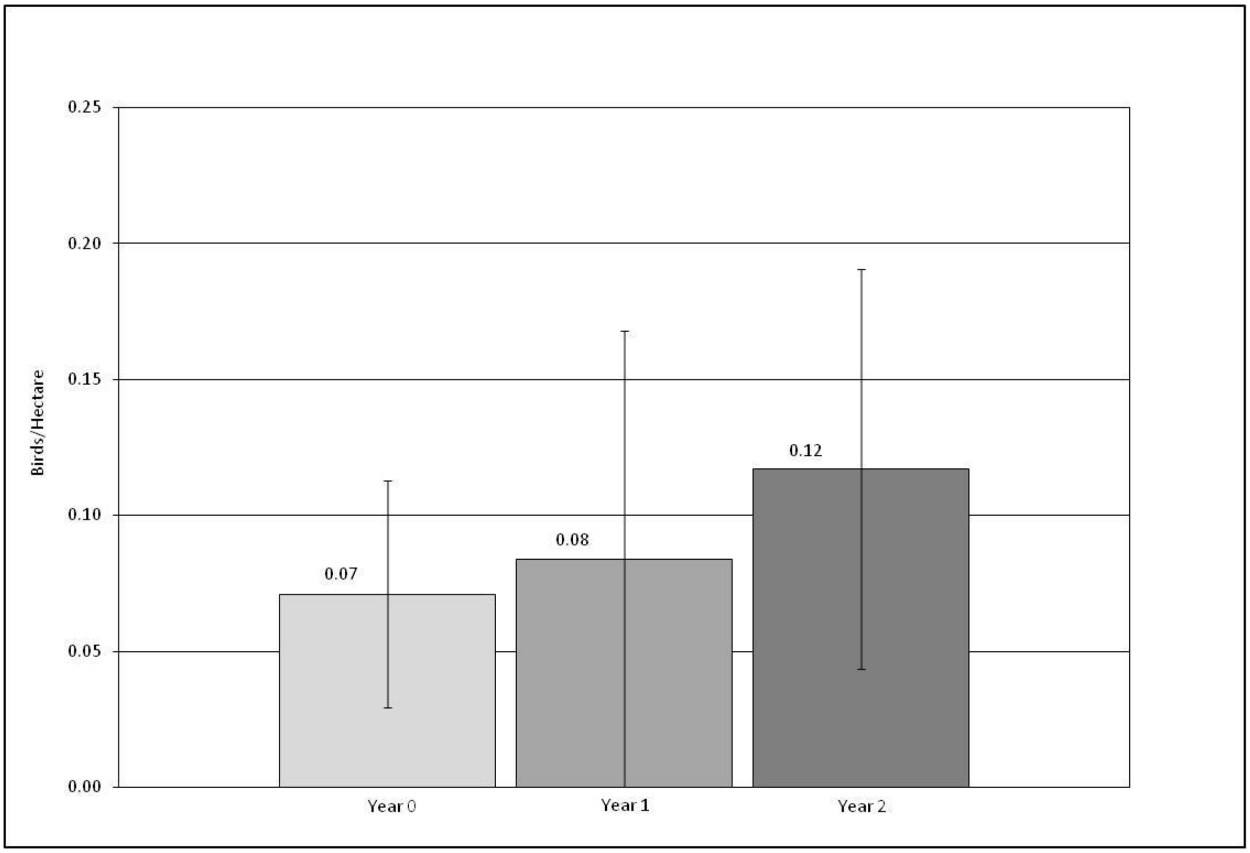


Figure 2c. Eastern Meadowlark estimated density by years since burning with 95% confidence intervals.

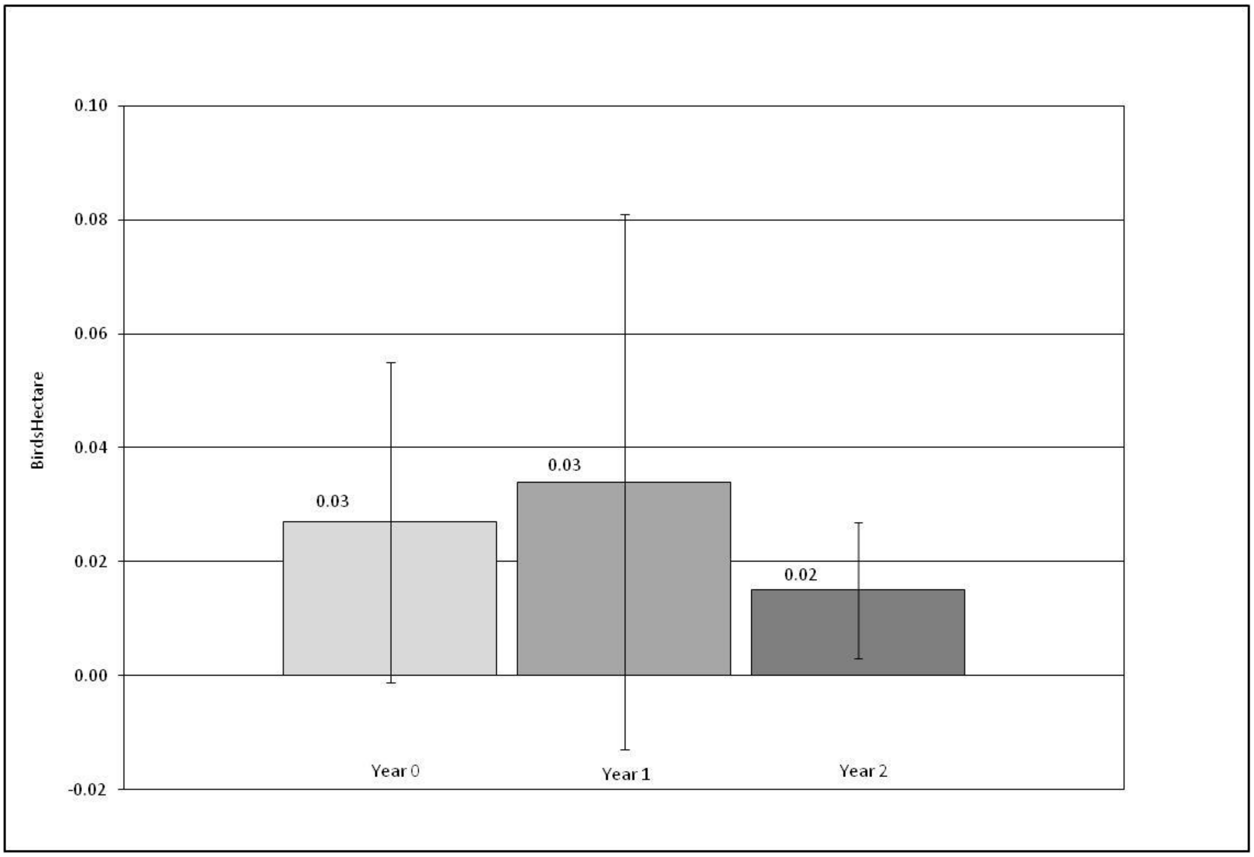


Figure 2d. Grasshopper Sparrow estimated density by years since burning with 95% confidence intervals.

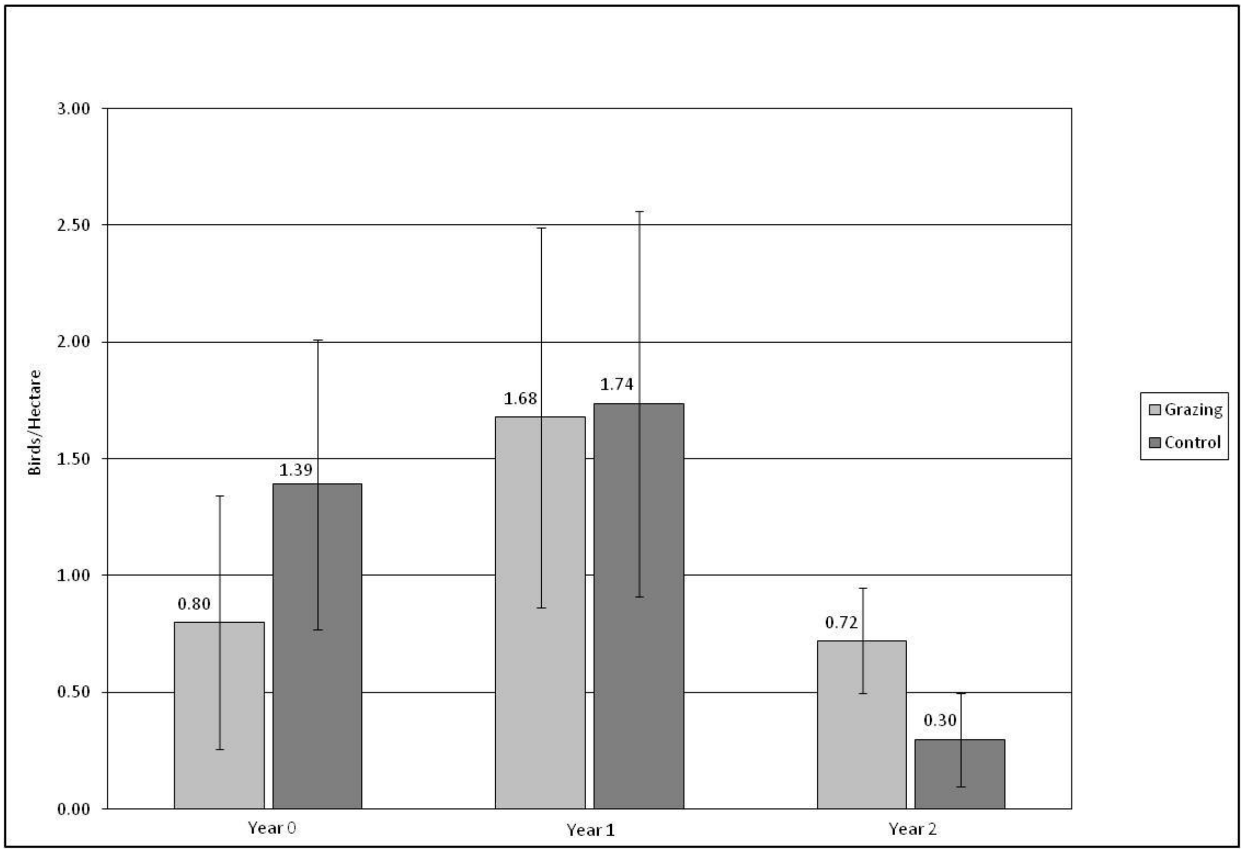


Figure 3a. Dickcissel estimated density by years since burning plus treatment with 95% confidence intervals.

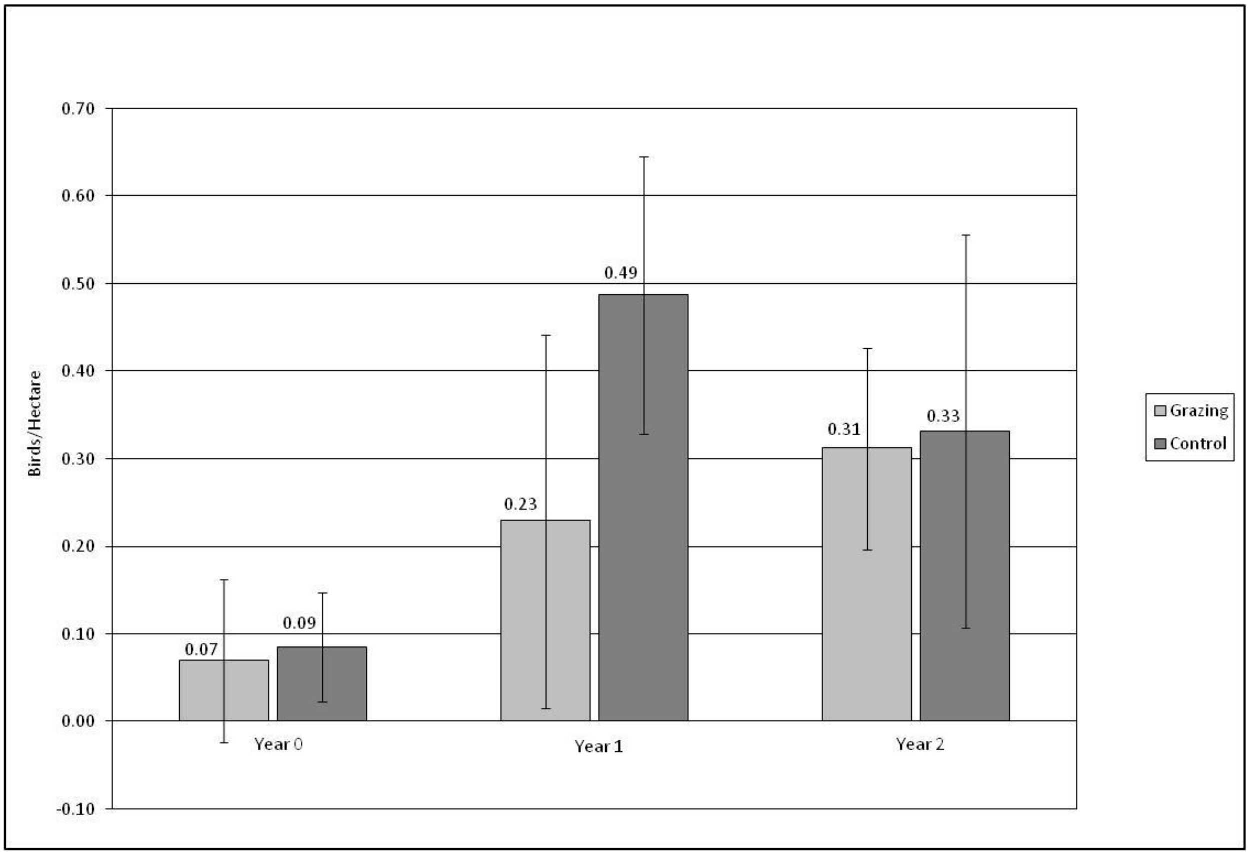


Figure 3b. Henslow's Sparrow estimated density by years since burning plus treatment with 95% confidence intervals.

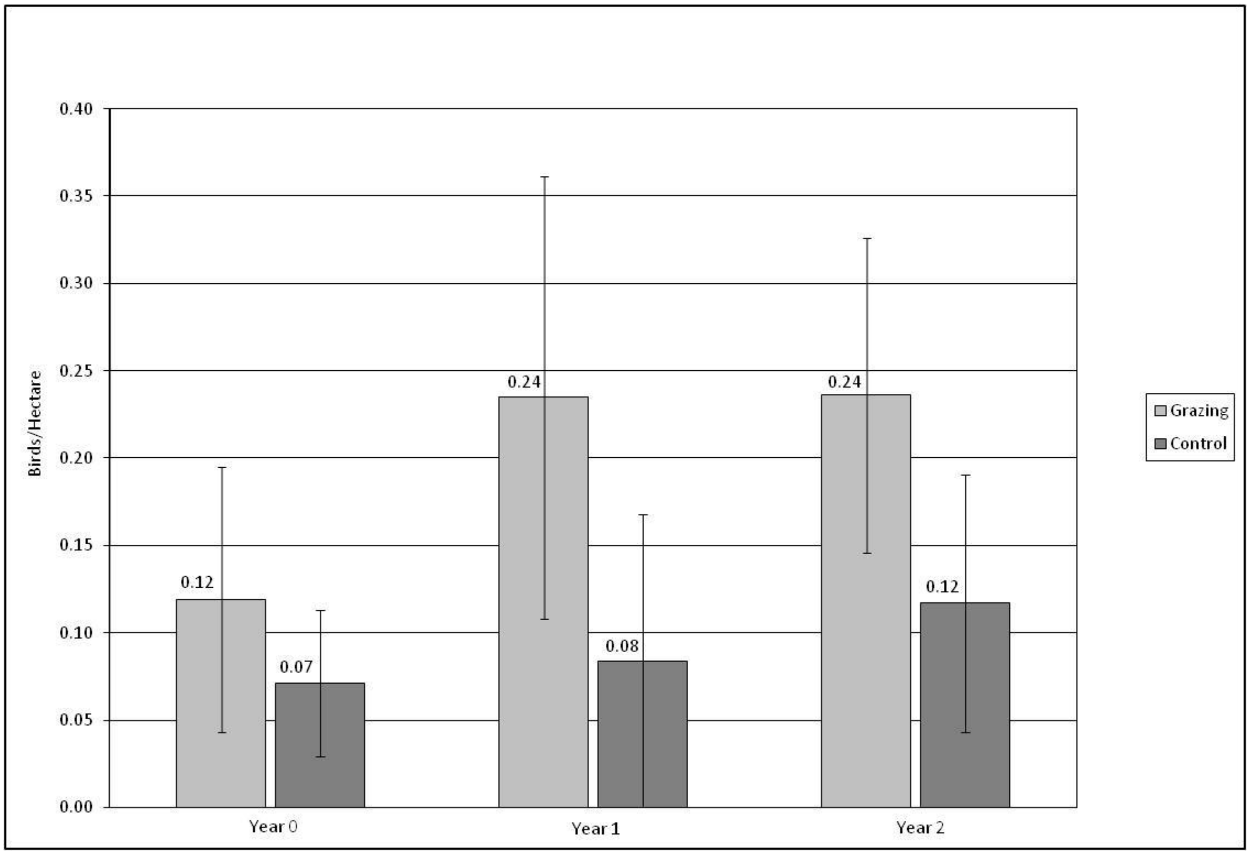


Figure 3c. Eastern Meadowlark estimated density by years since burning plus treatment with 95% confidence intervals.

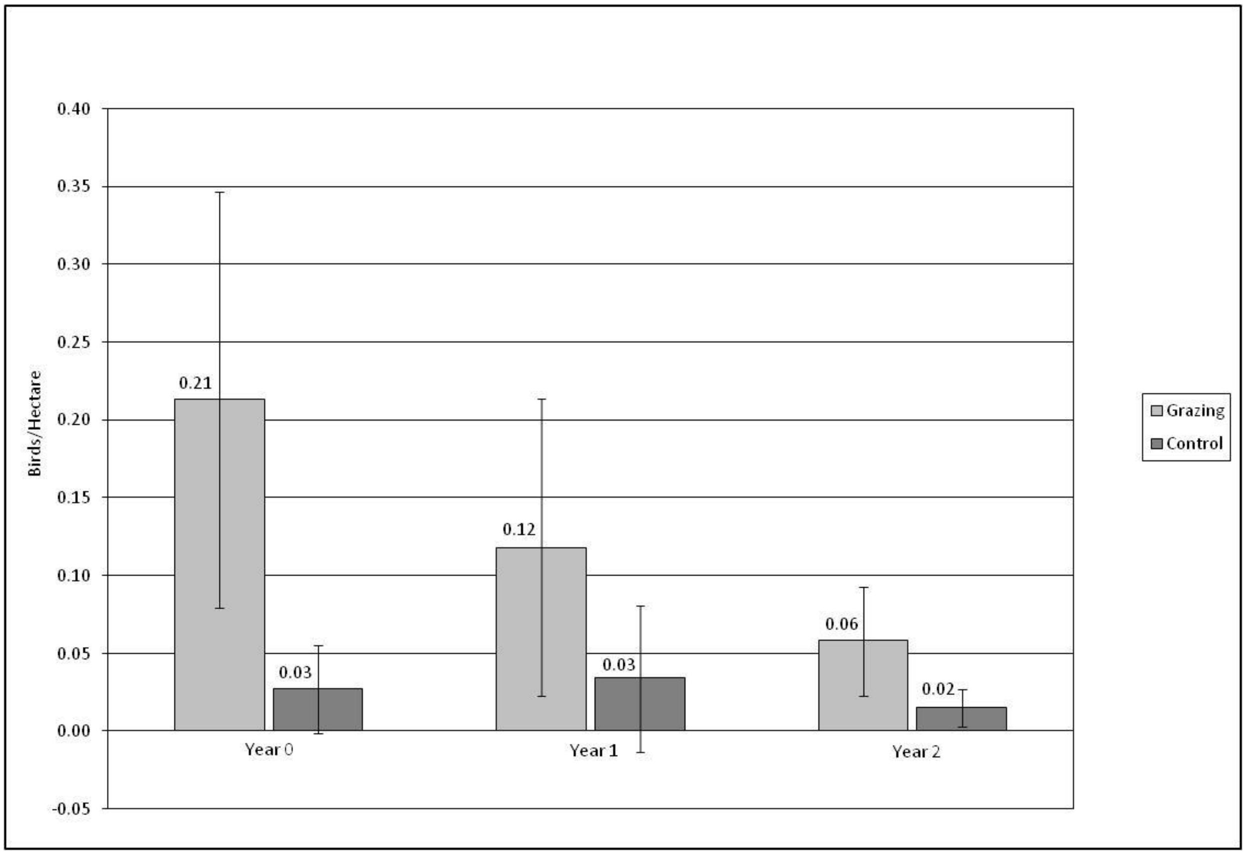


Figure 3d. Grasshopper Sparrow estimated density by years since burning plus treatment with 95% confidence intervals.

Appendix A. Description and representation of a priori models based on treatment effects on density of grassland birds in southwest Missouri.

Treatment Models

Model	Model Structure	Expected Result **DICK**	Expected Result **HESP**	Expected Result **EAME**	Expected Result **GRSP**
Yrs Since Burn	$B_o + B_1$(Burn Year)	+	+	+/-	-
Grazing	$B_o + B_1$(G vs. C)	–	–	+	+
Years Since Burning and Grazing	$B_o + B_1$(Burn Year) + B_2 (G vs. C)	+/-	+/-	+/-	+/-

Appendix B. Total number of detections for each species, detection probabilities with SE, and estimated densities (birds/ha) with SE at the treatment unit scale.

Species	**Total detections Treatment**	**Total detections Control**	**Detection probability w/ SE – Treatment**	**Detection probability w/ SE - Control**
DICK	**3789**	**4084**	**.93 (.039)**	**1.00 (.049)**
HESP	**675**	**651**	**.67 (.073)**	**.87 (.064)**
EAME	**909**	**262**	**.87 (.069)**	**1.1 (.152)**
GRSP	**513**	**53**	**1.12 (.111)**	**1.97(.874)**

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