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FINAL PERFORMANCE REPORT

FEDERAL AID GRANT NO. T-30-P-1

PATCH BURN MANAGEMENT: ENHANCING HABITAT FOR IMPERILED GRASSLAND BIRD SPECIES

OKLAHOMA DEPARTMENT OF WILDLIFE CONSERVATION

September 1, 2005 through August 31, 2008

FINAL REPORT

STATE: OKLAHOMA PROJECT NUMBER: T-30-P-1

GRANT PROGRAM: State Wildlife Grants

GRANT NAME: Patch Burn Management: Enhancing Habitat for Imperiled Grassland Bird Species

GRANT PERIOD: 1 September 2005 through 31 August 2008

PRINCIPAL INVESTIGATOR: Samuel D. Fuhlendorf

I. ABSTRACT

This report summarizes three years of avian, macroinvertebrate, and vegetation sampling on Cooper Wildlife Management Area (WMA) in Woodward, Oklahoma conducted from 2006 through 2008. We detected 55 species of birds during the three summer sampling periods. Over the three year period, we monitored 333 nests of 24 species and sampled nest vegetation for 229 of those nests. Avian diversity peaked at three years post-burn and then subsequently declined which suggests a need for frequent disturbance to prevent senescence in vegetation. Invertebrate biomass, as well as other invertebrate metrics such as diversity and abundances, for Hemiptera, Homoptera, Lepidoptera, Diptera, and Hymenoptera was higher in 2007 than in 2006 or 2008. Araneae, Coleoptera, and Orthoptera were most abundant in 2006. Lark sparrow (*Chondestes grammacus*) densities were highest in the most recently disturbed patches ($\leq$ 1 year post-burn), while Cassin's sparrow (*Aimophila cassinii*) densities were highest in patches that were 2–3 years post-burn and declined rapidly afterwards. However, field sparrow (*Spizella pusilla*) densities increased as the time since burn increased, with densities being the highest in the unburned patches. Patch-burn management altered the structure of

vegetation. Vegetation heights and percent cover were typically reduced immediately following a patch-burn, but they returned to levels comparable to the control pastures within 3–5 years. In some instances, patch-burned vegetation had cover values higher than cover values in control pastures after 3–5 years post-burn. Sand sagebrush (*Artemisia filifolium*) density did not decline following the application of patch-burn management, suggesting that sand sagebrush is not harmed by prescribed burning as applied in this study. Our results suggest that patch-burn management in sand sagebrush ecosystems can be beneficial to avian and invertebrate communities. Patch-burn management provides a diversity of habitats not provided by traditional rangeland management practices for a wide variety of avian species and invertebrates. Further, our results demonstrate that patch-burn management provides a useful tool to alter the vegetation structure, at large scales, of sand sage prairie. Using patch-burn management, managers may be able to provide suitable habitat for wildlife species that require habitat conditions that differ from that which is found in sand sage prairie that is not treated with patch-burning.

II. OBJECTIVES

1. Determine effects of patch fires on vegetation composition and other habitat parameters, including the ability of the vegetation to recover from the patch disturbance.
2. Determine responses of Tier I and Tier II grassland birds in patches that vary in time since fire. We expect that some species will be dominant on the most

recently disturbed patches while others will be dominant on the patches that have not been disturbed for several years.

III. NEED

Historically grasslands were heterogeneous as a result of fire and bison (*Bison bison*) grazing. This historical disturbance regime has been replaced with practices that decrease the inherent patchiness of grasslands. Current range management techniques have not benefited grassland birds as their populations continue to decline even though rangeland quality continues to improve (Holechek et al. 1998). As a result, patch-burn management has been proposed as a strategy to mimic the historical disturbance pattern in these systems that was created by the interaction between fire and bison grazing (Fuhlendorf and Engle 2001, 2004). Patch-burning seeks to restore the historical disturbance patterns by creating structural diversity through grazing and fire interactions. This particular management technique restores heterogeneity by creating areas that have been recently burned, others that vary by time since burn, and areas that have not been burned (Fuhlendorf and Engle 2001, 2004). Additionally, the introduction of grazers such as cattle or bison add an additional layer of complexity as grazers tend to focus on the most recently burned areas and less on the others (Vermeire et al. 2004, Wallace and Crosthwaite 2005). By burning one-third of a pasture each year on a rotational basis and introducing cattle at moderate stocking rates, patchiness is generated, which throughout time, shifts across the landscape creating a mosaic of habitat conditions (Fuhlendorf et al. 2006). Patch-burn sites

had four times greater avian diversity than grazed sites demonstrating that these patch-burned habitats can serve a wider variety of grassland birds (Harrell 2004). Therefore, heterogeneity can positively benefit grassland avifauna.

Grazing and Fire Effects on Grassland Birds.—Most often, the effects of grazing on grassland birds are quantified by the changes in vegetation structure (e.g., litter, forb, and visual obstruction) (Bock and Webb 1984, Fondell and Ball 2004, Atkinson et al. 2005, Davis 2005, Sutter and Ritchison 2005). Grazing directly affects the habitat available to different species. Species such as horned larks (*Eremophila alpestris*) prefer short, sparse grass (Beason 1995), while others such as Henslow's sparrows (*Ammodramus henslowii*) prefer unburned, decadent grass (Zimmerman 1997). Additionally, the use of herbicides to promote grass for cattle grazing also may influence grassland birds (Fuhlendorf and Engle 2004) by decreasing forb availability and therefore, invertebrate abundance. Plants classified as forbs are associated with higher invertebrate populations compared to grass, bare ground, and shrubs (Hill 1985, Jamison et al. 2002), and lesser prairie-chickens (*Tympanuchus pallidicinctus*) are positively associated with invertebrate biomass, suggesting hens will choose habitats based on invertebrate abundance (Jamison et al. 2002). In addition, Sutter and Ritchison (2005) determined that grazing decreased nest success and clutch size of grasshopper sparrows (*Ammodramus savannarum*), which they related to lower insect-prey availability in the grazed areas. These results suggest any decrease in forb cover will likely be a detriment to grassland birds.

In addition to affecting vegetation structure, grazing may also influence nest predation rates. Authors have suggested that management practices are as equally important as landscape context in affecting nest predation rates (Davison and Bollinger 2000, Shochat et al. 2005). In tallgrass prairie in Oklahoma, Shochat et al. (2005) determined nest predation in burned and grazed grasslands to be higher than grazed or undisturbed grasslands separately. Working in the same study area as Shochat et al. (2005), Churchwell (2005) determined that 52-60% of nest failures were the result of predation, and nest success varied depending on the species and whether the site had been burned. For example, dickcissels (*Spiza americana*) had higher nest success on unburned sites, while grasshopper sparrows had higher nest success on burned sites.

Grazing may also increase nest parasitism rates by increasing foraging opportunities for brown-headed cowbirds (*Molothrus ate*; Goguen and Mathews 1999). The cattle themselves may harbor important arthropod-prey such as ticks (Acarina) and flies (Diptera), and cattle also flush invertebrates as they move through the habitat, possibly increasing prey availability for cowbirds. The act of grazing itself may increase certain species of insects (particularly Formicidae and Scarabaeidae) which are important insect-prey for several species of birds (Hutchinson and King 1980). However, working in the Kansas Flint Hills, Jensen and Cully (2005) determined nest parasitism rates were best predicted by cowbird abundance and were not associated with local landscape variables including vegetation type, distance to edge, grassland bird density, and a host of other variables. This suggests the factors which determine local cowbird abundances

may not be the same across geographical ranges, and management strategies such as patch-burning may have contradictory results in different habitats.

Similar to grazing, fire effects on grassland birds are usually quantified by determining changes in vegetation structure available to birds (Madden et al. 1999, Smucker et al. 2005). The influences of fire have varying impacts on the grassland avifauna depending on the species. Certain species such as common yellowthroat (*Geothlypis trichas*) and Henslow's sparrows have lowered abundances after a recent fire, while others such as upland sandpipers (*Bartramia longicauda*) and red-winged blackbirds (*Agelaius phoeniceus*) may have higher abundances. Still, others such as dickcissels and eastern meadowlarks (*Sturnella magna*) showed little to no response to fire in tallgrass prairies in the Flint Hills (Zimmerman 1997). In North Dakota mixed-grass prairie, many grassland bird species showed a decrease immediately after fire but rebounded within two years (Johnson 1997). Likewise, Henslow's sparrows in Louisiana returned to pre-burn abundances within a year after burning (Bechtoldt and Stouffer 2005). However, habitat requirements for individual speices may change through the year (e.g., lesser prairie-chicken nesting vs. brooding) and each habitat requirement may be impacted differently by fire (Schroeder and Robb 1993, Boyd and Bidwell 2001). Therefore, the heterogeneity created by patch-burning should benefit many of these species compared to the homogenous habitat created by traditional range management which may benefit only a few.

Applied simultaneously, fire and grazing can positively influence grassland bird populations. Heterogeneity created by the interactions between

management, time, and space provide a diversity of habitats to benefit grassland birds with varying life history requirements (Harrison et al. 2003, Fuhlendorf and Engle 2004, Churchwell 2005). However, authors have suggested cowbird parasitism may increase on burned and grazed grasslands (Danley et al. 2004). Working in tallgrass prairie in Oklahoma, Churchwell (2005) noted brood parasitism was minimal (<3%) on patch-burned sites. As mentioned before, this suggests the factors that determine local cowbird abundances may not be the same across geographical ranges and management strategies such as patch-burning may have contradictory results in different habitats.

Role of Heterogeneity in Grassland Invertebrate Ecology.—Female grassland birds require a large proportion of protein in their diet during the egg laying and nestling-rearing periods (Wiens and Rotenberry 1979). Females acquire their protein through increased intake of invertebrates (Moreby 2003). Insects have more than four times the protein as compared to plants and contain essential amino acids not present in plant protein. Insect protein is also more easily assimilated compared to plant protein (Potts 1986, Moreby 2003). Little information is known about the diet composition of many grassland songbirds, although it has been reported that the nestling diet of western meadowlark may include Orthoptera, Lepidoptera, Diptera, Hymenoptera, Coleoptera, Hemiptera, and Arachnida (Orians and Horn 1969, Maher 1979, Kobal et al. 1998). Much more information is known about the diet of other grassland birds such as northern bobwhite (*Colinus virginianus*), which may select Lepidoptera, Hemiptera, Coleoptera, Homoptera, Araneae, Orthoptera, Hymenoptera, and

Diptera (Handley 1931, Landers and Mueller 1986, Jackson et al. 1987). Because little is known about the food habits of many grassland birds, it is difficult to assess management impacts on these birds. Therefore, a more fine-scaled examination of their feeding ecology is required to best determine how patch-burning may influence invertebrate selection and foraging behavior.

Role of Fire in Sand-sage Community.—Sand sagebrush (*Artemisia filifolium*) is a short stature shrub that is distributed throughout the western Great Plains (Stubbendieck et al. 1979). Throughout the Great Plains, sand sagebrush canopy cover can range from 20 to 50% (Collins et al. 1987). Studies have shown that at high densities and dominance sand sagebrush can negatively impact herbaceous plant communities as well as reduce forage production and overall habitat suitability for a variety of wildlife species. For example, lesser prairie chicken densities were correlated with low to moderate cover of sand sagebrush (Cannon and Knopf 1981) and avian diversity and abundance were highest in areas with moderate cover (Rodgers and Sexson 1990). Management of sand sagebrush requires maintaining moderately dense stands (Vermeire 2002). Because sand sagebrush-mixed prairie ecosystems have evolved under the influence of fire and grazing interactions (Wright and Bailey 1982), patch-burn management may be an appropriate management strategy for maintaining a diversity of habitats for wildlife. However, our knowledge of effects of fire-grazing interactions on sand sagebrush, especially at patch-level scales, is limited.

IV. APPROACH

Experimental Design.—The study site is the Hal and Fern Cooper Wildlife Management Area (Cooper WMA) in Woodward County, Oklahoma (Fig. 1). Vegetation of the study site is considered a sand sagebrush grassland with the dominant life-form being sand sagebrush (Berg 1994, Collins et al. 1987, Gillen and Sims 2004). The herbaceous vegetation in this community consists of a variety of annual and perennial forbs and tall, mid-height, and short perennial grasses, including both sod-forming and bunch-forming grasses (Berg 1994, Collins et al. 1987, Gillen and Sims 2004).

At Cooper WMA, five pastures were used for sampling. For sampling purposes, each study pasture was divided into three patches (15 patches total). Two pastures (North Pasture and South Pasture) were control pastures where cattle grazing during the growing season occurred on an annual basis but no patch-burning occurred. Three pastures (Middle Pasture, East Pasture and Bodwell Pasture) were categorized as treatment pastures where cattle grazing during the growing season occurred on an annual basis, and where patch-burning had occurred in the past and during the study. Cattle grazing consisted of steers stocked from 1 April to 15 September at 6.88 ha/AU (17 acres/AU; 4.04 ha/steer, 10 acres/steer). In treatment pastures, patch boundaries corresponded with previously-existing firebreaks. Figure 1 illustrates the study pastures at Cooper WMA. Table 1 lists the pastures used in the study, the size of all patches in each pasture and the year each patch in the treatment pastures was burned.

Vegetation Sampling.—Four permanent transects were randomly located in each patch of each pasture (60 transects total) in areas characterized by soils in the Pratt-Tivoli complex. Soils in this complex are loamy–fine sands characterized by steep dunes and valleys between the dunes (Nance et al. 1963). Slopes range from 1–12% in the Pratt series and 8–20% in the Tivoli series. Soils in the Pratt-Tivoli complex are the most prevalent soil type in Woodward County, representing approximately 12.2% of all soils (Nance et al. 1963). Each permanent transect was 100 m long, marked at each end by a steel t-post located 10 m from the end of the transect (120 m between each t-post).

In 2006, 2007 and 2008, vegetation data were collected along the permanent transects in May and June. At 10-m intervals along each transect, vegetation height and visual obstruction were measured using a visual obstruction pole modified from Robel et al. (1970). Percent bare ground, percent cover of litter, percent cover of vegetation structural groups (live and dead vegetation, grasses, forbs, shrubs and trees) were also estimated to the nearest 5% within 0.10-m^2 rectangular (20 x 50 cm) plots placed at 10-m intervals along each transect.

Sagebrush plants were counted within 10.0-m^2 plots (1.0 x 10.0 m), oriented parallel to each transect and any plants with multiple stems arising from the ground surface were considered a single plant if no stem was >20 cm from another stem at the ground surface. Stems >20 cm from another stem at the ground surface, and were not known to be connected below the ground surface, were considered separate plants. At each 10-m interval of a transect, the

sagebrush plant nearest to the 10-m interval was selected for sampling and its height was measured.

Avian Sampling. —We determined abundance and diversity estimates of grassland birds using distance sampling on point transects during the summers of 2006, 2007, and 2008. We established point transects 300 m apart and >150 m from fences and roads. Each patch contained four point transects (12 point transects per pasture; 60 point transects on Cooper WMA) that we sampled three times (mid-May, mid-June, and mid-July). Groups of two observers recorded all bird species identified by sight and sound for 10 min and recorded the distances to each bird using laser range finders (Ransom and Pinchak 2003). At each point, all birds seen and heard were counted and the distance to the bird was measured using a laser rangefinder. Weather variables including temperature and wind speed were taken at the onset of each sampling event with a handheld anemometer, and percent cloud cover was visually estimated. We conducted counts on days with low wind (<8 km/hour) and no inclement weather (e.g., rain or fog) between 0630 and 1000 hours CDT. We calculated avian species diversity using Shannon's diversity index (Nichols et al. 1998, Chao and Shen 2003).

To examine productivity of grassland birds, we located nests using three methods: intensive, weekly nest searching of 11 established plots; observing behavioral cues such as adults approaching the nest with nest-building materials and food; and incidentally while collecting other data. At each nest location, we recorded UTM coordinates, species, number of eggs, and incidences of

parasitism. We monitored the nests until successful fledging of chicks or nest failure.

Nest vegetation variables were collected for the majority of nests within one week of fledging or failure. We collected nest vegetation variables in the four cardinal directions with a Daubenmire frame located on the center of the nest bowl and again from a distance of 4-m away from the nest. In each frame, we determined the percent green and dead cover and the percent cover of grass, forbs, shrubs, litter, and bare ground. We used a Robel pole to determine the height-density index for surrounding vegetation at each nest. Additionally, we recorded the height of each nest and the distance to the nearest eastern redcedar (*Juniperus virginiana*).

Using mixed linear models, we tested the effects of year since burn and year × year since burn on avian diversity using pasture and year as random effects. We used distance modeling to determine effective distances and subsequent detection rates for each species in Program Distance (Buckland et al. 2001, 2004). We based all analyses on cluster size using the size-bias regression method. We truncated all analyses at 10%, and we used a divisor of three for the final density estimates because we visited each site three times in each year. To make inferences about abundances, we determined the model (or set of models) which was < 2 ΔAIC for each group of species. We then created a new model which incorporated these closely supported models and bootstrapped the data to determine variance estimates which allowed us to account for model uncertainty. All analyses were done in SAS 9.1.

Macroinvertebrate Sampling.—We sampled invertebrates in four 25-m line transects in each patch of each pasture once during each period of mid-May, mid-June, and mid-July 2006 and 2007 (Dietrick 1961). We were only able to sample in mid-May and mid-June, 2008. Excessive rainfall during mid-June 2007 precipitated sampling into early July by four days. We collected invertebrate samples by holding the intake cone of the vacuum sampler 15 cm above the ground and walking at a slow, constant place along the transect collecting invertebrates in a collection bag attached to the vacuum (Jackson et al. 1987, Burger et al. 1993). All invertebrate sampling was performed by one person in all years. After sampling was completed, we put the collection bag on ice until the sample could be frozen. Invertebrates were stored in a freezer until identification. Vacuum sampling may be biased towards lighter-bodied insects and underestimate heavier-bodied insects such as grasshoppers (Orthoptera: Acrididae) (Buffington and Redak 1998); however, vacuum sampling was appropriate in this case due to its ability to more efficiently sample insects along the ground and on low vegetation (Cooper and Whitmore 1990). We identified invertebrates to unique categories of morphospecies (Oliver and Beattie 1996, Derraik et al. 2002) as fine taxonomic identification is cumbersome (T. Joern, personal communication). Digital photographs and specimens were taken for reference. We dried invertebrates at 75°C for 24 hrs in a lab oven and weighed the dried biomass to the nearest 0.0001g. In addition, we sampled invertebrates around the nest area using vacuum sampling (Dietrick-vacuum) and sweep netting. We sampled along four 25-m line transects with each method for a total

of eight samples per nest. We additionally sampled four points within each of the 11 nest search areas using sweep nets once during mid-May, mid-June, and mid-July. However, these additional samples are still being processed.

Using repeated measures general linear models, we compared the effects of year and month interactions for species diversity, invertebrate biomass, and abundances of Orthoptera, Hemiptera, Homoptera, Diptera, Hymenoptera, Coleoptera, and Araneae. Using mixed linear models, we tested the effects of year since burn on species diversity, invertebrate biomass, and abundances of Orthoptera, Hemiptera, Homoptera, Diptera, Hymenoptera, Coleoptera, and Araneae using pasture and year as random effects. All analyses were done in SAS 9.1.

V. RESULTS AND DISCUSSION

Vegetation. — A description of the response of each vegetation functional group is listed below.

1. Recently-burned patches within treatment pastures were characterized by higher amounts of bare ground and lower amounts of litter compared to patches located in control pastures (Figs. 2a and 2b). In treatment pastures, the amount of bare ground decreased with time since fire and the amount of litter increased with time since fire.

2. Percent cover of live vegetation was lower in recently-burned patches compared to patches in the control pastures but higher in treatment patches that had been burned 3 to 5 years previously (Fig. 3a). Compared to patches in

control pastures, the percent cover of dead vegetation in treatment patches was reduced in recently-burned patches, but increased to levels comparable to those in control pastures with increasing time since fire (Fig. 3b).

3. Recently-burned patches in treatment pastures had lower amounts of live grass compared to patches in control pastures, while patches within treatment pastures that had burned 3 to 5 years previously had higher amounts of live grass compared to patches in control pastures (Fig. 4a).

4. Recently-burned patches in treatment pastures had higher amounts of live forbs compared to patches within control pastures while patches within treatment pastures that had been burned 3 to 5 years previously had lower amounts of live forbs compared to patches in control pastures (Fig. 5a).

5. Compared to patches in control pastures, percent cover of live forbs in treatment pastures that had recently been burned was lower, but higher within treatment patches that had been burned 3 to 5 years previously (Fig. 6a).

6. Vegetation height and visual obstruction within treatment pastures was lower in recently-burned patches than in patches located in treatment pastures, but were at levels comparable to, or higher than, those in control patches by five years after being burned (Figs. 7a and 7b).

7. Sagebrush height was reduced by burning but returned to pre-burn levels within five years (Fig. 8a). Burning had little, if any, effect on sagebrush density (Fig. 8b).

Avian Overall.—During 2006-2008, we detected a total of 53 bird species (Table 2). We describe responses of the three most common species (Cassin's

sparrow [*Aimophila cassinii*], lark sparrow [*Chondestes grammacus*], and field sparrow [*Spizella pusilla*]) to patch-burning. Cassin's sparrows were more abundant in 2006 and 2008 and were rarely detected in 2007 (Fig. 9). In 2006, the highest densities of Cassin's sparrows were in the one year post-burn and unburned patches. However, we only had one patch that was in the one year post-burn category and this patch had higher numbers of Cassin's sparrows in general. In 2007, the highest densities of Cassin's sparrows were in two and three years post-burn patches, while in 2008, densities of Cassin's sparrows increased as the number of years since the burning of the patch increased with years three and four post-burn having higher densities than unburned patches (Fig. 10). This suggests that Cassin's sparrows are focusing on the older patches, but still require some disturbance to prevent the vegetation from becoming too dense.

Field sparrows were more abundant in 2007 compared to 2006 and 2008, but were rarely detected in 2008 (Fig. 11). In 2006, the highest densities of field sparrows were in three year post-burn patches and unburned patches. In 2007, the highest densities of field sparrows were in four year post-burn and unburned patches, and the highest densities were in five year post-burn and unburned patches in 2008 (Fig. 12). This suggests that Field sparrows are focusing on the older patches, but unlike Cassin's sparrows, are not as dependent on disturbance to prevent the vegetation from becoming too dense. This may help explain why we did not detect many Cassin's sparrows in 2007; the thick vegetation growth caused by the high levels of precipitation may have prevented the Cassin's

sparrows from utilizing those habitats, whereas field sparrow were able to take advantage of the dense vegetation growth.

Lark sparrows were most abundant in 2008 (Fig. 13). In 2006, lark sparrows were most abundant in two year post-burn patches. However, in 2007 and 2008, lark sparrows were most abundant in the most recently burned patches (current year burn and two year post-burn in 2007 and current year burn and one year post-burn in 2008). Overall, Lark sparrows were most abundant in the most recently disturbed patches (Fig. 14).

Avian Diversity.—We did not detect any significant differences in avian diversity among years ($F_{2,3} = 0.67$; $p = 0.5$); however, we did find several significant differences among years since burn ($F_{6,7} = 2.83$; $p = 0.01$) (Fig. 15). In particular, the diversity in the year-of-burn patches was significantly lower than remaining patches, including the unburned reference patches. However, excluding the year-of-burn patches, all burn treatments were significantly higher than the unburned patches. The diversity peaked three years post-burn after which it began to decline. Average diversity for the burned patches was 1.64 ± 0.34 compared to 1.53 ± 0.37 for unburned patches. Excluding current year burn, patch-burn treatments overall had higher diversity and evenness. However, we did detect a peak during the third year post-burn and a subsequent decline which suggests a need for frequent disturbance to prevent senescence in vegetation.

Avian Nest Searching.—We located 99 nests representing 13 species in 2006 (Table 3). Field sparrow nests were the most common nests located (22.2%), followed by Cassin's sparrow (19.1%), lark sparrow (12.1%), and

mourning dove (*Zenaida macroura*) (10.1%). Of the 99 nests we located, 45 nests hatched young and 27 of those nests successfully fledged at least one chick.

We located 114 nests representing 18 bird species in 2007 (Table 4). Lark sparrow nests were the most common nests located (22.8%), followed by scissor-tailed flycatcher (*Tyrannus forficatus*) (15.7%) and northern mockingbird (*Mimus polyglottos*) (13.1%). Some of the other less common nests included western kingbird (*Tyrannus verticalis*), eastern kingbird (*Tyrannus tyrannus*), wild turkey (*Meleagris gallopavo*), blue grosbeak (*Passerina caerulea*), and red-headed woodpecker (*Melanerpes erythrocephalus*). Forty-seven of 114 nests hatched young and 24 of those nests successfully fledged at least one chick.

We located 220 nests representing 24 bird species in 2008 (Table 5). Lark sparrow nests were the most common nests located (31.0%), followed by northern mockingbird (18.2%), Cassin's sparrow (10.0%), and scissor-tailed flycatcher (10.0%). Of the 220 nests observed, 107 nests hatched and 58 of those nests successfully fledged at least one chick.

Invertebrate Biomass.—Invertebrate biomass was significantly higher in 2007 compared to 2006 and 2008 ($F_{2,3} = 11.18$; $P<0.0001$; Fig. 16). In all three years, biomass changed significantly from May to June ($F_{1,2}=43.2$; $P<0.0001$; $F_{1,2}=11.06$; $P=0.01$; $F_{1,2}=44.8$; $P<0.0001$, 2006, 2007, and 2008, respectively), but did not change significantly from June to July in either 2006 ($F_{1,2}=0.32$; $P=0.5$) or 2007 ($F_{1,2}=0.65$; $P=0.4$). However, different patterns emerged among the three years. In 2006 and 2008, biomass decreased from May to June, whereas biomass increased between May and June in 2007. We did not find an overall

significant year since burn trend ($F_{6,7}$ = 1.98; P=0.06); however, there was a significant year × year since burn interaction ($F_{8,9}$ = 3.51; P=0.007; Fig. 17). In 2006 and 2008, there was a general trend of increasing biomass since the patch was burned; however, the trend was less distinct in 2007. In 2006, invertebrate biomass in the 3-year post-burn treatment had increased to the same level as unburned treatment. In 2008, all patch-burn patches had higher biomass than unburned areas. In addition, there was little difference in biomass between the patch-burn patches. In 2008, the current year burn had lower biomass than the other patches and the older patch-burn patches had higher biomass than the unburned, control treatment.

Monthly Invertebrate Sampling.—Using vacuum sampling, we caught a total of 10,990 individuals of 331 morphospecies representing 14 orders in 2006 and 52,250 individuals of 519 morphospecies representing 17 orders in 2007. In May and June 2008, we caught a total of 20,790 individuals in 489 morphospecies representing 16 orders. In 2006, the most abundant orders sampled were Homoptera (26%), Diptera (23%), Orthoptera (10%), and Hemiptera (8.3%). The most abundant orders collected in 2007 were Diptera (40%), Homoptera (26%), Hymenoptera (17%), and Hemiptera (9.5%). In 2008, the most abundant orders sampled were Diptera (38.8%), Homoptera (23.2%), Hymenoptera (16.7%), and Hemiptera (7.8%). In 2006, patches averaged 21.3 morphospecies per sample (range 2-70) with an average of 83.1 individuals per sample (range 2–508). In 2007, patches averaged 58.4 morphospecies per sample (range 7–149) with an average of 395.8 individuals per sample (range 38-992). In 2008, patches

averaged 42.0 morphospecies per sample (range 5–86) with an average of 236.2 individuals per sample (range 6-788).

There was a very strong year effect ($F_{2,3}$=91.63; P<0.0001) with diversity for 2007 being 1.5x higher than diversity for 2006 and 1.0x higher than diversity for 2008 (Fig. 18). Diversity for 2008 was 1.3x higher than diversity for 2006. In 2006, monthly invertebrate diversity differed between May and June ($F_{1,2}$=53.56; P<0.0001), but was not significantly different between June and July ($F_{1,2}$=0.23; P=0.6). In 2007, monthly invertebrate diversity did not differ between May and June ($F_{1,2}$=0.37; P=0.5) or June and July ($F_{1,2}$=0.78; P=0.3). In 2008, monthly invertebrate diversity did not differ between May and June ($F_{1,2}$=0.5; P=0.4).

There was higher diversity in patches which had longer periods since burned for all three years (2006, $F_{3,4}$=6.02; P=0.001; 2007, $F_{4,5}$=13.04; P<0.0001; 2008, $F_{5,6}$=12.94; P<0.0001; Fig. 19). There was also a significant year × year since burn interaction ($F_{6,7}$=2.74; P=0.01; Fig. 20).

Invertebrate Abundance by Order.—We determined Orthoptera abundance was much higher in 2006 compared to either 2007 or 2008 ($F_{2,3}$=7.74; P<0.0001; Fig. 21). We did not detect a difference in Orthoptera abundance by month in 2006 ($F_{2,3}$=2.4; P=0.1) or 2007 ($F_{2,3}$=0.1; P=0.9). In 2008, we detected a significant difference between May and June ($F_{1,2}$=34.4; P<0.0001). There was a marginally non-significant year × year since burn interaction ($F_{6,7}$=2.12; P=0.056; Fig. 23). In 2006, there was a general trend of increasing grasshopper abundance since the patch was burned; however, the trend was less distinct in

2007 (Fig. 23). Conversely in 2008, there was a general trend of decreasing grasshopper abundance since the patch was burned (Fig. 22).

We determined Hemiptera abundance was higher in 2007 compared to 2006 and 2008 ($F_{2,3}$=33.9; P<0.0001; Fig. 24). In 2006, we found significant higher numbers of Hemiptera in May than June ($F_{1,2}$=5.3; P=0.02) but not June and July ($F_{1,2}$=2.7; P=0.1). In 2007, we found significant higher numbers of Hemiptera in June than July ($F_{1,2}$=8.52; P=0.005) but not May and June ($F_{1,2}$=0.7; P=0.3). In 2008, there was not a significant difference between May and June ($F_{1,2}$=0.1; P=0.9). Although Hemiptera abundance was much higher in year of burn and one year post-burn, there was not a significant year since burn effect ($F_{6,7}$=1.31; P=0.2; Fig. 25) nor was there a significant year × year since burn effect ($F_{6,7}$=1.59; P=0.1; Fig. 26). In 2006, there was a general trend of increasing Hemiptera abundance since the patch was burned; however, in 2007, Hemiptera abundance was very high in patches burned the current year, but abundance was low in patches one year since burn with a general trend of increasing Hemiptera abundance since the patch was burned. In 2008, Hemiptera abundance was much higher in year of burn and one year post-burn than unburned or older burned patches.

We determined Homoptera abundance was higher in 2007 compared to 2006 and 2008 ($F_{2,3}$=37.4; P<0.0001; Fig. 27). In 2006, there were no significant differences between May and June ($F_{1,2}$=2.7; P=0.1) or June and July ($F_{1,2}$=0.5; P=0.4). In 2007, there were significant higher numbers of Homoptera in May than June ($F_{1,2}$=21.1; P<0.0001) but not June and July ($F_{1,2}$=3.1; P=0.08). In

2008, there was not a significant difference between May and June ($F_{1,2}$=0.1; P=0.7). There was not a significant year since burn effect ($F_{6,7}$=0.3; P=0.9; Fig. 28) nor was there a significant year × year since burn effect ($F_{6,7}$=0.8; P=0.5; Fig. 29).

Although we determined Lepidoptera abundance was higher in 2007 compared to 2006 and 2008, this finding was not significant ($F_{2,3}$=0.4; P=0.6; Fig. 30). In 2006, there were no significant differences between May and June ($F_{1,2}$=2.3; P=0.1), but there was a significant difference between June and July ($F_{1,2}$=7.4; P=0.009). In 2007, there were no significant differences between May and June ($F_{1,2}$=0.8; P=0.3) or June and July ($F_{1,2}$=2.1; P=0.1). In 2008, there was a significant difference between May and June ($F_{1,2}$=4.4; P=0.04). There was not a significant year since burn effect ($F_{6,7}$=1.7; P=0.1; Fig. 31) nor was there a significant year × year since burn effect ($F_{6,7}$=1.6; P=0.1; Fig. 32).

Diptera abundance was significantly higher in 2007 compared to 2006 and 2008 ($F_{2,3}$=90.2; P<0.0001; Fig. 33). In 2006, there was not a significant difference between May and June ($F_{1,2}$=3.5; P=0.06), but there was a significant difference between June and July ($F_{1,2}$=4.2; P=0.04). In 2007, there were significant higher numbers of Diptera in May than June ($F_{1,2}$=28.3; P<0.0001) as well as a significantly higher numbers in June compared to July ($F_{1,2}$=6.5; P=0.01). In 2008, there were significant higher numbers of Diptera in May than June ($F_{1,2}$=7.6; P=0.008). There was a significant year since burn effect ($F_{6,7}$=2.9; P=0.01; Fig. 34), however, there was not a significant year × year since burn effect ($F_{6,7}$=1.5; P=0.1; Fig. 35).

Hymenoptera abundance was significantly higher in 2007 compared to 2006 and 2008 ($F_{2,3}$=32.6; P<0.0001; Fig. 36). In 2006, there were significant higher numbers of Hymenoptera in May than June ($F_{1,2}$=26.2; P<0.0001) but not June and July ($F_{1,2}$=3.2; P=0.07). In 2007, there was not a significant difference between May and June ($F_{1,2}$=1.4; P=0.2), but there was a significant difference between June and July ($F_{1,2}$=7.9; P=0.007). In 2008, there was not a significant difference between May and June ($F_{1,2}$=1.1; P=0.2). There was a significant year since burn effect ($F_{6,7}$=2.9; P=0.009; Fig. 37), however, there was not a significant year × year since burn effect ($F_{6,7}$=1.7; P=0.1; Fig. 38).

Coleoptera abundance was significantly higher in 2006 compared to 2007 and 2008 ($F_{2,3}$=6.9; P=0.001; Fig. 39). In 2006, there were significant higher numbers of Coleoptera in May than June ($F_{1,2}$=5.5; P=0.02) but not June and July ($F_{1,2}$=1.2; P=0.2). In 2007, there were significant higher numbers of Coleoptera in May than June ($F_{1,2}$=4.4; P=0.04) but not June and July ($F_{1,2}$=0.2; P=0.7). In 2008, there was not a significant difference between May and June ($F_{1,2}$=0.7; P=0.7). There was a significant year since burn effect ($F_{6,7}$=2.8; P=0.01; Fig. 40), however, there was not a significant year × year since burn effect ($F_{6,7}$=0.8; P=0.5; Fig. 41).

Araneae abundance was significantly higher in 2006 compared to 2007 and 2008 ($F_{2,3}$=22.8; P<0.0001; Fig. 42). In 2006, there were significant differences between May and June ($F_{1,2}$=9.3; P=0.003) and June and July ($F_{1,2}$=4.7; P=0.03). In 2007, there were significant differences between May and June ($F_{1,2}$=8.1; P=0.006) and June and July ($F_{1,2}$=12.8; P=0.0009). In 2008, there

was not a significant difference between May and June ($F_{1,2}$=2.8; P=0.1). There was a significant year since burn effect ($F_{6,7}$=2.3; P=0.03; Fig. 43), as well as a significant year × year since burn effect ($F_{6,7}$=2.2; P=0.04; Fig. 44).

Conclusions and Management Recommendations.—Our results suggest that patch-burn management does improve overall avian diversity. Excluding the patches that were burned in the current year, we detected higher diversity in patch-burned patches than in the unburned control patches. Species such as lark sparrows were more abundant in the recently burned patches. However, certain species, particularly Cassin's and field sparrows, were more abundant in the older patches. Both of these species are shrub-nesting species that did not readily nest in the recently burned areas. Field sparrows were more abundant in the unburned patches and Cassin's sparrows were more abundant in the older, burned patches. With these findings, we suggest maintaining unburned areas to benefit those species which do not respond well to disturbance or lengthening the period of disturbance.

Our results suggest that patch-burn management may improve macroinvertebrate diversity; however, these results were closely related to sampling year and were most likely the result of very different weather patterns among years. In 2006, diversity was highest in the unburned patches, whereas in 2008, patches that had been burned one year previously had the highest diversity. In the wet year of 2007, diversity was similar among the patches, but slightly higher in the oldest patches (3–4 years after burn and unburned patches). These trends are similar in the other invertebrate metrics. However, the response of the

various invertebrate orders we examined varied. Certain orders such as Orthoptera were more abundant in unburned patches, whereas orders such as Hemiptera were more abundant in the recently burned patches. Still, orders such as Lepidoptera showed little to no trend among the years since burned, while orders such as Araneae peaked at 3–4 years post-burn. Like the management implications for the birds, these results suggest the importance of maintaining unburned areas as refuge areas for certain invertebrate taxa. Moreover, these results also demonstrate the importance of providing a diversity of habitats for invertebrates, as not all taxa will respond in the same way to disturbance.

Patch-burn management at Hal and Fern Cooper WMA altered the structure of vegetation within pastures where this treatment was applied. Vegetation heights and percent cover were typically reduced immediately following a patch burn but returned to levels which were comparable to those found in control pastures within 3–5 years. In some instances, vegetation that had been treated with patch-burning within treatment pastures had cover values higher than in the control pastures after a period of 3–5 years post-fire. Sagebrush density did not decline following the application of patch-burn management, suggesting that sand sagebrush is not harmed by prescribed fires of the type applied in this study.

The results of this research demonstrate that patch-burn management provides a useful tool for managers who want to alter the vegetation structure, at large scales, of sand sage prairie. Using patch-burn management, managers may

be able to provide suitable habitat for wildlife species that require a diversity of habitat conditions that are not provided by traditional rangeland management.

VI. SIGNIFICANT DEVIATIONS

The vacuum sampling machine which we utilized for invertebrate sampling broke before we were able to sample the July sampling points in 2008. Therefore, the monthly invertebrate analyses for 2008 could only be conducted on May and June. However, we still were able to collect the sweep net samples in July 2008. Weather conditions varied greatly among the years. We experienced extreme drought conditions in 2006 which were so severe that we were unable to burn any of the patches in that year.

VII. TOTAL COSTS: $ 199,999.99

FEDERAL SHARE: $149,999.99

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Table 1. Pastures used in the study, the size of all patches in each pasture and the year each patch was burned at Cooper Wildlife Management Area in Woodward County, Oklahoma.

Pasture	Patch	Size	Year of Patch-burn
North	A	160 ha (395 ac)	unburned control
	B	198 ha (489 ac)	unburned control
	C	156 ha (385 ac)	unburned control
South	A	300 ha (741 ac)	unburned control
	B	211 ha (521 ac)	unburned control
	C	247 ha (610 ac)	unburned control
Middle	A	101 ha (249 ac)	2008
	B	198 ha (489 ac)	2007
	C	204 ha (503 ac)	2005
East	A	154 ha (380 ac)	2008
	B	352 ha (869 ac)	2007
	C	350 ha (864 ac)	2004
Bodwell	A	167 ha (412 ac)	2004
	B	144 ha (355 ac)	2003
	C	96 ha (237 ac)	2008

Table 2. Fifty-five species of birds were detected on Cooper WMA, May-July 2006-2008.

Species	Scientific Name
American Crow	*Corvus brachyrhynchos*
American Kestrel	*Falco sparverius*
Baltimore Oriole	*Icterus galbula*
Bell's Vireo	*Vireo bellii*
Bewick's Wren	*Thryomanes bewickii*
Blue-gray Gnatcatcher	*Polioptila caerulea*
Blue Grosbeak	*Guiraca caerulea*
Brown-headed Cowbird	*Molothrus ater*
Brown Thrasher	*Toxostoma rufum*
Carolina Chickadee	*Poecile carolinensis*
Carolina Wren	*Thryothorus ludovicianus*
Cassin's Sparrow	*Aimophila cassinii*
Chipping Sparrow	*Spizella passerina*
Chuck-will's-widow	*Caprimulgus carolinensis*
Clay-colored Sparrow	*Spizella pallida*
Cliff Swallow	*Petrochelidon pyrrhonota*
Common Grackle	*Quiscalus quiscula*
Cooper's Hawk	*Accipiter cooperii*
Dickcissel	*Spiza americana*
Downy Woodpecker	*Picoides pubescens*
Eastern Bluebird	*Sialia sialis*
Eastern Kingbird	*Tyrannus tyrannus*
Eastern Meadowlark	*Sturnella magna*
Eastern Phoebe	*Sayornis nigricans*
Field Sparrow	*Spizella pusilla*
Grasshopper Sparrow	*Ammodramus savannarum*
Great Horned Owl	*Bubo virginianus*
Greater Roadrunner	*Geococcyx californianus*
Hairy Woodpecker	*Picoides villosus*
Lark Sparrow	*Chondestes grammacus*
Loggerhead Shrike	*Lanius ludovicianus*
Killdeer	*Charadrius vociferus*
Mallard	*Anas platyrhynchos*
Mississippi Kite	*Ictinia mississippiensis*
Mourning Dove	*Zenaida macroura*
Northern Bobwhite	*Colinus virginianus*
Northern Cardinal	*Cardinalis cardinalis*
Northern Flicker	*Colaptes auratus*
Northern Harrier	*Circus cyaneus*
Northern Mockingbird	*Mimus polyglottos*
Painted Bunting	*Passerina ciris*
Red-bellied Woodpecker	*Melanerpes carolinus*
Red-headed Woodpecker	*Melanerpes erythrocephalus*
Ring-necked Pheasant	*Phasianus colchicus*
Rock Dove	*Columba livia*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Blackbird	*Agelaius phoeniceus*
Scissor-tailed Flycatcher	*Tyrannus forficatus*
Swainson's Hawk	*Buteo swainsoni*
Turkey Vulture	*Cathartes aura*
Upland Sandpiper	*Bartramia longicauda*
Western Kingbird	*Tyrannus verticalis*
Wild Turkey	*Meleagris gallopavo*
Western Meadowlark	*Sturnella neglecta*
Yellow-billed Cuckoo	*Coccyzus americanus*

Table 3. We located 99 nests representing 13 species of birds were located on Cooper WMA, May 2006–July 2006.

Species	Scientific Name
Brown Thrasher	*Toxostoma rufum*
Cassin's Sparrow	*Aimophila cassinii*
Dickcissel	*Spiza americana*
Field Sparrow	*Spizella pusilla*
Grasshopper Sparrow	*Ammodramus savannarum*
Lark Sparrow	*Chondestes grammacus*
Eastern Meadowlark	*Sturnella magna*
Western Meadowlark	*Sturnella neglecta*
Mourning Dove	*Zenaida macroura*
Northern Bobwhite	*Colinus virginianus*
Northern Mockingbird	*Mimus polyglottos*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Blackbird	*Agelaius phoeniceus*

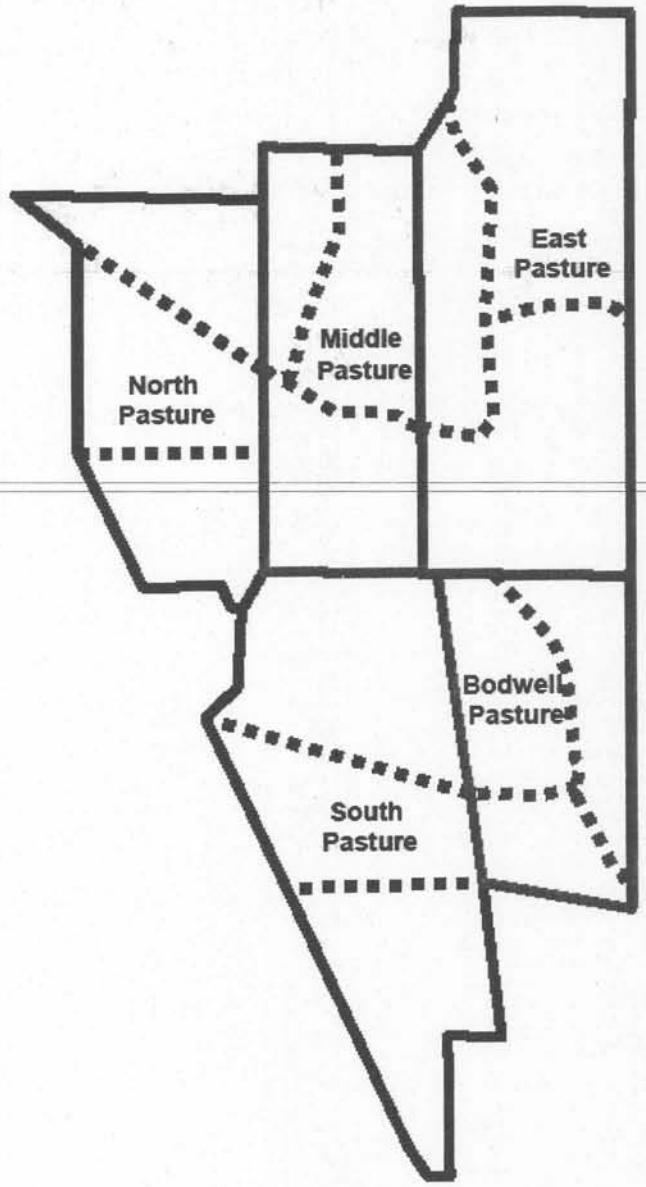
Table 4. We located 114 nests representing 18 species of birds on Cooper WMA, May 2007–July 2007.

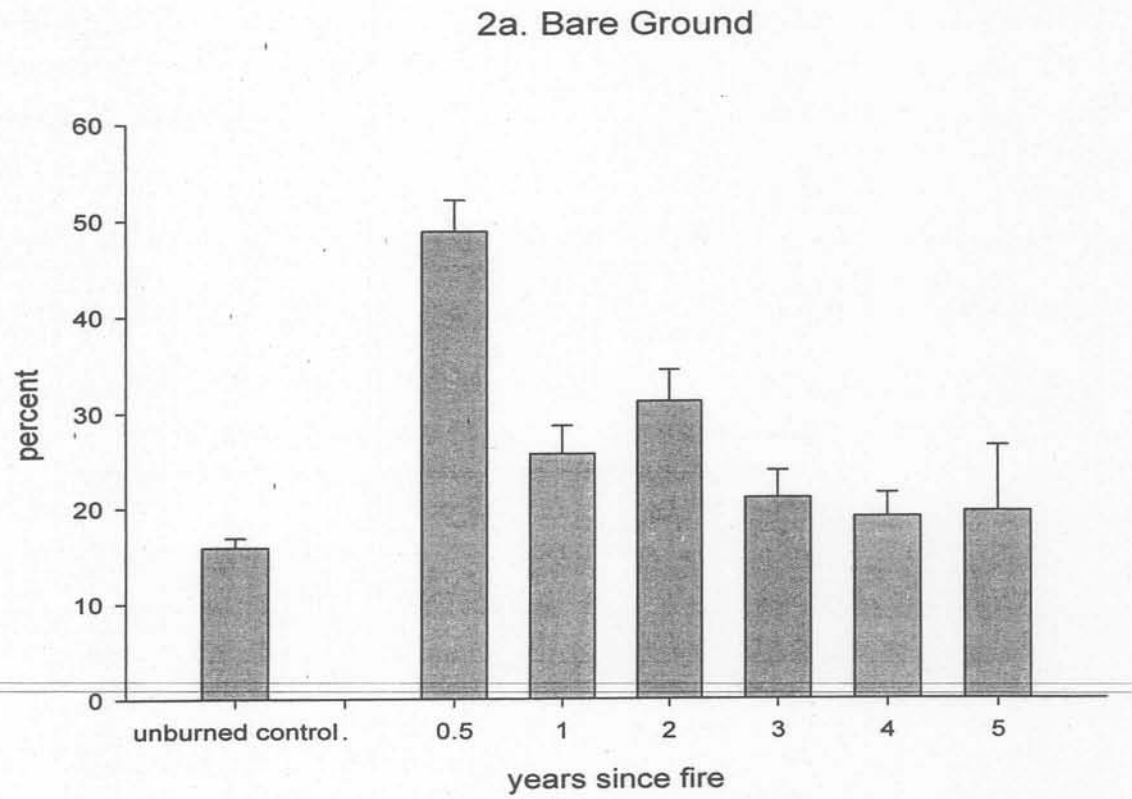
Species	Scientific Name
Brown Thrasher	*Toxostoma rufum*
Cassin's Sparrow	*Aimophila cassinii*
Dickcissel	*Spiza americana*
Field Sparrow	*Spizella pusilla*
Grasshopper Sparrow	*Ammodramus savannarum*
Lark Sparrow	*Chondestes grammacus*
Western Meadowlark	*Sturnella neglecta*
Mourning Dove	*Zenaida macroura*
Northern Bobwhite	*Colinus virginianus*
Northern Mockingbird	*Mimus polyglottos*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Blackbird	*Agelaius phoeniceus*
Scissor-tailed Flycatcher	*Tyrannus forficatus*
Western Kingbird	*Tyrannus verticalis*
Eastern Kingbird	*Tyrannus tyrannus*
Wild Turkey	*Meleagris gallopavo*
Blue Grosbeak	*Passerina caerulea*
Red-headed Woodpecker	*Melanerpes erythrocephalus*

Table 5. We located 220 nests representing 24 species of birds on Cooper WMA, May 2008–July 2008.

Species	Scientific Name
Blue Grosbeak	*Passerina caerulea*
Brown Thrasher	*Toxostoma rufum*
Cassin's Sparrow	*Aimophila cassinii*
Cliff Swallow	*Petrochelidon pyrrhonota*
Common Grackle	*Quiscalus quiscula*
Common Poorwill	*Phalaenoptilus nuttallii*
Dickcissel	*Spiza americana*
Eurasian Collared-Dove	*Streptopelia decaocto*
Field Sparrow	*Spizella pusilla*
Grasshopper Sparrow	*Ammodramus savannarum*
Greater Roadrunner	*Geococcyx californianus*
Killdeer	*Charadrius vociferus*
Lark Sparrow	*Chondestes grammacus*
Loggerhead Shrike	*Lanius ludovicianus*
Western Meadowlark	*Sturnella neglecta*
Mourning Dove	*Zenaida macroura*
Northern Bobwhite	*Colinus virginianus*
Northern Mockingbird	*Mimus polyglottos*
Red-headed Woodpecker	*Melanerpes erythrocephalus*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Blackbird	*Agelaius phoeniceus*
Scissor-tailed Flycatcher	*Tyrannus forficatus*
Western Kingbird	*Tyrannus verticalis*
Wild Turkey	*Meleagris gallopavo*

Figure 1. Hal and Fern Cooper Wildlife Management Area study pastures and patches.





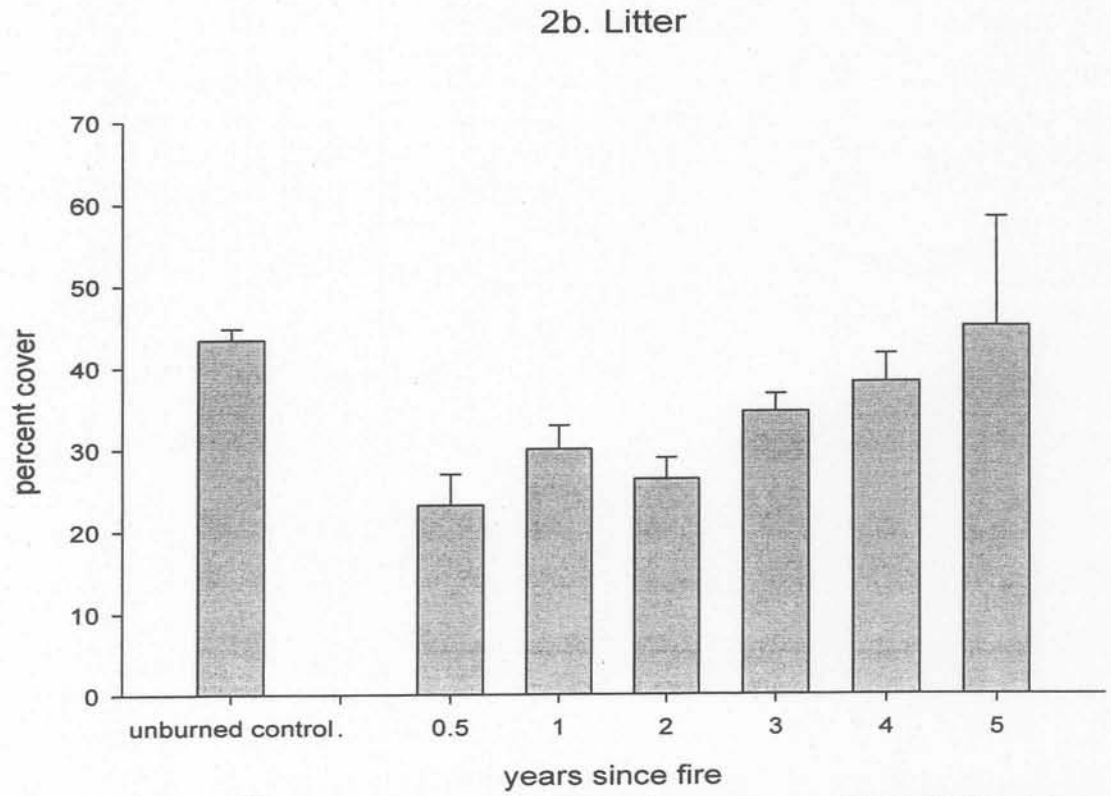
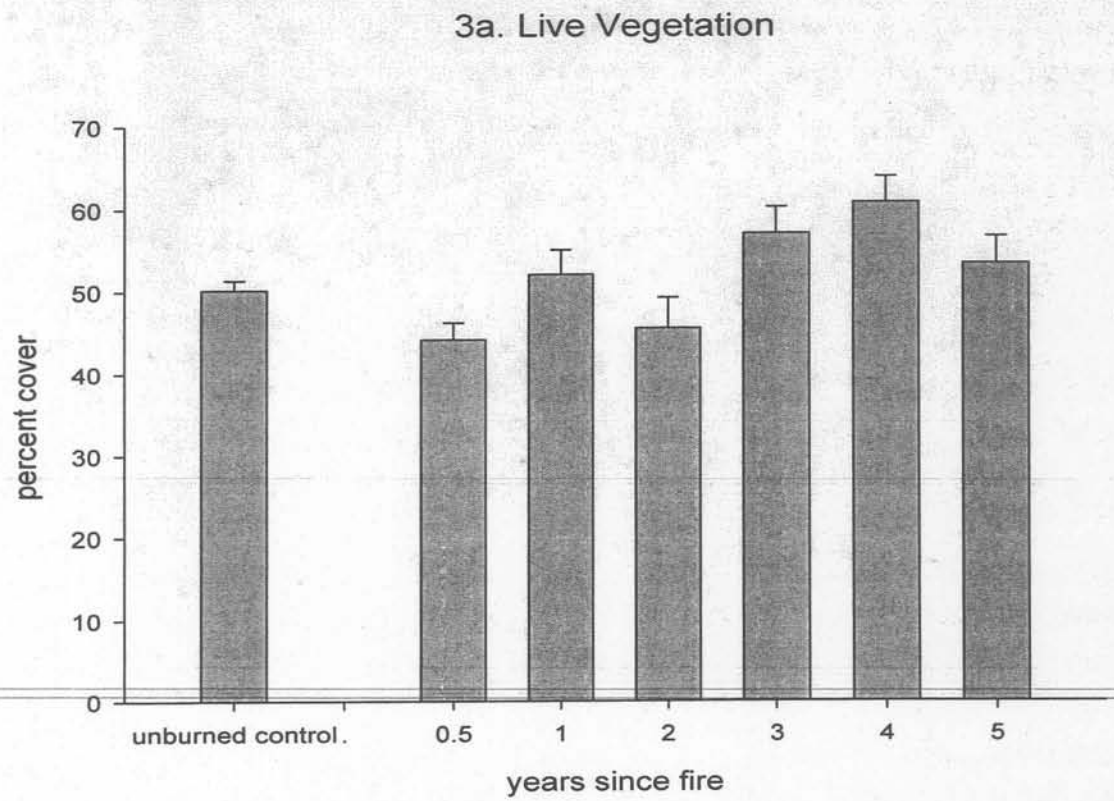


Figure 2. Mean ± SE percent bare ground (2a) and percent cover litter (2b) in pastures managed by patch-burning at Cooper WMA.



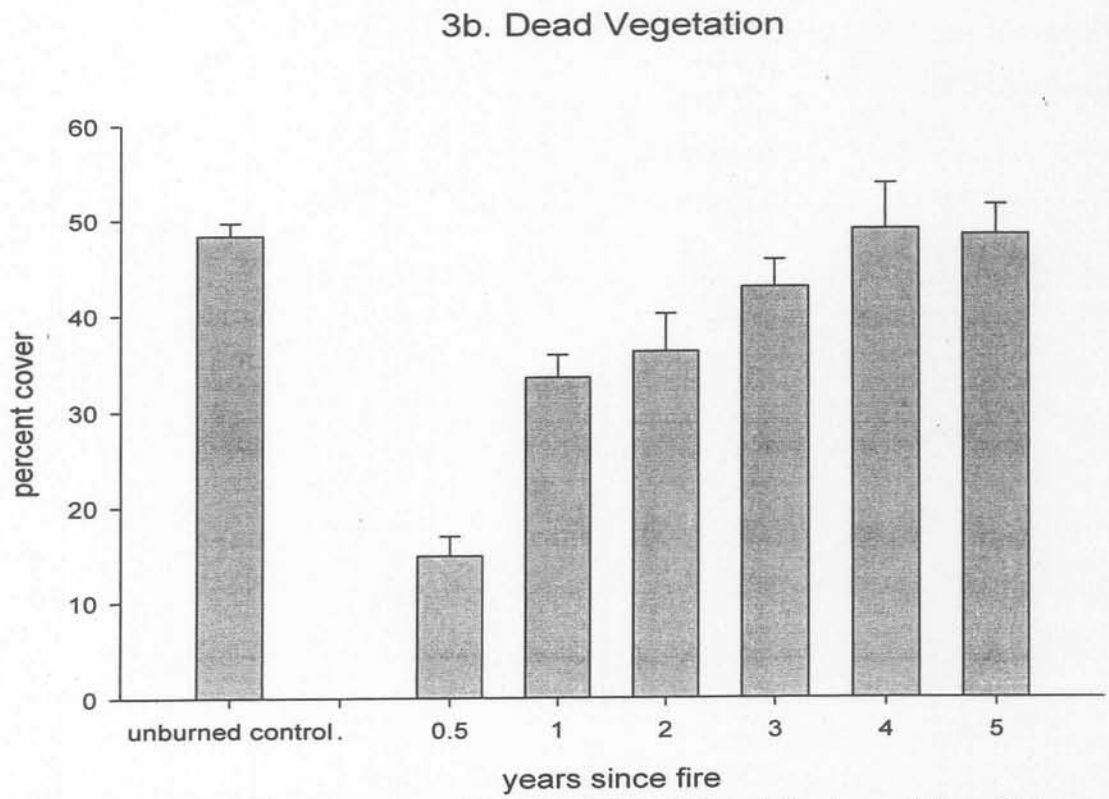
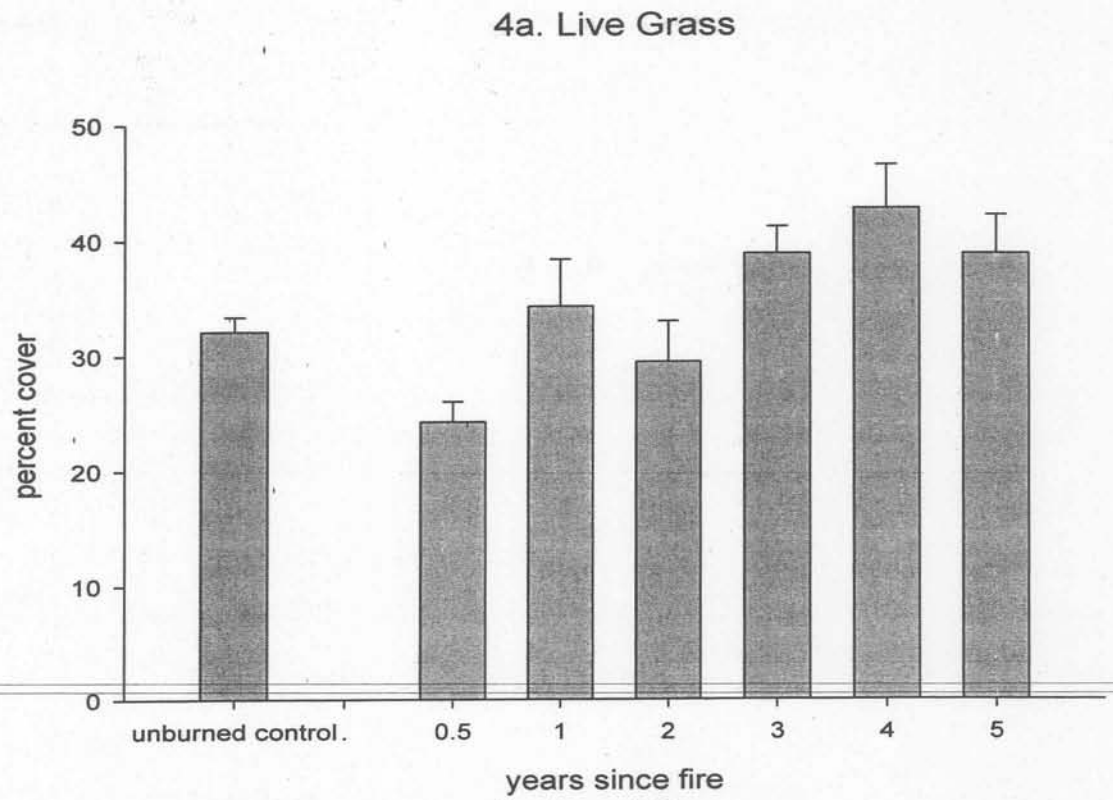


Figure 3. Mean ± SE percent cover live vegetation (3a) and dead vegetation (3b) in pastures managed by patch-burning at Cooper WMA.



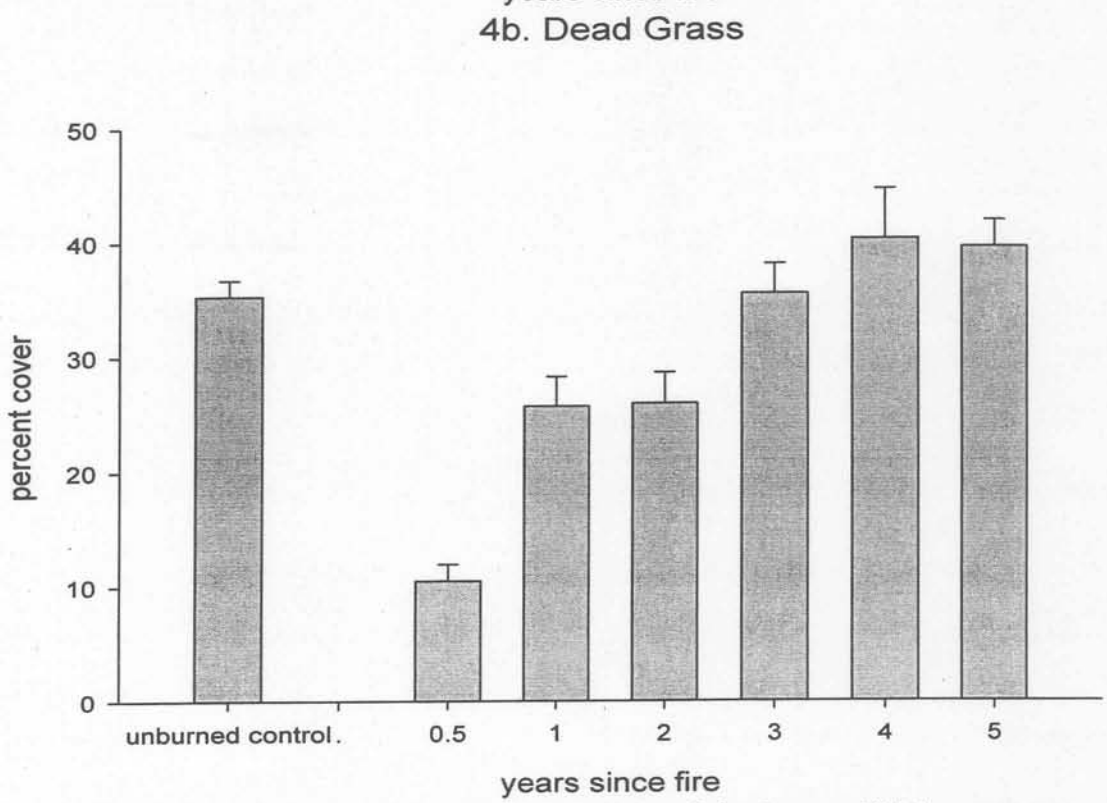
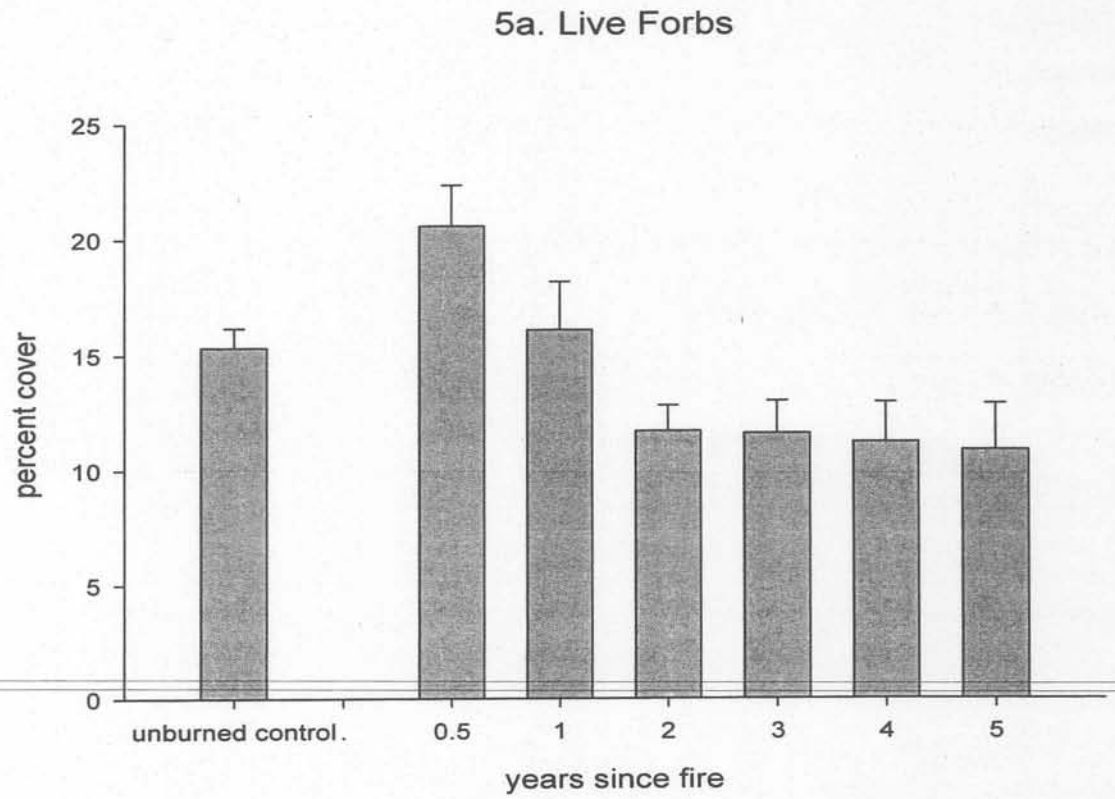


Figure 4. Mean ± SE percent cover live grass (4a) and dead grass (4b) in pastures managed by patch-burning at Cooper WMA.



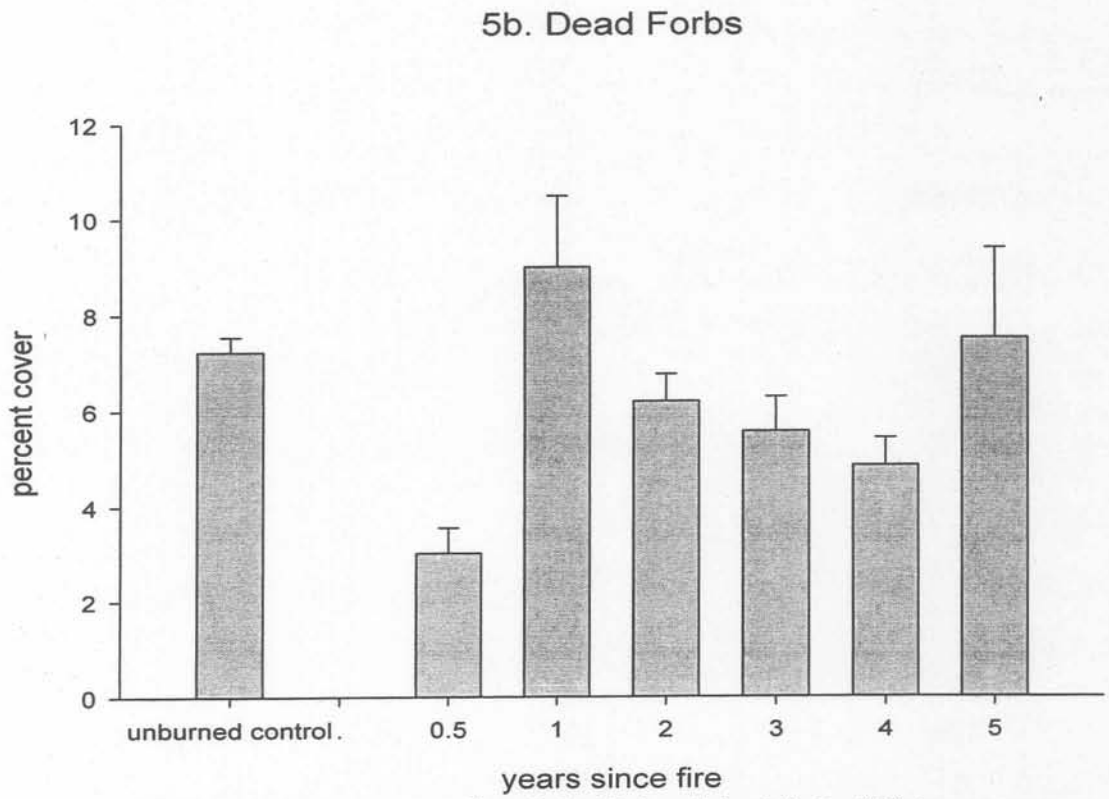
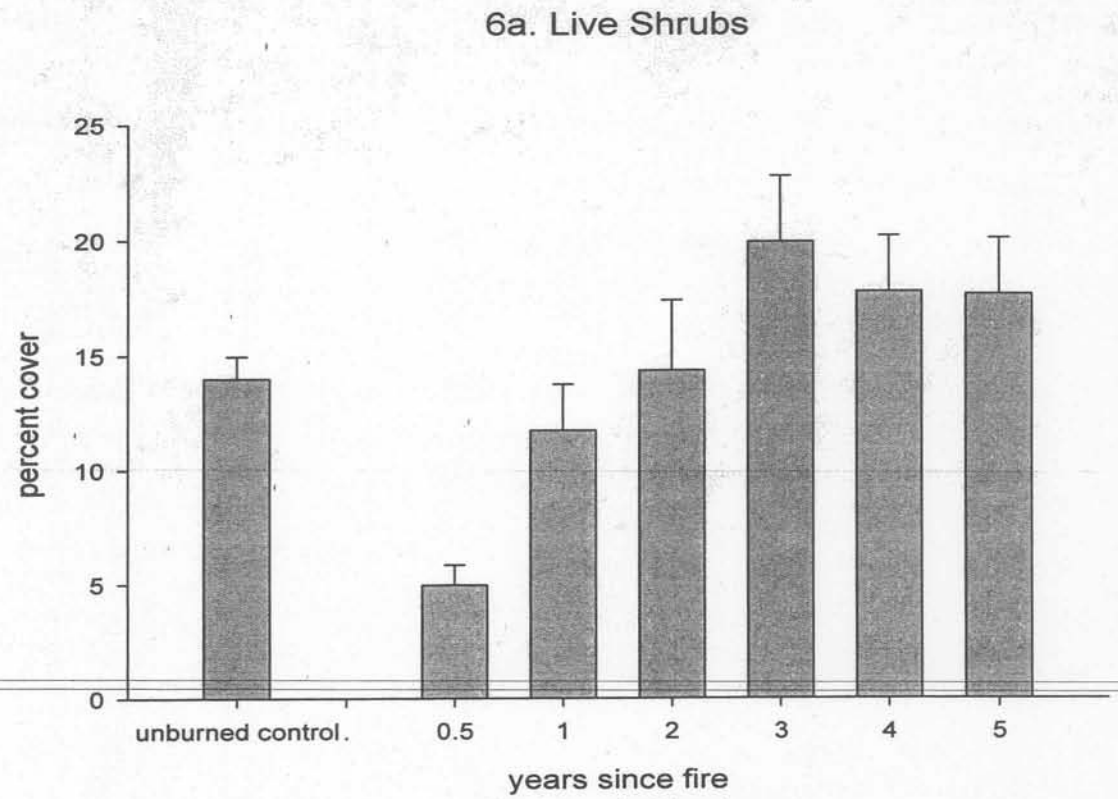


Figure 5. Mean ± SE percent cover live forbs (5a) and dead forbs (5b) in pastures managed by patch-burning at Cooper WMA.



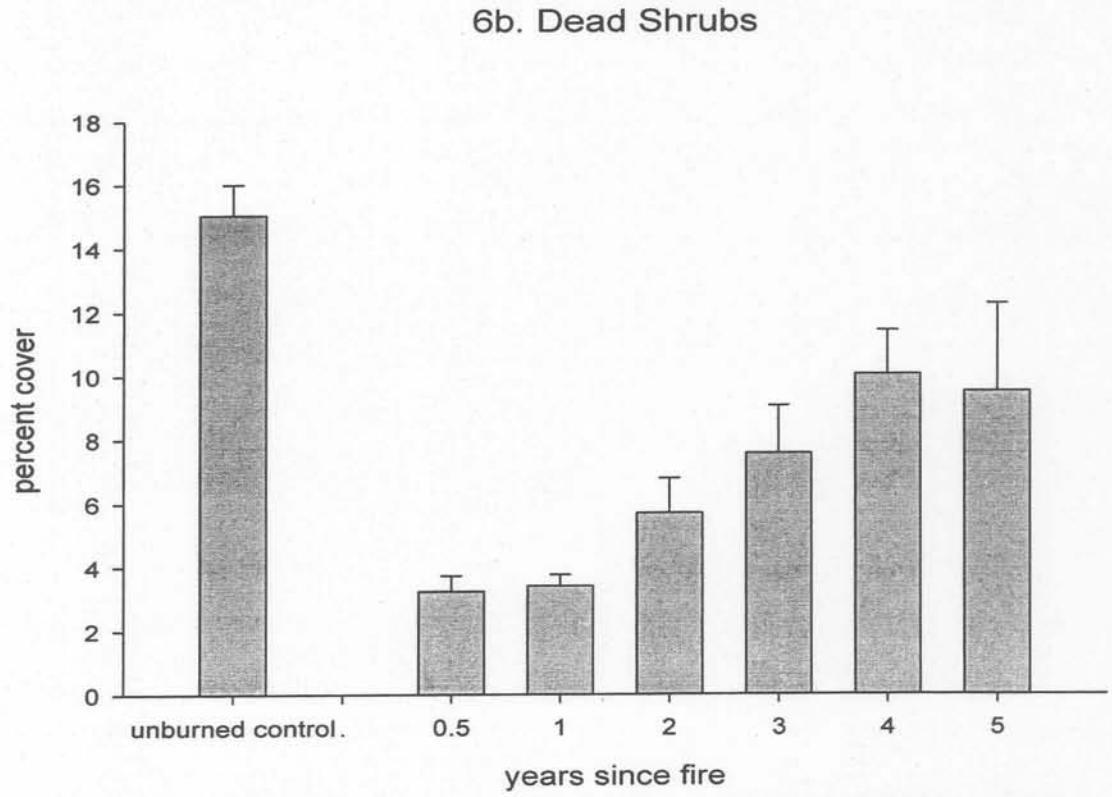
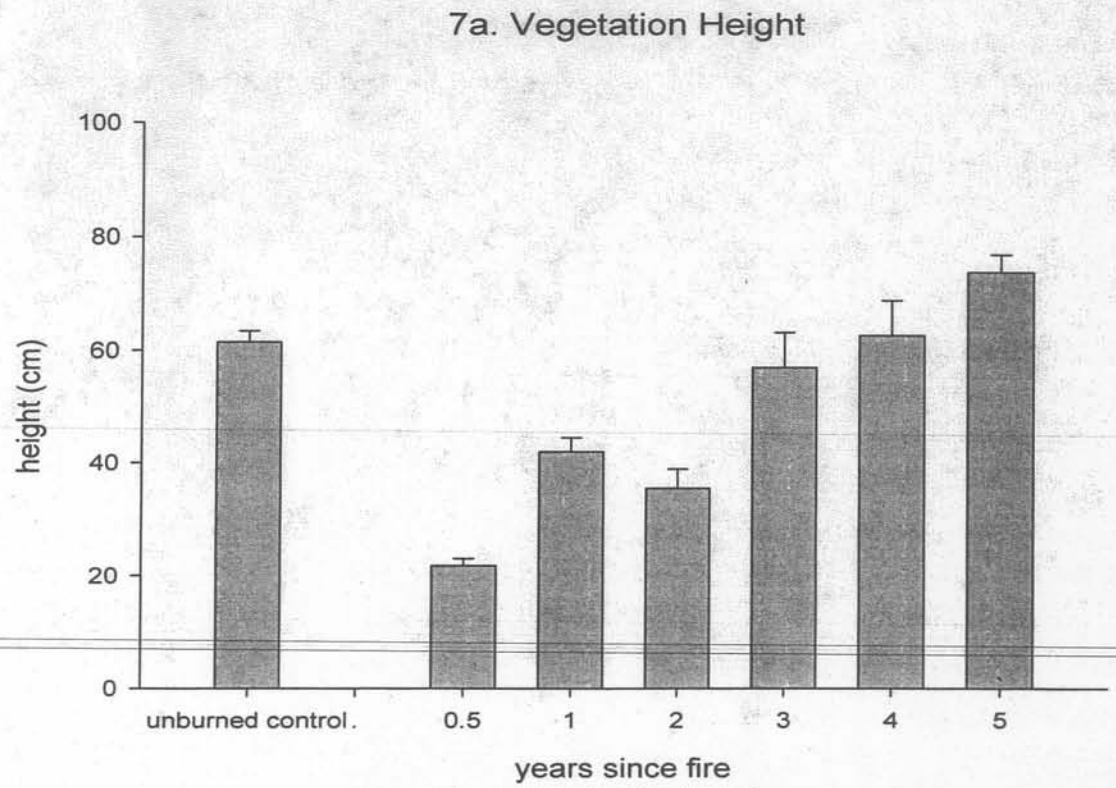


Figure 6. Mean ± SE percent cover live shrubs (6a) and dead shrubs (6b) in pastures managed by patch-burning at Cooper WMA.



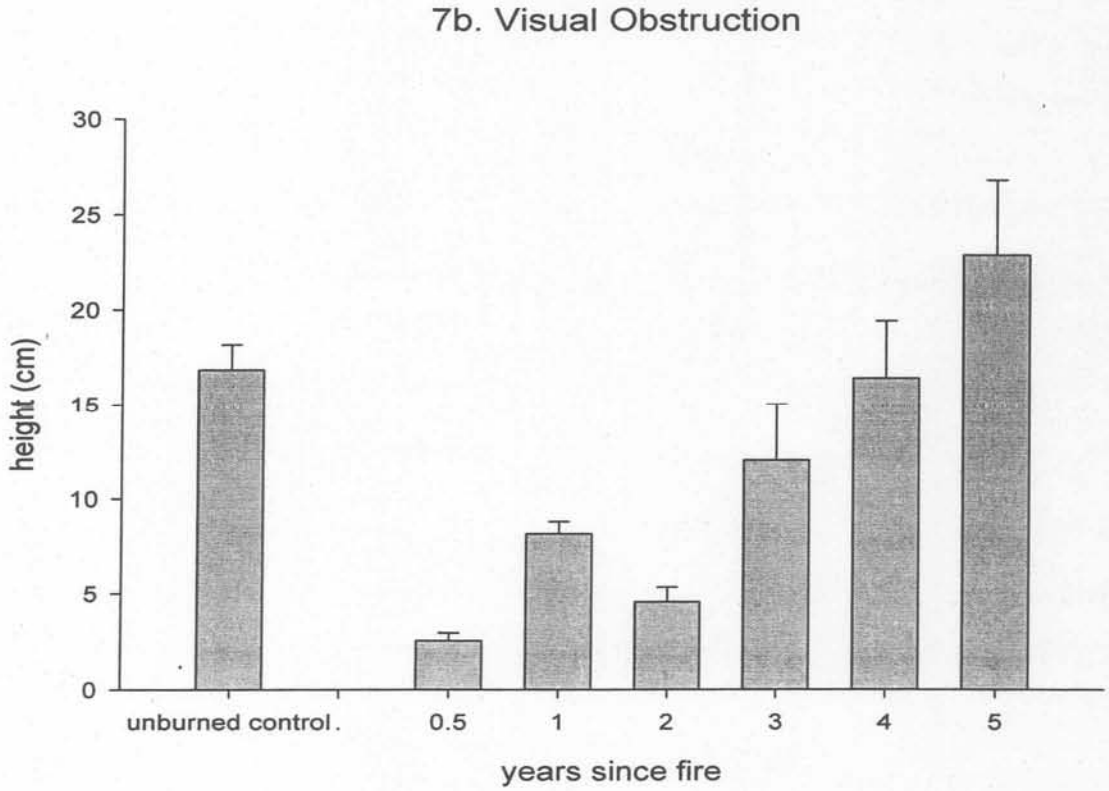
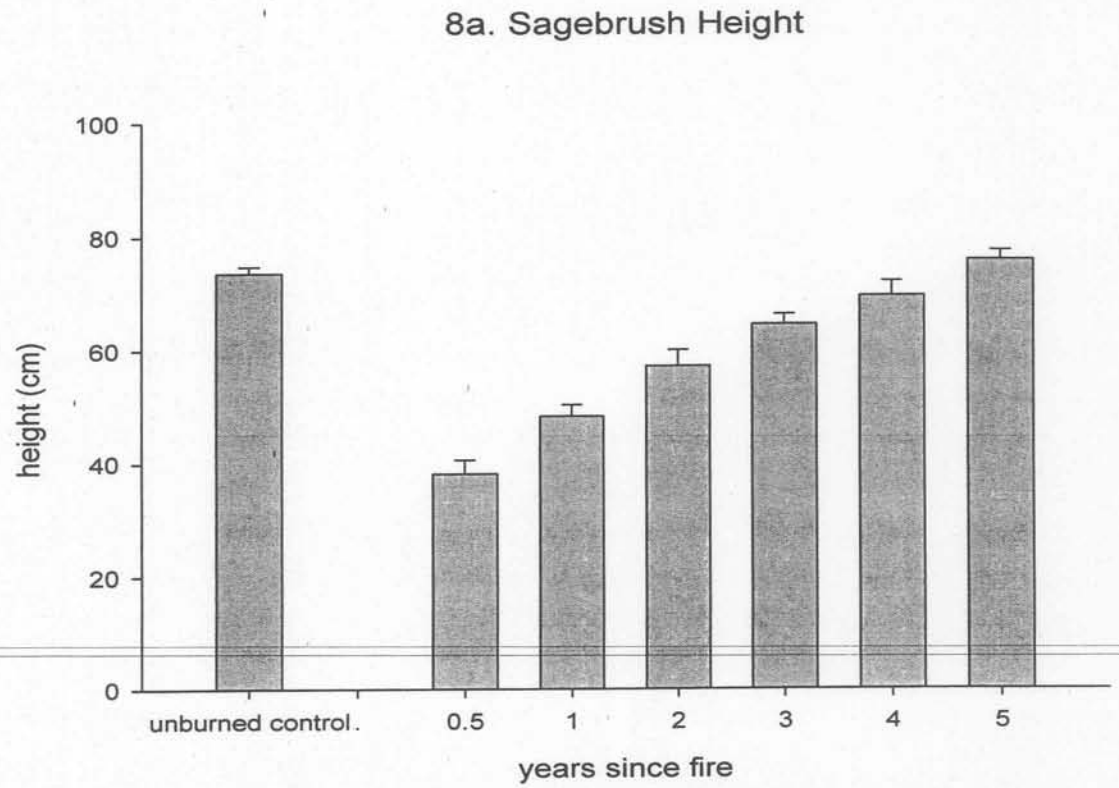


Figure 7. Mean ± SE vegetation height (7a) and visual obstruction (7b) in pastures managed by patch-burning at Cooper WMA.



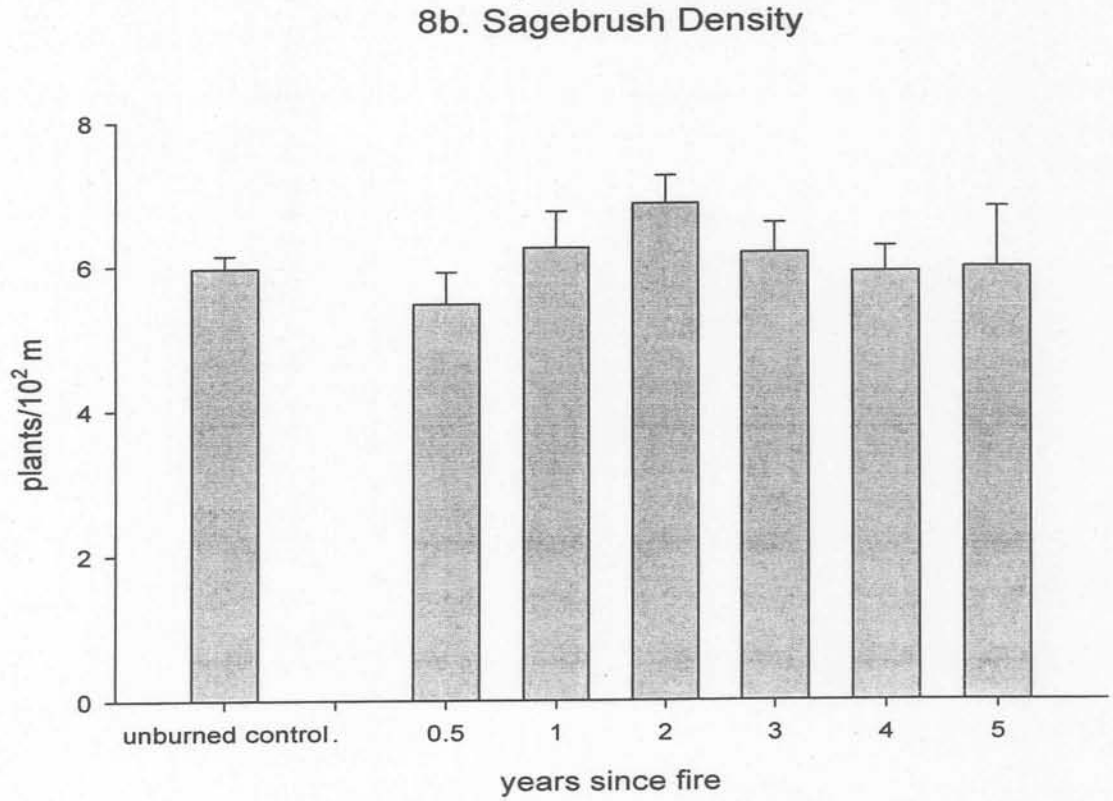


Figure 8. Mean ± SE sagebrush height (8a) and sagebrush density (8b) in pastures managed by patch-burning at Cooper WMA.

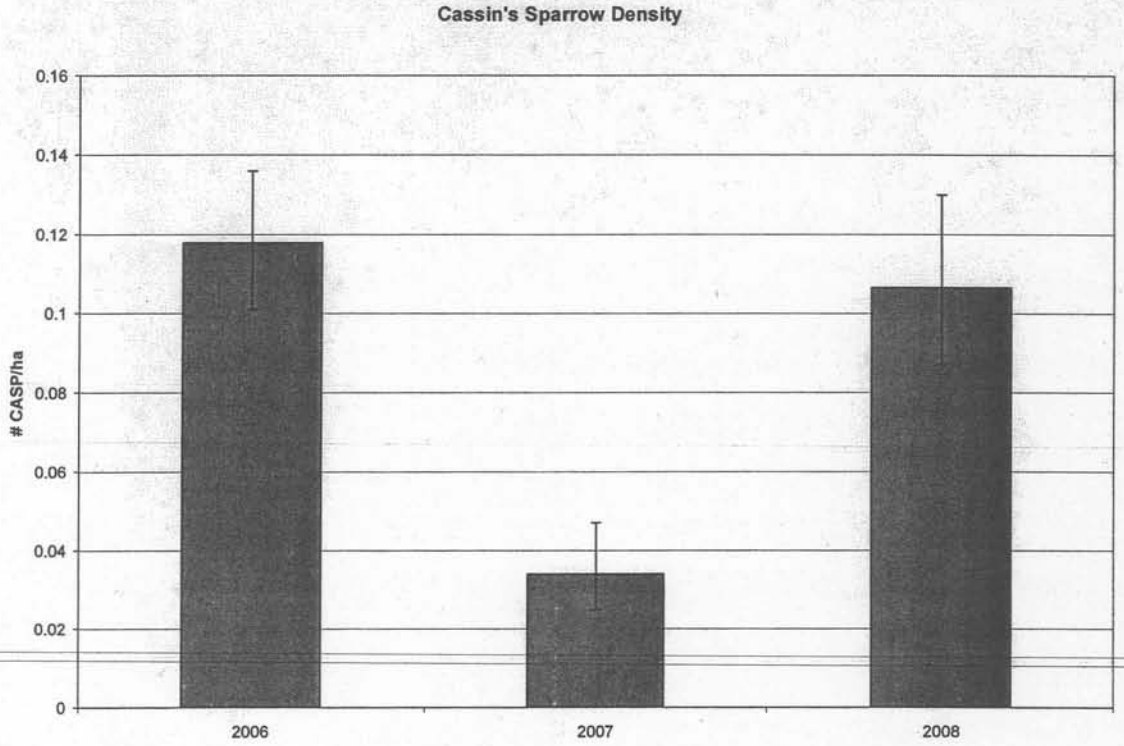


Figure 9. Cassin's sparrow densities (± 95% CI) at Cooper WMA 2006-2008.

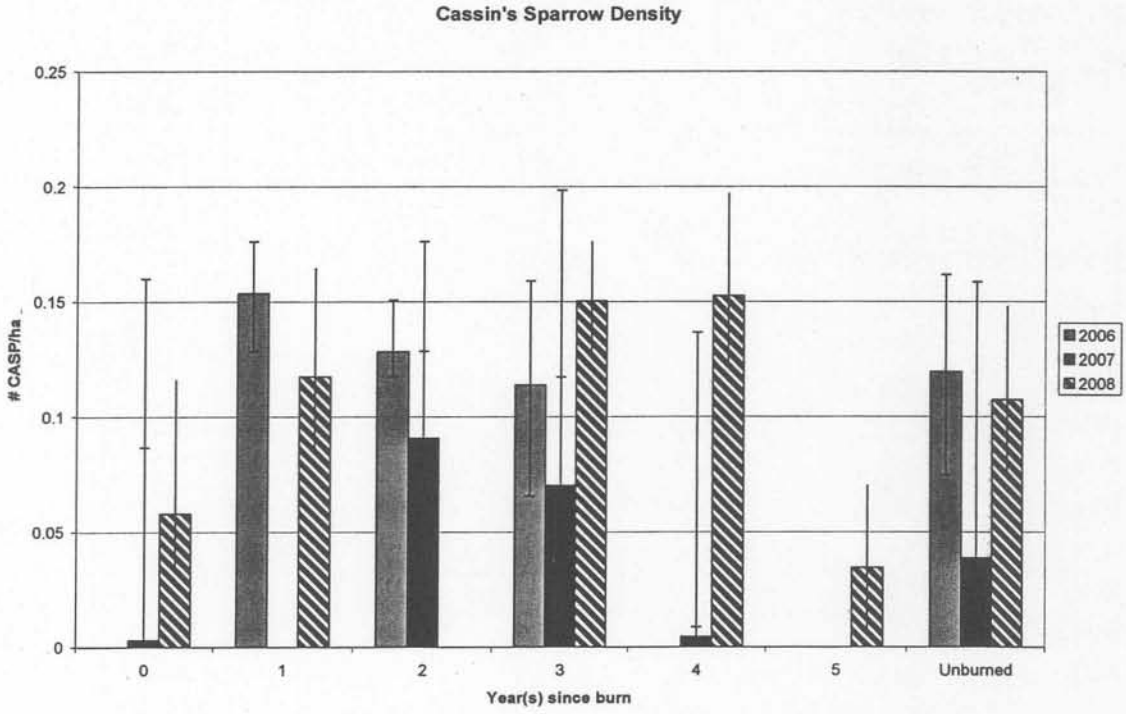


Figure 10. Cassin's sparrow densities (± 95% CI) in response years since burning in patch-burn managed pastures at Cooper WMA.

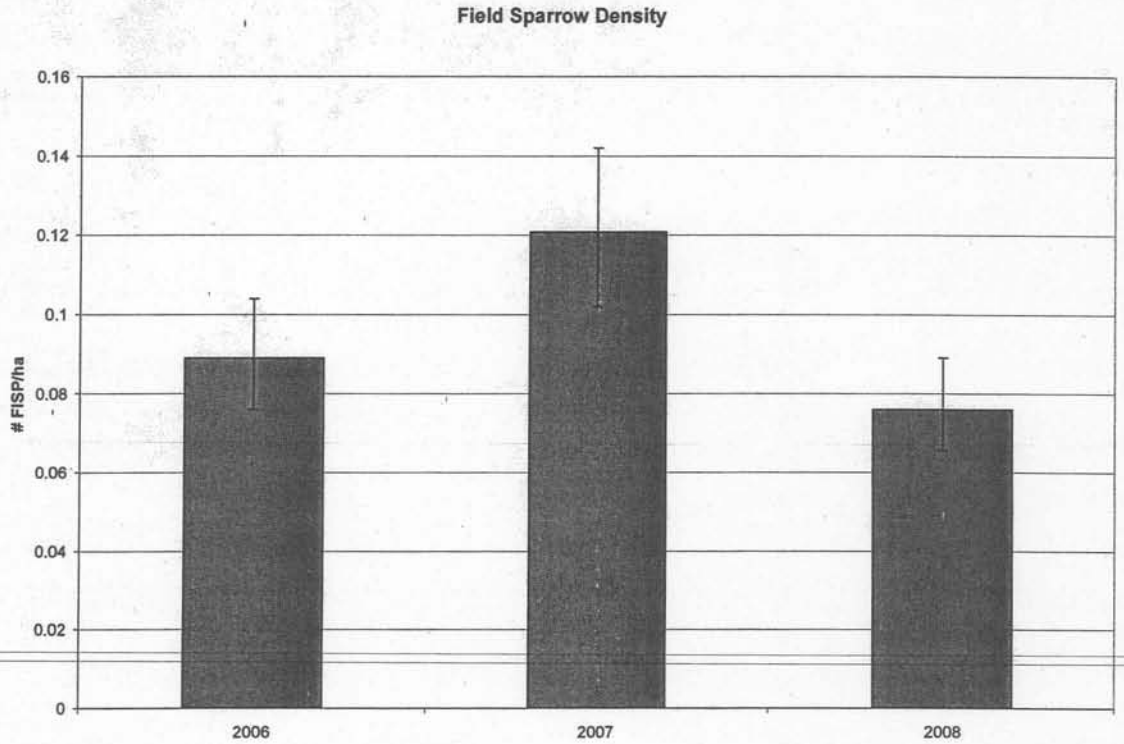


Figure 11. Field sparrow densities (± 95% CI) at Cooper WMA 2006-2008.

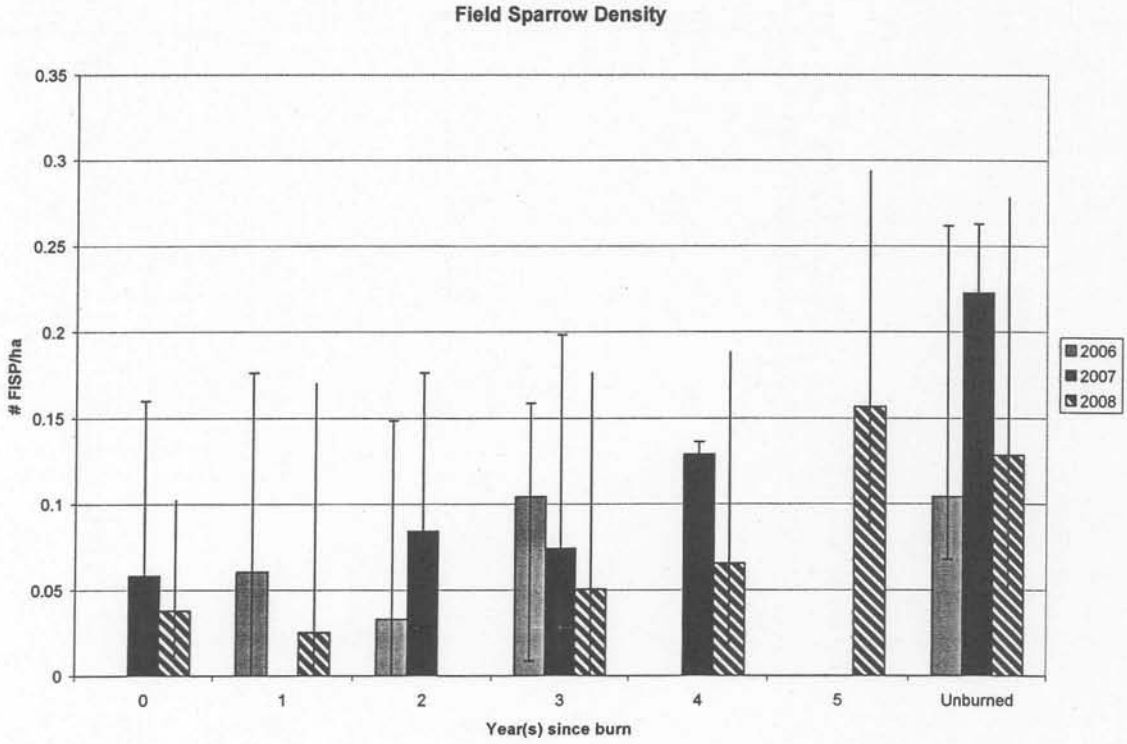


Figure 12. Field sparrow densities (± 95% CI) in response years since burning in patch-burn managed pastures at Cooper WMA.

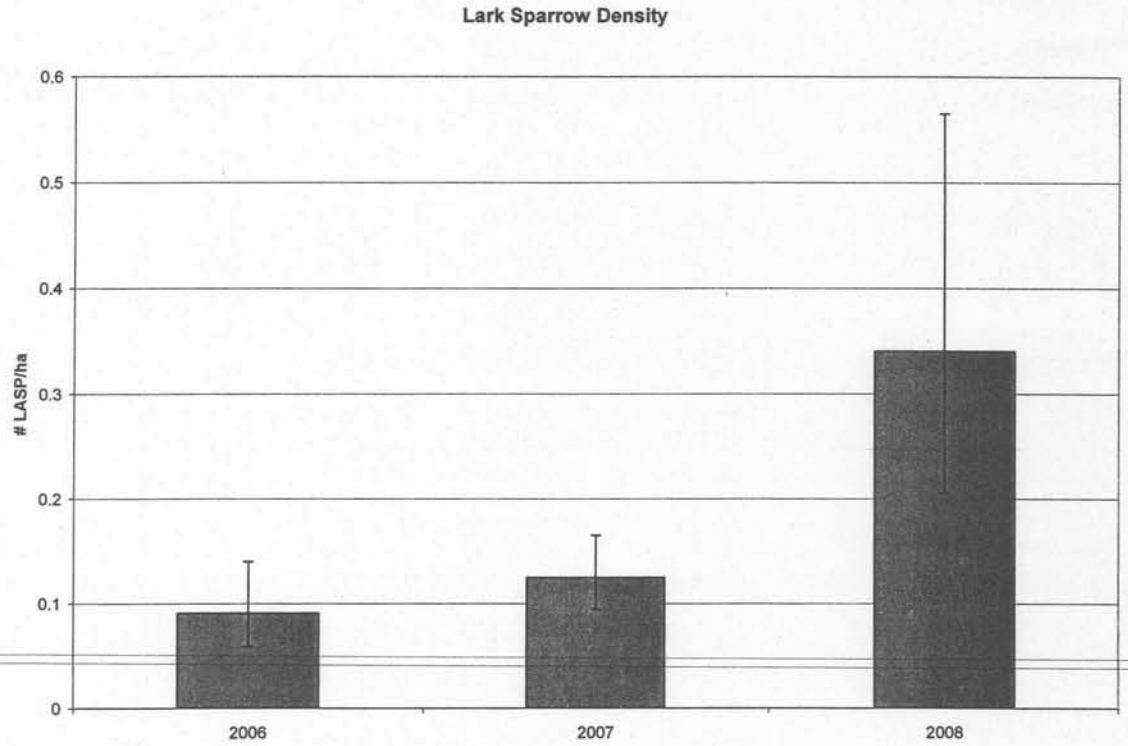


Figure 13. Lark sparrow densities (± 95% CI) at Cooper WMA 2006-2008.

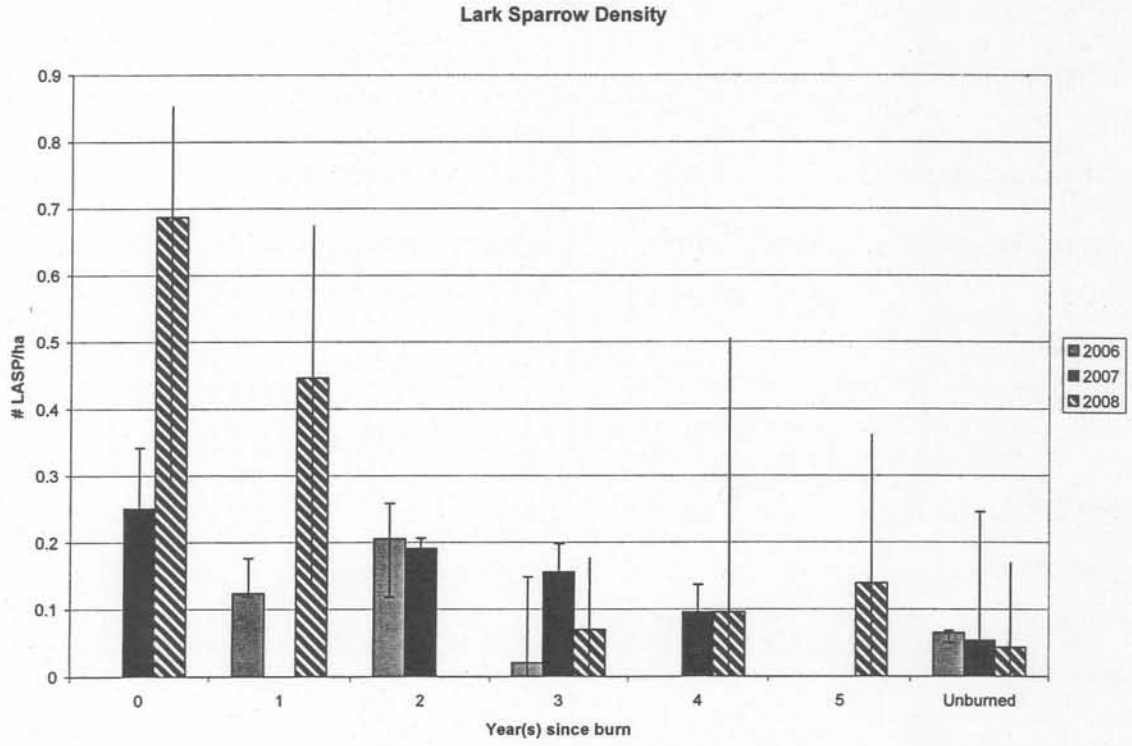


Figure 14. Lark sparrow densities (± 95% CI) in response years since burning in patch-burn managed pastures at Cooper WMA.

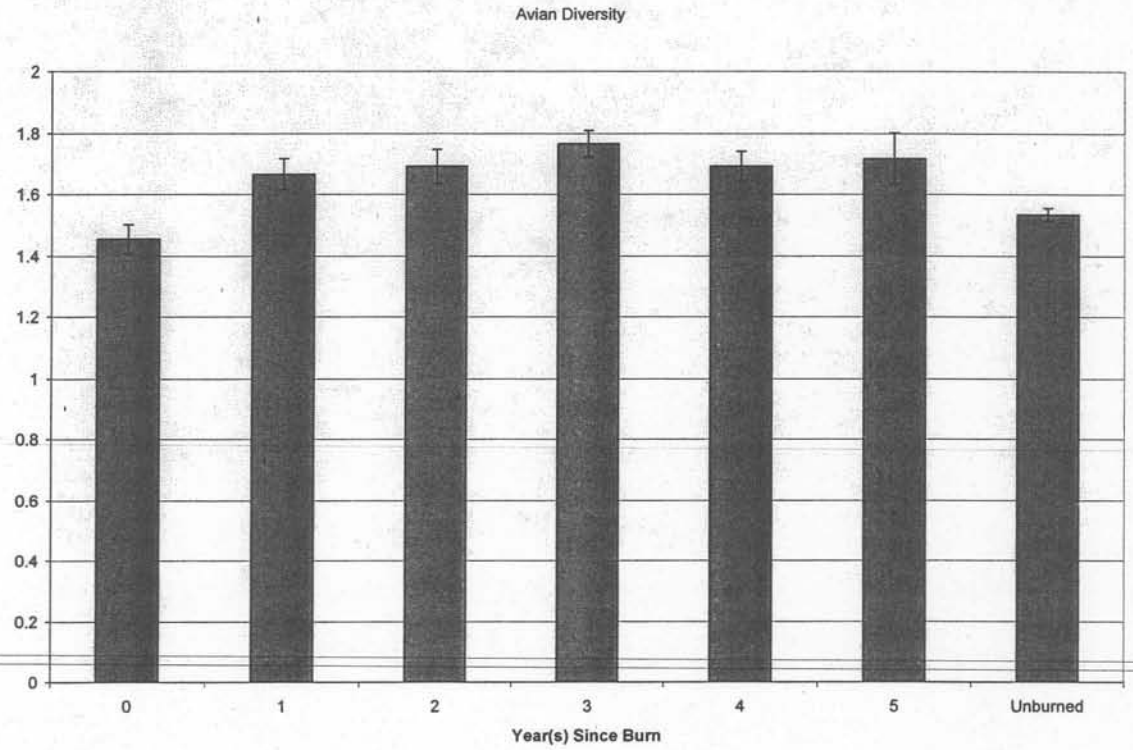


Figure 15. Avian diversity (± SE) response to years since burning in patch-burn managed pastures at Cooper WMA.

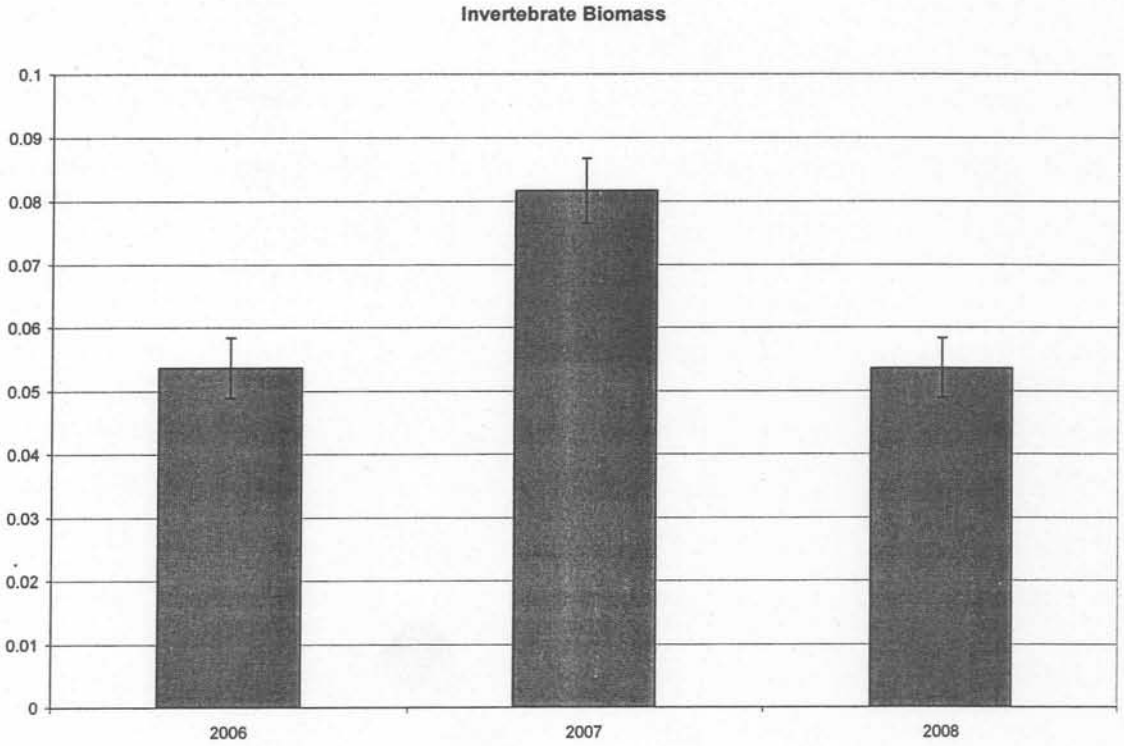


Figure 16. Invertebrate biomass (± SE) at Cooper WMA.

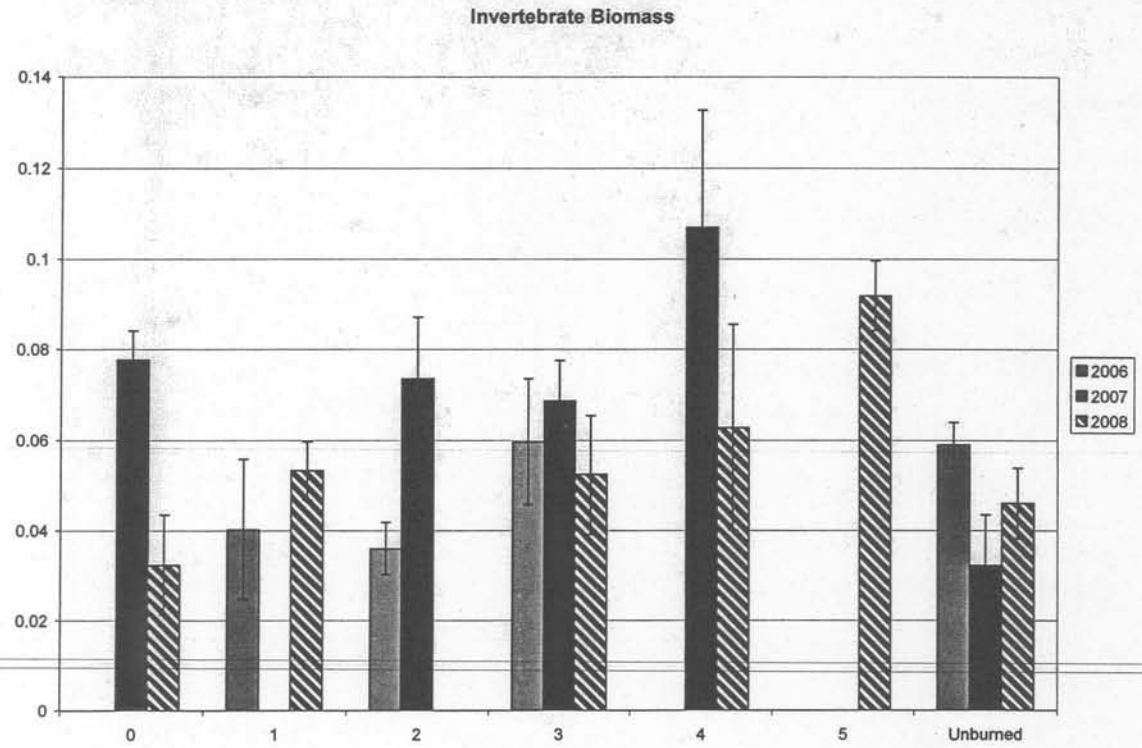


Figure 17. Invertebrate biomass (± SE) in response to years since burning in patch-burn managed pastures at Cooper WMA.

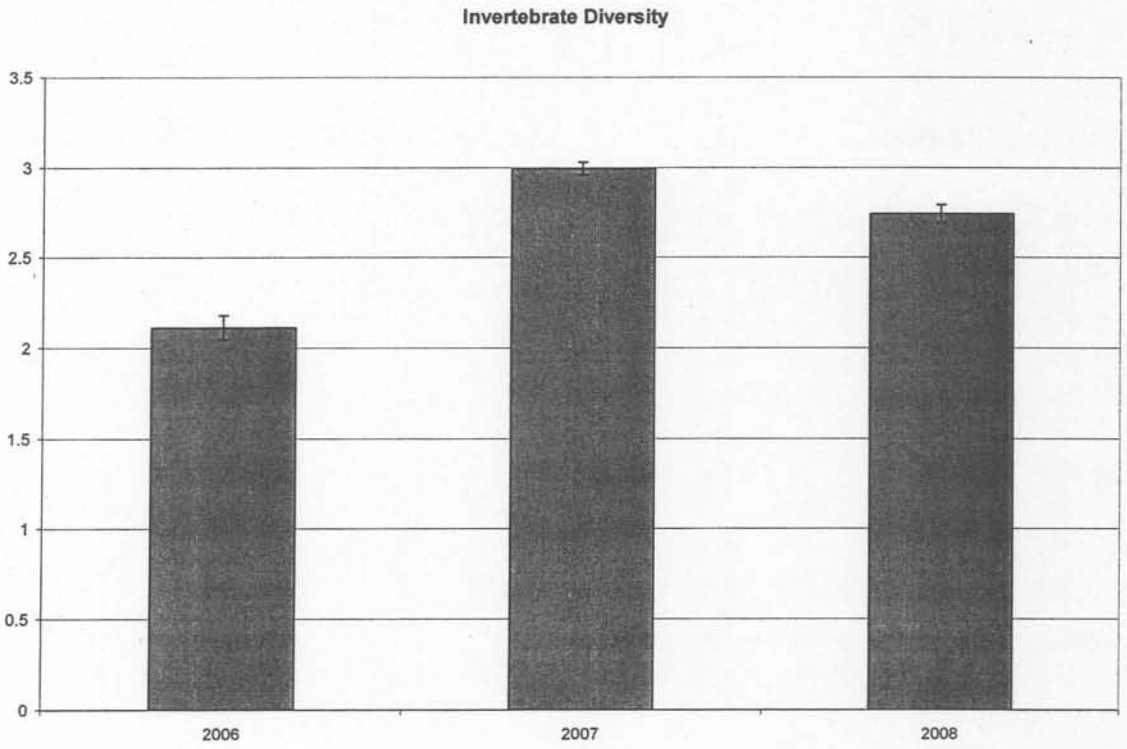


Figure 18. Invertebrate diversity (± SE) at Cooper WMA.

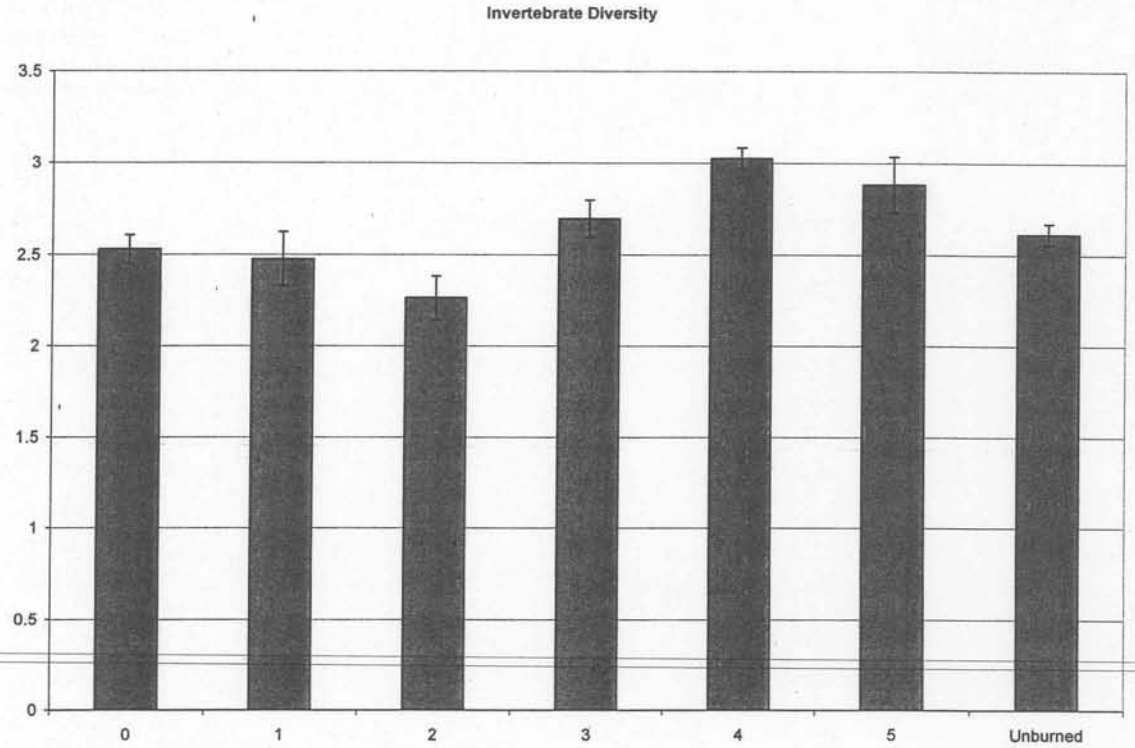


Figure 19. Overall invertebrate diversity (± SE) in response to years since burning in patch-burn managed pastures at Cooper WMA.

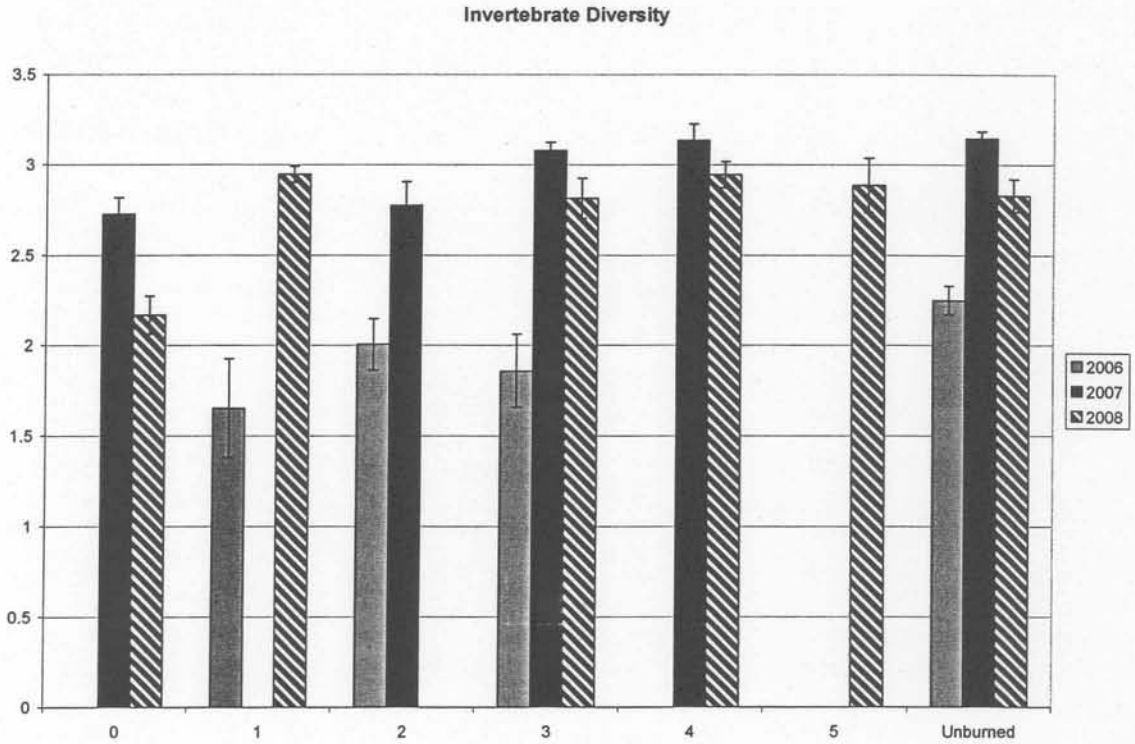


Figure 20. Invertebrate diversity (± SE) in response to years since burning during each year at Cooper WMA.

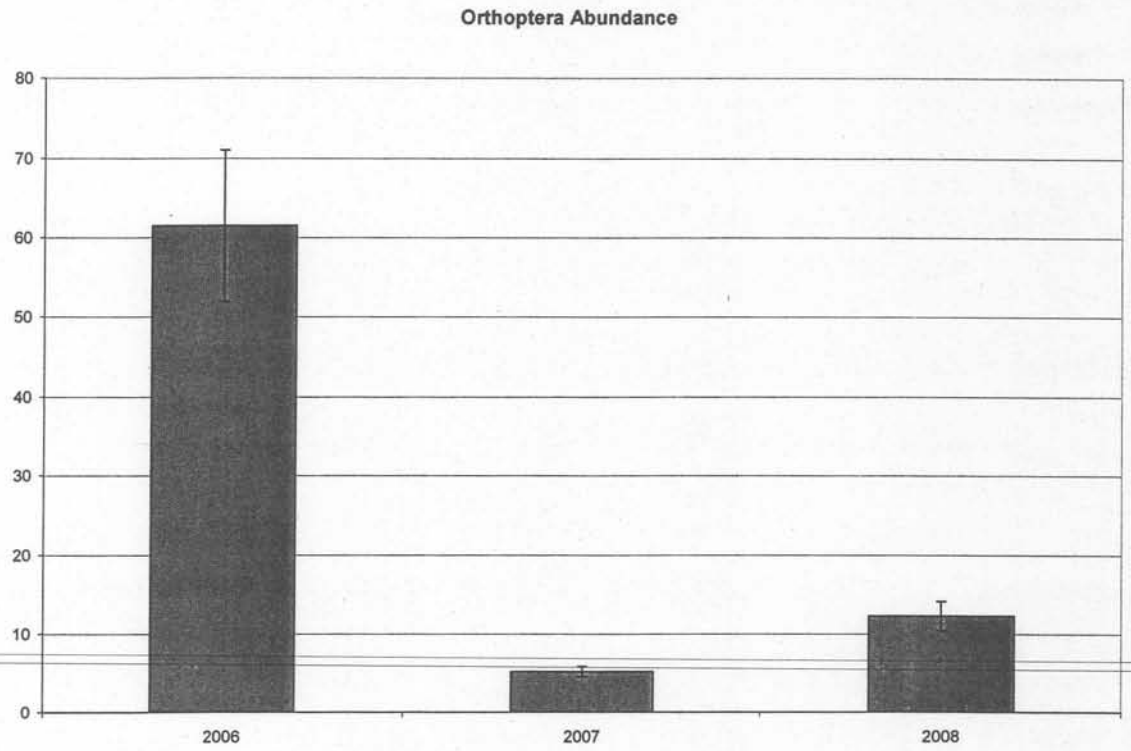


Figure 21. Orthoptera abundance (± SE) at Cooper WMA.

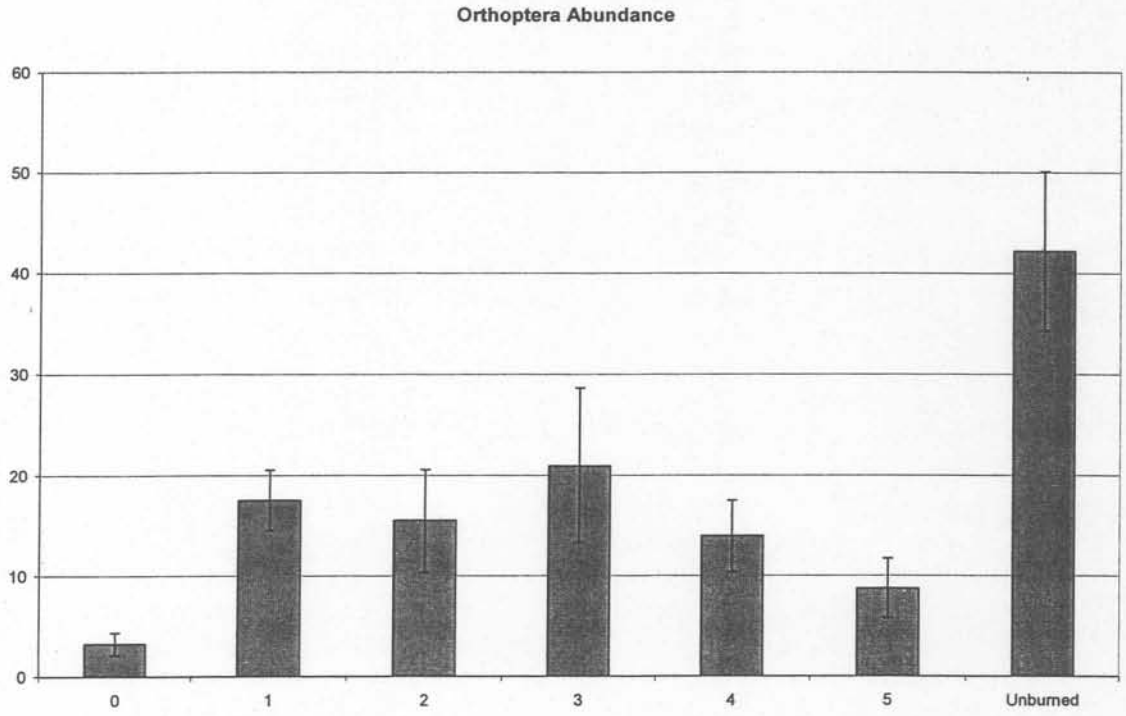


Figure 22. Overall Orthoptera abundance (± SE) in response to years since burn at

Cooper WMA.

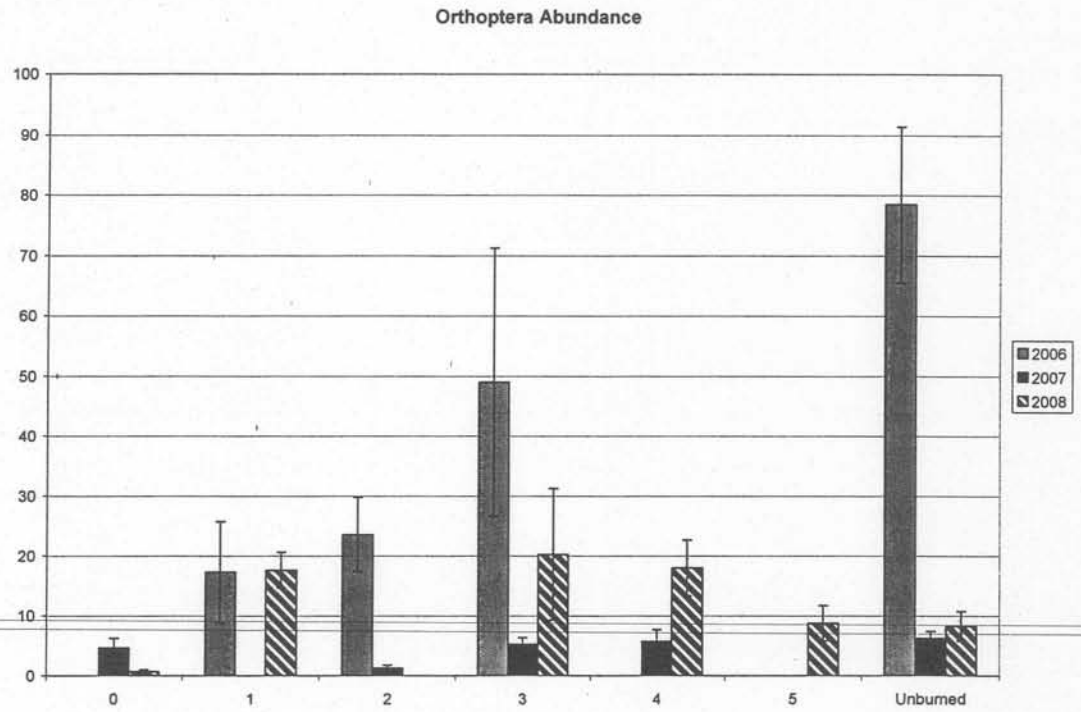


Figure 23. Orthoptera abundance (± SE) in response to years since burning during each year at Cooper WMA.

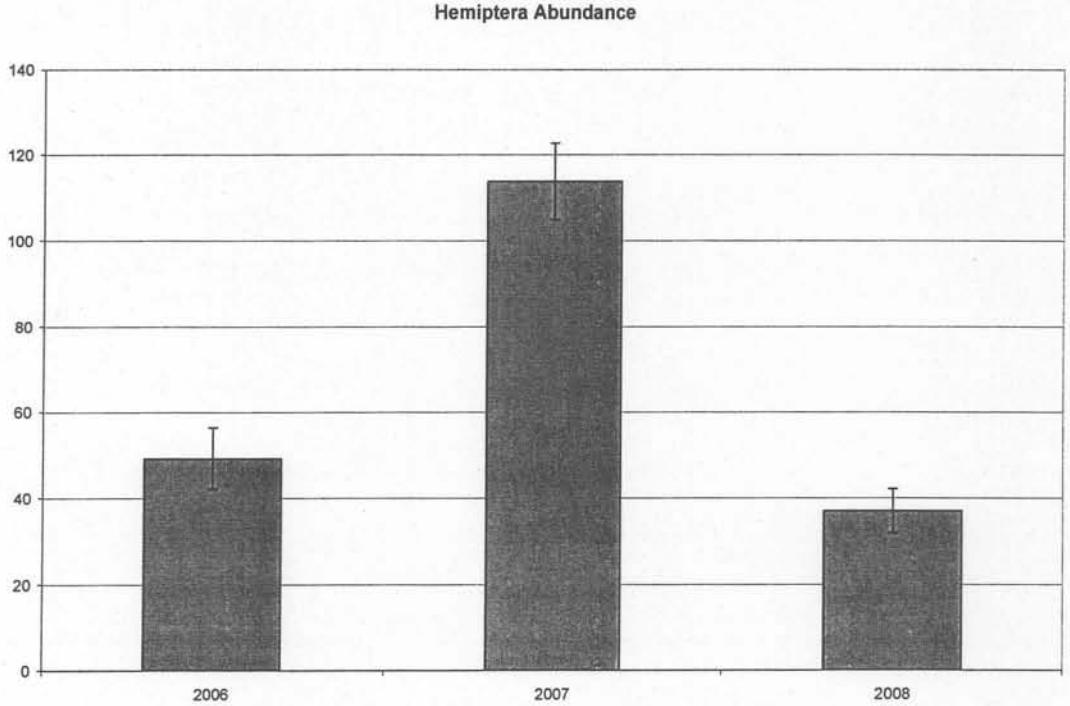


Figure 24. Hemiptera abundance (± SE) at Cooper WMA.

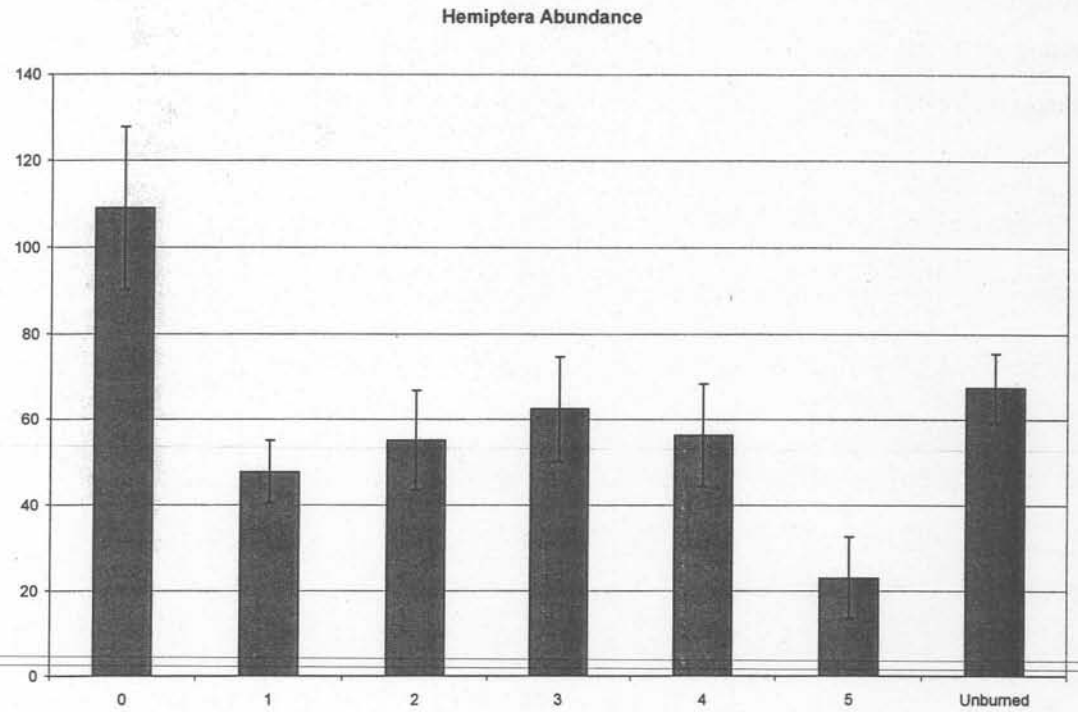


Figure 25. Overall Hemiptera abundance (± SE) in response to years since burn at Cooper WMA.

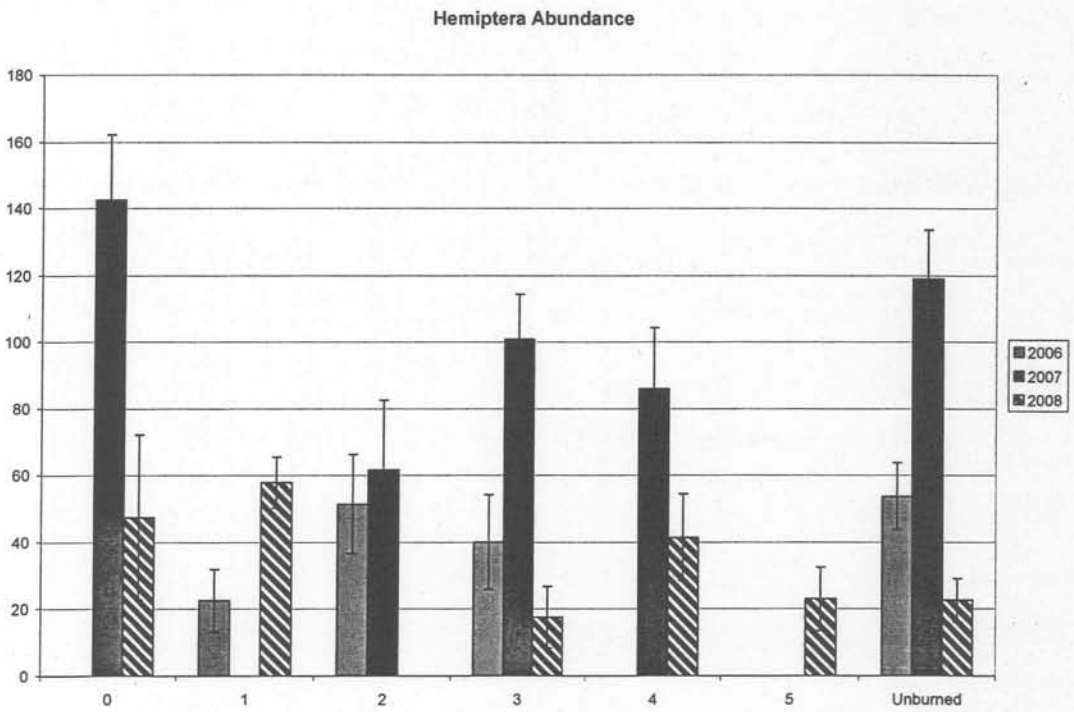


Figure 26. Hemiptera abundance (± SE) in response to years since burning during each year at Cooper WMA.

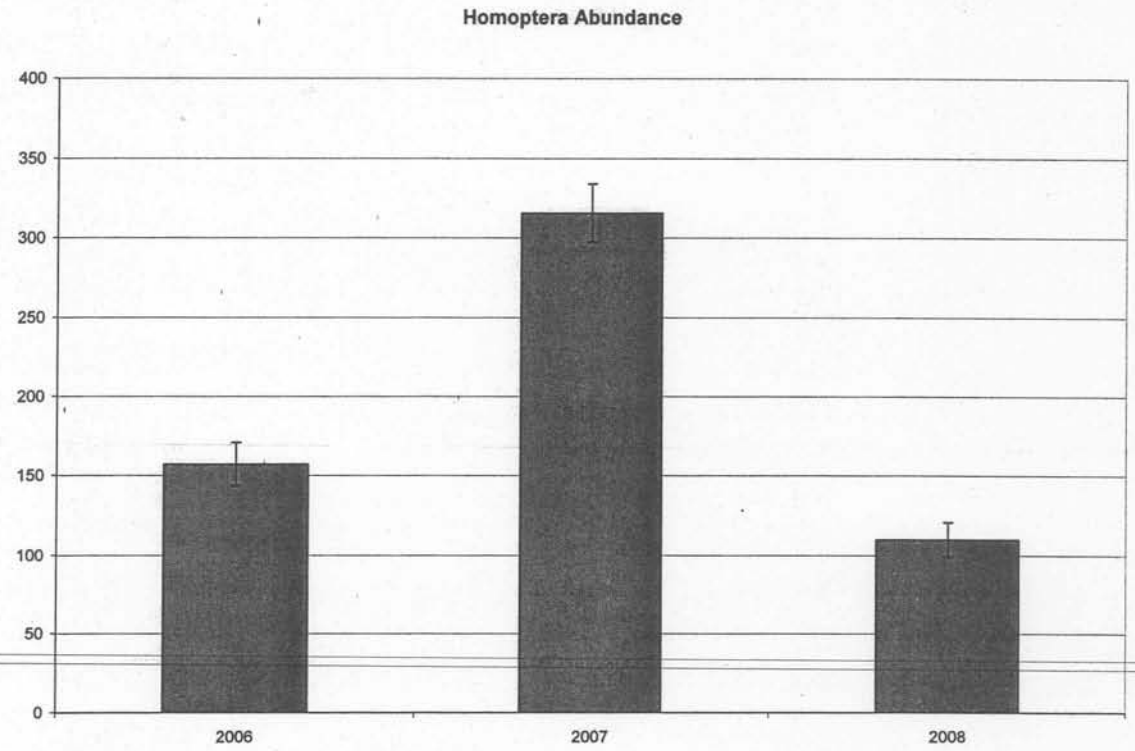


Figure 27. Homoptera abundance (± SE) at Cooper WMA.

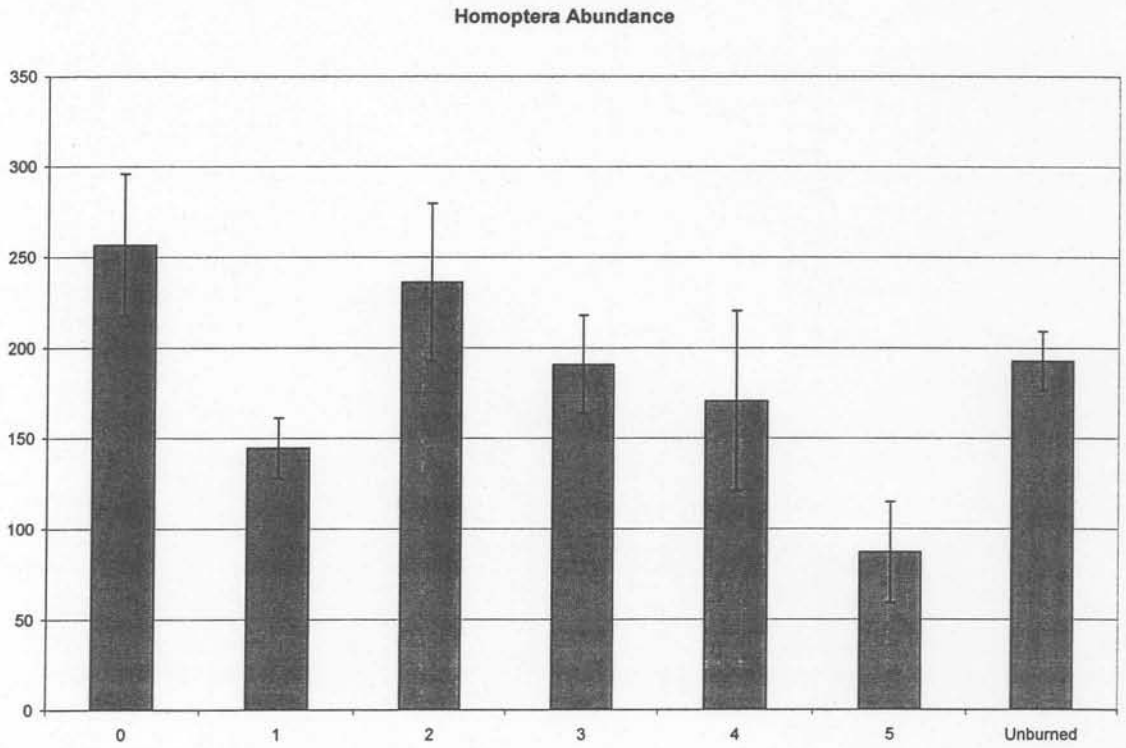


Figure 28. Overall Homoptera abundance (± SE) in response to years since burn at Cooper WMA.

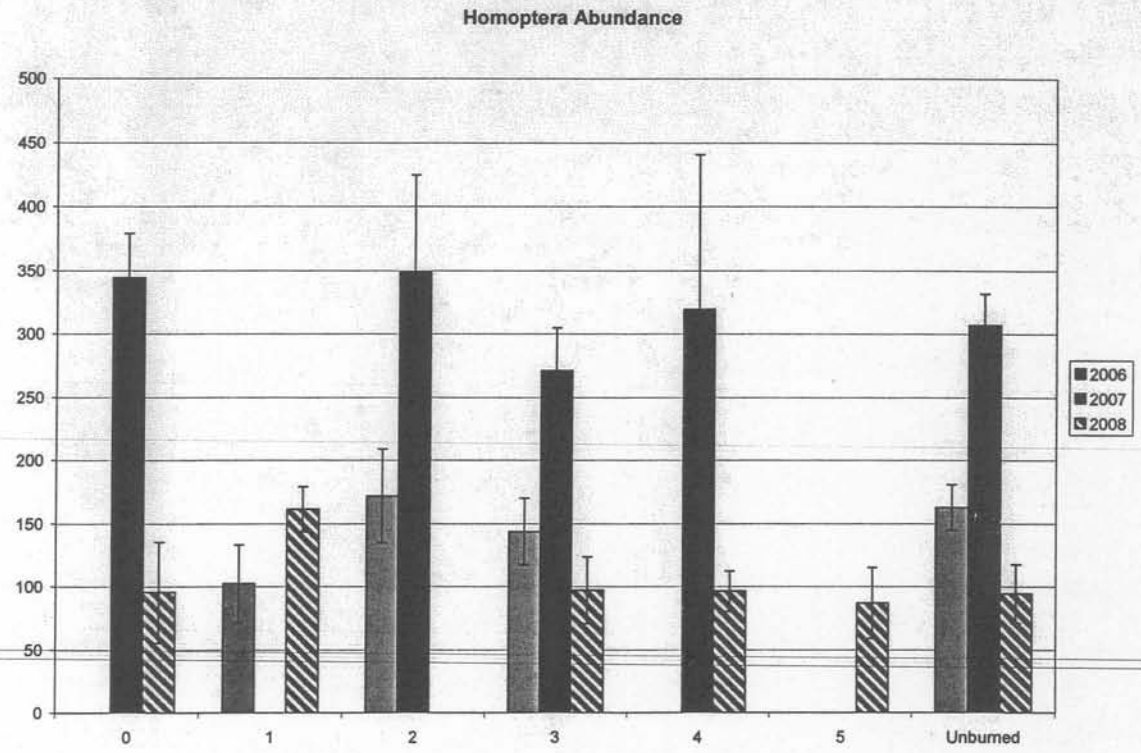


Figure 29. Homoptera abundance (± SE) in response to years since burning during each year at Cooper WMA.

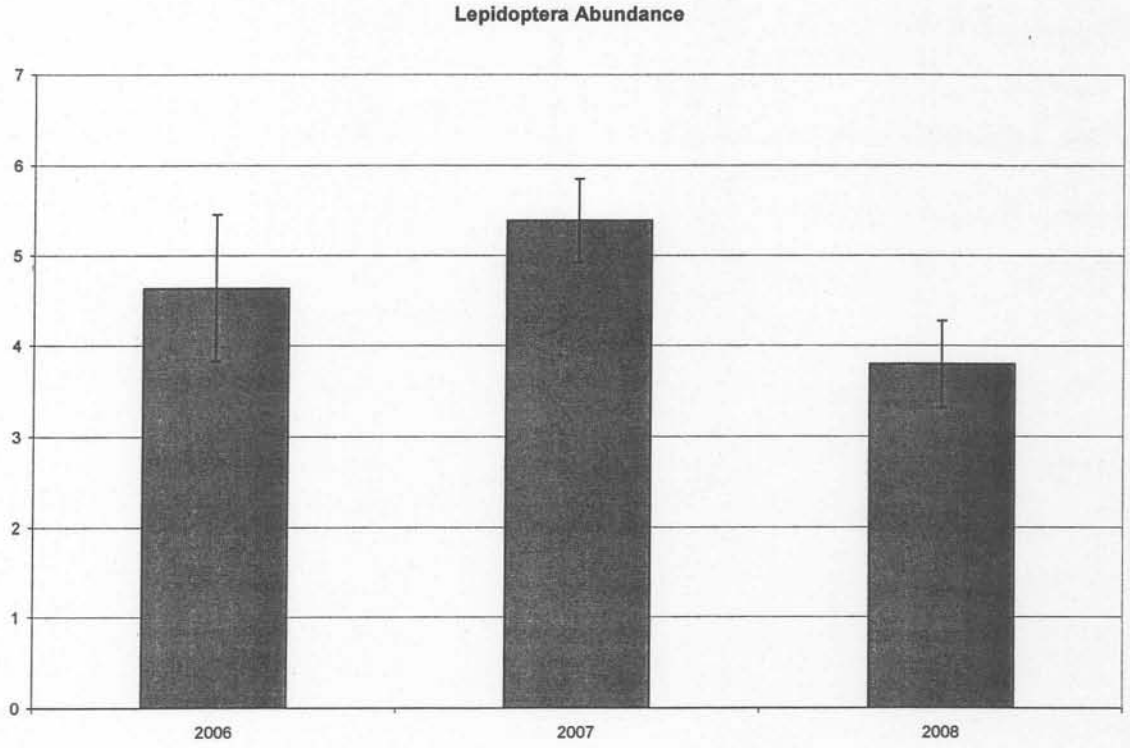


Figure 30. Lepidoptera abundance (± SE) at Cooper WMA.

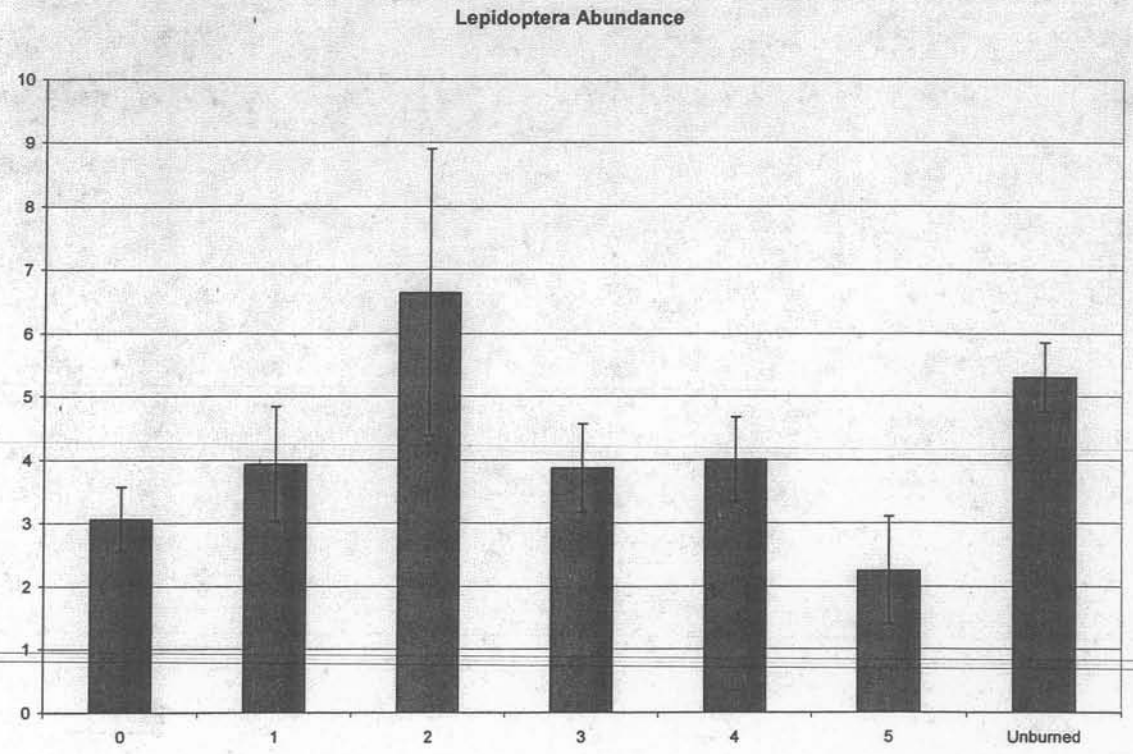


Figure 31. Overall Lepidoptera abundance (± SE) in response to years since burn at Cooper WMA.

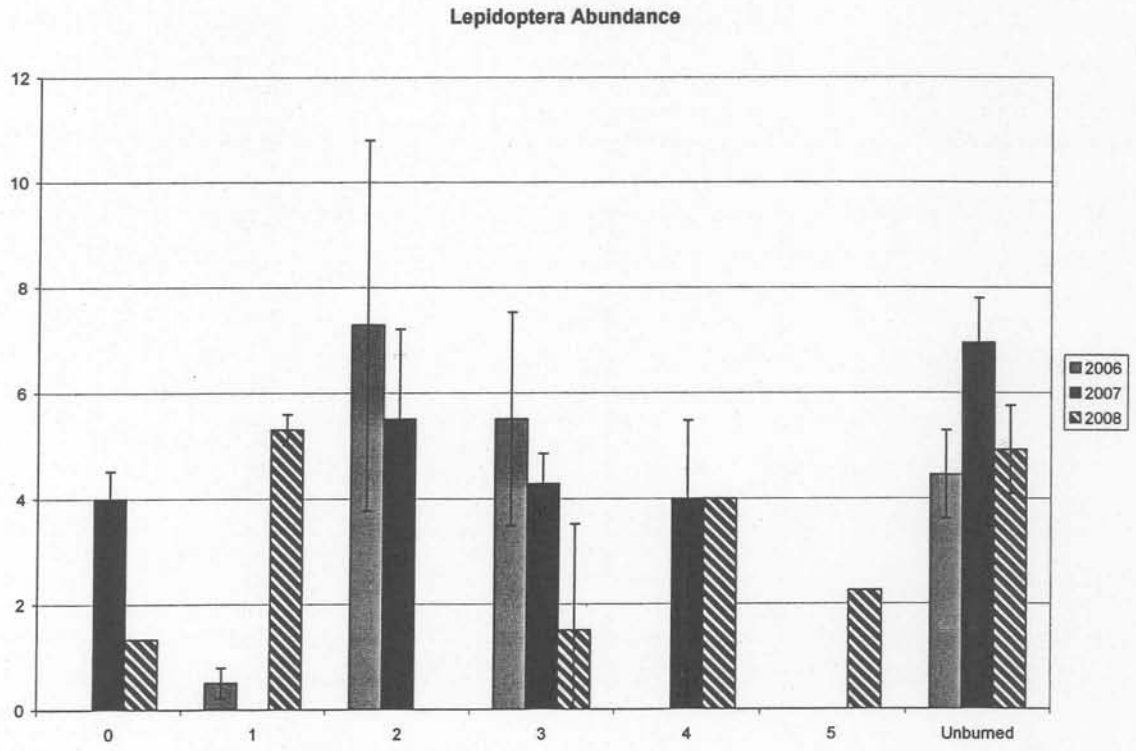


Figure 32. Lepidoptera abundance (± SE) in response to years since burning during each year at Cooper WMA.

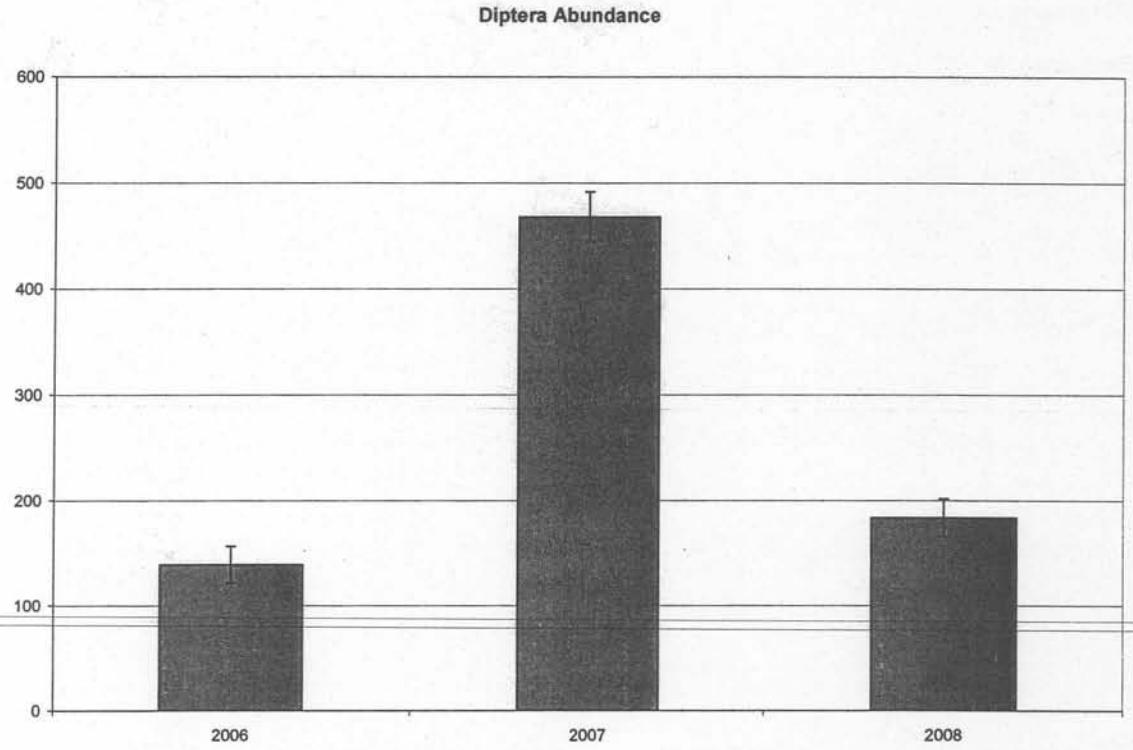


Figure 33. Diptera abundance (± SE) at Cooper WMA.

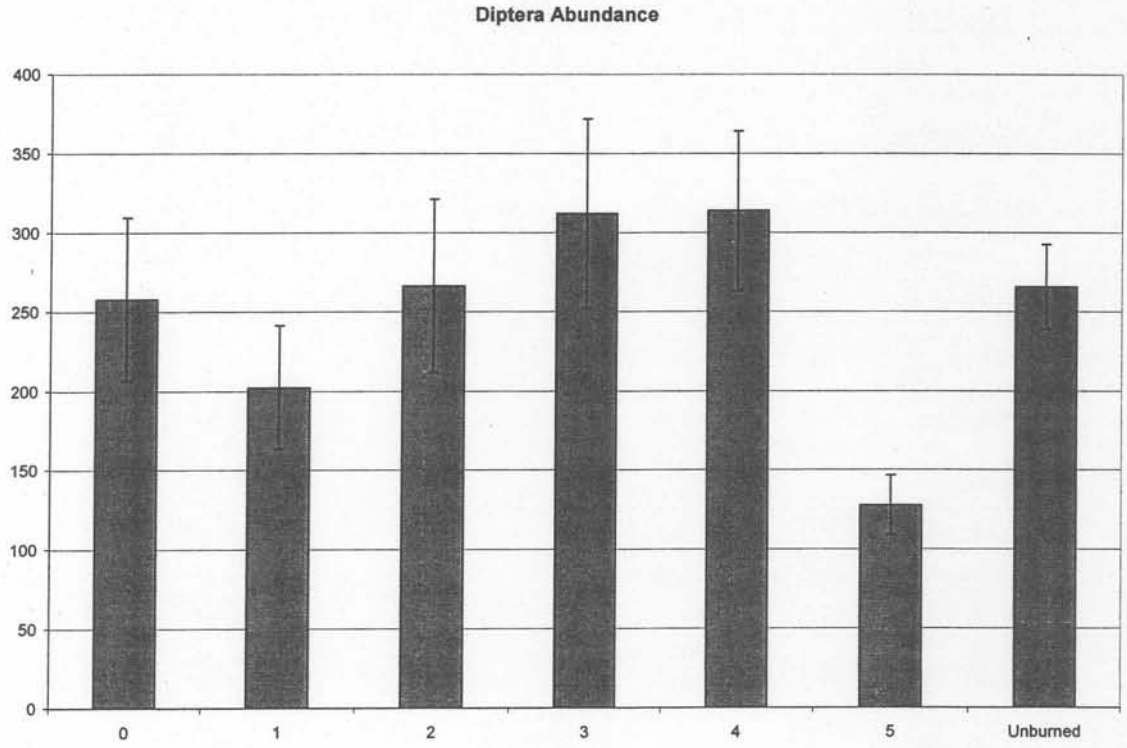


Figure 34. Overall Diptera abundance (± SE) in response to years since burn at Cooper WMA.

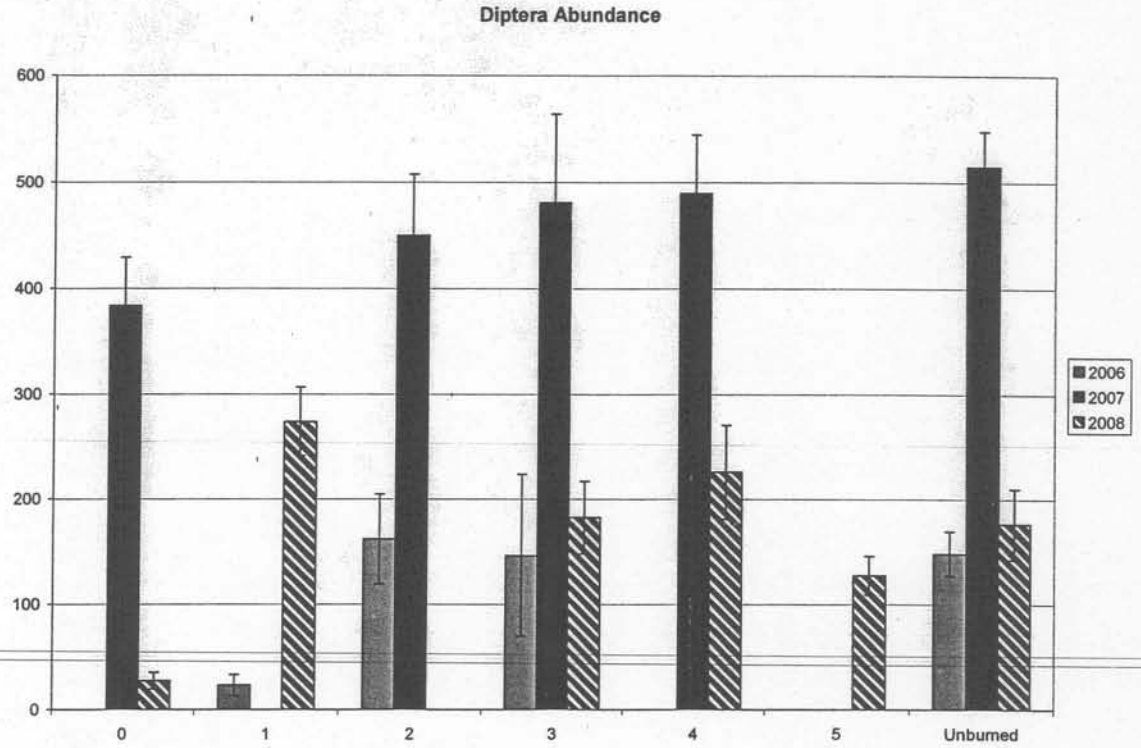


Figure 35. Diptera abundance (± SE) in response to years since burning during each year at Cooper WMA.

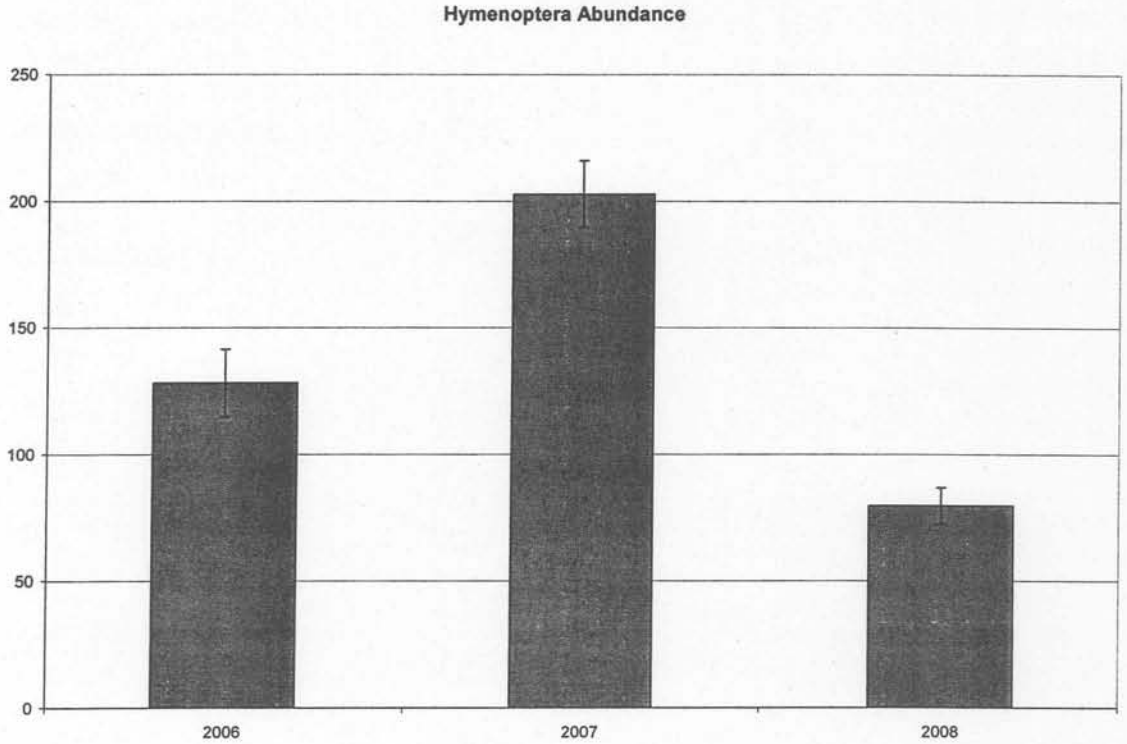


Figure 36. Hymenoptera abundance (± SE) Cooper WMA.

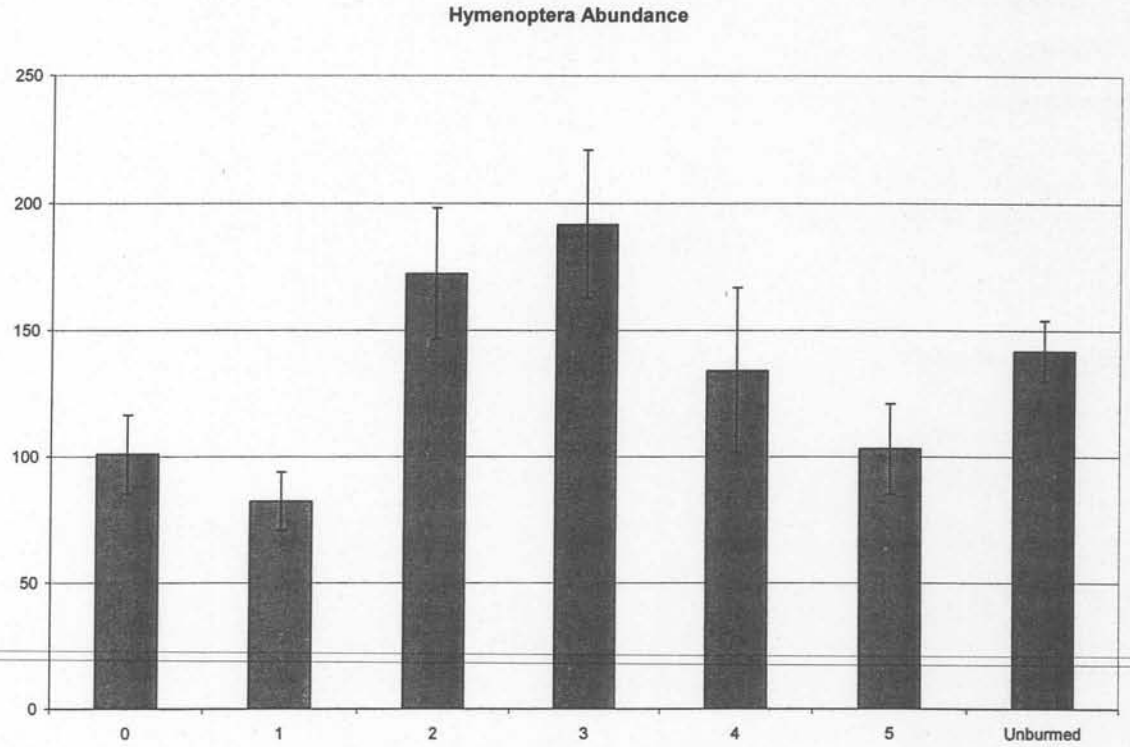


Figure 37. Overall Hymenoptera abundance (± SE) in response to years since burn at Cooper WMA.

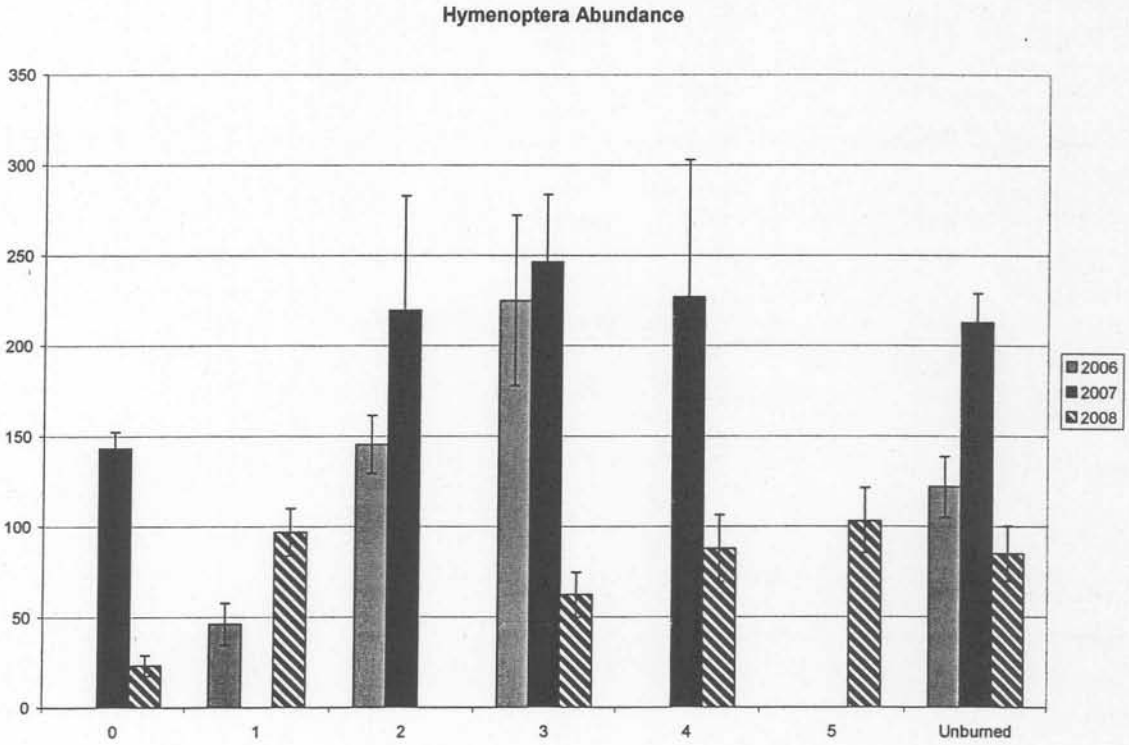


Figure 38. Hymenoptera abundance (± SE) in response to years since burning during each year at Cooper WMA.

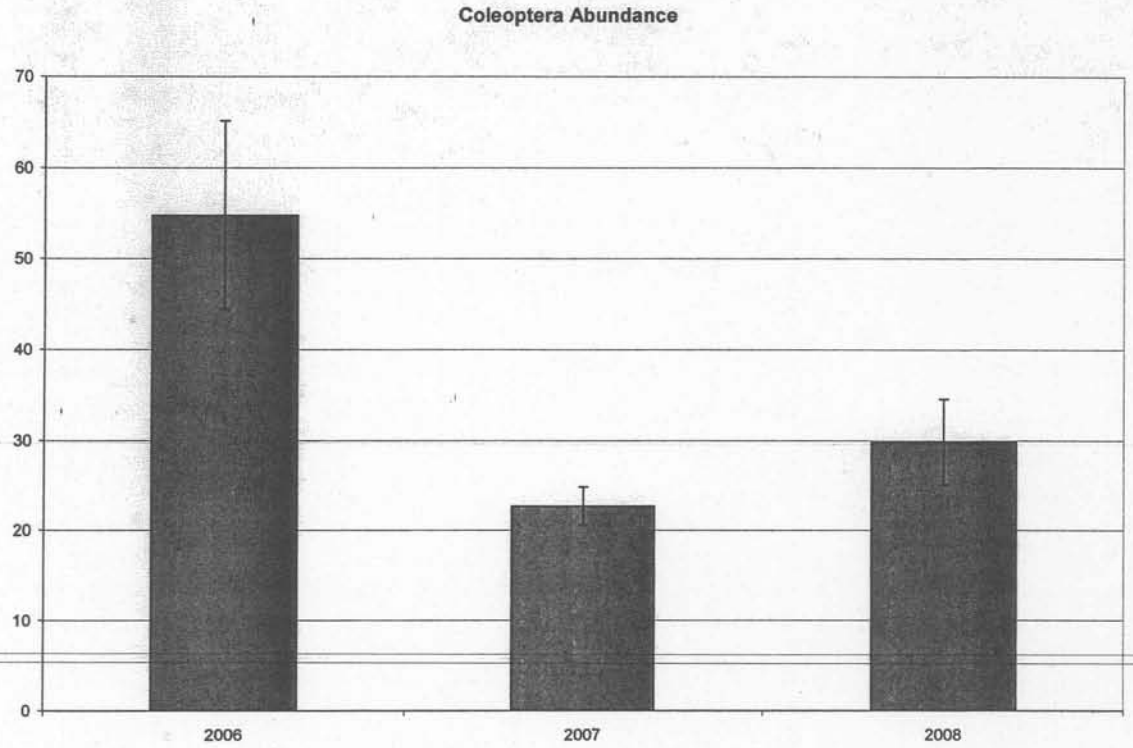


Figure 39. Coleoptera abundance (± SE) at Cooper WMA.

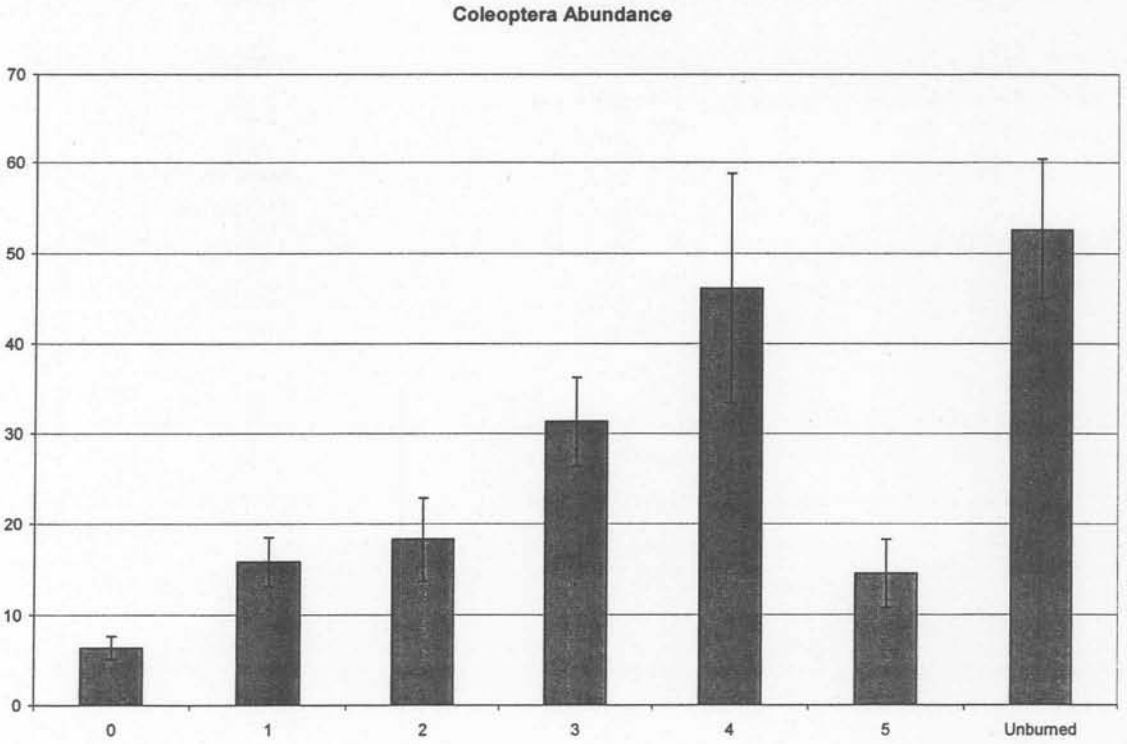


Figure 40. Overall Coleoptera abundance (± SE) in response to years since burn at Cooper WMA.

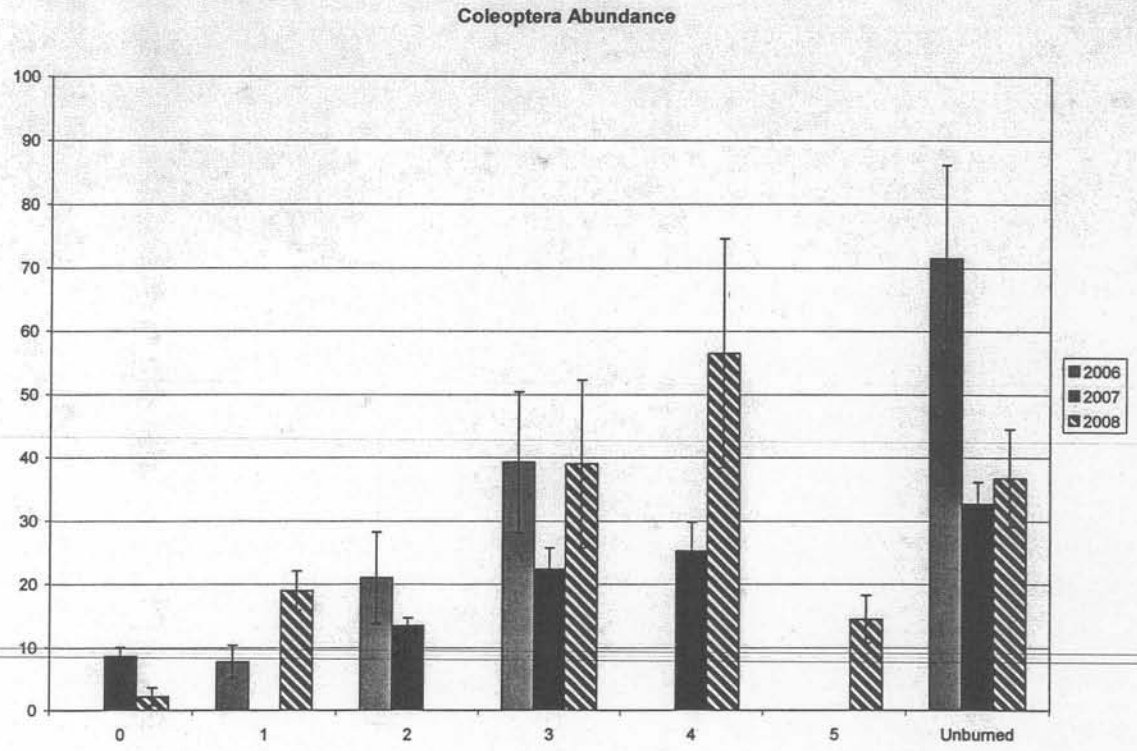


Figure 41. Coleoptera abundance (± SE) in response to years since burning during each year at Cooper WMA.

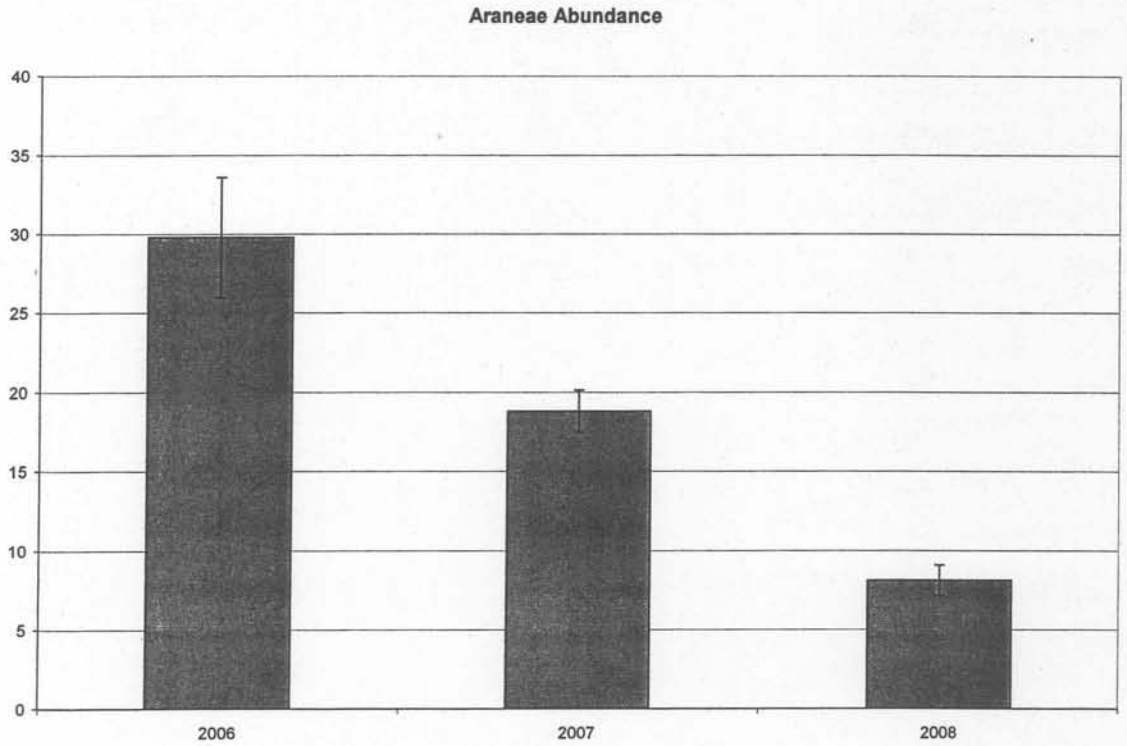


Figure 42. Araneae abundance (± SE) at Cooper WMA.

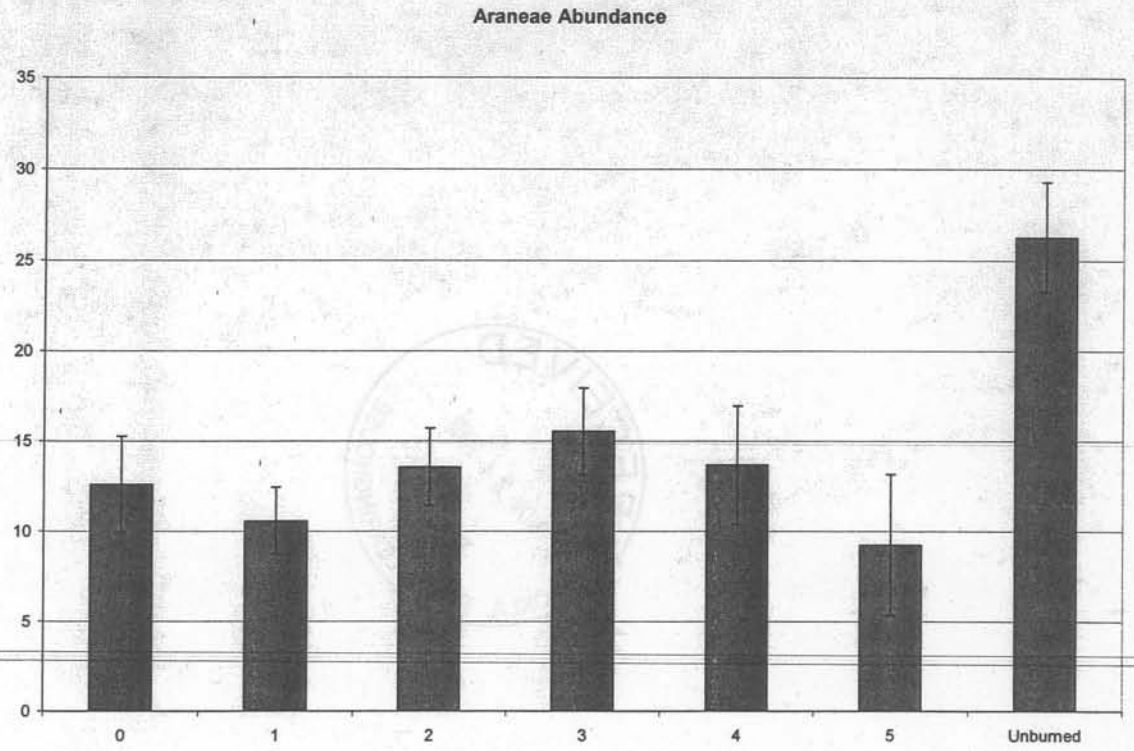


Figure 43. Overall Araneae abundance (± SE) in response to years since burn at Cooper WMA.

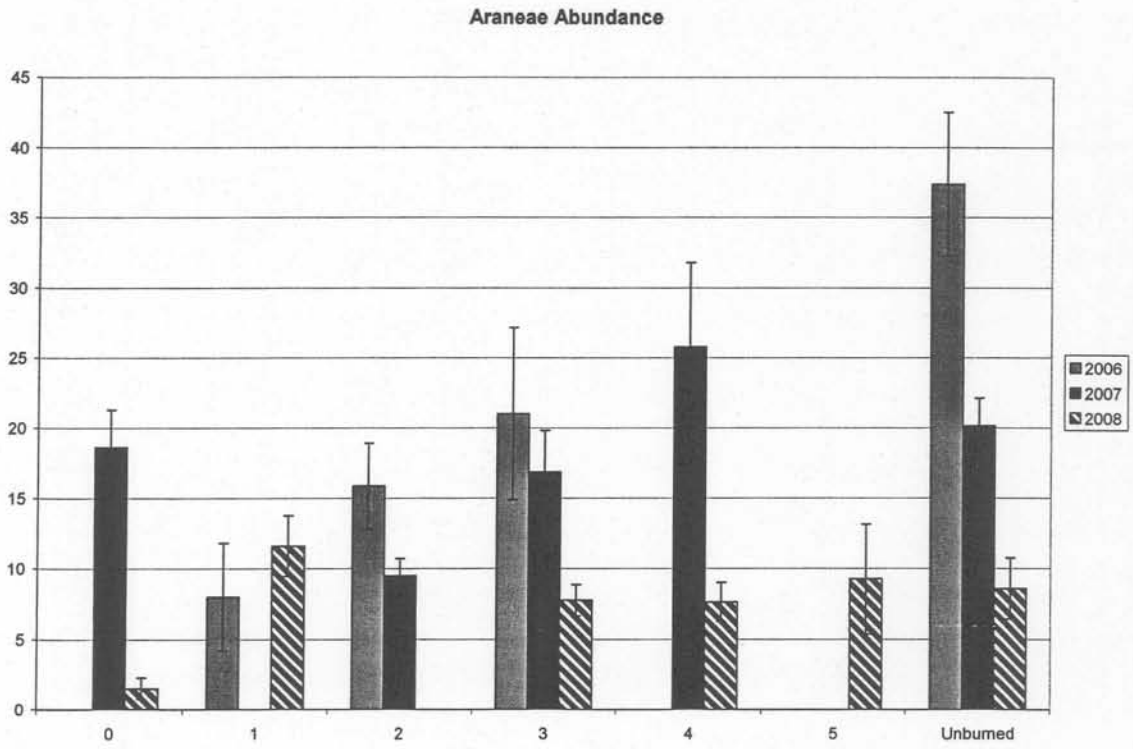


Figure 44. Araneae abundance (± SE) in response to years since burning during each year at Cooper WMA.

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