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TRENDS IN GRASSLAND BIRD ABUNDANCE FOLLOWING PRESCRIBED
BURNING IN SOUTHERN ARIZONA

by

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ABSTRACT

I examined trends in relative abundance and species richness of breeding and wintering grassland birds before (1996) and after (1997, 1998) a spring prescribed burn in a mesquite-dominated desert grassland at Buenos Aires National Wildlife Refuge, Arizona. The burn was moderate in intensity, patchy in extent, and affected ground cover more strongly than shrub cover, smaller shrubs more strongly than larger shrubs, and killed 1% of velvet mesquite (*Prosopis velutina*). Species richness of breeding birds decreased in the first year post-burn. Of breeding species, black-throated sparrows (*Amphispiza bilineata*) and mourning doves (*Zenaida macroura*) increased; whereas Botteri's sparrows (*Aimophila botterii*), Cassin's sparrows (*Aimophila cassinii*), and pyrrhuloxias (*Cardinalus sinuatus*) decreased in relative abundance. Breeding species characterized as not shrub-dependent exhibited changes that were more pronounced than those for shrub-dependent species. Of wintering birds, ladder-backed woodpeckers (*Picoides scalaris*) and vesper sparrows (*Pooecetes gramineus*) increased, and cactus wrens (*Campylorhynchus brunneicapillus*) decreased in relative abundance.

INTRODUCTION

Populations of grassland birds have declined at a rate greater than any other avian guild in North America over the last quarter century (Knopf 1995, Sauer et al. 1995), and the destruction of native grasslands has been implicated as a major cause in the decline (Johnson and Schwartz 1993, DeSante and George 1994). Conservation efforts to recover declining populations of grassland birds have focused on restoration and maintenance of remnant grasslands. Prescribed burning has become an important tool in meeting these management objectives (Herkert 1994, Johnson and Miyanishi 1995). Despite the recent use of prescribed burning, relatively little is known about the efficacy and potential impacts of this management technique on grassland bird communities (Herkert 1994).

In the southwestern United States, native desert grasslands have been substantially degraded during the last century as a result of overgrazing, drought, invasion of exotic species, and fire suppression (Bahre 1995). Before the arrival of Anglo settlers, fire was a regular and integral component of the desert grassland ecosystem (Humphrey 1974, McPherson 1995). Historical accounts and indirect evidence suggest that before the 1880s fires were extensive and occurred at least every 10 years (McPherson 1995). The interruption of the natural fire regime contributed to pronounced structural, compositional, and functional changes to native desert grasslands (Bahre 1991). No change has been more apparent than the dramatic increase of shrubs, primarily mesquite (*Prosopis* spp.), in upland areas that were formerly dominated by native grasses

(Humphrey 1974, Brown 1994).

Shrub-dominated grasslands currently support a very different avian community than was present historically (Lloyd et al. 1998). Encroachment of shrubs seems to have facilitated the movement of shrub-dependent bird species into areas that were formerly open grasslands (Lloyd 1997). As a result, grassland obligate species are now less common and shrub-dependent species dominate the composition and structure of these avian communities (Maurer 1985, Lloyd et al. 1998).

The Buenos Aires National Wildlife Refuge, located in the Altar Valley of southern Arizona, was established in 1985 and protects the largest ungrazed grassland in Arizona. The principal goal of the Refuge is to provide habitat for the reintroduction of the endangered masked bobwhite quail (*Colinus virginianus ridgwayi*). However, Refuge grasslands are an important resource for many other species of breeding and wintering grassland birds. Over 100 species of birds have been recorded on Refuge grasslands including 11 species considered to be of management concern in the U.S. Fish and Wildlife southwestern region (USFWS 1995). Despite the recent protected status, Refuge grasslands suffer the effects of decades of overgrazing and fire suppression, and mesquite is well established in most of the upland areas (Lloyd 1997).

Managers have adopted an ecosystem-level approach to restoration and long-term management of desert grasslands on the Refuge and prescribed burning has become the principal technique used to meet this goal. Specific objectives of the prescribed burning program are to (1) reintroduce fire as a regular component of the desert grassland

ecosystem, (2) encourage growth of native grasses, and (3) reduce the density of shrubs, sub-shrubs, and introduced grasses in upland areas. Under the current fire regime, approximately 5,700 ha are burned each spring on a 5-year rotation.

Although several studies have measured the effect of wildfire on avian communities in open grasslands of the southwestern United States (Bock et al. 1976; Bock and Bock 1988, 1992), few studies have examined the effect of prescribed burning on grassland birds, especially in grasslands with a substantial shrub component (see Reynolds 1997). Refuge managers have started a monitoring program designed in part to assess the long-term effects of prescribed burning on the breeding bird community. Despite this effort, no attempt has been made to quantify the short-term effect of prescribed burning on both the breeding and wintering avian communities.

Lloyd (1997), in a precursor to this study, predicted that prescribed burning at the Refuge will likely affect breeding grassland birds differentially. Abundances of species negatively correlated with increasing mesquite density such as cactus wren (*Campylorhynchus brunneicapillus*) and loggerhead shrike (*Lanius ludovicianus*) were predicted to increase, and abundances of species positively correlated with increasing mesquite density such as pyrrhuloxia (*Cardinalis sinuatus*) and Lucy's warbler (*Vermivora luciae*) were predicted to decrease. Furthermore, total bird abundance was predicted to decline after fire (Lloyd 1997). The current study was designed to test these predictions while examining the effects of a spring prescribed burn on a grassland bird community in a mesquite-dominated desert grassland.

My specific objectives were to (1) quantify fire-related characteristics, such as pre-burn fuel conditions, fire behavior, and extent of burn; (2) determine the direction, magnitude, and duration of short-term trends in relative abundance of breeding and wintering grassland birds following fire; (3) determine the influence of prescribed fire on grassland birds at the community level as evidenced by changes in species richness and total relative abundance; (4) measure changes in percent cover of ground and shrub vegetation after fire; (5) measure fire damage and structural change to mesquite; and (6) test predictions set forth by Lloyd (1997).

METHODS

Study Area

The 46,537 ha Buenos Aires National Wildlife Refuge is located in the Altar Valley of south-central Pima County, Arizona and is bounded by the Baboquivari and Palo Verde mountains to the west, the Colorado and Las Guijas mountains to the east, and the Mexican border to the south. Elevations within the Refuge range between 914-1,463 m. Climate is semi-arid and is characterized by high summer temperatures, low humidity, and low precipitation. Temperatures range from 44°C in the summer to -13°C in winter with a monthly mean of 17°C. Precipitation varies greatly from year to year, but on average the valley floor receives 300 mm annually. Annual precipitation is bimodal with a brief summer season of localized thunderstorms and a longer winter season of widespread frontal storms.

The Altar Valley of southern Arizona once supported Sonoran savanna grassland, a subtropical fire-climax ecosystem dominated by native grasses (Brown 1994). During the last century, however, these grasslands were largely converted to semi-desert grassland through historic land-use practices (Madsen 1996). Relative to Sonoran savanna grassland, semi-desert grasslands have a lower percent cover of grasses and a higher percent cover of shrubs, sub-shrubs, and bare ground.

The study site was located at an elevation of 1,100 m in an upland area of mesquite dominated semi-desert grassland on the northern half of the Refuge (Fig.1). Ground cover was characterized by a mix of perennial bunchgrasses, sub-shrubs, forbs, and large patches

of bare ground. Lehmann lovegrass (*Eragrostis lehmanniana*), a fire-resistant grass introduced from South Africa, was the dominant grass species and often formed large monotypic stands with a dense litter layer. Common native grasses included Arizona cottontop (*Digitaria californica*), grama grasses (*Bouteloua* spp.), three-awns (*Aristida* spp.), cane beardgrass (*Andropogon barbinodis*), and plains lovegrass (*Eragrostis intermedia*). Snakeweed (*Gutierrezia sarothrae*) and burroweed (*Isocoma tenuisecta*) were the common sub-shrub species. Velvet mesquite (*Prosopis velutina*) was the dominant shrub/tree species and densities of mesquite varied markedly across the study area from scattered individuals in open grasslands to relatively dense stands associated often with small washes and gulleys that cut across the site. Some mesquite trees were >5.0 m in height; however, most were small enough (<3.0 m) to be classified as shrubs. Cholla cactus (*Opuntia* spp.) and soap tree yucca (*Agave elata*) were uncommon.

Experimental Design

In early 1996, I chose 2 experimental units from a selection of Fire Management Units (FMUs) on the Refuge. The 1,596-ha Blanco FMU was designated as the burn unit and the 1,818-ha Road Camp, Secundino, and Middle FMUs were designated as the control unit. I matched the control to the burn unit based on similarities in size, burn histories, physiognomies, and distances to major landscape features (mountains, reservoirs, streams, recent burns). The control unit was located immediately south of the burn unit (Fig. 1).

Seven 25-ha plots (1,000 x 250 m) were placed randomly within each treatment unit. I eliminated plots that were <200 m from a road to reduce potential disturbance and edge effect, where possible, and I selected plots that were >500 m apart to ensure adequate dispersion. Due to space constraints in the burn area, however, ends of 3 plots abutted sides of adjacent plots. I bisected each plot with a 1-km long transect, marked every 50 m by a metal stake and colored flagging, around which birds were surveyed and vegetation was sampled.

Fire Monitoring

One week before the prescribed burn, I measured the pre-burn load of surface fuels by collecting 6 biomass samples from within the treatment plots. At the end of 6 transects, I placed a 0.5 m² wooden sampling frame on the ground at 4 sub-sampling points located 5 m N, S, E, and W from the transect endpoint. Within the sampling frame, I clipped all standing grasses, forbs, and sub-shrubs and collected all organic surface litter. I placed standing vegetation and litter samples into separate storage bags, dried the samples at 70°C for 48 hours in a lab oven, and weighed each sample to the nearest gram using a 200-g Pesola scale.

On the morning of the prescribed burn, I clipped 6 grass samples (~100 g) to determine the moisture content of the fine-fuel source. I calculated the moisture content by subtracting the weights of the samples dried at 70°C for 48 hours in a lab oven from the weights of samples taken in the field immediately after clipping. Measurements were made to the nearest 0.5 g using a 100-g Pesola scale.

Prescribed burns on the Refuge were conducted under the following burning prescription: temperatures of 21-35 °C, relative humidity of 5-25%, and winds of 16-40 kph. During the burn, I measured these variables hourly from the Sasabe remote automatic weather station located about 2 km southeast of the burn unit.

Following the burn, I quantified the areal extent of the fire by sampling 40 points located randomly within each burn plot. At each point, I noted whether vegetation had been burned or unburned. In addition, I collected post-burn samples of litter biomass 15 months after the fire from 4 of the original sample points that were affected by the fire. I followed the pre-burn protocol during post-burn collection and processing of samples; however, instead of collecting litter from the same locations, I shifted each sub-sampling point 45° (i.e., NE, SE, SW, and NW) in relation to the transect endpoint.

Vegetation sampling

Using the line intercept method (Canfield 1941), I quantified percent cover of total grasses, exotic grass, native grasses, annual grass, forbs, sub-shrubs, mesquite, and bare ground. Vegetation variables were defined broadly to facilitate analyses and to reflect larger-scale vegetation features that may have influenced the abundance and distribution of grassland birds within the study area (Appendices A and B).

On each of the 14 plots, I randomly selected 6 of 21 metal stakes demarcating the survey transect, and from these stakes I extended a 20-m sample line in a random direction. I measured to the nearest 0.5 cm the interception of plants through the vertical plane of the sampling line. Grasses and forbs were measured basally and shrubs and

sub-shrubs were measured aerially on the crown-spread intercept (Canfield 1941). I collected pre-burn (late March 1997) and post-burn (late March 1998) measurements of percent cover from the same sample lines on the burn and the control plots.

To detect changes in the percent cover of vegetation through time, I compared percent cover values before and after the prescribed fire on the control and burn units using a repeated measures analysis of variance (ANOVA). I used a square root transformation to correct for non-constant variances for some but not all of the variables. I blocked for between-subjects effects and computed a Wilks' Lambda F statistic for within-subjects, time \times treatment interactions. I calculated the magnitude of changes by finding the difference between pre- and post-burn cover on the burn unit. To this value, I added a correction factor which was the difference between pre- and post-burn cover on the control unit.

Because of the importance of mesquite in determining the distribution of grassland birds on the study site (Lloyd et al. 1998), I collected post-burn measurements of mesquite density, structure, and fire-damage to better document the effect of fire on this important vegetation component. In the summer of 1998 (15 months post-burn), I placed 10 10-m radius quadrats at 100-m intervals and at a random distance (<60 m) perpendicular to the transects in each of the burn plots. Within these quadrats, I recorded the total number of mesquite, and for each mesquite, I recorded the maximum height (to the nearest 0.5 m) of live and dead vegetation using a modified range pole. I used height as a single measure of mesquite structure because height and width of the mesquite canopy were correlated

strongly ($r = 0.95$, $P < 0.01$) (Lloyd et al. 1998). I recorded the maximum height of dead structure (usually fire-killed limbs and twigs) as an estimate of the pre-burn height of the mesquite canopy. I then calculated the change in mesquite structure resulting from the burn as the difference between the estimated pre-burn height and the maximum height of live vegetation (remnant or regrowth). I categorized mesquite into 3 height classes (< 1.5 m, $1.5 - 3.0$ m, and > 3.0 m).

I also classified the fire-damage to each mesquite into 1 of 4 categories: (1) no visible impact of fire, (2) partial crown kill, (3) complete crown kill with resprouting from base or stems, and (4) crown and roots killed (DeBano et al. 1996). I combined classes 1 and 2 into a single category to denote mesquite with less severe fire damage and classes 3 and 4 into a single category to denote mesquite with severe fire damage.

Bird Surveys

I quantified species relative abundance, total relative abundance, and species richness (absolute number of species) for the breeding and wintering communities of grassland birds by surveying birds along 1-km line transects (Lancia et al. 1994). As an index of relative abundance, I recorded all visual and auditory observations of birds within a band 125-m wide on either side of the transect. This lateral distance constituted the boundary of the plot and the limit of detectability for most species. I used relative abundance instead of an estimate of density for 2 reasons: (1) relative abundance is often used to compare the same populations through time (Temple 1981), and (2) my attempts to generate density estimates were successful for only 3 species due to a lack of

independent distance observations.

Birds were surveyed for 2-3 hours beginning 1/2 hour after sunrise and typically 2-3 surveys were completed by a single observer during the morning. In 1996, 2 observers surveyed birds by alternating their survey efforts on the same plots to minimize observer bias. In 1997 and 1998, only 1 of the observers surveyed birds. On average, a survey was completed in 25 minutes by walking at a moderate pace and stopping occasionally to look, listen, and record observations. Observations of raptors, bird flyovers, and flocks of quail were noted but were not included in analyses. No surveys were performed during periods of rain or strong winds (>32 kph).

Breeding birds were surveyed from 1 April to 30 July in 1996 (pre-burn) and in 1997 and 1998 (post-burn). I divided the relative abundance data collected during the breeding season into 2 time periods (April/May and June/July) in an effort to better detect intraseasonal trends and to better capture peak detectabilities of birds that breed in spring following the winter rains and those that breed in late summer with the advent of the summer monsoon (Maurer 1985).

Because density of mesquite influenced the distribution of grassland birds more than any other vegetation variable on our study site, I classified breeding species into guilds (shrub-dependent or not shrub-dependent) based on the importance of shrubs as a component of each species' habitat (Table 4) (Lloyd et al. 1998). I searched for patterns not evident at the community level through examination of these smaller, more homogenous subsets of species (Wiens 1989).

Winter birds were surveyed from 1 February to 30 March in 1997 (pre-burn) and 1998 (post-burn). Because wintering sparrows often formed mixed-species flocks, I estimated the size and species composition of flocks with >5 individuals and analyzed these data separately from the relative abundance data for wintering species.

I analyzed relative abundance data only for species that were widely distributed (present on >50% of plots). For breeding species that showed a clear intraseasonal peak in detectability (>40% difference in frequency of observations between time periods), I limited my analyses to the time period with the greater number of observations. I estimated a mean relative abundance value for each species on both the control and burn units for the 2 breeding season time periods in 1996, 1997, and 1998 and the wintering season time period in 1997 and 1998 using the average number of observations per plot per survey (total number of observations/total number of survey efforts). For each species, I examined residual plots for outlying values and to assess the need for transformations. A square root transformation +0.1 helped to stabilize non-constant variances in the data for many but not all of the species. I report untransformed summary statistics in tables and figures for all species but used transformed data for analyses.

To detect trends through time, I compared species relative abundance, total relative abundance, and species richness across 3 breeding and 2 wintering seasons on control and burn units using a repeated measures ANOVA. I blocked for between-subjects effects and computed a Wilks' Lambda F statistic for within-subjects, time×treatment interactions for breeding and wintering data. For species that showed

time×treatment interactions during the breeding season, I made follow-up comparisons of group means using linear contrasts. I compared pre-burn values of relative abundance on the burn and the control units. In addition, I compared differences in mean relative abundance between burn and control units in 1996 and 1997 and in 1996 and 1998. I calculated the magnitude of change by finding the difference between pre- and post-burn relative abundance on the burn unit. To this value, I added a correction factor which was the difference between pre- and post-burn relative abundance on the control unit.

RESULTS

Prescribed burn

Study plots were burned on 1 April 1997. On the morning of the burn, the moisture content of fine fuels was 28% (SE = 1.3, $n = 6$) and biomass of surface fuels was 3,361 kg/ha (SE = 926, $n = 6$), of which 2,308 kg/ha (SE = 949, $n = 6$) was litter. Temperatures were cool, relative humidity was high, and winds were strong and gusty (Table 1).

Back-fires were hand-lit at 0900 along the north and east sides of the burn unit and the head fire was set late in the afternoon along the south and west sides. The 1,596-ha unit took most of the night to burn out. Fire behavior was variable, but average flame lengths were small and the rate of fire spread was rapid. Several large washes acted as firebreaks during the burn, and unaffected plots were spot-burned on 7 and 14 April (Table 1).

A total of 72% (SE = 2.8, $n = 7$) of the area within the burn unit was affected by the fires. Fifteen months after the burn, litter biomass had decreased by 59% or 1,369 kg/ha (SE = 509, $t = 2.69$, 1-tailed $P = 0.04$, $df = 3$) in areas that were burned.

Vegetation

One year after the fire, percent cover of total grasses was reduced on the burn unit by 28%, sub-shrubs by 92%, and exotic grass by 37% (Table 2). Percent cover of forbs increased across the study area, but more so on the control, resulting in a 500% decrease of forbs on the burn unit. Percent cover of mesquite was reduced by 49% and percent

cover of bare ground increased by 12%. Finally, no change was detected in the percent cover of native or annual grasses (Table 2). Appendices A and B summarize the pre- and post-burn values of percent cover for vegetation variables on the control and burn units respectively.

Velvet mesquite was the dominate shrub species and comprised 93% (SE = 1.9, $n = 7$) of all woody plants on the burn unit. Fifteen months after the burn, mesquite density was 295 shrubs/ha (SE = 27.0, $n = 7$). Average post-burn height of mesquite shrubs was 1.6 m (SE = 0.09, $n = 7$). From the height of dead structure, I estimated that the average pre-burn height of mesquite was 2.0 m (SE = 0.04, $n = 7$). One percent (4 shrubs/ha, SE = 0.91, $n = 7$) of all mesquite were killed outright by the prescribed burn, 56% (164 shrubs/ha, SE = 25.3, $n = 7$) were top-killed but resprouted basally or from stems, 23% (64 shrubs/ha, SE = 13.6, $n = 7$) showed evidence of a partial crown kill, and 20% (63 shrubs/ha, SE = 22.0, $n = 7$) were unaffected by the burn.

Twenty-four percent (9 mesquite/ha, SE = 2.8, $n = 7$) of large mesquite (>3.0 m) were top-killed by the fire and 76% (28 mesquite/ha, SE = 2.3, $n = 7$) were either unaffected or showed evidence of a partial crown kill. Large mesquite lost an average of 0.6 m (SE = 0.19, $n = 7$) in height, a 17% decrease of pre-burn structure. No mesquite >3.5 m in height was killed outright.

Forty-eight percent (58 mesquite/ha, SE = 9.2, $n = 7$) of medium-sized mesquite (1.5-3.0 m) were top-killed or killed outright and 52% (64 mesquite/ha, SE = 15.6, $n = 7$) were unaffected or partially crown-killed. Medium-sized mesquite lost an average of 0.7

m (SE = 0.13, $n = 7$) in height, a 27% decrease of pre-burn structure.

Seventy-four percent (101 mesquite/ha, SE = 15.6, $n = 7$) of small mesquite (<1.5 m) were either top-killed or killed outright, and 26% (35 mesquite/ha, SE = 10.2, $n = 7$) were unaffected or partially crown killed. Small mesquite lost an average of 0.3 m (SE = 0.06, $n = 7$) in height, a 30% decrease of pre-burn structure.

Breeding birds

During the 3 years of the study, I completed 373 breeding bird surveys (Table 3) and detected 5,266 individuals of 45 species (Table 4). Of the 45 species, 60% were year-round residents, 27% were summer residents, and 13% were transients. Black-throated sparrow was the most abundant and widely distributed species, accounting for 24% of total sightings. Other common year-round residents included eastern meadowlark, cactus wren, loggerhead shrike, and verdin. The most commonly observed summer resident species were Lucy's warbler, mourning dove, Cassin's sparrow, ash-throated flycatcher, and northern mockingbird.

I had sufficient data for statistical analyses of trends in relative abundance for 12 species of breeding grassland birds; 8 species in April/May (Table 5) and 12 species in June/July (Table 6). These species comprised 85% of all bird observations. I observed increasing trends for 2 species (black-throated sparrow and mourning dove), decreasing trends for 3 species (Botteri's sparrow, Cassin's sparrow, and pyrrhuloxia), and a mixed trend for cactus wren. I found suggestive, but inconclusive, evidence for trends in 3 other species; possible increases in relative abundance of loggerhead shrike and Lucy's warbler,

and a possible decrease in relative abundance of eastern meadowlark. No change was detected for ash-throated flycatcher, northern mockingbird, and blue grosbeak following fire (Tables 5 and 6).

Black-throated sparrows (Fig. 2) showed a sequential increase in 2 post-fire breeding seasons. By the end of the study, relative abundance had increased by 103% (absolute increase of 3.2 birds/25 ha) on the burn unit as compared with the control (linear contrast, $t = 2.74$, $P = 0.01$).

Mourning doves (Fig. 3) also increased in 2 post-fire breeding seasons. The increase was twice as large in the first year post-burn (955%, 2.7 birds/25 ha) (linear contrast, $t = 7.57$, $P < 0.01$) as compared with the second year post-burn (433%, 1.2 birds/25 ha) (linear contrast, $t = 3.44$, $P < 0.01$).

Botteri's sparrows (Fig. 4) decreased by 288% (1.2 birds/25 ha) in 1997 (linear contrast, $t = -4.52$, $P < 0.01$) and by 229% (0.9 birds/25 ha) in 1998 (linear contrast, $t = -3.84$, $P = 0.01$). In the first post-fire breeding season I observed only 2 singing males on the burn plots; both birds displayed from remnant patches of unburned cover and were observed only once at the beginning of the breeding season. I detected no Botteri's sparrows in the second year post-burn making this the only breeding species to disappear from the burn unit after the fire.

Cassin's sparrow (Fig. 5) also declined after fire. During the first post-fire breeding season, I observed only 3 individuals on the burn plots; however, this apparent decrease was not significant due the variable pre-burn abundance of this species (linear

contrast, $t = -1.54$, $P = 0.14$). During the second post-fire breeding season, I observed an increase in relative abundance of Cassin's sparrow across the study area. This increase was substantially greater on the control as opposed to the burn unit, and as a result, Cassin's sparrow declined by 217% (1.7 birds/25 ha) on the burn unit in 1998 (linear contrast, $t = -3.27$, $P < 0.01$).

Pyrrhuloxia (Fig. 6) declined by 103% (0.7 birds/25 ha) in the first post-burn breeding season on the burn unit (linear contrast, $t = -2.31$, $P = 0.03$).

Cactus wren (Fig. 7) was the only species to exhibit a mixed pattern of trends. In the first post-fire breeding season there was an apparent increase in abundance across the study area in June/July, but the increase was greater on the control unit than on the burn unit. Therefore, cactus wren decreased by 619% (1.3 birds/25 ha) on the burn unit as compared with the control (linear contrast, $t = -2.30$, $P = 0.03$). In contrast to this decline, relative abundance of cactus wren increased by 142% (0.4 birds/25 ha) in April/May of the second post-fire breeding season. Cactus wren was the only species to exhibit substantial differences in pre-burn relative abundance between experimental units. Relative abundance was less on the burn than on the control unit; 75% (0.8 birds/25 ha) in April/May 1996 (linear contrast, $t = 2.93$, $P = 0.01$) and 79% (0.8 birds/25 ha) in June/July 1996 (linear contrast, $t = 3.54$, $P < 0.01$).

Four of 5 (80%) species classified as not shrub-dependent and 2 of 7 (28%) species classified as shrub-dependent exhibited changes. Furthermore, species that were not shrub-dependent generally displayed trends that were more immediate, of greater

magnitude, and of longer duration as compared with trends for shrub-dependent species (Table 7).

At the community level, I found no change in total abundance of breeding birds for any time period during the study. Species richness declined by 30% (1.4 species/25 ha), however, on the burn unit in the first year post burn ($F_{2,11} = 13.57$, $P < 0.01$; linear contrast, $t = -3.42$, $P < 0.01$) (Fig. 8).

Several anecdotal observations of trends in less abundant and sparsely distributed species deserve mention. After the burn, horned larks were observed for the first time on the burn unit. One territorial male was present in the first post-fire breeding season and 3 territorial males in the second post-fire breeding season. Horned Larks were present on the control unit in low numbers and did not appear to fluctuate greatly in numbers through time.

American kestrel (*Falco sparveius*), an uncommon pre-burn species across the study area, increased dramatically on the burn unit immediately after the fire. Kestrels were present on all 7 burn plots with a maximum of 6 individuals/25 ha. Kestrels showed no corresponding increase on the control unit and after 2 weeks kestrel numbers had returned to pre-burn levels on the burn unit.

Relative abundance of vermilion flycatcher increased on the burn unit following the fire. During the pre-burn breeding season, vermilion flycatchers were present on 3 of 6 burn plots with an average relative abundance of 0.1 birds/25 ha (SE = 0.06, $n = 7$). During the post-burn breeding season, vermilion flycatchers were observed on 6 of 7

plots with an average relative abundance of 1.1 birds/25 ha (SE = 0.44, $n = 7$). I observed no vermilion flycatchers on the control unit during the study.

Wintering birds

During 2 years of winter surveys, I completed 161 surveys (Table 3) and detected 1,425 individuals (not including individuals in flocks >5 birds) of 28 species (Table 8). Of the 28 species, 71% were year-round and 29% were winter residents. Black-throated sparrow, eastern meadowlark, and verdin were the most commonly observed year round residents. Vesper sparrow, Brewer's sparrow, and grasshopper sparrow were the most commonly observed winter residents.

Wintering sparrows often formed mixed-species flocks with an average of 21 birds/flock (SD = 16.7, $n = 57$). Vesper sparrow, the most common flocking species, was observed in 70% of all flocks, Brewer's sparrow in 60%, white-crowned sparrows in 16%, and black-throated sparrow and chipping sparrow (*Spizella passerina*) in 12%. I found suggestive evidence for a 147% increase (0.5 flocks/25 ha) in the average number of mixed flocks on the burn unit following the fire (ln transformed, $F_{1,12} = 2.42$, $P = 0.14$).

Following the fire, I detected trends for 3 of the 10 wintering species analyzed (Table 9). Ladder-backed woodpecker and vesper sparrow (not including individuals in flocks) increased in relative abundance by 260% (0.5 birds/25 ha) and 68% (1.8 birds/25 ha) respectively, and cactus wren decreased by 274% (0.6 birds/25 ha) on the burn relative to the control unit.

At the community level, there was no change in total abundance (including individuals in flocks) (sqrt transformed, $F_{1,12} = 1.46$, $P = 0.25$) or species richness ($F_{1,12} = 0.80$, $P = 0.39$) of wintering birds on the burn unit as compared with the control following the fire. However, total abundance of wintering birds declined substantially across the study area from an average of 19 birds/25 ha in 1997 to an average of 9 birds/25 ha in 1998 (sqrt transformed, $F_{1,12} = 11.86$, $P < 0.01$).

DISCUSSION

Effects of fire on vegetation

The spring prescribed burn was moderate in intensity and produced a mosaic of burned and unburned vegetation patches, affecting $<3/4$ of the treatment unit. I attributed this outcome to the highly variable distribution of the fuel source (range 655-6,435 kg/ha) and to the presence of intervening washes which acted as natural firebreaks during the burn. Unlike most grassland fires that burn with relative uniformity, fires in shrubby environments that lack a continuous fuel source often result in a distinct pattern of burned and unburned patches (Reynolds and Bohning 1956, Petersen et al. 1987, DeBano et al. 1998).

Within the areas that were burned, the effect of fire on vegetation was dependent on both the severity of the fire and the growth forms of the affected vegetation. The prescribed burn occurred near the lower end of the Refuge's burning prescription. The combination of weather conditions and fire behavior observed during the burn (Table 1) are indicative of a fire of only moderate intensity (Johnson and Miyanishi 1995, DeBano et al. 1998). Because no two fires are identical, the subsequent spot-burning of unaffected areas likely complicated the outcome. Nevertheless, damage to the vegetation community as a whole was limited and of short duration, an outcome typical of fires that burn with lesser severity (DeBano et al. 1998). In the path of the fire, most ground cover was destroyed and many shrubs were top-killed. However, perennial grasses and shrubs resprouted quickly, and by the end of the study the physiognomy on the burn unit closely

resembled that of the control.

Despite this general pattern, burning affected vegetation differentially depending on the growth forms of the affected plants. Sub-shrubs, for instance, were impacted most strongly by the fire with a >90% decrease in percent cover 1 year after the fire, no evidence of regrowth from affected plants, and minimal recruitment from seed by the end of the study. Sub-shrubs are particularly susceptible to fire and mortality rates of 88% (Reynolds and Bohning 1956) and 92-98% (Cable 1967) have been recorded for burroweed in desert grasslands.

Perennial grasses are more resistant to fire and generally return to pre-burn levels within 1-4 years (Reynolds and Bohning 1956, Cable 1967, McPherson 1995). I found that perennial grasses recovered quickly, resprouting from charred root crowns within 3 months of the fire. Nevertheless, total grass cover was reduced by almost 1/3 1 year after fire. Although percent cover of exotic grass declined substantially, I detected no change in percent cover of native grasses after the fire. This pattern suggested a differential susceptibility to fire of exotic and native grasses. Instead, I believe that the difference was attributable to the areal dominance of exotic grass (85% of total grass cover) and the correspondingly small and variable amount of native grasses sampled on the burn unit (Appendix B). Previous research indicates that fire reduces both exotic and native grasses in desert grasslands (Bock and Bock 1992).

Generally, forb cover increases following fire in desert grasslands (Bock et al. 1976, Bock and Bock 1992), although Reynolds (1997) found no effect of winter burning

on forb cover. In this study, the substantially greater increase in forbs observed on the control unit was most likely the result of increased rainfall from 1997 to 1998 and possibly the result of a larger seed bank in the control unit from relatively recent (1990, 1991, and 1992) prescribed burns in the area.

Mesquite mortality was only 1% and smaller shrubs were more strongly affected by the burning than larger shrubs. Several factors most likely contributed to the observed pattern: (1) the burn occurred in the early spring, outside of the principal growing season and period of vulnerability of woody species like mesquite (McPherson 1995); (2) flame lengths, although variable, averaged <1.5 m during the burns; flame length is an important factor governing tree mortality (Johnson and Miyanishi 1995); and (3) there may have been insufficient fuel to carry fire underneath large mesquite trees which compete with and exclude grasses from their vicinity (Reynolds and Bohning 1956, Burgess 1995). Previous research in desert grasslands also indicates that smaller mesquites suffer greater fire damage and mortality than larger mesquites (Reynolds and Bohning 1956, Blydenstein 1957, Cable 1965, Alonso 1967, Reynolds 1997)

Because I did not collect pre-burn measurements of mesquite, I likely overlooked some smaller mesquite shrubs and seedlings that were killed by the fire but were not evident during post-burn sampling. Therefore, I may have underestimated mesquite mortality. Because the fire was not severe enough to completely destroy the above-ground structure of larger mesquite, I accounted for most mesquite that were >1.0 m in height. Thus, I inferred that there was little change in mesquite density, at least for

larger shrubs, on the burn unit following the fire. Nevertheless, most mesquite (80%) were affected to some extent by the fire and canopy cover was reduced by 1/2 and height by 1/4 in the year after the fire.

Patterns of change in the avian community

Given the simple community measures used during this study, the avian community as a whole changed little following spring burning. For instance, I observed no change in either species richness or total abundance for wintering birds during the study. Previous research shows that total abundance of wintering birds generally increases after fire in southwestern grasslands (Bock and Bock 1988, 1992; Reynolds 1997). Many wintering birds forage on grass and forb seeds, a resource that increases in abundance and availability in burned areas (Bock 1976, Bock and Bock 1992). Some changes may have been obscured by the 50% decline in wintering bird abundance observed across the study area from 1997 to 1998, a decrease that may have been related to the abnormally wet weather during the second year of winter surveys. For instance, the average number of mixed flocks of wintering birds appeared to increase by almost 150% on the burn unit following the fire.

As for the breeding community, species richness declined on the burn unit after the fire. The change was short-lived, however, and species richness returned to pre-burn levels by the second year post-burn. The loss of protective and thermal cover immediately after fire may temporarily displace many species from burned areas. Conversely, the absence of cover may attract other species to the same areas (DeBano et al. 1998).

American kestrels, mourning doves, and vermilion flycatchers, for example, were quick to invade burned areas following the fire. Changes for these species may reflect increased foraging opportunities as a result of decreased ground cover.

I detected no other community-level changes for breeding birds and, thus, did not confirm the prediction that total abundance of breeding birds would decrease after fire (Lloyd 1997). In general, the response of birds to fire is closely related to the effect of fire on vegetation (McPherson 1995). Following disturbance, plant species diversity often increases because of increased production, standing crop biomass, and structural complexity (Wiens 1989). In this study, burning had only a moderate impact on the vegetation community, and I believe this contributed to the limited and ephemeral changes observed in the breeding community as a whole. Two other factors may have influenced this pattern: (1) the breeding community was dominated, both numerically (70%) and compositionally (75%), by shrub-dependent species, few of which exhibited change after fire; and (2) post-burn sampling may not have been of sufficient duration to detect a long-term increase in species diversity. Further, patterns of change to breeding assemblages of grassland birds are often difficult to detect because grassland birds display species-specific responses of varying directions (Wiens 1989, Ganey et al. 1996).

In fact, the patterns of change that I observed were primarily species-level trends of varying directions. Previous research in desert grasslands suggests that the response of breeding birds to the effects of fire is closely related to the natural history requirements of each species (Bock and Bock 1992, Ganey et al. 1996). In general, the abundances of

birds that show a positive response to fire are negatively associated with grass and litter cover, whereas the abundances of birds that avoid burned areas are positively associated with shrub or grass cover (Bock and Bock 1992).

I observed patterns of association between populations of breeding birds and the effects of fire on vegetation that were consistent with this pattern. For example, both Cassin's and Botteri's sparrows prefer grasslands with sufficient grass and litter cover for nesting (Bock and Webb 1984, Rising and Beadle 1996, Webb and Bock 1996), resources that decreased after fire during the present study. Cassin's and Botteri's sparrows are known to decline immediately after fire and appear to require >2 years to recover to pre-burn abundance levels (Bock and Bock 1988, 1992). I observed similar patterns, although the trend for Cassin's sparrow was confounded somewhat by the considerable annual fluctuations observed for this species across the study area. Populations of Cassin's sparrow exhibit large fluctuations in annual abundance, primarily in response to changes in precipitation levels (Monson and Phillips 1981, Rising and Beadle 1996).

Mourning doves prefer to forage on bare ground where excessive ground litter does not conceal food items (Mirarchi and Baskett 1996) and this species is often associated with recently burned areas from which grass and litter cover has been effectively removed (Bock et al. 1976, Bock and Bock 1988, Reynolds 1997). In the present study, mourning doves displayed the greatest relative change in magnitude of any species, increasing by >950% immediately after the fire, and maintaining increased abundance on the burn unit until the end of the study.

The mixed trend observed for cactus wren was harder to decipher. Cactus wrens are more abundant in mesquite-grasslands with lower mesquite density (Maurer 1985, Lloyd et al. 1998), and this species was predicted to increase after burning that reduced mesquite density on the Refuge (Lloyd 1997). Instead, I found that relative abundance of cactus wrens decreased on the burn unit relative to the control after the fire and increased only at the end of the study. Although cactus wrens were widely distributed across the study area, they were much more abundant on the control area. Thus, trends for cactus wren in both the breeding and wintering seasons should be viewed with reserve.

Before the study, I grouped Cassin's sparrow, Botteri's sparrow, mourning dove, cactus wren, and eastern meadowlark into the category of birds that were not shrub-dependent. With hindsight, it appeared that some of these species (e.g., eastern meadowlark, Cassin's sparrow, and cactus wren) used shrubs to some degree as a component of their habitat on the Refuge. Nevertheless, as a group, these species displayed trends that were more immediate, of greater magnitude, and of greater duration than trends for shrub-dependent species (Table 7). This pattern was consistent with the observation that ground cover was more strongly impacted by fire than shrub cover.

The effect of prescribed fire on species I classified as shrub-dependent was more variable. Less than 1/3 of shrub-dependent species exhibited trends and these trends were generally less immediate, of lesser magnitude, and of shorter duration than trends exhibited by species that were not shrub-dependent (Table 7). Information is scarce, but previous research suggests that fire has only a minimal effect on shrub-dwelling birds,

especially when fires are patchy or moderate in intensity (Petersen and Best 1987, Breininger and Smith 1992, Reynolds 1997).

I found that prescribed fire did not reduce the density of mesquite, but it did reduce the effective density of mesquite by top-killing or otherwise reducing the structure of mesquite shrubs. Thus, nesting and foraging opportunities for some shrub-dependent species may have been reduced temporarily by the burn. Following fire in a Texas grassland, nests of shrub-dwelling species were found only in the leafy portion of shrub crowns with >50% living material (Renwald 1978). Pyrrhuloxia, a shrub-nesting species (Ehrlich et al. 1988) whose abundance is positively associated with increasing mesquite density (Lloyd 1997), declined temporarily in relative abundance during the first breeding season after the fire. Pyrrhuloxia is considered an opportunistic species, moving from one area to another if conditions become unfavorable (Russell and Monson 1998). In addition, post-burn trends for two other shrub-dependent species, Lucy's warbler and loggerhead shrike, tended to match predictions made by Lloyd (1997).

Black-throated sparrows are associated with shrubby areas with increased bare ground and are found infrequently on rangelands that lack a substantial shrub component (Phillips et al. 1964, Rising and Beadle 1996). The strong decrease in ground cover and relatively moderate decrease in shrub cover may help to explain the increase in relative abundance of this species on the burn unit after the fire. I observed many family groups of black-throated sparrow on the burn unit in July of 1998 which initially suggested that productivity of this species had increased on the burn unit after fire. However, small

groups of adults and young wander extensively at the end of the breeding season (Russell and Monson 1998), and the increase may reflect a temporary influx of black-throated sparrows into the burn unit from adjacent, unburned areas. No other study has examined the effect of fire on black-throated sparrow.

Changes in relative abundance of wintering bird species concurred with the general habitat affinities of each species. For instance, relative abundance of ladder-backed woodpeckers increased following fire and woodpeckers are known to move into burned areas, possibly in response to the increased abundance or availability of insects that inhabit dead trees (Bendell 1974, Horton and Mannan 1988, Breininger and Smith 1992). Vesper sparrows prefer areas of open ground (Russell and Monson 1998, Rising and Beadle 1996) and show a strong affinity for burned areas both in the winter (Bock and Bock 1988, 1992) and during the breeding season (Petersen and Best 1987, Pylypec 1991). Vesper sparrow was the most common flocking species, and considering the suggestive increase of mixed flocks on the burn after the fire, vesper sparrows likely increased more than was indicated by the analyses of individuals alone.

Strengths and limitations of study

The current study had both strengths and limitations that are important when considering the results. On the positive side, the study was planned and I collected a full season of breeding and wintering bird data before the application of the treatment. The experimental manipulation involved a relatively large area (>1,500 ha), larger than manipulations reported for similar bird/fire studies in desert grasslands (Bock et al. 1976,

Bock and Bock 1988, 1992, Reynolds 1998). Large manipulations are considered appropriate for detecting broader-scale patterns such as population-level trends (Wiens 1986). The size and number of independent plots was sufficiently large to show general trends in populations, and I examined trends both at the community and species-level. Finally, I examined trends of breeding birds for 2 years post-burn unlike many studies that examine effects for only 1 year post-burn (Bendell 1974, Wiens 1989). However, a third post-fire year of bird surveys would have been beneficial because several species, such as mourning dove and Cassin's sparrow, appeared to be returning to pre-burn levels and other species, such as black-throated sparrow, were just beginning to show trends at the end of the study.

On the negative side, the experimental units were unreplicated and inferences to populations beyond the study area at the Buenos Aires Refuge are limited. In addition, I made no attempt to establish causal relations between fire effects and the observed changes in bird populations due to the lack of interspersed control and burn plots. Any suggestions to this extent are merely speculative and will require further research. Following fire, measures of relative abundance may be biased because fire alters vegetation structure and birds are generally more conspicuous on burned as opposed to unburned areas (Dawson 1981). Overall, I do not believe this was a substantial problem during the current study for 2 reasons: (1) most species that exhibited change showed declining trends despite the increased probability of detecting individuals in burned areas, and (2) those species that showed increasing trends did so even at the end of the study

when post-burn vegetation structure had recovered substantially. However, the increased relative abundance of mourning dove immediately after the fire may have been due, in part, to the increased conspicuousness of individuals within the burn. Finally, I did not attempt to quantify avian productivity during the study. Because bird abundance may not be a good indicator of habitat quality (Van Horne 1983), the associations I have demonstrated between bird populations and the effects of fire should be considered with caution.

Summary

The spring prescribed burn was moderate in intensity and produced a mosaic of burned and unburned vegetation patches. The effects of prescribed fire on the vegetation community as a whole were relatively moderate and of short duration. The burn impacted ground cover, especially sub-shrub cover, more strongly than shrub cover, and impacted smaller shrubs more strongly than larger shrubs. Mesquite mortality and change in density were minimal, although mesquite structure was reduced by fire.

The avian community, as a whole, changed little following spring burning on the Refuge. Species richness of breeding birds declined by almost 1/3 after the fire but returned to pre-burn levels by the second year post-burn. I observed no community-level changes for wintering birds; however, further research may be required to confirm this pattern. I observed mixed patterns of associations between fire and changes in bird populations at the species level. These patterns conform to the general habitat associations of each species, and may reflect changes in cover and resources.

Furthermore, species that were not shrub-dependent exhibited patterns of change that were more pronounced than shrub-dependent species. This pattern was consistent with the observation that ground cover was more strongly impacted by fire than shrub cover.

Results from this study add another component to the understanding of the effects of fire on birds in desert grasslands. The present study confirmed patterns observed in similar studies for several grassland species following fire. In addition, the study tested and confirmed several predictions about the effect of fire on shrub-dependent species, a group of birds for which information on fire effects is lacking. Future research should focus on clarifying the observed relationships between bird populations and fire, ideally through the use of randomized experimental designs. In addition, future studies should go beyond the tracking of bird abundance and focus on measures of avian productivity.

MANAGEMENT IMPLICATIONS

Spring prescribed burns that are patchy in extent and moderate in intensity are not uncommon in mesquite-dominated grasslands on the Refuge. In 1998, for example, spring burning affected an estimated 75% of the area within treated FMUs (R. Madsen pers. comm.). In addition, most burning on the Refuge occurs in the early spring when weather conditions favor fires of lesser intensity. I believe that the prescribed burn in the present study was representative of a typical spring burn on the Refuge. However, no 2 fires are identical and predictions based on data collected from a single fire event must be viewed with caution.

Spring prescribed burns of moderate intensity are likely to impact ground cover, especially sub-shrub cover, more strongly than shrub cover. Although moderate intensity burns are capable of reducing mesquite recruitment (Blydenstein 1957, Humphrey 1974), hotter, more intense fires will be required to reduce the presence of large mesquite on Refuge grasslands. Experimentation with burns at the higher end of the Refuge's burning prescription has shown that fires of higher intensity kill larger mesquite trees and result in increased mesquite mortality (~50%) (W. Shifflett, pers. comm.). Such fires would likely have a greater impact on species associated with shrubs, and consequently, a greater impact on the avian community as a whole.

In the short-term, continued application of spring burns will likely have only a minimal impact on the avian community. However, species-level changes of varying direction, magnitude, and duration will occur. Several species, including species of

management concern like Cassin's and Botteri's sparrows, are likely to be affected negatively in the short-term. Managers currently use a rotational system of burning that creates a mosaic of post-fire vegetation types across Refuge grasslands. Thus, habitat loss is localized and species that are displaced by fire may find suitable habitat in adjacent unburned areas, assuming unoccupied habitat is available.

Many species of management concern are native to desert grasslands and evolved with fire as a regular component of their environment (Bock et al. 1976). Despite the short-term negative effects of fire to some of these species, the continued use of prescribed burning will likely have a long-term positive effect by helping to remove the invasive shrub component and by improving conditions for species that were historically more abundant. However, until prescribed burns are conducted at the higher end of the burning prescription, these changes will be slow in coming.

I predict that the avian community as a whole will remain relatively unchanged in the foreseeable future given the current fire regime employed by the Refuge. Because community change is likely to be minimal, I recommend that managers focus their monitoring efforts on species-level trends, especially for species of management concern. However, long-term monitoring of the avian community as a whole provides an invaluable tool in determining the efficacy of the prescribed burning program and in assessing the long-term impact of fire on grassland birds at the Refuge. I encourage managers to continue with the annual monitoring of breeding birds and suggest that a similar program be established to monitor wintering birds.

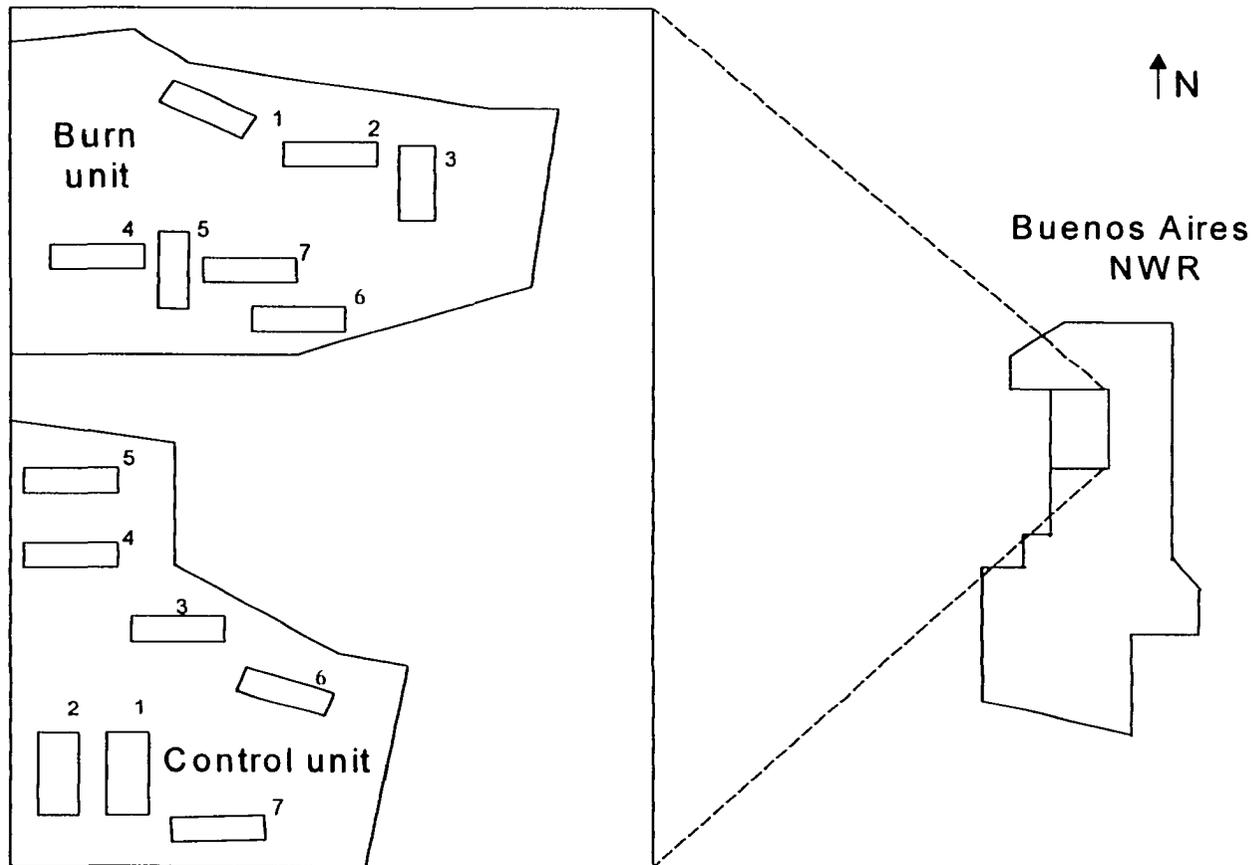


Figure 1. Map of study area at the Buenos Aires National Wildlife Refuge, Arizona showing the general location of the 14 25-ha plots within burn and control units. Map is not drawn to scale.

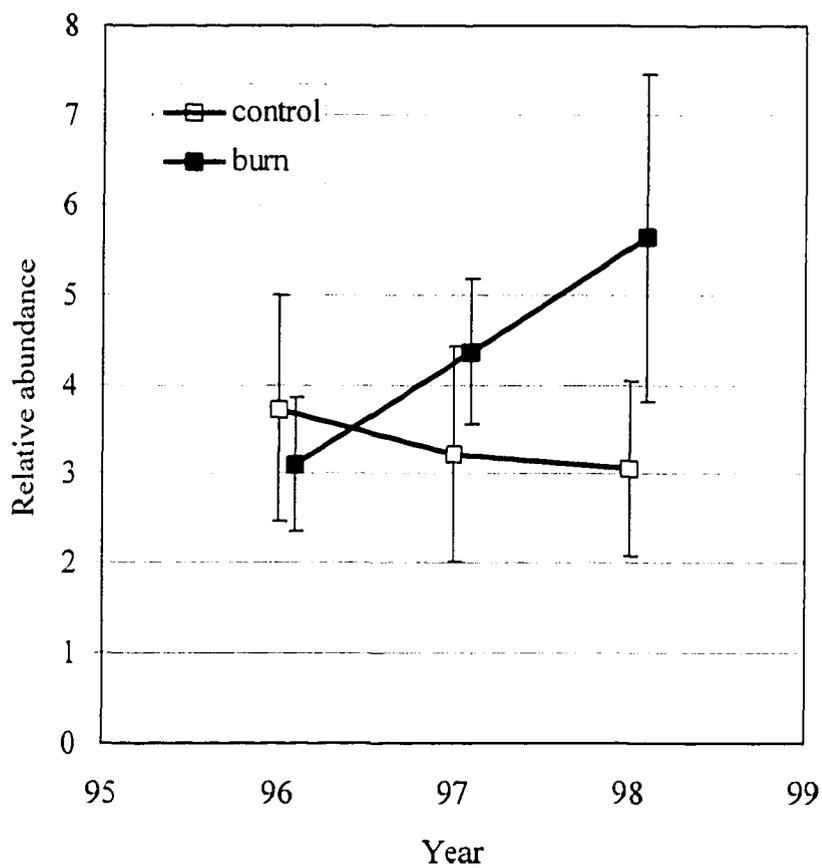


Figure 2. Trends in relative abundance (mean no./survey effort with 90% CI) of black-throated sparrows on control and burn units in June/July from 1996 to 1998 at the Buenos Aires National Wildlife Refuge, Arizona. Prescribed burn occurred 1 April 1997.

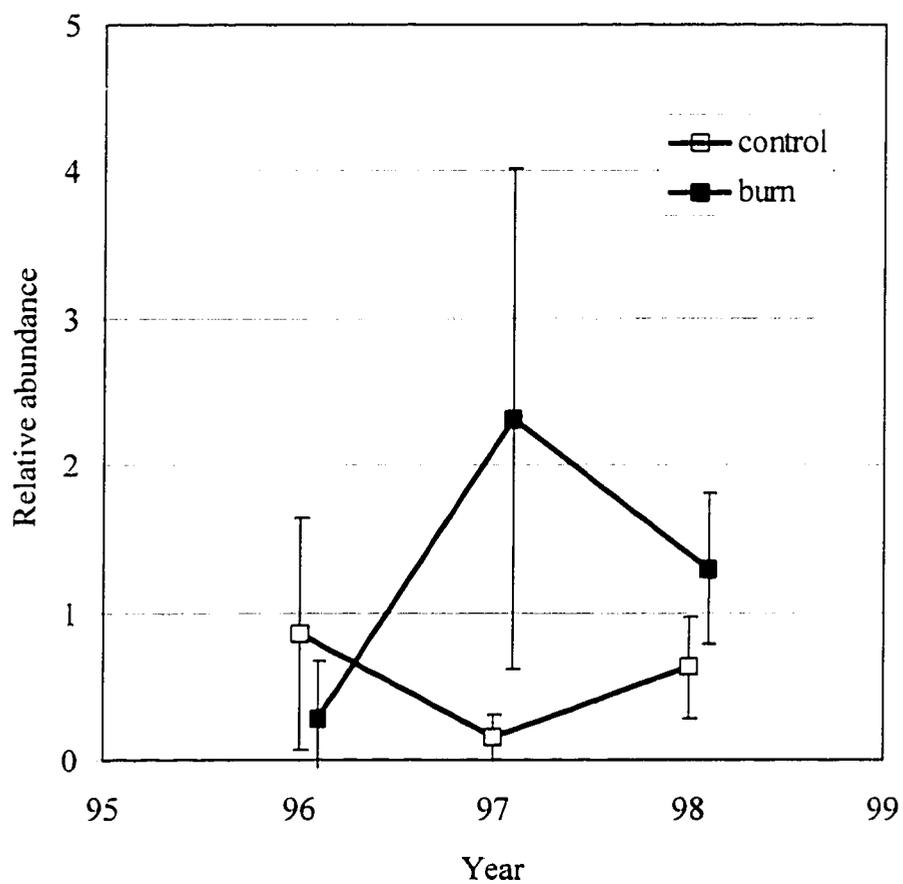


Figure 3. Trends in relative abundance (mean no./survey effort with 90% CI) of mourning doves on control and burn units in April/May from 1996 to 1998 at the Buenos Aires National Wildlife Refuge, Arizona. Prescribed burn occurred 1 April 1997.

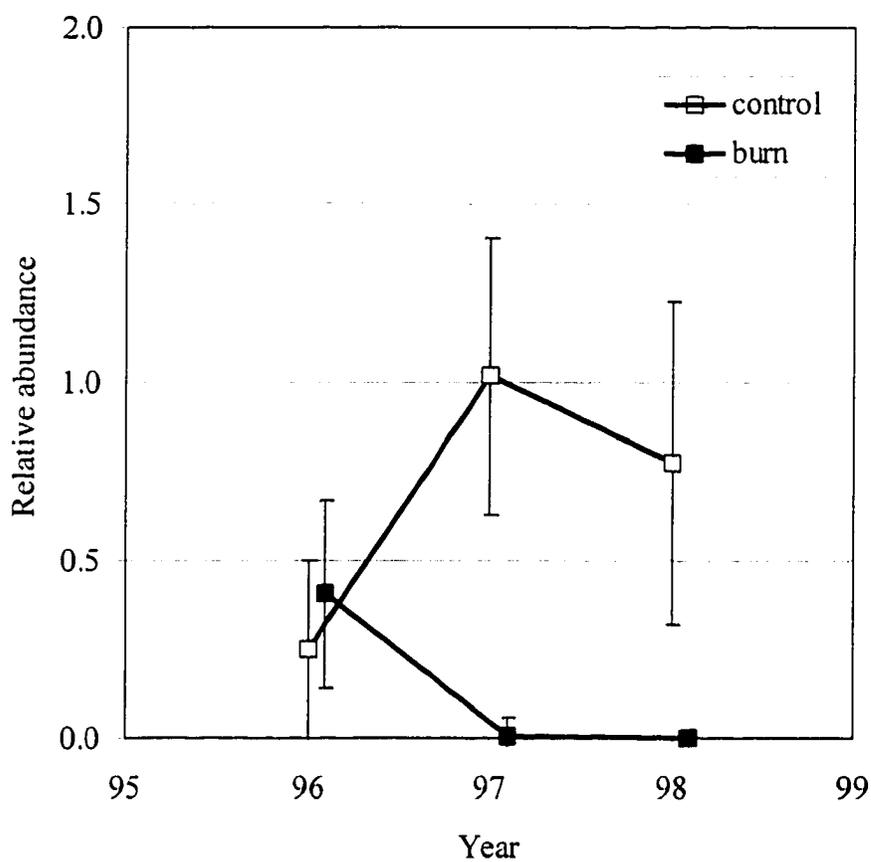


Figure 4. Trends in relative abundance (mean no./survey effort with 90% CI) of Botteri's Sparrows on control and burn units in June/July from 1996 to 1998 at the Buenos Aires National Wildlife Refuge, Arizona. Prescribed burn occurred 1 April 1997.

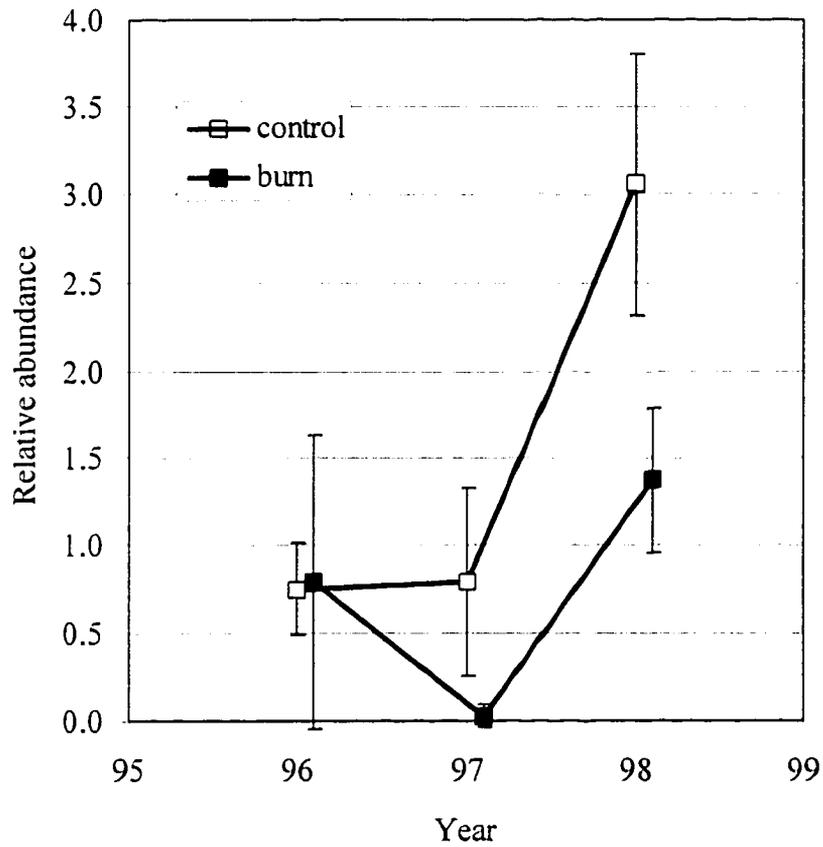


Figure 5. Trends in relative abundance (mean no./survey effort with 90% CI) of Cassin's sparrows on control and burn units in June/July from 1996 to 1998 at the Buenos Aires National Wildlife Refuge, Arizona. Prescribed burn occurred 1 April 1997.

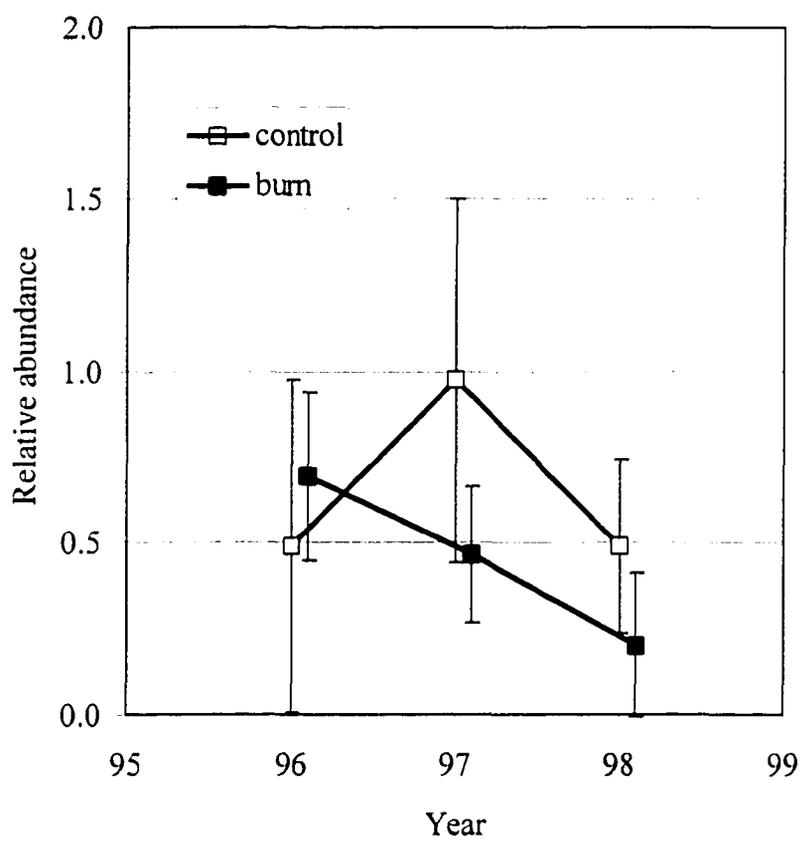


Figure 6. Trends in relative abundance (mean no./survey effort with 90% CI) of pyrrhuloxias on control and burn units in June/July from 1996 to 1998 at the Buenos Aires National Wildlife Refuge, Arizona. Prescribed burn occurred 1 April 1997.

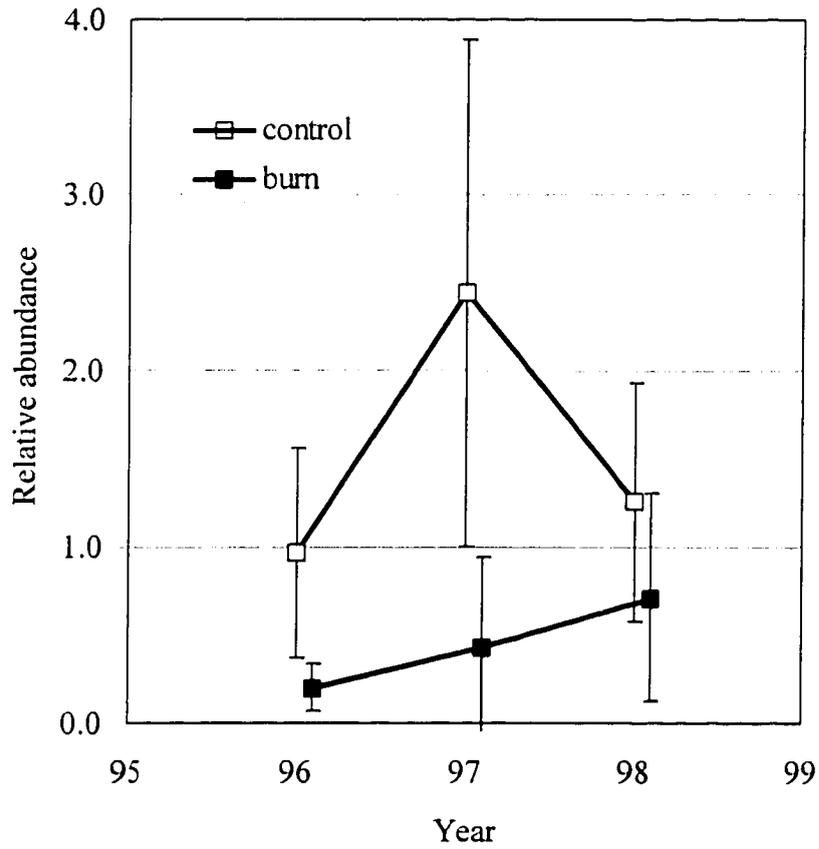


Figure 7. Trends in relative abundance (mean no./survey effort with 90% CI) of cactus wrens on control and burn units in June/July from 1996 to 1998 at the Buenos Aires National Wildlife Refuge, Arizona. Prescribed burn occurred 1 April 1997.

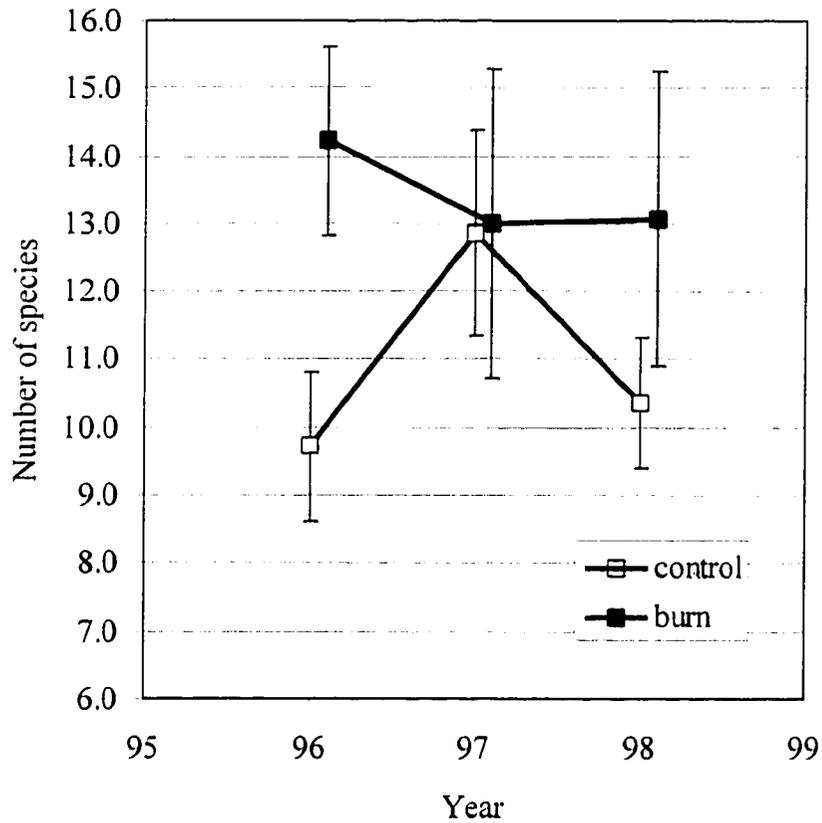


Figure 8. Trends in species richness (mean no. species/3 survey efforts with 90% CI) of breeding birds on control and burn units from 1996 to 1998 at the Buenos Aires National Wildlife Refuge, Arizona. Prescribed burn occurred 1 April 1997.

Table 1. Weather conditions, fuel conditions, and fire behavior during the prescribed burn (1 April 1997) and subsequent spot-burns (7 and 14 April 1997) of burn plots on the Buenos Aires National Wildlife Refuge, Arizona.

Burn Event	Start Time	% H ₂ O Fine Fuels	Fuel Temp. (° C)	Air Temp. (° C)	Rel. Humid (%)	Wind Speed (km/h)	Gust Speed (km/h)	Ave. Wind Direct. (°)	Ave. & (max) Flame lngth(m)	Ave. & (max) Rate Fire Spread (m/min)	Plots Burned
1 April	0900	27.9 ^a	26.4 ^b	20.1 ^b	37 ^b	32 ^b	53 ^b	212 ^b	1 (6)	8 (38)	1,5,6,7
7 April	0915	22.1 ^c	32.8	21.1	30	11	22	248	1 (1.5)	3 (6)	5,7
7 April	1350	22.1 ^c	37.8	23.3	24	14	29	266	3 (6)	5 (15)	1,2,3
14 April	0930	28.8 ^d	33.9	21.6	19	3	13	291	1 (3)	1.5 (3)	4,5

^a n = 6.

^b average of 9 readings taken from 0900-1700.

^c n = 4.

^d n = 3; grass samples collected 16 April.

Table 2. Mean differences (burn - control), 90% confidence intervals, and outcomes of repeated measures ANOVA (time×treatment interactions) of percent cover of vegetation on 7 control and 7 burn plots in late-March 1997 (pre-burn) and late-March 1998 (post-burn) on the Buenos Aires National Wildlife Refuge, Arizona.

Vegetation variable	1997 (pre-burn)		1998 (post-burn)		$F_{1,12}$	P
	\bar{d}	90% CI	\bar{d}	90% CI		
total grasses	-0.54	-1.15, 0.07	-1.33	-2.00, -0.65	7.38	0.02
native grasses	-1.19	-1.79, -0.60	-1.04	-1.56, -0.52	0.92	0.35
exotic grass	0.66	0.16, 1.15	-0.29	-0.83, 0.26	19.33	<0.01
annual grasses	0.00	0.00, 0.00	0.08	-0.02, 0.18	2.00	0.18
forbs ^a	-0.06	-0.20, 0.07	-0.57	-0.93, -0.21	5.84	0.03
sub-shrubs ^a	3.78	0.25, 7.31	-1.89	-4.73, 0.95	42.19	<0.01
mesquite ^a	1.48	-5.12, 8.08	-2.53	-7.10, 2.05	3.58	0.08
bare ground ^a	-4.24	-10.21, 1.74	5.46	0.47, 10.46	11.35	<0.01

^a F statistic and p -value generated from square root transformed data.

Table 3. Total number of bird surveys conducted on 7 control and 7 burn plots for two time periods during the breeding season (April - August) in 1996, 1997, and 1998 and during the wintering season (February - March) in 1997 and 1998 on the Buenos Aires National Wildlife Refuge, Arizona.

Time period	Control plots							Burn plots						
	1	2	3	4	5	6	7	1	2	3	4	5	6	7
April/May 1996	3	3	3	2	3	4	4	3	3	3	3	3	3	3
June/July 1996	4	4	4	4	4	3	3	4	4	4	3	4	3	3
Feb./March 1997	5	5	5	5	5	5	5	6	6	6	6	6	6	6
April/May 1997	6	6	5	5	5	5	6	5	5	5	5	5	5	5
June/July 1997	4	4	5	5	5	5	4	5	5	5	4	5	5	5
Feb./March 1998	6	6	6	6	6	6	6	6	6	6	6	6	6	6
April/May 1998	5	5	5	5	5	5	5	5	5	5	5	6	6	6
June/July 1998	5	5	5	5	5	5	5	5	5	5	5	5	5	5

Table 4. Total number detected, residence and guild status of grassland bird species observed during 373 breeding bird surveys (April-August, 1996, 1997 and 1998) on the Buenos Aires National Wildlife Refuge, Arizona.

Species	Total number detected	Res. status ^a	Shrub-dependent? ^b
black-throated sparrow (<i>Amphispiza bilineata</i>)	1,249	YR	Y
eastern meadowlark (<i>Sturnella magna</i>)	429	YR	N
Lucy's warbler (<i>Vermivora luciae</i>)	329	SR	Y
cactus wren (<i>Campylorhynchus brunneicapillus</i>)	318	YR	N
mourning dove (<i>Zenaida macrura</i>)	306	SR	N
Cassin's sparrow (<i>Aimophila cassinii</i>)	277	YR ^c	N
ash-throated flycatcher (<i>Myriarchus cinerascens</i>)	247	SR	Y
northern mockingbird (<i>Mimus polyglottus</i>)	230	SR	Y
loggerhead shrike (<i>Lanius ludovicianus</i>)	212	YR	Y
verdin (<i>Auriparus flaviceps</i>)	200	YR	Y
pyrrhuloxia (<i>Cardinalis sinuatus</i>)	184	SR	Y
canyon towhee (<i>Pipilio fuscus</i>)	138	YR	Y
western kingbird (<i>Tyrannus verticalis</i>)	148	SR	Y
vermillion flycatcher (<i>Pyrocephalus rubinus</i>)	108	YR	Y
Bewick's wren (<i>Thryomanes bewickii</i>)	104	YR	Y
ladder-backed woodpecker (<i>Picoides scalaris</i>)	94	YR	Y

^a Residence status: YR = year-round; SR = summer.

^b Adapted from Lloyd et al. 1998.

^c Distinct winter and summer populations.

^d Combined count of 17 species with ≤ 10 observations each.

Table 4 Cont.

Species	Total number detected	Res. status ^a	Shrub-dependent? ^b
Botteri's sparrow (<i>Aimophila botterii</i>)	91	SR	N
white-winged dove (<i>Zenaida asiatica</i>)	82	SR	Y
brown-headed cowbird (<i>Molothrus ater</i>)	80	SR	Y
blue grosbeak (<i>Guiraca caerulea</i>)	73	SR	Y
Bullock's oriole (<i>Icterus bullockii</i>)	73	SR	Y
horned lark (<i>Eremophila alpestris</i>)	57	YR	N
lark sparrow (<i>Chondestes grammacus</i>)	51	YR	N
Bell's vireo (<i>Vireo bellii</i>)	44	SR	Y
black-tailed gnatcatcher (<i>Polioptila melanura</i>)	30	YR	Y
rufous-winged sparrow (<i>Aimophila carpalis</i>)	23	SR	Y
lesser nighthawk (<i>Chordeiles acutipennis</i>)	18	SR	Y
rufous-crowned sparrow (<i>Aimophila ruficeps</i>)	12	YR	Y
other ^d	62		
Total	5,266		

^a Residence status: YR = year-round; SR = summer.

^b Adapted from Lloyd et al. 1998.

^c Distinct winter and summer populations.

^d Combined count of 17 species with ≤10 observations each.

Table 5. Mean differences (burn - control), 90% confidence intervals, and outcomes of repeated measures ANOVA (time×treatment interactions) for mean relative abundance/25 ha of breeding grassland bird species on 7 control and 7 burn plots in April/May during 3 years of study on the Buenos Aires National Wildlife Refuge, Arizona.

Species	April/May 1996 (pre-burn)		April/May 1997 (1 yr post-burn)		April/May 1998 (2 yrs post-burn)		$F_{2,11}$	P
	\bar{d}	90% CI	\bar{d}	90% CI	\bar{d}	90% CI		
mourning dove ^a	-0.57	-1.37, 0.23	2.16	0.59, 3.73	0.67	0.10, 1.23	5.18	0.03
ash-throated flycatcher	0.81	0.20, 1.42	0.51	-0.09, 1.10	0.98	0.41, 1.56	1.33	0.30
cactus wren ^a	-0.86	-1.47, -0.25	-1.11	-1.53, -0.69	-0.45	-0.99, 0.08	2.88	0.10
loggerhead shrike ^a	0.31	-0.18, 0.80	0.28	-0.09, 0.66	0.33	0.04, 0.61	0.19	0.83
Lucy's warbler ^a	1.20	0.39, 2.07	1.48	0.41, 2.51	1.11	0.29, 1.93	0.23	0.80
eastern meadowlark ^a	-0.45	-1.38, 0.47	-1.03	-1.92, -0.14	-0.99	-1.70, -0.28	2.31	0.14
pyrrhuloxia ^a	0.29	-0.12, 0.69	-0.24	-0.73, 0.25	-0.20	-0.51, 0.10	1.37	0.30
black-throated sparrow	0.12	-1.57, 1.81	0.94	-0.34, 2.22	0.48	-0.11, 1.07	0.51	0.62

^a F statistic and p -value generated from square root transformed data.

Table 6. Mean differences (burn - control), 90% confidence intervals, and outcomes of repeated measures ANOVA (time×treatment interactions) for mean relative abundance/25 ha of breeding grassland bird species on 7 control and 7 burn plots in June/July during 3 years of study on the Buenos Aires National Wildlife Refuge, Arizona.

Species	June/July 1996 (pre-burn)		June/July 1997 (1 yr post-burn)		June/July 1998 (2 yrs post-burn)		$F_{2,11}$	P
	\bar{d}	90% CI	\bar{d}	90% CI	\bar{d}	90% CI		
mourning dove ^a	0.17	-0.20, 0.53	0.39	-0.05, 0.84	0.94	0.03, 1.85	0.38	0.69
ash-throated flycatcher	0.12	-0.38, 0.62	0.51	-0.04, 1.07	0.29	-0.06, 0.64	0.35	0.71
cactus wren ^a	-0.76	-1.20, -0.32	-2.01	-3.13, -0.90	-0.54	-1.19, 0.12	6.08	0.02
northern mockingbird ^a	-0.16	-0.93, 0.62	-0.25	-0.59, 0.09	0.23	-0.46, 0.92	1.74	0.22
loggerhead shrike ^a	0.12	-0.25, 0.47	0.18	-0.36, 0.72	1.06	0.38, 1.73	2.52	0.12
Lucy's warbler ^a	0.92	-0.11, 1.95	0.19	-0.35, 0.74	0.66	0.14, 1.17	2.45	0.13
eastern meadowlark ^a	-0.74	-1.40, 0.08	-1.30	-2.05, -0.55	-0.89	-1.69, 0.08	0.62	0.56

^a F statistic and p -value generated from square root transformed data.

Table 6 cont.

Species	June/July 1996 (pre-burn)		June/July 1997 (1 yr post-burn)		June/July 1998 (2 yrs post-burn)		$F_{2,11}$	P
	\bar{d}	90% CI	\bar{d}	90% CI	\bar{d}	90% CI		
pyrrhuloxia ^a	0.20	-0.30, 0.70	-0.51	-1.03, 0.01	-0.29	-0.59, 0.01	3.36	0.07
blue grosbeak ^a	-0.04	-0.55, 0.48	-0.16	-0.54, 0.23	0.08	-0.16, 0.33	0.34	0.72
Botteri's sparrow ^a	0.16	-0.18, 0.49	-0.99	-1.34, -0.63	-0.77	-1.19, -0.36	8.18	0.01
Cassin's sparrow	0.05	-0.76, 0.85	-0.76	-1.26, -0.27	-1.69	-2.49, -0.90	3.66	0.06
black-throated sparrow	-0.63	-1.98, 0.72	1.14	-0.18, 2.47	2.57	0.67, 4.47	2.92	0.10

^a F statistic and p -value generated from square root transformed data.

Table 7. Magnitude (% and absolute), direction, and duration of trends in relative abundance of breeding species for either time period in 1997 and 1998 following a spring prescribed burn on the Buenos Aires National Wildlife Refuge, Arizona. Species are listed in order of decreasing magnitude of change in relative abundance.

Species	Shrub-dependent?	1997 (1 yr post-burn)		1998 (2 yrs post-burn)	
		Magnitude of change	Direction of change	Magnitude of change	Direction of change
mourning dove	N	955% (2.7)	∧	433% (1.2)	∧
cactus wren	N	619% (1.2)	∨	142% (0.4)	∧
Botteri's's sparrow	N	288% (1.2)	∨	229% (0.9)	∨
Cassin's sparrow	N	-	-	217% (1.7)	∨
loggerhead shrike ^a	Y	-	-	148% (1.0)	∧
pyrrhuloxia	Y	122% (0.5)	∨	-	-
black-throated sparrow	Y	-	-	103% (3.2)	∧
eastern meadowlark ^a	N	51% (0.6)	∨	-	-
Lucy's warbler ^a	Y	41% (0.7)	∨	-	-

^a Suggestive, but statistically inconclusive, evidence of trend.

Table 8. Total number detected and residence status of grassland bird species observed during 164 winter bird surveys (February - March, 1997 and 1998) on the Buenos Aires National Wildlife Refuge, Arizona.

Species	Total number detected	Residence status ^a
vesper sparrow (<i>Pooecetes gramineus</i>)	333	WR
black-throated sparrow (<i>Amphispiza bilineata</i>)	217	YR
eastern meadowlark (<i>Sturnella magna</i>)	201	YR
verdin (<i>Auriparus flaviceps</i>)	102	YR
cactus wren (<i>Campylorhynchus brunneicapillus</i>)	100	YR
Brewer's sparrow (<i>Spizella breweri</i>)	82	WR
grasshopper sparrow (<i>Ammodramus savannarum</i>)	71	YR ^c
loggerhead shrike (<i>Lanius ludovicianus</i>)	54	YR
ladder-backed woodpecker (<i>Picoides scalaris</i>)	44	YR
canyon towhee (<i>Pipilio fuscus</i>)	42	YR
Bewick's wren (<i>Thryomanes bewickii</i>)	36	YR
white-crowned sparrow (<i>Zonotrichia leucophrys</i>)	23	WR
horned lark (<i>Eremophila alpestris</i>)	23	YR
vermillion flycatcher (<i>Pyrocephalus rubinus</i>)	21	YR
other ^b	76	
Total	1,425	

^a Residence status: YR = year-round; WR = winter resident.

^b Combined count of 12 species with ≤ 10 observations each.

^c Distinct winter and summer populations.

Table 9. Mean differences (burn - control), 90% confidence intervals, and outcomes of repeated measures ANOVA (time×treatment interactions) for mean relative abundance/25 ha of wintering grassland bird species on 7 control and 7 burn plots in February/March during 2 years of the study on the Buenos Aires National Wildlife Refuge, Arizona.

Species	Feb./March 1997 (pre-burn)		Feb./March 1998 (post-burn)		$F_{1,12}$	P
	\bar{d}	90% CI	\bar{d}	90% CI		
ladder-backed woodpecker ^a	0.09	-0.08, 0.26	0.59	0.09, 1.09	5.88	0.03
cactus wren ^a	-0.43	-0.84, -0.01	-1.04	-1.70, -0.37	3.90	0.07
loggerhead shrike	0.13	-0.14, 0.40	0.18	-0.26, 0.62	0.04	0.83
eastern meadowlark ^a	-1.03	-1.74, -0.32	-1.87	-3.62, -0.12	1.32	0.27
verdin ^a	0.57	0.18, 0.95	0.35	-0.16, 0.86	1.56	0.24
canyon towhee	-0.07	-0.40, 0.25	-0.10	-0.39, 0.18	0.02	0.88
grasshopper sparrow ^a	-0.76	-1.61, 0.09	-0.09	-0.17, -0.01	1.17	0.30

^a F statistic and p -value generated from square root transformed data.

Table 9 cont.

Species	Feb./March 1997 (pre-burn)		Feb./March 1998 (post-burn)		$F_{1,12}$	P
	\bar{d}	90% CI	\bar{d}	90% CI		
vesper sparrow	-0.99	-2.25, 0.27	0.82	-0.12, 1.76	3.29	0.09
black-throated sparrow	-0.40	-1.20, 0.39	0.15	-0.40, 0.69	1.15	0.30
brewer's sparrow	-0.23	-0.98, 0.52	-0.14	-0.44, 0.16	0.04	0.85

^a F statistic and p -value generated from square root transformed data.

Appendix A. Percent cover (mean, SE) of vegetation variables on 7 control plots in 1997 (pre-burn) and 1998 (post-burn) on the Buenos Aires National Wildlife Refuge, Arizona. Grasses and forbs were measured basally and shrubs and sub-shrubs were measured aerially.

Vegetation variable	Control plots 1997 (pre-burn)		Control plots 1998 (post-burn)	
	\bar{Y}	SE	\bar{Y}	SE
total grasses	3.35	0.29	2.93	0.32
exotic grass	1.80	0.22	1.68	0.24
native grass	1.55	0.33	1.26	0.29
annual grass	0.00	0.00	0.13	0.01
forbs	0.17	0.04	1.05	0.15
sub-shrubs	2.41	1.36	2.91	1.26
mesquite	6.67	2.12	7.18	1.73
bare ground	87.98	2.56	86.51	2.54

total grasses = native and exotic grasses

native grasses = *Digitaria californica* (25.1%), *Bouteloua* spp. (23.7%),
Andropogon barbinodis (22.6%), *Eragrostis intermedia* (16.4%),
Panicum sp.(5.0%), *Aristida* spp. (4.8%), *Heterotheca contortous*
(<2.5%), *Setaria* sp.(<2.5%), *Chloris* sp.(<2.5%)

exotic bunchgrass = *Eragrostis lehmanniana*

annual grass = *Vulpia octoflora*

forbs = assorted spp.

mesquite = *Prosopis velutina*

sub-shrubs = *Isocoma tenuisecta* (49.3%), *Gutierrezia sarothrae* (50.7%).

Appendix B. Percent cover (mean, SE) of vegetation variables on 7 burn plots in 1997 (pre-burn) and 1998 (post-burn) on the Buenos Aires National Wildlife Refuge, Arizona. Grasses and forbs were measured basally and shrubs and sub-shrubs were measured aerially.

Vegetation variable	Burn plots 1997 (pre-burn)		Burn plots 1998 (post-burn)	
	\bar{Y}	SE	\bar{Y}	SE
total grasses	2.81	0.18	1.61	0.19
exotic grass	2.46	0.18	1.39	0.18
native grass	0.36	0.03	0.22	0.04
annual grass	0.00	0.00	0.21	0.05
forbs	0.10	0.04	0.48	0.06
sub-shrubs	6.20	0.88	1.02	0.33
mesquite	8.16	2.16	4.66	1.19
bare ground	83.74	2.17	91.97	1.20

total grasses = native and exotic grasses

native grasses = *Aristida* spp. (30.2%), *Digitaria californica* (28.2%), *Bouteloua* spp. (17.2%), *Heterotheca contortous* (9.4%), *Andropogon barbinodis* (6.8%), *Setaria* sp. (6.4%), *Eragrostis intermedia* (<2.5%)

exotic grass = *Eragrostis lehmanniana*

annual grass = *Vulpia octoflora*

forbs = assorted spp.

mesquite = *Prosopis velutina*

sub-shrubs = *Isocoma tenuisecta* (57.9%), *Gutierrezia sarothrae* (42.1%)

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