

ABUNDANCE OF NON-BREEDING HORNED LARKS AND CHESTNUT-COLLARED LONGSPURS ON GRAZED AND RESTED SEMIARID GRASSLAND

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ABSTRACT—We counted birds monthly from October through April of 1999–2000 and 2000–2001 on regularly grazed and rested (since 1973) semiarid grassland of central New Mexico. Horned larks (*Eremophila alpestris*) and chestnut-collared longspurs (*Calcarius ornatus*) accounted for 66% and 10% of all birds detected, respectively. We examined variation in counts of these species relative to grazing history, site, and vegetation characteristics. We used an information-theoretic approach to evaluate a set of 11 candidate models to determine which best described the variation in abundance of horned larks and chestnut-collared longspurs. There was little agreement between models for the 2 species or the 2 years. While grazing was a factor in the most likely models of abundance in the 1999–2000 sampling season for both bird species, this factor was not an important variable in explaining abundance in 2000–2001. These results suggest that abundance of horned larks and chestnut-collared longspurs are not as closely tied to grazing use during the non-breeding season in the semiarid grasslands of the northern Chihuahuan Desert, which contrasts with literature from breeding sites in the shortgrass prairie.

RESUMEN—Contamos las aves cada mes de octubre a abril de 1999–2000 y 2000–2001 en pastizales semiáridos de Nuevo México central que fueron pastoreados con regularidad o sin pastorear (desde 1973). Las alondras cornudas (*Eremophila alpestris*) y los escribanos de collar (*Calcarius ornatus*) fueron 66% y 10% de las aves vistas, respectivamente. Examinamos la variación en los conteos de estas especies con relación a la historia del pastoreo, el sitio y las características vegetales. Usamos un enfoque información-teórico para evaluar un grupo de 11 modelos para determinar cuál describió mejor la variación en la abundancia de las alondras cornudas y de los escribanos de collar. Hubo poco acuerdo entre los modelos de las dos especies o los dos años. Aunque el pastoreo fue un factor para los modelos más probables de abundancia durante la temporada de la muestra 1999–2000 para ambas especies de aves, este factor no fue una variable importante en explicar la abundancia en 2000–2001. Estos resultados sugieren que la abundancia de las alondras cornudas y de los escribanos de collar no se vincula con el pastoreo durante la temporada no reproductiva en los pastizales semiáridos en el norte del desierto Chihuahuan, y esto contrasta con la literatura proveniente de los sitios reproductivos en la pradera de hierba corta.

Semiarid grasslands in the western USA typically support an avifauna of 2 to 6 dominant bird species that are widely distributed (Wiens, 1973, 1974). These species tend to be either grassland endemics (Mengel, 1970) or obligates (Knopf, 1994; Vickery et al., 1999). During the non-breeding season in the shortgrass prairie-Chihuahuan Desert transition zone of central New Mexico, the numerically dominant bird species are horned lark (*Eremophila alpe-*

tris) and chestnut-collared longspur (*Calcarius ornatus*).

Over 70% of the land area in the western USA is used for livestock grazing, including 87% of New Mexico (Engle and Bidwell, 2000). Debate over impacts of current grazing practices on western grasslands has become increasingly contentious (Noss, 1994; Brown and McDonald, 1995; Donahue, 1999; Curtin, 2002). Consequently, there is an increasing

need for good description of current and historical impacts of grazing, if any, on these grasslands and the avifauna they support.

The majority of research on semiarid grassland birds has focused on breeding populations (Vickery et al., 1999:23), which has been valuable for understanding effects of habitat management on reproductive success and population dynamics. However, similar understanding of relationships between grazing management and non-breeding bird abundance and distribution has lagged far behind. Non-breeding avifauna of semiarid grasslands remains poorly quantified, even at the level of distribution and abundance (Luekering and Bradley, 1997). For example, we know of no study that has quantified the distribution and abundance of non-breeding horned larks and chestnut-collared longspurs in desert grasslands. There have, however, been numerous studies of these relationships during the breeding season (e.g., in shortgrass prairie, Kantrud, 1981).

As is typical of southwestern rangelands, the number of domestic livestock on open range in New Mexico reached its all-time peak at 8.9 million animal units in 1888 (Wildeman and Brock, 2000:18). Nearly all grasslands in New Mexico are grazed by domestic livestock and have been for hundreds of years, but the short-lived and ecologically unsustainable cattle boom of the 1880s was a particularly important event in the history of these grasslands. Erosion of 10 to 15 cm of topsoil, interruption of fire regimes, and acceleration of woody plant encroachment were all byproducts of the severe overgrazing that ended when 50 to 85% of the livestock died in the drought of 1891–1893 (Wildeman and Brock, 2000:18–19). As a consequence of the cattle boom, grasslands were degraded and shrublands expanded throughout the arid Southwest. The resultant increase in shrub-dominated landscapes has since been a persistent feature of the American Southwest (Bestelmeyer et al., 2003; Bestelmeyer et al., 2004). Despite the probable ubiquity of historical grazing effects, arid-grassland bird research is rarely designed to distinguish these effects from those of current grazing management.

This study was designed to describe the effect of current grazing management on abundance of horned larks and chestnut-collared

longspurs. We analyzed counts of horned larks and chestnut-collared longspurs on grazed and rested grasslands in the transition zone between the shortgrass prairie and Chihuahuan Desert grasslands. Previous research suggests that ongoing grazing creates vegetation structure and composition that is associated with high abundance of these grassland-obligate birds (Wiens, 1973; Beason, 1995; Hill and Gould, 1997; Pidgeon et al., 2002). Therefore, we expected the abundance of these birds to be positively associated with ongoing grazing. We evaluated the relative importance of recent grazing in comparison to measures of vegetation structure using an information-theoretic approach to model selection.

METHODS—Study Area—Sevilleta National Wildlife Refuge (hereafter, Sevilleta NWR) is about 80 km south of Albuquerque, New Mexico at the northern edge of the Chihuahuan Desert, where it meets Great Plains and Great Basin biotic communities (Gosz, 1992). The 91,500 ha of the refuge span the width of the Rio Grande Valley. Previously, the refuge was a privately owned cattle ranch, which The Nature Conservancy acquired from the Campbell Foundation in 1973 and subsequently donated to the United States Fish and Wildlife Service (USFWS, 1998). No livestock grazing has been permitted on the refuge since 1973. The study sites range from loamy and sandy soils, are between 1,500 and 1,600 m in elevation, and lie within predominantly open grassland vegetation. The area receives 250 mm of precipitation per year on average, most (about 150 mm) during the monsoon season (July through September).

We compared bird numbers in grazed plots to that in plots rested from grazing since 1973. The rested plots were located on the Sevilleta NWR and the grazed plots were on adjacent private or Bureau of Land Management (BLM) land. Grazed plots were stocked primarily in late summer, fall, and winter, although we saw some cattle year-round. The plots were dominated by black grama (*Bouteloua eriopoda*) and blue grama (*Bouteloua gracilis*) grasses. Other common grasses included galleta grass (*Pleuraphis jamesii*), dropseeds and sacatons (*Sporobolus*), threeawns (*Aristida*), and love grasses (*Muhlenbergia*). The dominant shrubs were creosote bush (*Larrea tridentata*), western honey mesquite (*Prosopis glandulosa*), Mormon tea (*Ephedra viridis*), four-winged saltbush (*Atriplex canescens*), winterfat (*Krascheninnikovia lanata*), tree cholla (*Opuntia imbricata*), and small soapweed (*Yucca glauca*).

Field Methods—We sought open grasslands dominated by perennial grass cover with relatively few

shrubs. We established 60 3-ha (173×173 m) plots and censused them (primarily by DLH) on a monthly basis from October 1999 through April 2000 and from October 2000 through April 2001, for a total of 420 plot surveys in each sampling season. Half of these plots were located on Mackenzie Flats at the northern boundary of the Sevilleta NWR ($34^{\circ}70'N$, $106^{\circ}65'W$) and were divided into grazed (15 on adjacent private land) and rested (15) plots. The other 30 plots were located at South Gate about 20 km south of Mackenzie Flats near the southern boundary of the Sevilleta NWR ($34^{\circ}20'N$, $106^{\circ}80'W$). The South Gate plots were also divided into rested (15) and grazed plots (15 on BLM land). At both sites, all plots were at least 300 m apart and at least 300 m from the refuge boundary fence that separated grazed from rested grassland. Mackenzie Flats typically receives about 15 mm more precipitation per year than South Gate (USFWS, 1998).

We censused each plot for 20 minutes between 0600 and 1100 h by systematically moving through the plot in a zigzag pattern. We employed this fixed-area search method rather than a sampling point approach because horned larks and chestnut-collared longspurs often do not vocalize from the ground. Only birds detected on the ground and identified to species were included in the analysis. The surveyor flushed most birds counted on plots. It is possible that some birds were counted more than once, although we attempted to keep track of birds that were counted and found that, when flushed, they usually did not return to the plot during the census. Moreover, we think it unlikely that the degree of sampling error associated with our counts is biased among plots.

We collected vegetation measurements on all 60 plots in February or March 2000 and 2001. All measurements were collected from a diagonal that bisected each plot. At each of 10 equally spaced sampling points along the diagonal, we established a perpendicular transect and measured vegetation characteristics at 10 points spaced at 1-m intervals on these transects. The ground cover was recorded by point intercept method at 1-m intervals in 1 of 5 categories: bare ground, litter, grass, shrub, and forb. We identified grasses, shrubs, and most forbs to species. At the beginning and end of each transect, we estimated the average grass height with a measuring rod and counted the number of contacts by grass with a 6-mm-diameter dowel held perpendicular to the ground (hereafter called structural grass diversity; Wiens, 1973). We counted shrubs by species and height class (0.5-m intervals) in a 0.04-ha circle centered at each of the 10 sampling points; shrubs were counted only in 2000.

Data Analysis—Our primary approach to data analysis was to formulate a limited set of general linear models that we thought were most likely to ex-

plain the distribution and abundance of horned larks and chestnut-collared longspurs on Sevilleta NWR (11 models). Because the dependent variable was birds counted, we initially fit these models assuming a Poisson distribution. However, these Poisson models resulted in substantial over-dispersion ($\hat{c} = \chi^2/df > 10$; Burnham and Anderson, 2002). Consequently, we assumed a negative binomial distribution, which we found to better fit the data ($\hat{c} < 1.5$; White and Bennetts, 1996). Because we did not randomly select plots or randomly assign them to grazing treatments, a hypothesis-testing framework was unwarranted (Johnson, 1999; Anderson et al., 2000). Rather, we applied an information-theoretic approach to model selection (Burnham and Anderson, 2002), whose terminology and calculations we followed throughout. In brief, we used the log-likelihood value for each model to calculate Akaike's Information Criterion with a correction for small sample size (AIC_c). We also employed a correction for over-dispersion ($QAIC_c$). We considered models with $\Delta QAIC_c$ values < 2 to be well supported and those with $\Delta QAIC_c < 4$ to have some support. Those models with $\Delta QAIC_c$ values > 4 were treated as unsupported models (Burnham and Anderson, 2002: 70). We interpreted Akaike weights for each model (w) as likelihood of a model being the best approximation of the data.

Monthly counts on individual plots were not independent (i.e., they were autocorrelated). Rather than choosing one monthly count to represent each plot, we summed the total number of horned larks and chestnut-collared longspurs counted on each plot over the 7 months of each sampling season. We used this sum as the dependent variable in our general linear models. We constructed 11 models with various combinations of 7 variables to describe processes that we thought might drive the distribution and abundance of these grassland birds. The variables were combinations of: site (Mackenzie Flats or South Gate), grazing history (grazed or rested), and percent bare ground, percent grass cover, structural grass diversity, and total number of shrubs per ha. We expected that horned larks and chestnut-collared longspurs would be more common where there was: 1) grazing; 2) more bare ground; 3) low percent basal grass cover; 4) short grass; 5) low grass structural diversity; and 6) low shrub density. To evaluate whether any of the 11 models were useful, we included a null model in which we fit only the intercept and no independent variables (Table 1).

RESULTS—Vegetation Description—Grazed and rested grasslands differed in both grass and shrub components of vegetation structure. Percent grass cover tended to be greater on rested than on grazed grassland (Table 2). In addi-

TABLE 1—General linear models constructed to explain variation in of horned larks (*Eremophila alpestris*) and chestnut-collared longspurs (*Calcarius ornatus*) wintering on grazed and rested grasslands at Sevilleta National Wildlife Refuge, New Mexico.

Model name	Model construction
Global	site + grazing + site $\times$ grazing + % bare ground + % grass cover + grass height + grass structural complexity + shrub density + intercept
Site and Grazing	site + grazing + site $\times$ grazing + intercept
Site, Grazing, and Grass	site + grazing + site $\times$ grazing + % grass cover + grass height + grass structural complexity + intercept
Site, Grazing, and Bare Ground	site + grazing + site $\times$ grazing + % bare ground + intercept
Site, Grazing, and Shrub	site + grazing + site $\times$ grazing + shrub density + intercept
Site	site + intercept
Grazing	grazing + intercept
Not Grazing	site + % bare ground + % grass cover + grass height + grass structural complexity + shrub density + intercept
Grass	% grass cover + intercept
Bare	% bare ground + intercept
Shrub	% shrubs + intercept
Intercept Only	intercept

tion to having greater grass cover, rested grasslands also tended to have taller grass and higher structural grass diversity at both Mackenzie Flats and South Gate. The patterns between sites were similar between the 2 sampling seasons. There was little difference in percent bare ground in 1999–2000 sampling season. However, there appeared to be relatively more bare ground on the grazed grasslands in the 2000–2001 season.

Patterns in shrub density varied between Mackenzie Flats and South Gate. Overall, there was a slight tendency for higher shrub density on rested than grazed plots, but there was much variation in the data (Table 2). There were greater densities of Mormon tea and mesquite on grazed plots than on rested plots at South Gate (Table 3). In contrast, there tended to be greater densities of four-winged saltbush and winterfat on rested than on grazed plots (Table 4). At Mackenzie Flats, only Mormon tea differed obviously in density with respect to grazing; it was more abundant on rested plots (Table 3).

Bird Counts—Of all birds detected, 65% and 68% were horned larks during 1999–2000 and 2000–2001, respectively. Chestnut-collared longspurs comprised 10% of all detections in each sampling season. No other species accounted for >5% of observations in either year. Although horned larks and chestnut-col-

lared longspurs were the numerically dominant species, they were still sparsely distributed, with densities usually <1 bird/ha (Table 2). Horned larks seemed to be somewhat more common on the grazed plots than rested plots at Mackenzie Flats. There was no discernable pattern at South Gate.

General Linear Models—Of the 11 models that we examined, no single model received consistent support between sampling seasons or bird species (Table 4). Abundance of horned larks in the 1999–2000 sampling season was best explained by the SITE AND GRAZING model. The GLOBAL model was nearly as well supported, but no other model had a $\Delta\text{QAIC}_c < 4$. In the October 2000 through April 2001 sampling season the NOT GRAZING model was the best explanation of horned lark abundance, while the SITE, GRAZING, AND GRASS model and GLOBAL model also seemed to be relatively good fits to the data. Models used to explain variation in chestnut-collared longspur data were more equivocal. In each sampling season, 7 models from the candidate set of 11 models had ΔQAIC_c that were <4. For the 2000–2001 sampling season the best model set included the INTERCEPT ONLY model (Table 4).

DISCUSSION—*Patterns in Vegetation Structure*—Reduced basal grass cover on grazed grasslands

TABLE 2—Median (minimum–maximum) values for bird counts on 3-ha plots and measures of vegetation structure on those plots at Sevilleta National Wildlife Refuge, New Mexico. All sample sizes are 15 3-ha plots.

Variable	1999–2000				2000–2001			
	Mackenzie Flats		South Gate		Mackenzie Flats		South Gate	
	Grazed	Rested	Grazed	Rested	Grazed	Rested	Grazed	Rested
HOLA ^a	24 (5–65)	7 (1–17)	4 (0–16)	11 (0–65)	5 (0–99)	5 (0–41)	2 (0–44)	4 (0–37)
CCLO ^b	1 (0–16)	2 (0–19)	0 (0–4)	0 (0–11)	0 (0–15)	0 (0–13)	0 (0–42)	0 (0–9)
Structural								
Grass diversity	1.1 (0.6–1.8)	1.5 (0.6–4.2)	1.0 (0.1–2.8)	1.6 (0.7–4.1)	0.7 (0–1.7)	1.1 (0.4–2.2)	1.0 (0.4–1.9)	1.3 (0.6–2.1)
Grass height (cm)	8 (5–17)	15 (11–49)	5 (4–7)	12 (9–17)	6 (4–9)	13 (10–17)	5 (4–6)	11 (6–14)
Shrubs ^c (number/ha)	108 (25–423)	490 (43–1048)	55 (18–2043)	90 (8–700)				
Percent bare ground	26 (11–61)	25 (12–52)	34 (23–59)	32 (14–51)	33 (17–45)	28 (18–39)	44 (34–56)	38 (25–55)
Percent grass cover	18 (11–27)	24 (17–30)	15 (7–25)	26 (16–32)	14 (10–19)	21 (10–24)	18 (4–25)	20 (12–26)

^a HOLA = Sum total number of horned larks (*Eremophila alpestris*) counted in 7 monthly surveys from October through April.
^b CCLO = Sum total number of chestnut-collared longspurs (*Calcarius ornatus*) counted in 7 monthly surveys from October through April.
^c Shrubs were counted only in the 1999–2000 sampling season.

TABLE 3—Median (minimum–maximum) number of shrubs/ha on grazed and rested plots at 2 sites on the Sevilleta National Wildlife Refuge, New Mexico, and adjacent lands, as measured in the winter of 1999–2000.

Species	Mackenzie Flats		South Gate	
	Grazed	Rested	Grazed	Rested
Four-winged saltbush (<i>Atriplex canescens</i>)	0 (0–25)	0 (0–8)	0 (0–130)	50 (0–345)
Mormon tea (<i>Ephedra viridis</i>)	80 (0–360)	276 (0–1,018)	15 (0–988)	0 (0–263)
Winterfat (<i>Krascheninnikovia lanata</i>)	0 (0–30)	0 (0–8)	0 (0–3)	0 (0–420)
Creosote (<i>Larrea tridentata</i>)	0 (0–13)	0 (0–15)	0 (0–38)	0 (0–25)
Tree cholla (<i>Opuntia imbricata</i>)	0 (0–5)	3 (0–28)	3 (0–18)	3 (0–45)
Western honey mesquite (<i>Prosopis glandulosa</i>)	0 (0–5)	0 (0–3)	5 (0–28)	0 (0–3)
Small soapweed (<i>Yucca glauca</i>)	13 (0–73)	18 (0–195)	13 (0–213)	8 (0–328)

TABLE 4—Models that best approximated count data from horned larks (*Eremophila alpestris*) and chestnut-collared longspurs (*Calcarius ornatus*) at Sevilleta National Wildlife Refuge, New Mexico, in the winters of 1999–2000 and 2000–2001. Of 11 models included as candidates (Table 1), only those models with $\Delta\text{QAIC}_c < 4$ are shown.

Model ^a	Log likelihood	$\hat{c}^b$	K ^c	QAIC _c ^d	ΔQAIC_c^e	w ^f
Horned lark: 1999–2000						
Site and Grazing*	1,644.41	1.06	5	–3,093.26	0.00	0.47
Global	1,645.84	1.06	10	–3,092.69	0.57	0.35
Horned lark: 2000–2001						
Not Grazing*	1,642.90	1.35	8	–2,898.39	0.00	0.51
Site, Grazing, and Grass	1,641.19	1.35	8	–2,895.36	3.03	0.11
Global	1,643.03	1.35	10	–2,895.35	3.04	0.11
Chestnut-collared longspur: 1999–2000						
Site, Grazing, and Bare Ground*	82.75	1.13	6	–116.19	0.00	0.21
Not Grazing	84.43	1.13	8	–115.68	0.51	0.16
Site, Grazing, and Shrubs	82.38	1.13	6	–115.64	0.55	0.16
Site	79.30	1.13	3	–115.08	1.11	0.12
Site, Grazing, and Grass	83.87	1.13	6	–114.85	1.34	0.11
Site and Grazing	80.59	1.13	5	–114.39	1.80	0.09
Global	85.71	1.13	10	–114.31	1.88	0.08
Chestnut-collared longspur: 2000–2001						
Shrub*	139.29	0.59	3	–277.38	0.00	0.34
Bare Ground	138.24	0.59	3	–275.28	2.11	0.12
Grass	138.21	0.59	3	–275.22	2.17	0.12
Intercept Only	138.07	0.59	2	–274.94	2.44	0.10
Grazing	138.60	0.59	3	–274.80	2.58	0.09
Area	138.29	0.59	3	–274.18	3.20	0.07
Area, Grazing, and Shrub	140.10	0.59	6	–273.80	3.58	0.06

^a Model structure is defined in Table 1.

^b $\hat{c} = \Pi^2/df$. Values > 1 indicate over-dispersion of data.

^c K = number of parameters in the model + 2.

^d QAIC_c = Akaike's Information Criterion (AIC) correct for overdispersion ($\hat{c}$).

^e $\Delta\text{QAIC}_c = \text{QAIC}_c$ of each model minus the minimum QAIC_c among the models tested.

^f w = Akaike weights for each model. Interpreted as likelihood of a model being the best approximation of the data.

* Model with the best fit to the data as indicated by $\Delta\text{QAIC}_c = 0.0$.

relative to rested grasslands was more consistent with previous results from desert grasslands (Gardner, 1950; Bock and Bock, 1993; Holecheck et al., 1994; Loftin et al., 2000) than with those from the shortgrass prairie (Milchunas et al., 1989; Lauenroth et al., 1994). Relationships between recent grazing and shrub density were variable at our 2 sites. At Mackenzie Flats, there was little apparent association between densities of individual shrub species and grazing. Total shrub density, however, seemed to be greater on rested grasslands than on the recently grazed grasslands. The lack of clear association of shrub density with recent

grazing or the possibility that shrub density is greater on rested sites is more consistent with findings from shortgrass prairie (Milchunas et al., 1989; Bestelmeyer and Wiens, 2001) than Chihuahuan Desert.

At South Gate, there was a similar pattern. With the possible exception of Mormon tea and honey mesquite, shrubs tended to be denser on rested than on recently grazed plots. These patterns were driven primarily by trends in winter fat and four-winged saltbush, which were more abundant on rested plots than grazed plots at South Gate. The greater abundance of mesquite on grazed grasslands is con-

sistent with literature from the Chihuahuan Desert (Loftin et al., 2000) and southern Great Plains (Brown and Archer, 1989), where this species has become a primary problem for grassland managers (Engle and Bidwell, 2000).

Models of Bird Abundance—Our model selection procedures generally suggested that site and grazing effects were the most important variables in explaining variation in abundance of both horned larks and chestnut-collared longspurs. There were, however, exceptions to this pattern. For instance, model selection also suggested that vegetation structure in the form of both grass structure (horned larks in the 2000–2001 season) and shrub density (chestnut-collared longspurs in the 2000–2001 season) can be more important than effects of recent grazing. Moreover, at some times grazing was the least important explanatory variable (horned larks in the 2000–2001 season). However, it is important to note that in this instance our INTERCEPT ONLY model was well within our group of best models, suggesting that none of these models might have been particularly informative. That is, the best explanatory models were not substantially better at predicting the number of birds counted than a model with no independent variables.

As anticipated, horned larks tended to be more abundant on grazed than rested grassland at Mackenzie Flats in the 1999–2000 sampling season (Table 2). Also, horned larks were more abundant at Mackenzie Flats than at South Gate. There was no obvious pattern in the abundance of chestnut-collared longspurs at Mackenzie Flats or South Gate in either year, nor for horned larks at South Gate. Studies from the shortgrass prairie, mixed-grass prairie, and desert grassland have shown that the abundance of horned larks is positively associated with grazing (Wiens, 1973; Ryder, 1980; Bock et al., 1984; Hill and Gould, 1997).

Most studies have reported chestnut-collared longspurs to be positively associated with grazing during the breeding season (Owens and Myers, 1973; Maher, 1979; Kantrud, 1981), a relationship that has also been shown for non-breeding populations in Sonoran Desert grasslands (Bock and Bock, 1988). However, there also have been studies that reported longspurs to be neutral to grazing (Ryder, 1980; Kantrud and Kogoliski, 1982).

Overall, our results suggested that, while the

abundances of horned larks and chestnut-collared longspurs were related to recent grazing, this relationship was not consistent through time or between sites. We propose that the distribution of shrubs, which reflects historical grazing, climate, and fire (e.g., Buffington and Herbel, 1965), is an important correlate of the distribution of these 2 grassland obligate birds. We think this result illustrates the potentially complex interactions between historical and current grazing management. One interpretation of the low densities of obligate grassland birds in the Chihuahuan Desert is that large tracts of desert grassland are required to support populations during the non-breeding period. We advocate future studies that partition the effects of recent and historical grazing on vegetation and bird distributions. Continuing to ignore this distinction implies that current land condition is primarily the direct result of current management practices.

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