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Authors: Guzmán-Ojeda, Rocío Jazmín, Castillo-Gámez, Reyna Amanda, Macías-Duarte, Alberto, and Alvarado-Castro, José Andrés

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Rocío Jazmín Guzmán-Ojeda,* Reyna Amanda Castillo-Gámez, Alberto Macías-Duarte, and José Andrés Alvarado-Castro

Licenciatura en Ecología, Universidad Estatal de Sonora, Ley Federal del Trabajo S/N, Colonia Apolo, C.P. 83100, Hermosillo, Sonora, México (RJGO)
Departamento de Investigaciones Científicas y Tecnológicas de la Universidad de Sonora, Luis Donaldo Colosio S/N, Colonia Centro, C.P. 83000, Hermosillo, Sonora, México (RACG)
Cuerpo Académico de Recursos Naturales, Universidad Estatal de Sonora, Ley Federal del Trabajo S/N, Colonia Apolo, C.P. 83100, Hermosillo, Sonora, México (AMD, JAAC)
Correspondent: rocio.j.guzman@gmail.com

Abstract—Buffelgrass (*Cenchrus ciliaris*, syn. *Pennisetum ciliare*) is exotic to the Sonoran Desert and has caused drastic changes in this ecosystem. Yet, little research has documented the impact of buffelgrass on wildlife in the region, including small mammals. The objective of this study was to estimate abundance of native nocturnal rodents and characterize habitat structure in an area with native vegetation and buffelgrass prairies to assess the effect of this exotic grass on the diversity and population density of this community in the Sonoran Desert, located in the municipality of La Colorada, in central Sonora, Mexico. We trapped nocturnal rodents in a buffelgrass prairie site and in a site with native vegetation (with two trapping plots at each site), before and during the summer rainy season, and observed the differences in species composition between sites. We used capture-mark-recapture and distance sampling models to estimate population density of the two most common species, *Dipodomys merriami* and *Chaetodipus penicillatus*. Despite differences in habitat structural diversity between buffelgrass prairies and native vegetation, our results suggest that buffelgrass does not seem to affect dominance or population density of these two heteromyids, whereas evenness and species richness showed some differences between buffelgrass prairies and native vegetation sites.

Resumen—El zacate buffel (*Cenchrus ciliaris*, syn. *Pennisetum ciliare*) es una especie exótica al desierto Sonorense que ha ocasionado cambios drásticos en este ecosistema. Sin embargo, poco se ha documentado sobre el impacto del zacate buffel en la fauna silvestre de la región, incluyendo a los mamíferos pequeños. El objetivo del presente estudio fue estimar la abundancia de roedores nocturnos nativos y caracterizar la estructura del hábitat en un área con vegetación nativa y praderas de zacate buffel para evaluar el efecto de este pasto exótico sobre la diversidad y densidad poblacional de esta comunidad en el desierto Sonorense, ubicado en el municipio de La Colorada, en el centro de Sonora, México. Se realizaron trampeos en un sitio con pradera de zacate buffel y en un sitio con vegetación nativa. Se colocaron dos parcelas de trampeo en cada uno, antes y durante la temporada de lluvias de verano y se observaron las diferencias en la composición de especies entre sitios. Se utilizaron modelos de captura-marcaje-recaptura y muestreo por distancia para estimar la densidad poblacional de las dos especies más comunes, *Dipodomys merriami* y *Chaetodipus penicillatus*. A pesar de las diferencias en diversidad estructural de hábitat entre las praderas de zacate buffel y la vegetación nativa, nuestros resultados sugieren que no existe efecto del zacate buffel ni sobre la dominancia ni sobre la densidad poblacional de los dos heterómidos, mientras que la uniformidad y la riqueza de especies mostraron algunas diferencias entre los sitios de pradera de zacate buffel y vegetación nativa.

Humans introduced buffelgrass (*Cenchrus ciliaris*, syn. *Pennisetum ciliare*) to the state of Sonora in northern Mexico in the 1960s to increase forage availability in cattle ranches (Castellanos-Villegas et al., 2010). However, the high competitiveness and fire tolerance of buffelgrass, the bulldozing of native vegetation for its establishment, and dispersal facilitated by roadsides has transformed large portions of Sonoran Desert scrubland into exotic grassland, leading to the reduction of plant biomass and biodiversity (Van Devender et al., 1997; Búrquez and Martínez-Yrízar, 2006; Aguirre Muñoz and Mendoza Alfaro, 2009; Olsson et al., 2012; Morales-Romero et al., 2019).

Changes in plant communities and ecosystem function introduced by the establishment of buffelgrass in arid and semiarid lands are likely to have bottom-up effects on

animal communities. For instance, several studies have shown that buffelgrass has negative effects on wildlife in Australia and the United States. Native quail populations have declined in the southwestern United States due to habitat loss after establishing buffelgrass and other exotic grasses prairies for cattle (Kuvlesky et al., 2002). Buffelgrass has changed the composition of reptile, bird, and ant communities in Australia, even at a low ground-cover percentage (Eyre et al., 2009; Smyth et al., 2009; Schlesinger et al., 2020). Moreover, eradication of buffelgrass can improve vegetation and seed bank diversity, in addition to increasing seed mass availability (Wright et al., 2020). In Sonora, ant assemblages and the rufous-winged sparrow (*Peucaea carpalis*) seem resilient to changes in their habitats caused by the introduced buffelgrass (Franklin, 2012; Macías-Duarte et al., 2019). However, there is little information that documents the impact of buffelgrass on the wild fauna of the Sonoran Desert, and more specifically, on the mammalian fauna. Rodents often constitute the most important food item of carnivores in food webs. Therefore, adverse effects of disturbance by buffelgrass on rodent communities may have bottom-up effects in wildlife community assemblage. Even though we are not aware of published studies that focus on the direct effects of buffelgrass on desert rodent communities, there is evidence that areas with native vegetation are more favorable for rodents such as the fossorial Maritime pocket gopher in the southern United States (*Geomys personatus maritimus*; Cortez et al., 2015) and the Chacoan cavy in Argentina (*Pediolagus salinicola*; Marinaro and Grau, 2015) than areas with a combination of buffelgrass and other exotic grasses with native vegetation. Conversely, understanding mechanisms of habitat selection in nocturnal rodents in the Sonoran Desert ecosystems under disturbance by buffelgrass may allow predicting buffelgrass range expansion in North American deserts, as nocturnal desert rodents disperse and cache seeds (Sommers and Chesson, 2016).

Given that habitat complexity is positively associated with species diversity (MacArthur and MacArthur, 1961), including desert rodents (Rosenzweig and Winakur, 1969), we hypothesize that the reduced vegetation complexity produced by the establishment of buffelgrass decreases habitat diversity and subsequently decreases the diversity and abundance of nocturnal rodents in the Sonoran Desert ecosystem. In this regard, the objectives of this study were to (1) obtain abundance estimates of the members of the native nocturnal rodent community in native vegetation and buffelgrass prairies in the Plains of Sonora subdivision of the Sonoran Desertscrub (Brown, 1994) of Mexico, and (2) characterize and compare vegetation cover and composition between these two habitat types. We predicted that abundance and diversity of nocturnal rodents is lower in the buffelgrass prairies than in native vegetation because of the low structural diversity of the former.

MATERIALS AND METHODS—*Study Area*—The study area was located in the municipality of La Colorada, in central Sonora, Mexico (Fig. 1). The study area is a mosaic of buffelgrass prairies and native vegetation. The buffelgrass prairie site (buffelgrass site hereafter) was located on the El Pozo ranch (28°42′05.23″N, 110°34′11.14″W). This site is an old prairie where humans introduced buffelgrass into the area after a selective clearing that involved removing most of the native vegetation. The site with native vegetation (native vegetation site hereafter) was located on the El Cajon de la Uvalama ranch (28°41′07.97″N, 110°33′49.04″W), 2.5 km from the buffelgrass site. Our sites in native vegetation and buffelgrass prairies were located in a 22-km^2 and an 8-km^2 patch, respectively. Plots within the buffelgrass and native vegetation sites were approximately 300 m apart from each other. Native vegetation of the area includes mezquital and subtropical scrub (Instituto Nacional de Estadística, Geografía e Informática [INEGI], 2013). Mezquital or Sonoran Desert shrublands are dominated by legume trees such as desert ironwood (*Olneya tesota*), velvet mesquite (*Prosopis velutina*), and foothill paloverde (*Parkinsonia microphylla*), and shrubs such as brittlebush (*Encelia farinosa*), piojito (*Caesalpinia pumila*), and heartleaf limberbush (*Jatropha cardiophylla*), as well as columnar cacti. Climate is dry and semi-warm with cool winters (sensu García, 2004). Temperatures vary from 6 to 26°C from November to April, and from 20 to 38°C from May to October (INEGI, 2009*a*, 2009*b*).

Rodent Trapping—We estimated species abundance and species richness of the nocturnal rodent community by trapping at four plots: two in the buffelgrass site and two in the native vegetation site. Each trapping plot consisted of 42 traps arranged in six radial transects, separated by 10 m from each other in each transect (Fig. 1). Each plot had a diameter of 140 m, covering 1.54 ha per plot, 3.08 ha per site, and a total of 6.16 ha. We carried out the sampling biweekly from June to September 2014 with each trapping occasion consisting of two consecutive days (Table 1) setting Sherman® traps (H. B. Sherman Traps Inc., Tallahassee, Florida) in each plot. We sampled before (June) and during (July–September) the summer rainy season, to account for structural changes in vegetation cover dominance. We used three trap sizes: small (5.8 × 6.35 × 16.51 cm), medium (7.62 × 8.89 × 22.86 cm), and large (10.16 × 11.43 × 38.1 cm). Slade et al. (1993) documented that the trigger mechanism sensitivity of a trap is directly associated with its size. To mitigate the effect of this source of variation on capture probability, we distributed the number of traps of each size evenly across plots and randomly set them within each plot on the first night of consecutive trapping sessions. We set traps at sunset (1700–1900 h) and checked at sunrise (0500–0900 h). We baited traps with peanut butter and oatmeal. We regarded it as unnecessary to include bedding material as night temperatures were benign during the study period. We believe that thermal stress was minimal because captured individuals readily moved away after release. Total trapping effort was 1,806 trap-nights, computed as of the number of trapping nights times the constant number of traps used each night (42 traps; Table 1). We marked captured individuals with a numbered monel ear tag (Mouse Ear Tags, Style 1005-1, National Band and Tag Company, Newport, Kentucky) if the ear measured ≥10 mm, or we dyed a spot on the back of the individual with gentian violet if the ear measured <10 mm, and immediately released them after marking at the capture site. Gentian violet produced indistinctive individual marks. These activities were authorized by the
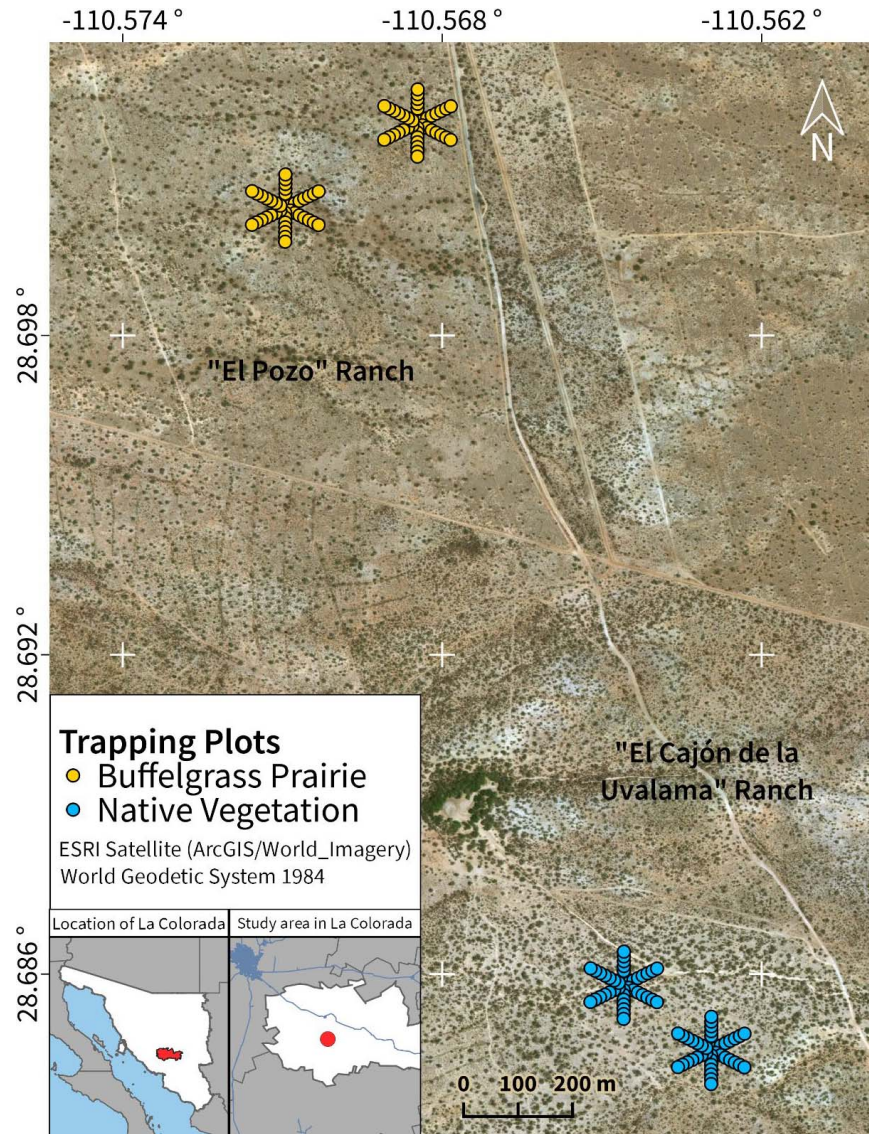
Fig. 1—Location of nocturnal rodent trapping plots set in buffelgrass prairie and native vegetation in the municipality of La Colorada, Sonora, Mexico. (Color version is available online.)

Secretariat of Environment and Natural Resources of Mexico (SEMARNAT) under the scientific collection permit Oficio No. SPGA/DGVS/04775/14.

Vegetation Cover—We used the line-intercept method (Canfield, 1941) to determine differences in vegetation composition and structure between native vegetation and buffelgrass sites. We traced a 100-m line that crossed the center of each trapping plot (from NE to SW) and we estimated ground cover per plant species as the percentage of the line covered per plant species. We sampled vegetation at the end of the dry season (4 July) and during the summer rainy season (10 August) of 2014.

Statistical Analysis—We estimated two diversity indexes from rodent trapping data for both native vegetation and buffelgrass sites: Shannon-Wiener's index (H' using $\log_2$; Shannon, 1948; Magurran, 1988) and Simpson's index, which indicates dominance (D; Simpson, 1949; Magurran, 1988). We also performed hypothesis tests to evaluate statistical differences in these indexes between the native vegetation and the buffelgrass sites:

Hutcheson's t test for Shannon-Wiener's index (Hutcheson, 1970) and Student's t test for Simpson's index (Simpson, 1949). We also estimated Pielou's evenness index ($J' = H'/H'_{max}$, where $H'_{max} = \log_2 S$ and S is the species richness; Pielou, 1966).

We estimated population density of the only two species with sufficient capture-recapture events: *Dipodomys merriami* (Merriam's kangaroo rat; 53 individuals marked, 32 recaptured) and *Chaetodipus penicillatus* (desert pocket mouse; 282 individuals marked, 150 recaptured). Because we marked *D. merriami* with ear tags and could identify each individual, we estimated population density from individual encounter histories from our sampling using the capture-mark-recapture model of Cormack-Jolly-Seber (Cormack, 1964; Jolly, 1965; Seber, 1965) in program MARK (White and Burnham, 1999). Due to the difficulty of identifying individuals using gentian violet, we estimated population density of *C. penicillatus* using the trapping web implementation of distance sampling (Buckland et al., 2001). We used distances from the center of each plot to traps

TABLE 1—Trapping effort per plot and sampling dates in a buffelgrass prairie and in native vegetation at the study area in the municipality of La Colorada, Sonora, Mexico, in 2014. 1 = nights when traps were set at the plot; 0 = nights when traps were not set at the plot.

| Date | | Trapping plot | | | |
| | | Buffelgrass | | Native vegetation | |
Month	Day	1	2	1	2
6	01	1	0	0	0
	21	1	1	0	0
	22	1	1	1	1
7	05	1	1	1	1
	06	1	1	1	1
	19	1	0	1	1
	20	1	1	1	1
8	02	1	1	1	0
	03	1	1	1	0
	16	1	1	0	0
	17	1	1	1	1
	30	1	0	1	1
	31	1	0	1	1
9	20	1	0	0	0
	21	1	0	1	0
Trap-nights per plot		630	378	462	336

TABLE 2—Absolute and relative frequencies of the nocturnal rodent species captured at the study area in the municipality of La Colorada, Sonora, Mexico. D = population density estimates (individuals·ha^{-1}); C = total counts in traps (individuals captured); % = percentage of total counts; — = population density could not be estimated due to low sample size.

| Family | Species | Native vegetation | | | Buffelgrass | | |
		D	C	%	D	C	%
Heteromyidae	Chaetodipus baileyi	—	2	1	—	8	4
	Chaetodipus penicillatus	12.3	156	88	13.4	126	70
	Dipodomys merriami	6.2	15	8	8.5	38	21
Cricetidae	Onychomys torridus	—	1	1	—	4	2
	Peromyscus eremicus	—	4	2	—	1	1
	Sigmodon arizonae	—	0	0	—	3	2
Total		—	178	100	—	180	100

with capture events to model a half-normal detection function using the package Distance (Miller, 2014) in program R (R Development Core Team, 2014).

RESULTS—*Rodent Trapping*—We captured 358 individuals from six species and two families (Table 2). In the family Heteromyidae, *C. penicillatus* had the greatest relative abundance with 78.8% (282 individuals), followed by *D. merriami* with 14.8% (53 individuals) and *Chaetodipus baileyi* (Bailey's pocket mouse) with 2.8% (10 individuals) of the captures. Family Cricetidae constituted only 3.6% of the total captures, which included *Peromyscus eremicus* (cactus mouse; five individuals), *Onychomys torridus* (southern grasshopper mouse; five individuals), and *Sigmodon arizonae* (Arizona cotton rat; three individuals). The latter species was only present in the buffelgrass site, whereas the former species had individuals captured in both sites. All individuals captured were solely found in one trapping plot in subsequent recaptures (i.e., there was no migration between trapping plots or between sites during the trapping period). We had more captures of *C. baileyi*, *D. merriami*, and *O. torridus* in the buffelgrass site than the native vegetation site (Table 2). The International Union for Conservation of Nature (IUCN) lists all species captured as Least Concern (Álvarez-Castañeda et al., 2016; Lacher et al., 2016, 2019; Linzey et al. 2016a, 2016b; Timm et al. 2016). Although *C. baileyi*, *C. penicillatus*, *D. merriami*, and *P. eremicus* have threatened subspecies in Mexico (NOM-059-SEMARNAT-2010), none were present in our study area.

Chaetodipus penicillatus had 88% and 70% of the relative frequencies in the native vegetation and buffelgrass sites, respectively, whereas *D. merriami* had 8% and 21% (Table 2). Thus, we captured 55.3% of *C. penicillatus* individuals in the native vegetation site, whereas we captured 71.7% of the individuals of *D. merriami* in the buffelgrass site. Relative abundance (counts) of these two species differed between buffelgrass and native vegetation sites (χ^2 proportion test, $\chi^2 = 14.5$, $df = 1$, $P < 0.001$). We captured only 22 new individuals in both sites (15 and 7 individuals of *C. penicillatus* and *D. merriami*, respectively) before the rainy season, whereas we captured 313 of the individuals during the rainy season (267 and 46 individuals of *C. penicillatus* and *D. merriami*, respectively; Fig. 2). We did not find differences in sex ratio in either site for *C. penicillatus* ($\chi^2 = 1.237$, $df = 1$, $P = 0.27$) and *Dipodomys merriami* ($\chi^2 = 1.297$, $df = 1$, $P = 0.25$).

Rodent Diversity—The Shannon-Wiener index was $H' = 1.30$ ($H'_{max} = 2.58$) in the buffelgrass site and $H' = 0.71$ ($H'_{max} = 2.32$) in the native vegetation site. Hutcheson's t test showed that the diversity was greater in the buffelgrass prairie site ($t = 4.00$, $df = 359.38$, $P < 0.001$). Furthermore, evenness was higher at the buffelgrass sites ($J' = 0.51$) than at the native vegetation sites ($J' = 0.31$). Simpson's index was $D = 0.53$ in the buffelgrass site and $D = 0.77$ in the native vegetation site, showing no statistical difference between sites (Student's t test, $t = 1.57$, $df = 9$, $P = 0.15$).

Population Density—We estimated a population density of 12.3 individuals·ha^{-1} for *C. penicillatus* in the native vegetation site (95% confidence interval [95% *CI*] of 6.2–26 individuals·ha^{-1}) and 13.4 individuals·ha^{-1} in the buffelgrass site (95% *CI*: 5.6–37.2 individuals·ha^{-1}). The population density of *C. penicillatus* did not differ between native vegetation and buffelgrass sites (Wald test, $t = -0.34$, $df = 4$, $P = 0.75$). We estimated an average population density for *D. merriami* of 6.2 individuals·ha^{-1}

Vegetation cover was greater in the natural vegetation site than in the buffelgrass site during this period. The effects of summer rainfall on vegetation physiognomy were evident in both habitat types. Estimates of vegetation cover increased by 1.63 and 2.56 times from dry to rainy season in native vegetation and buffelgrass sites, respectively. However, vegetation cover was similar between the native vegetation site and buffelgrass site during the rainy season with 99.3 and 98.05% ground cover, respectively, due to perennial foliage growth, and the presence of annual species and seedlings in both sites.

We did not find buffelgrass in the native vegetation site, but found 10 plant species typical of the subtropical desertscrub in the area (Table 3). During the dry season, *Olneya tesota* was the dominant species occupying 22.05% of the cover, followed by *J. cardiophylla* with 12.40%, Arizona pencil cholla (*Cylindropuntia arbuscula*) with 7.41%, bitter snakewood (*Condalia globosa*) with 6.47%, and Arizona mimosa (*Mimosa laxiflora*) with 6.23%. Five species (*Cylindropuntia fulgida*, *Merremia palmeri*, *Havardia mexicana*, *Lophocereus schottii,* and *Randia thurberi*) occupied the remaining 6.2% of the covered ground. We only found five perennial plant species in the buffelgrass site during the dry season: buffelgrass (*Cenchrus ciliaris*) with 35.12% of ground cover, followed by the honey mesquite (*Prosopis glandulosa*) with 1.25%, *Olneya tesota* with 0.83%, *M. laxiflora* with 0.72%, and jumping cholla (*Cylindropuntia fulgida*) with 0.28%.

During the rainy season, annual plants and seedlings occupied most of the cover in the native vegetation site (46.62%), followed by *Olneya tesota* (16.64%), *M. laxiflora* (10.82%), *R. thurberi* (9.68%), and *J. cardiophylla* (9.46%). The remaining 6.07% of the cover was occupied by *C. arbuscula*, *Cylindropuntia fulgida*, *H. mexicana*, *Condalia globosa*, and *L. schottii*. Buffelgrass was the dominant species in the buffelgrass site, with 55.50% of the relative cover, followed by annuals and seedlings (36.76%). *Olneya tesota*, *M. laxiflora*, *P. glandulosa*, *J. cardiophylla*, Berlandier's wolfberry (*Lycium berlandieri*), and *Cylindropuntia fulgida* occupied the remaining 5.79% of the covered ground.
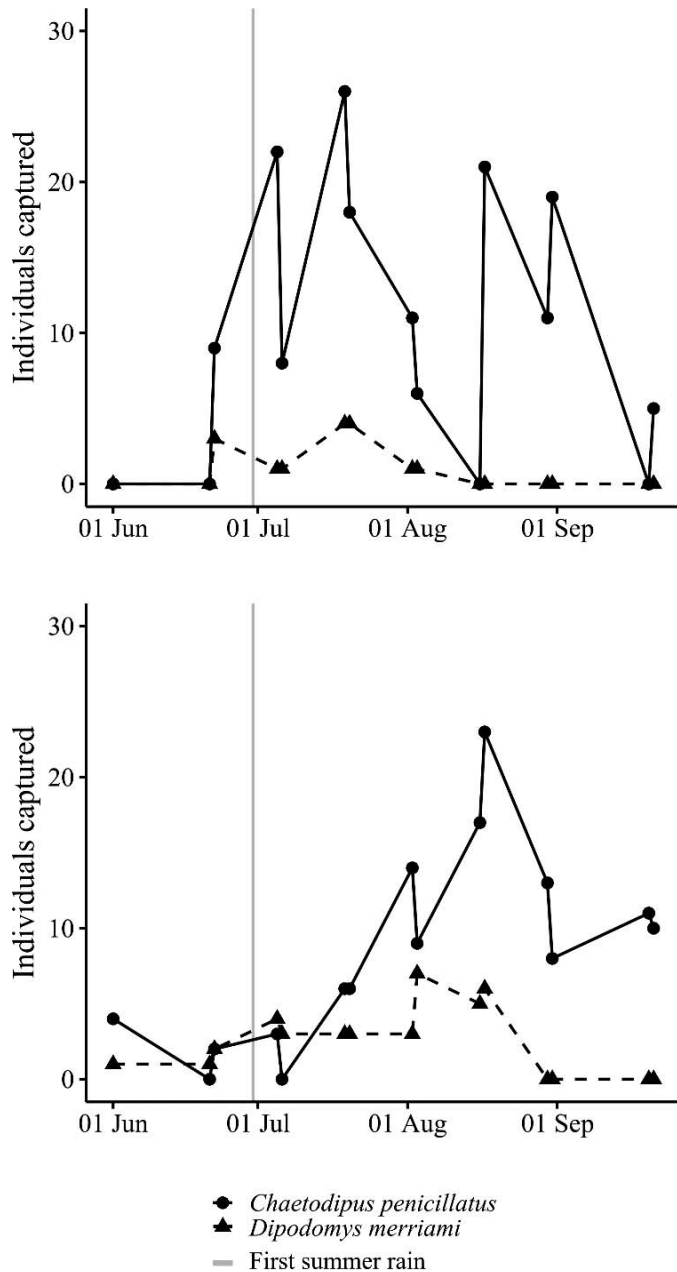
FIG. 2—Capture counts of *Chaetodipus penicillatus* and *Dipodomys merriami* before and during the rainy season in a native vegetation (upper) and a buffelgrass (lower) sites in the municipality of La Colorada, Sonora, Mexico. Vertical gray lines indicate the onset of the rainy season.

(95% *CI*: 2.4–10.2 individuals·ha^{-1}) in the native vegetation site and 8.5 individuals·ha^{-1} (95% *CI*: 5.3–11.7 individuals·ha^{-1}) in the buffelgrass site using a Cormack-Jolly-Seber model. Similarly, we did not find differences in population density between the buffelgrass and native vegetation sites for *D. merriami* (Wald test, $t = -0.77$, $df = 4$, $P = 0.48$).

Vegetation Cover—Before summer rains (dry season), we estimated 60.76 and 38.20% of vegetation cover in the native vegetation and the buffelgrass sites, respectively.

DISCUSSION—We found inconclusive evidence that the lower habitat complexity of buffelgrass prairies had an effect on abundance and diversity (Simpson's index) of the desert nocturnal rodent community in central Sonora. Nevertheless, both the Shannon-Wiener and the evenness indices were higher in the buffelgrass site. Higher diversity and evenness among species at buffelgrass habitat may be due to weakened interspecific interactions by an intermediate disturbance effect (Connell, 1978) of a patchy, not continuous, distribution of buffelgrass prairies in the study area. A higher Shannon-Wiener index also may be an artifact of a relatively low sample size. Difference in relative frequency of captures between the two most abundant species (*C. penicillatus* and *D. merriami*) between sites provides greater evenness

TABLE 3—Plant cover percentages per species at native vegetation and buffelgrass sites during the dry and rainy season at the study area the municipality of La Colorada, Sonora, Mexico. — = a species was not found in a site and/or season.

	Dry season		Rainy season	
Species	Buffelgrass	Native vegetation	Buffelgrass	Native vegetation
Perennial grasses				
Cenchrus ciliaris	35.12	—	55.50	—
Cacti				
Cylindropuntia arbuscula	—	7.41	—	2.83
Cylindropuntia fulgida	0.28	0.24	0.57	0.11
Lophocereus schottii	—	1.77	—	0.05
Shrubs				
Condalia globosa	—	6.47	—	0.76
Havardia mexicana	—	1.71	—	2.32
Lycium berlandieri	—		0.80	—
Merremia palmeri	—	0.34	—	—
Mimosa laxiflora	0.72	6.23	1.34	10.82
Randia thurberi	—	2.14	—	9.68
Trees				
Jatropha cardiophylla	—	12.40	0.52	9.46
Olneya tesota	0.83	22.05	1.39	16.64
Prosopis glandulosa	1.25	—	1.18	—
Annuals and seedlings	—	—	36.76	46.62
Covered ground	38.20	60.76	98.05	99.30
Uncovered ground	61.80	39.24	1.95	0.70

to the importance values of the species in the buffelgrass prairie, which tends to increase Shannon-Wiener's index. Another reason for the greater Shannon-Wiener index in the buffelgrass prairie is the presence of one more species in this site, *Sigmodon arizonae*, which was possibly absent in the native vegetation site because it occupies habitats mainly composed with grasses (Guthery et al., 1979). In both sites, *C. penicillatus* was the most abundant species, followed by *D. merriami*; the other four species (*C. baileyi*, *P. eremicus*, *O. torridus*, and *S. arizonae*) had relatively lower abundance. Since Simpson's index gives more weight to each species' dominance, the fact that both sites had a similar pattern in species evenness may be the reason why this index was not statistically different between sites. Our estimates of the Shannon-Wiener index for buffelgrass ($H' = 1.30$) and native vegetation ($H' = 0.71$) are comparable to other studies of desert rodent communities, including grasslands and shrublands (range, $H' = 0.62$–0.68; Hernández et al., 2005) and creosotebush (range, $H' = 0.35$–2.0; Whitford, 1976) communities in the Chihuahuan Desert, and across habitats in North American deserts (range, $H' = 0.26$–0.73; Patterson and Brown, 1991).

Our sample size (captures) before the rainy season (three trapping nights) is not large enough to provide precise estimates of seasonal differences in the density of *C. penicillatus* and *D. merriami* using mark-recapture techniques. Nevertheless, the increase in captures of both species during the rainy season could be related to their sensitivity to seasonal changes in temperature and humidity, as well as to the changes in vegetation structure

(Kenagy, 1973; Stamp and Ohmart, 1978). *Onychomys torridus* was the only species found with a carnivorous diet (mainly arthropods; McCarty, 1975), while the rest of the species are generalists, eating seeds, insects, and green vegetation in different proportions (Reichman, 1975; Veal and Caire, 1979; Paulson, 1988; Nagy and Gruchacz, 1994; Mantooth and Best, 2005). Annual plants play an important role in the diets of heteromyids during and after the monsoon (Stamp and Ohmart, 1978): *Chaetodipus baileyi* feeds mainly on forb's seeds (as *Perognathus baileyi*; Reichman, 1975); *D. merriami* feeds mainly on seeds of annual plants during the summer, and its diet changes to more arthropods during winter (Nagy and Gruchacz, 1994); and *C. penicillatus* has been found to occur sympatrically and feeds on annual plants such as *Aristida* species, *Bouteloua* species, *Boerhavia* species, and *Erodium* species (Mantooth and Best, 2005). Additionally, Reichman and Van De Graaff (1973) have found that several Sonoran Desert rodents, including *D. merriami*, *C. baileyi* (as *P. baileyi*) and *P. eremicus*, have peaks in their reproductive activity mainly during the months of the rainy season (June–October). Our results showed no difference in operational sex ratio in the captures of *C. penicillatus* and *D. merriami*. There is evidence of differential activity between sexes during breeding season in heteromyids, with males moving greater distances than females of *D. merriami* and *D. spectabilis* (Randall, 1993), and *Liomys irroratus* (Santos-Moreno and Santiago-Marcial, 2012), which could bias towards more male captures in sampling. However, researchers have not found male-biased dispersal distances in species of *Chaetodipus* and

Dipodomys, including *D. merriami,* with both sexes dispersing more during breeding season (Allred and Beck, 1963; Behrends et al., 1986; Daly et al., 2000).

Habitat preferences of desert nocturnal rodents, including *C. penicillatus* and *D. merriami,* have been well documented in desert environments in the southwestern United States while testing mechanisms on species coexistence (e.g., Price, 1978; Thompson, 1982; Kotler and Brown, 1988). Small heteromyids, such as *C. penicillatus,* tend to competitively exclude large heteromyids, such as *D. merriami,* because the smaller species are better at exploiting seeds (Rosenzweig, 1973). In addition, *D. merriami* counteracts its disadvantage in foraging in open habitats by having a greater chance to escape from predators (Rosenzweig, 1973). Food limitation and predation then produce habitat segregation between these two species: *D. merriami* occupies open areas, such as buffelgrass prairies, whereas *C. penicillatus* occupies areas with greater shrub cover for foraging (Mantooth and Best, 2005), such as desertscrub. In this regard, the relative frequency of captures of *C. penicillatus* was greater at the native vegetation site (88 vs. 70% in the buffelgrass site) where the vegetation cover percentage is higher, whereas the relative frequency of captures of *D. merriami* was higher in the buffelgrass site (21 vs. 8% in the native vegetation site) where the vegetation cover was only 38.2% before the rainy season. Nevertheless, population densities of *C. penicillatus* and *D. merriami* showed no differences between the native vegetation site and the buffelgrass site. Therefore, our results suggest that the openness of buffelgrass prairies does not affect *C. penicillatus* negatively but also does not affect *D. merriami* positively in this area. We hypothesize that the observed vegetation growth in buffelgrass prairies after the monsoon season may buffer the structural differences between native vegetation and buffelgrass prairies.

We found differences in the relative abundance of *C. penicillatus* and *C. baileyi.* Rosenzweig and Winakur (1969) found that, although *C. penicillatus* and *C. baileyi* (as *Perognathus penicillatus* and *P. baileyi*) can be equally abundant in southern Arizona, they are not sympatric species. These authors suggest that these species may have similar requirements in slightly different habitats, with *C. penicillatus* preferring shrubby areas and grasslands with bushes and trees. We found that *C. penicillatus* was much more abundant than *C. baileyi* in both sites, which was possibly due to the presence of native shrubs and trees in the buffelgrass prairies in our study area and the difference in habitat requirements of these species, thus promoting a greater abundance of *C. penicillatus* over *C. baileyi.*

In both native vegetation and buffelgrass sites, we found four native plant species in common (*Olneya tesota, M. laxiflora, J. cardiophylla,* and *Cylindropuntia fulgida*). It is possible that the relatively small and discrete size of the buffelgrass prairies and the presence of these common native plants favored the occurrence of rodent species in both sites by providing food (Paulson, 1988; Castillo, 2005; Mantooth and Best, 2005), regardless of the presence of buffelgrass. Our results suggest that the presence of buffelgrass, as an introduced species to the Sonoran Desert, does not seem to negatively affect the population density of *C. penicillatus* and *D. merriami,* and promotes the presence of *S. arizonae* in the region in the spatio-temporal scale addressed in our study.

Similar to other studies about the effect of buffelgrass on wildlife in Sonora (Franklin, 2012; Macías-Duarte et al., 2019), and in contrast to studies in other countries (Kuvlesky et al., 2002; Eyre et al., 2009; Smyth et al., 2009; Schlesinger et al., 2020) and studies about the effect exotic grasses, including buffelgrass, on rodents of other countries (Cortez et al., 2015; Marinaro and Grau, 2015), we conclude that, at our spatial scale, buffelgrass does not seem to affect the community dynamics of nocturnal desert rodents in central Sonora. However, readers must take this conclusion with caution due to our relatively low sample size. The methods used for the establishment and maintenance of a buffelgrass prairie can affect the composition and vegetation cover, which in turn can influence the distribution and abundance of rodent species in the site and change diversity. In this regard, further study should include long-term observational or experimental studies to determine the effects of buffelgrass on rodent communities with different methods of establishment, maintenance, and use of buffelgrass prairies (e.g., control of other invasive plant species and areas with natural invasion). Also, extensive mark-recapture studies would allow researchers to draw more general conclusions about the effects of buffelgrass introduction on the rodent community of the Sonoran Desert, because we did not capture other species that were probably present in our study area (e.g., *Neotoma albigula*). Moreover, research should focus on the possibility of differences in rodent activity during the reproductive period in buffelgrass-invaded areas, including range and movement. Finally, as land conversion and invasion in the region threaten to expand the current distribution of buffelgrass (Castellanos-Villegas et al., 2002; Lyons et al., 2013), expanding our understanding on the effects of buffelgrass on other wildlife species in the Sonoran Desert will be relevant to mitigation and biodiversity conservation.

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