

NOTES

EFFECTS OF CONSERVATION RESERVE PROGRAM SEEDING REGIME
ON HARVESTER ANTS (*POGONOMYRMEX*), WITH IMPLICATIONS FOR
THE THREATENED
TEXAS HORNED LIZARD (*PHRYNOSOMA CORNUTUM*)

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ABSTRACT—I compared the presence and abundance of nest-sites made by harvester ants (*Pogonomyrmex*), the primary prey for the endangered Texas horned lizard (*Phrynosoma cornutum*), among restored grassland plots planted in different grass species and indigenous prairie. The restored plots had been seeded as part of the Conservation Reserve Program (CRP) as exotic monocultures of either Old World bluestem (*Bothriochloa ischaemum*) or weeping lovegrass (*Eragrostis curvula*), or as mixtures of native grasses (both with and without buffalograss, *Buchloë dactyloides*). On average, the fewest ant mounds were found on Old World bluestem plots, whereas the indigenous grassland had the highest density of harvester ant mounds. However, there were no significant differences between native and exotic CRP plantings. Results obtained from a simultaneous visual survey for Texas horned lizards corroborate these findings. Thus, there is no evidence that CRP plots planted in exotic grasses are significantly poorer habitat for Texas horned lizards in terms of ant abundance than native grass plantings.

RESUMEN—Comparé la presencia y abundancia de nidos de hormigas *Pogonomyrmex*, la presa principal del camaleón texano (*Phrynosoma cornutum*), una lagartija en peligro de extinción, en parcelas restauradas de pastizales sembrados de especies diferentes y parcelas de pradera nativa. Dichos parcelas restauradas forman parte del Programa de Conservación de Reservas (CRP por sus siglas en inglés) y abarcan monocultivos de pastos introducidos (*Bothriochloa ischaemum* o *Eragrostis curvula*) o mezclas de pastos nativos (con y sin *Buchloë dactyloides*). En general, la menor densidad de nidos de hormigas *Pogonomyrmex* se encontró en las parcelas de *Bothriochloa ischaemum*, y la mayor densidad de los nidos se encontraron en los pastizales nativos. Sin embargo, no hubo diferencias significativas entre parcelas CRP de pastos exóticos y pastos nativos. Los resultados obtenidos de un muestreo visual simultáneo del camaleón texano corroboran estos resultados. Entonces, no hay ninguna evidencia de que los campos del Programa de Conservación de Reservas sembrados en pastos exóticos son un hábitat significativamente más pobre para *Phrynosoma cornutum* en términos de abundancia de hormigas que los sembrados en pastos nativos.

More than 99% of the Great Plains has been lost to anthropogenic habitat transformations, with resultant negative effects on many animal populations (Noss et al., 1995). Formed in 1985, the Conservation Reserve Program (CRP) of the United States Department of Agriculture leases private agricultural land for 10-year periods, whereby the landowner seeds the land and does not plow or graze it during that time (Young and Osborn, 1990; Olenbusch et

al., 1995). Approximately 14 million ha are currently enrolled in the CRP (United States Department of Agriculture, 2000). This program has effectively been a form of prairie restoration in the Great Plains for many wildlife species (Berthelsen et al., 1989; Johnson and Schwartz, 1993; Reynolds et al., 1994; Johnson and Igl, 1995; King and Savidge, 1995).

Most of the earliest CRP leaseholders planted less costly exotic grasses, especially *Eragrostis*

curvula (weeping lovegrass) and *Bothriochloa ischaemum* (Old World bluestem). Leases initiated since 1997, however, require landowners to plant at least 90% of their fields in a mix of native grasses, such as *Bouteloua curtipendula* (sideoats grama), *B. gracilis* (blue grama), *Panicum virgatum* (switchgrass), and *Buchloë dactyloides* (buffalograss). Lease renewals since 1997 require reseeding at least 51% of existing CRP land in native plants. Because buffalograss is relatively expensive compared to the other species, however, many CRP fields do not include it despite its abundance and importance in native shortgrass prairie (Berthelsen et al., 1989; Lauenroth and Milchunas, 1991). Thus, CRP lands comprise a mosaic of exotic monocultures and mixed native plantings.

Harvester ants (*Pogonomyrmex*) are abundant and conspicuous components of grassland ecosystems (Cole, 1968). These ants clear vegetation around their nest-sites (mounds) to form a disk of bare ground 1 to 2 m in diameter (Crist and Wiens, 1994; McIntyre, 1999). These granivorous insects alter plant community composition through differential seed predation, affect soil chemistry through burrowing, and alter the behavior of other arthropods by providing areas of bare soil for thermoregulation and oviposition (McIntyre, 1999). They are also the primary prey for the Texas horned lizard (*Phrynosoma cornutum*).

The Texas horned lizard is listed as a threatened species in Texas and Oklahoma. Although the species once occurred in semi-arid grasslands from Kansas to the northern half of Mexico and from eastern Texas to extreme southeastern Arizona, its numbers have declined precipitously in recent decades, and it has virtually disappeared from the eastern portion of its historic range (Donaldson et al., 1994). The rapid and recent decline in the Texas horned lizard makes a habitat assessment for the species vital for future conservation efforts to halt this downward trend (Donaldson et al., 1994). One key element of such a habitat assessment must be an evaluation of the occurrence of its primary prey, harvester ants. Ants can comprise 69% of the prey individuals consumed by the Texas horned lizard (and 61.2% of stomach volume; Pianka and Parker, 1975; Donaldson et al., 1994). Few organisms other than horned lizards consume these ants, and the occurrence of horned liz-

ards is tightly coupled to the occurrence of harvester ants (Whitford and Bryant, 1979; Donaldson et al., 1994).

The goals of this study were to assess the occurrence of harvester ants on CRP lands planted as exotic monocultures versus CRP planted in mixed native grasses, as compared to indigenous, unseeded grassland. Harvester ants were surveyed at 16 sites in the Texas Panhandle Plains: 4 plots were CRP weeping lovegrass; 3 were CRP Old World bluestem; 4 were CRP plots seeded with native species (with a dominance of sideoats grama, blue grama, and switchgrass) without buffalograss; 4 were CRP plots seeded with sideoats grama, blue grama, switchgrass, and buffalograss; and 1 was an unseeded, unplowed native grassland plot: Muleshoe National Wildlife Refuge (NWR). These plots were not associated with any particular soil types or textures, which consisted of various loams, clay and sandy loams, and fine sands (Newman, 1962; Sanders, 1962; Girdner, 1963; Blakley, 1974; Blackstock, 1979). Each of the CRP plots averaged 1 km² in size and was at least 3 years old. The Muleshoe NWR plot had never been plowed, and it had not been grazed by livestock for 3 years prior to study. Plots were separated by at least 1 km, with the maximum linear distance between sites being 97 km, indicating that the sites were largely independent and yet subject to a similar climate and agro-economic history. The red imported fire ant (*Solenopsis invicta*), which might competitively eliminate harvester ants (Hook and Porter, 1990), was not found at any of these sites; the species has not become widely established in the Texas Panhandle, primarily because of its intolerance of xeric environments (Moody et al., 1981).

Because horned lizard predation pressure can cause harvester ants to curb foraging activity (Whitford and Bryant, 1979), the density of ant nest-sites might be a more constant and reliable indicator of habitat suitability than forager activity or abundance. I measured the density of harvester ant nest-sites at each of the 16 study locations. Ant colonies are relatively long-lived and sessile, and depredation by horned lizards does not significantly depress numbers of colonies (Whitford and Bryant, 1979), meaning that mounds can persist for many years in a given area. The density of ant nest-sites was surveyed along a 100-m north-

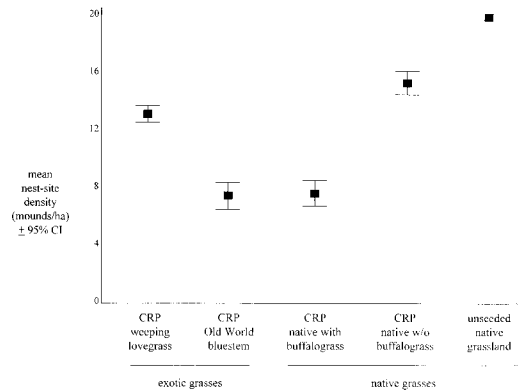


FIG. 1—Mean density (squares) with 95% confidence intervals (T-bars) of harvester ant mounds by site type in the Texas panhandle.

south belt transect (10 m in width) located at the center of each site. All sites were surveyed between 0700 and 1200 Central Daylight Time in July 2001 when the weather was sunny, clear, and dry, with temperatures of 22 to 35°C. Mean densities of nest-sites were compared among site types using 95% confidence intervals (CI) (Zar, 1999). Because there was only 1 unseeded, unplowed grassland plot, its density value is reported for comparison purposes and is not associated with 95% CI. A visual survey for Texas horned lizards was conducted simultaneously with the ant nest-site survey.

Harvester ant nest-site densities ranged from 0 to 20 mounds/ha. These values are consistent with previously reported densities from western Texas (0 to 25 mounds/ha; Donaldson et al., 1994). Significant differences were noted in mound density by site type (Fig. 1). On average, the fewest ant mounds were found on Old World bluestem plots (0 to 10.9 mounds/ha, with an average of 4.8); the unseeded native grassland had the highest density of harvester ant mounds (20 mounds/ha). No harvester ant mounds were found at 3 sites (1 Old World bluestem site, 1 weeping lovegrass site, and 1 native CRP plot planted without buffalograss). Even when these sites were excluded from analysis, however, the trends remained unchanged. The replicate sites within each site type differed from each other in terms of number of ant mounds present (evidenced by variable widths of the confidence intervals in Fig. 1). There was no evidence that CRP plots planted in exotic grasses were significantly

poorer habitat in terms of ant abundance than native grass plantings.

Texas horned lizards were observed at 2 Old World bluestem sites (1 lizard at each site), 1 native CRP plot with buffalograss (2 lizards), and 1 native CRP plot planted without buffalograss (1 lizard). Horned lizards were observed only at sites where ant mounds were recorded. These results add corroborative evidence to the findings from the ant nest-site survey. The relatively high cost of buffalograss did not render CRP plots that incorporated it significantly better ant or horned lizard habitat compared to plots without buffalograss, meaning that CRP leaseholders who originally planted less expensive grasses still provided valuable wildlife habitat. New leases and lease renewals require that native grasses be used over a majority of the land.

Incidence of harvester ant nests is only 1 factor in the estimation of the value of an area as habitat for the Texas horned lizard (Burrow et al., 2001). The potential influence of other factors, such as the land-use history of a field, soil type and texture, and landscape context, warrant further investigation. However, the presence and abundance of the ants is a critical factor, nonetheless, in determining whether a parcel of land could support the Texas horned lizard. This is a promising avenue for future research concerning conservation of the Texas horned lizard and the value of CRP.

I thank the numerous CRP leaseholders who permitted access to their property for this study. I thank T. Thompson (Department of Range, Wildlife, and Fisheries Management, Texas Tech University), C. Boal (Texas Cooperative Fish and Wildlife Research Unit), the Natural Resources Conservation Service, and Muleshoe NWR for help in formulating this study. B. Bestelmeyer (Department of Biology, New Mexico State University), C. Boal, and 2 anonymous reviewers made helpful comments on manuscript drafts. H. Stevens (Department of Biological Sciences, Texas Tech University) provided the Spanish translation of the abstract.

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Submitted 14 December 2001. Accepted 18 March 2002.
Associate Editor was Geoffrey C. Carpenter.

BASELINE SURVEY OF TEXAS HORNED LIZARDS, *PHRYNOSOMA CORNUTUM*, IN TEXAS

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ABSTRACT—I studied 102 Texas horned lizards (*Phrynosoma cornutum*) encountered along roads or fortuitously at 50 sites in northern, southern, eastern, western, and central Texas from May through September 1999. Female-to-male and juvenile-to-adult ratios were 58:44 and 56:46, respectively. Distribution of horned lizards by sites were 60%, 50%, 50%, 30%, and 0% for the western, southern, northern, central, and eastern portions of the state, respectively. Average number of lizards captured was greater in the southern region compared to the central and eastern regions. The size of juvenile lizards was similar by region and sex; however, adult lizards were heavier and larger in the southern region compared to the western region.

RESUMEN—Estudié 102 camaleones texanos (*Phrynosoma cornutum*) encontrados alrededor de caminos o en otros lugares en 50 localidades del norte, sur, este, oeste y centro de Texas de mayo a septiembre de 1999. La proporción hembra-macho y juvenil-adulto fue de 58:44 y 56:46, respectivamente. La distribución por localidades de los camaleones estudiados fue de 60%, 50%, 50%, 30%, y 0% para el oeste, sur, norte, centro, y este del estado, respectivamente. El promedio de capturas de lagartijas fue mayor en la región sur en comparación con las regiones centro y este. El tamaño de los camaleones juveniles fue similar por región y sexo; sin embargo, adultos fueron más pesados y más largos en el sur en comparación con el oeste.

Texas horned lizards (*Phrynosoma cornutum*) were once widespread and abundant throughout Texas, except in the far eastern portion of the state (Bigony, 1981); however, their population has experienced a dramatic decline over the past 40 years (Donaldson et al., 1994). Texas horned lizards probably are limited to a few, small fragmented populations in Texas east of a line from Fort Worth to Corpus Christi (Donaldson et al., 1994). Consequently, the species is listed as threatened by the state of Texas (Texas Parks and Wildlife Code, 1987).

Museum records contain numerous specimens of Texas horned lizards, but early researchers failed to provide details of their capture efforts (i.e., person-hours). Also, to my knowledge, recent surveys to assess density or abundance of Texas horned lizards have not been conducted. Without such information, population trends of Texas horned lizards cannot be assessed. Donaldson et al. (1994) revisited historical collection sites and concluded that Texas horned lizard populations were declining throughout most of Texas, except in

southern and northern regions, where populations appeared stable. Since the survey by Donaldson et al. (1994), assessments of the status of Texas horned lizard populations have not been published.

Because of their threatened status in Texas, populations of Texas horned lizards should be regularly assessed to determine if trends reported by Donaldson et al. (1994) continue. Therefore, my objectives were to determine regional population demographics and capture rates of Texas horned lizards to provide a population baseline for future assessments.

Texas horned lizards were captured by systematic searches and fortuitous encounters (Fair and Henke, 1997) throughout 5 study regions (i.e., north, south, east, west, and central) of Texas as delineated by Donaldson et al. (1994) (Fig. 1). Study regions included 10 major vegetative regions of Texas (Gould, 1962). Fifteen possible search sites within each region were selected based on anecdotal evidence that Texas horned lizards were observed on site within the last 2 years; 10 sites were chosen

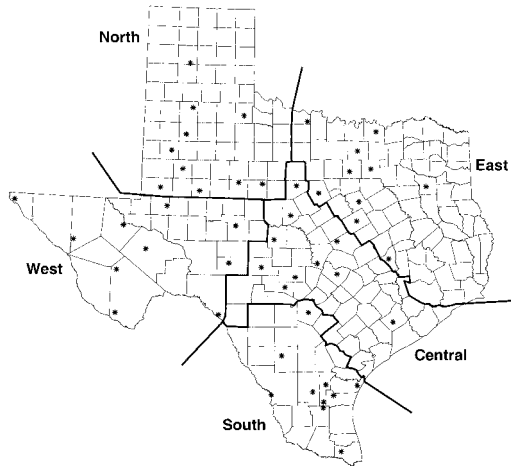


FIG. 1.—Map of Texas indicating sites surveyed for Texas horned lizards and regional boundaries of the state as delineated by Donaldson et al. (1994).

randomly (Table 1). Search sites were located within 8 km of city limits, but were not located within city or residential areas. Sites were searched sequentially from May through September 1999, with the majority of the sites ($n = 41$) searched during May and June to reduce seasonal variation in capture rates (Henke and Montemayor, 1998). Each site was searched for a minimum of 10 person-hours (1 person searching 1 hour = 1 person-hour) by 1 to 4 biologists. Participating biologists were trained in search methodologies for behaviors and evidence of Texas horned lizards. The same biologists were used for all sites and regions. Searches for Texas horned lizards and their sign (i.e., remains of Texas horned lizards, their scat, and track patterns) were conducted during 0800 to 1300 hr on days when temperatures were $\geq 27^{\circ}\text{C}$ and cloud cover was $< 50\%$. Upon capture, Texas horned lizards were weighed, measured for snout-vent length (SVL), categorized to age class according to size, sexed, marked with a passive integrated transponder to identify potential recaptures (Camper and Dixon, 1988), and released at the location of capture. Lizards were considered adults when $\text{SVL} \geq 68$ mm (Ballinger, 1974).

The experimental design was a completely randomized design. Analyses of variance were used to test the effect of capture rate on region and to test the effects of sex and region on size

TABLE 1.—Capture rates per site of Texas horned lizards collected throughout Texas during May through September 1999. Regions delineated by Donaldson et al. (1994). n = number of Texas horned lizards captured at site. Rate = number of Texas horned lizards captured/person-hours of search.

Site	North			South			East			West			Central		
	n	Rate	Site	n	Rate	Site	n	Rate	Site	n	Rate	Site	n	Rate	Site
Abilene	0	0.00	Kingsville	17	4.86	Waco	0	0.00	San Angelo	5	0.36	Junction	0	0.00	
Sweetwater	0	0.00	Premont	0	0.00	Bryan	0	0.00	Sonora	0	0.00	Kerrville	0	0.00	
Big Spring	0	0.00	Alice	0	0.00	Dallas	0	0.00	Del Rio	0	0.00	Fredericksburg	0	0.00	
LaMesa	4	0.31	Falfurrias	10	2.86	Fort Worth	0	0.00	Odessa	0	0.00	Austin	0	0.00	
Andrews	0	0.00	Benavides	10	2.50	Sherman	0	0.00	Fort Stockton	2	0.13	Killeen	5	0.23	
Brownfield	6	0.50	San Antonio	0	0.00	Wichita Falls	0	0.00	Pecos	1	0.08	Eastland	9	0.50	
Lubbock	4	0.29	Corpus Christi	0	0.00	Tyler	0	0.00	Alpine	0	0.00	Brownwood	4	0.21	
Plainview	3	0.27	Harlingen	0	0.00	Hillsboro	0	0.00	Terlingua	3	0.20	Brady	0	0.00	
Amarillo	4	0.33	Laredo	5	0.50	Stephensville	0	0.00	Van Horn	2	0.20	Marble Falls	0	0.00	
Paducah	0	0.00	Cotulla	5	0.50	Denton	0	0.00	El Paso	3	0.20	El Campo	0	0.00	
Total	21	0.19	Total	47	0.58	Total	0	0.00	Total	16	0.13	Total	18	0.13	

TABLE 2—Regional mean capture rate of Texas horned lizards collected throughout Texas during May through September 1999. Study regions defined by Donaldson et al. (1994). *n* = number of capture sites per region. Capture rates with the same letter were not significantly different ($P > 0.05$) between regions.

Region	<i>n</i>	Capture rate (mean ± SE)
South	10	1.036 ± 0.51 A
North	10	0.170 ± 0.06 AB
West	10	0.117 ± 0.04 AB
Central	10	0.094 ± 0.05 B
East	10	0.000 ± 0.00 B

(i.e., SVL and weight) of horned lizards (SAS Institute, 1989). Scheffé's test was conducted when a significant ($P \leq 0.05$) *F*-test was noted (Steel and Torrie, 1980). Analyses of size were separated by cohort (i.e., juvenile and adult) because of the obvious size differential between cohort groups. Distributions of residual errors were tested for normality using Shapiro-Wilk test (SAS Institute, 1989). Homogeneity of variances among treatments was evaluated with Bartlett's test (Steel and Torrie, 1980). Sex ratios and age ratios were analyzed using Chi-square analyses with the Yates correction factor of continuity to determine if ratios differed from a 1:1 relationship (Steel and Torrie, 1980). Chi-square analysis also was used to test if the proportion of sites with horned lizards differed between regions. Statistical significance was inferred at $P \leq 0.05$. Descriptive statistics are presented as the mean ± 1 SE.

I captured 102 Texas horned lizards during

1999 (Table 1). Scat from horned lizards was found on several sites, but was only located on sites where horned lizards were captured. Mean capture rates differed ($F_{4,45} = 3.34$; $P = 0.02$) by region, with mean capture rates of horned lizards being greater on sites within the southern region than on sites within the central or eastern regions (Table 2). Texas horned lizards were observed and caught on 60%, 50%, 50%, 30%, and 0% of the capture sites in the western, southern, northern, central, and eastern regions, respectively. However, the proportion of sites with horned lizards did not differ ($\chi^2 = 6.0$; $df = 4$; $P > 0.18$) between regions. Sex ratios (M:F) of Texas horned lizards were 19:28, 9:9, 7:9, and 9:12 on sites within the southern, central, western, and northern regions, respectively, and ratios did not deviate ($\chi^2 < 1.4$; $df = 1$; $P > 0.25$) from a 1:1 relationship. Juvenile-to-adult ratios of Texas horned lizards were 21:26, 9:9, 9:7, and 17:4 on sites within the southern, central, western, and northern regions, respectively. Age ratios did not deviate from a 1:1 relationship except on sites within the northern region, where more ($\chi^2 = 6.8$; $df = 1$; $P < 0.01$) juvenile than adult horned lizards were caught. Weights and SVL of juvenile Texas horned lizards did not differ ($F_{1,90} < 0.75$; $P > 0.39$; $F_{4,90} < 1.8$; $P > 0.14$) by region, sex, or by the interaction of main effects (Table 3). However, in adult Texas horned lizards, a regional difference in weight and SVL occurred with lizards from sites within the southern region being heavier ($F_{4,90} = 4.1$; $P < 0.004$) and longer ($F_{4,90} = 2.6$; $P < 0.04$) than lizards from sites within the western region (Table 3). Weights and SVL of adult Texas

TABLE 3—Snout-vent length (SVL) and weight of juvenile and adult Texas horned lizards collected throughout Texas from May through September 1999. Lizards considered adults at SVL ≥ 68 mm (Ballinger, 1974). Study regions defined according to Donaldson et al. (1994). Texas horned lizards with same letter were not significantly different ($P > 0.05$) in length or weight. Texas horned lizards not found in East region.

Region	Juveniles				Adults			
	SVL (mm)		Weight (g)		SVL (mm)		Weight (g)	
	mean ± SE	Range	mean ± SE	Range	mean ± SE	Range	mean ± SE	Range
South	54.0 ± 2.1 A	21–67	13.5 ± 2.1 A	1–34	81.9 ± 1.7 A	68–96	36.2 ± 2.0 A	15–54
Central	58.9 ± 1.5 A	52–65	17.4 ± 1.4 A	12–27	76.5 ± 2.8 AB	68–91	28.5 ± 1.1 AB	20–32
West	52.9 ± 2.9 A	44–67	9.8 ± 1.2 A	6–16	75.9 ± 2.6 AB	69–87	30.8 ± 0.8 AB	29–33
North	54.3 ± 0.9 A	50–65	12.5 ± 1.5 A	6–29	70.5 ± 0.9 B	68–72	24.0 ± 1.8 B	17–30

horned lizards were not affected by sex ($F_{1,90} < 2.8$; $P > 0.09$) or the interactive effect of sex and region ($F_{4,90} < 1.1$; $P > 0.39$).

Texas horned lizard populations have declined in Texas (Donaldson et al., 1994), but it is difficult to determine if the decline has halted or if it continues today. The only published assessment of the status of Texas horned lizards was conducted by Donaldson et al. (1994), but they, like past researchers of museum collections, failed to provide adequate information (e.g., person-hours, density estimates) to make temporal comparisons. From their study, Donaldson et al. (1994) concluded that abundances of Texas horned lizards were greatest in the southern and northern regions of Texas. Although the same regions provided me with the greatest numbers of horned lizards, the numbers collected at the sites within the northern region were not significantly greater than at sites within the other regions of the state. It is possible that the population of Texas horned lizards within the northern region of the state continued to decline after the assessment by Donaldson et al. (1994) to a level similar to sites within the western and central regions. Alternatively, it is equally possible that the populations in western and central Texas increased in horned lizard abundance, which also would account for no differences between sites with these regions.

Anecdotally, populations of Texas horned lizards were considered contiguous throughout the southern region of Texas. I found horned lizards on only 50% of the sites within the southern region, even though the majority of sites within this region were located in the South Texas Plains vegetative region and contained similar climates, soil, and plant communities (Gould, 1962). Although speculative, populations within this region might have become fragmented. The effects of habitat fragmentation on populations of Texas horned lizards are unexplored.

Although the western region had the greatest proportion of sites with Texas horned lizards, horned lizard abundance at these sites was low. Distribution and abundance of horned lizards were both low on sites within central Texas, and horned lizards might be extirpated from my sites in eastern Texas. I acknowledge that horned lizards could have been present but evaded capture in the search

areas. Although such evasions would affect my conclusion about horned lizard distribution, it would have a negligible effect on the regional trend of capture rates (i.e., abundance) that were observed.

Adult and juvenile horned lizards were found in each region, except at sites within the eastern region. This provides evidence that horned lizards are reproducing, but long-term population monitoring is needed to determine the stability of populations. Perhaps survival of Texas horned lizards was greatest on sites within the southern region of Texas because they were larger, on average, than on sites in the remainder of the state. Fair and Henke (1999) estimated that annual survival of Texas horned lizards in southern Texas ranged from 9% to 54%. Survival estimates of Texas horned lizards from other regions of Texas are unknown.

The demographic data provided in this study can serve as a reference point from which future studies can assess trends in the population status of Texas horned lizards in Texas. Without such assessments, potential recovery of this threatened species will remain unknown.

I thank L. Gaston, J. Feilds, and L. Harveson for assistance with the capture of horned lizards, and D. Hewitt, B. Ballard, and W. C. Sherbrooke for comments on an earlier draft of the manuscript. Financial support was provided by the Houston Livestock Show and Rodeo Association. Texas horned lizards were captured and handled in accordance with Texas Parks and Wildlife Department (Austin, Texas) Scientific Permit Number SPR-0993-636. This study was approved by the Texas A&M University-Kingsville Animal Care and Use Committee (#1-97-37). This manuscript is Contribution Number 02-107 of the Caesar Kleberg Wildlife Research Institute.

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- Submitted 8 January 2002. Accepted 10 June 2002.
Associate Editor was Geoffrey C. Carpenter.

FORAGING MODE OF THE RETICULATE COLLARED LIZARD, *CROTAPHYTUS RETICULATUS*

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ABSTRACT—The reticulate collared lizard (*Crotaphytus reticulatus*) is a large, crotaphytid lizard restricted to the lower Rio Grande Valley of southern Texas and northern Mexico. Our objectives were to determine the foraging mode of this species and test predictions made by other investigators about foraging behavior. We conducted focal observations on 10 male and 10 female lizards in Falcon State Park, Starr County, Texas, to quantify the number of movements per minute (MPM), the proportion of time spent moving (PTM), and the proportion of attacks made on prey items while moving (PAM). We removed foraging movements followed by social displays from the data and recalculated and reanalyzed foraging activity values. With “display” movements removed, there was a significant difference between MPM values and PTM values for males and for the sexes combined, but there was no significant difference in MPM values or PTM values for females. The foraging measures calculated for *C. reticulatus*, with and without “display” movements removed from the calculations, suggest that it is a “sit-and-wait” ambush forager like its congener *C. collaris*.

RESUMEN—La lagartija reticulada de collar (*Crotaphytus reticulatus*) es una lagartija grande restringida al valle bajo del Río Bravo del sur de Texas y del norte de México. Nuestros objetivos fueron determinar el modo de buscar comida y probar algunas hipótesis hechas por otros investigadores referentes al comportamiento de búsqueda de comida. Realizamos observaciones focales de 10 machos y 10 hembras de la especie en el Parque Estatal de Falcón, condado de Starr, Texas, para cuantificar el número de desplazamientos por minuto (MPM), la proporción del tiempo utilizada desplazándose (PTM), y la proporción de ataques a presas mientras se movían (PAM). Descartamos los desplazamientos seguidos de exhibiciones sociales de los datos y calculamos y analizamos nuevamente los valores de búsqueda de comida. Con estos desplazamientos removidos, hubo una diferencia significativa entre los valores de MPM y PTM para los machos y para ambos sexos juntos, pero no hubo una diferencia significativa de los valores ni de MPM o de PTM para las hembras. Las medidas de búsqueda de comida para *C. reticulatus*, con o sin los desplazamientos de “exhibición,” sugieren que es una especie que busca la comida por medio de emboscadas de “sentarse y esperar” como su congénere *C. collaris*.

The reticulate collared lizard (*Crotaphytus reticulatus*) is a large, crotaphytid lizard that exhibits strong sexual dimorphism in body size, with adult males reaching larger sizes than females (Smith, 1946). Male and female adult lizards are both tan in color, with a white, reticulated dorsal pattern enclosing 7 or 8 transverse lines of black spots (Montanucci, 1976; McGuire, 1996). Males have a bright yellow coloration on the front limbs and on the anterior dorsum and ventral region of the trunk during the breeding season (McGuire, 1996). The species is restricted to the lower Rio Grande Valley of southern Texas and northern Mexico. The reticulate collared lizard is primarily saxicolous and mainly occurs in rolling terrain with shallow, gravelly soils, but it has been known to utilize flatland habitats where rocks are not present (Montanucci, 1971). Because of its limited geographic distribution and because the area in which it lives is disturbed by humans and introduced grasses, *C. reticulatus* is listed as a state-threatened species in Texas (Dixon, 2000).

Little of the natural history of this species has been quantitatively studied. Smith (1946) commented that the reticulate collared lizard is 1 of the least known lizards in the United States, and little has been done since then to remedy that situation. Montanucci (1971) anecdotally reported diel and seasonal activity patterns. Accounts of diet suggest that adult and juvenile lizards are opportunistic foragers, consuming a wide variety of arthropods and plant tissue (Klein, 1951; Montanucci, 1971). Montanucci (1971) reported that individuals were observed foraging during early morning and evening hours, but no mention was made as to how foraging was conducted.

Two common lizard foraging strategies have been described as distinct "modes" (MacArthur and Pianka, 1966; Huey and Pianka, 1981), but it has been suggested that foraging modes might vary along a continuum between the 2 extremes from ambush (sit-and-wait) to active (wide) foraging (Magnusson et al., 1985; Pietruszka, 1986; Perry et al., 1990; Cooper and Whiting, 1999). There is still debate on which of these concepts is a better explanation of foraging modes (McLaughlin, 1989; Perry, 1999; Cooper et al., 2001). The foraging mode of *C. reticulatus* is of particular interest because this species is the basal taxon in this genus

(McGuire, 1996) and, therefore, represents a lineage that is the closest extant relative of all *Crotaphytus*, an ambush foraging group, and *Gambelia*, a genus composed primarily of more active foragers (Montanucci, 1978; Tollestrup, 1983). Reticulate collared lizards, like other *Crotaphytus* species, spend the majority of their time in stationary positions for thermoregulatory and social purposes (Husak and Ackland, unpubl. data), suggesting that they are ambush foragers. Our objectives were to determine foraging mode and test predictions made by other investigators about foraging behavior.

The study site was a sandstone outcropping in Falcon State Park, Starr County, Texas, vegetated by a honey mesquite (*Prosopis glandulosa*) and acacia (*Acacia rigidula* and *A. berlandieri*) thornscrub community. Other common plants on the site included huisache (*A. minuta*), retama (*Parkinsonia aculeata*), and prickly pear cactus (*Opuntia lindheimeri*). During 20 through 22 May 2001, 20-min focal observations were conducted on adult lizards throughout the study site to quantify foraging behavior. Lizards were located by walking the site slowly until a subject was found, waiting for 2 minutes to minimize disturbance effects, then initiating the focal observation. All activities of lizards were quantified from a distance of at least 20 m, using binoculars and a hand-held microcassette recorder, including (following Cooper et al., 2001) time (to the nearest second) spent moving and stationary, feeding attempts, and whether feeding attempts were made while sedentary or actively searching. Successful attempts at predation were not distinguished from unsuccessful attempts, because such observations are difficult to make from long distances. Postural adjustments of the body were not counted as movements; only unambiguous movements from 1 location to another were recorded.

To determine whether or not foraging mode calculations might be affected by social behavior, we also recorded whether or not movements were followed by displays. Observations were made when the substrate temperature was between 35 and 42°C, a temperature range similar to that used in a study of *Crotaphytus collaris* behavior (Baird and Timanus, 1998). Because the preferred temperature range for *C. reticulatus* is unknown, we used this range because it corresponded to the active behavior

of the lizards. On a few occasions, lizards left our field of view, so the total time of those focal observations was adjusted for the calculation of foraging measurements. Observations were made of the same population over several days, but our experience with the individually marked lizards in the population gave us confidence that no lizard was observed more than once. From focal observations, we calculated 3 measures of foraging activity as described in Cooper et al. (2001): 1) the number of movements per minute (MPM), 2) the proportion of time spent moving (PTM; amount of time spent moving divided by total time observed), and 3) the proportion of attacks made on prey items while moving (PAM; the number of attacks on prey discovered while actively searching divided by total recorded mobile or immobile attacks on prey).

Due to hot midday temperatures, the lizards had a bimodal daily activity pattern (Husak and Ackland, unpubl. data). Therefore, focal observations were conducted during morning and evening. Because there were no differences for any of the foraging activity measures during the morning and evening (Kolmogorov-Smirnov 2-sample test, $P > 0.75$), we combined these data for all analyses. Differences were tested between the sexes in the 3 measures using 2-sample t -tests. Because data were collected during breeding season, foraging movements followed by display were removed from the data and foraging activity values were recalculated and reanalyzed. Differences between foraging values with and without "display" movements were tested within each sex using paired t -tests, and differences between sexes were tested using 2-sample t -tests.

There were no significant differences between the sexes for any of the measures of foraging mode (Table 1). With and without "display" movements removed from the calculations, there was a significant difference between MPM values (Table 2) for males ($t = 2.81$, $P = 0.021$) and for the sexes pooled ($t = 2.62$, $P = 0.017$), and between PTM values (Table 2) for males ($t = 2.77$, $P = 0.022$) and the sexes pooled ($t = 2.46$, $P = 0.024$). There was no significant difference in MPM or PTM values for females when "display" movements were removed from or included in the analysis (Table 2). When "display" movements were removed, there were still no significant differ-

ences between the sexes in MPM or PTM values (Table 2).

TABLE 1—Mean ($\pm SE$) movements per minute (MPM), proportion of time spent moving (PTM), and proportion of attacks made on prey items while moving (PAM) for 10 adult male and 10 female reticulate collared lizards (*Crotaphytus reticulatus*) in southern Texas. No significant differences were found between sexes.

	MPM	PTM	PAM
Male	0.318 \pm 0.082	0.018 \pm 0.006	0.0
Female	0.318 \pm 0.073	0.014 \pm 0.003	0.0
Combined	0.318 \pm 0.054	0.016 \pm 0.003	0.0

ences between the sexes in MPM or PTM values (Table 2).

The foraging measures calculated for *C. reticulatus* are similar to those calculated for its congener *C. collaris* (MPM 0.086, PTM 0.004, PAM 0.0; Cooper et al., 2001) and are within the range of foraging mode values for classification as an ambush forager (Cooper et al., 2001). Ecomorphological analysis of crotaphytid feeding (Lappin, 1999) suggests that lizards with robust heads should prey primarily on arthropods and occasionally on vertebrates, which agrees with findings on *C. reticulatus* (Klein, 1951; Montanucci, 1971). We only saw predation attempts on arthropods. Opportunistic ambush foraging seems to be the ancestral state for iguanians (Perry, 1999), with *C. reticulatus* and *C. collaris* being no exception (this study; Best and Pfaffenberger, 1987; Husak and McCoy, 2000). The foraging strategy of *Gambelia wislizenii* (and potentially *G. copei*; Lappin and Swinney, 1999) seems to be derived within the crotaphytids.

Montanucci (1978) suggested that the cryptic dorsal pattern of leopard lizards (*Gambelia wislizenii*), similar to that of *C. reticulatus*, is an adaptation for efficient predation. Basically, a more cryptic predator is more likely to sneak up on prey than a more conspicuous predator, and this was supported by Montanucci (1978) with the observation that *G. wislizenii* exhibited a type of stalking behavior while pursuing prey. Montanucci (1978) alluded to the idea that reticulate collared lizards might also display this behavior. Indeed, *C. reticulatus* was observed by us to exhibit what might be considered stalking behavior. Before moving from a perch, lizards were in every case (with the exception of some movements by males that were clearly to chase

TABLE 2—Mean (\pm SE) movements per minute (MPM) and proportion of time spent moving (PTM), with and without “display” movements, for 10 adult male and 10 female reticulate collared lizards (*Crotaphytus reticulatus*) in southern Texas. Values with the same superscript were significantly different ($P < 0.025$).

	MPM			PTM		
	Male	Female	Combined	Male	Female	Combined
With “display” movements	0.318 \pm 0.082 ^a	0.318 \pm 0.073	0.318 \pm 0.054 ^b	0.018 \pm 0.006 ^c	0.014 \pm 0.003	0.016 \pm 0.003 ^d
Without “display” movements	0.178 \pm 0.061 ^a	0.308 \pm 0.071	0.243 \pm 0.048 ^b	0.011 \pm 0.005 ^c	0.014 \pm 0.003	0.012 \pm 0.003 ^d

off intruding rivals) observed to move slowly at first before making a dash to a new perch or to capture prey. However, this behavior could also be used to avoid predators. Moving slowly at first before quickly running to a new location would be strongly favored in an environment with high predation pressure, such as that estimated at our study site by our observations (0.442 predator sightings/hour).

There has been recent interest in the literature as to what measure of foraging mode is most appropriate for inter-species comparisons (see Cooper et al., 2001). For our data, MPM, PTM, and PAM all gave the same general conclusion that reticulate collared lizards are ambush foragers. However, we found that MPM and PTM were significantly affected by elimination of “social movements” but PAM was unaffected by definition. The modified measures of MPM and PTM, however, did not change our conclusions about foraging mode. We agree with Cooper et al. (2001) that PAM might have the least problems among measurements of foraging mode.

We would like to thank the staff of Falcon State Park and M. Lockwood and D. Riskind at Texas Parks and Wildlife for allowing us access to the study site and helping us to obtain permits. We would also like to thank M. Husak, S. Fox, and 2 anonymous reviewers for comments on earlier versions of the manuscript and S. Fox for writing the Spanish abstract. All work was conducted under Texas Parks and Wildlife Department Scientific Permit SPR0299-001 and Texas Parks and Wildlife State Park Scientific Permit 57-01.

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- Submitted 14 November 2001. Accepted 13 March 2002.
Associate Editor was Geoffrey C. Carpenter.

SURVEY OF BLOOD PARASITES IN ROSS' AND WHITE-FRONTED GEESE IN SOUTHERN TEXAS

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ABSTRACT—Blood parasites have been known to cause morbidity and mortality in waterfowl, particularly in Canada geese (*Branta canadensis*). However, little is known about blood parasites infecting Ross' geese (*Chen rossii*) and greater white-fronted geese (*Anser albifrons*). This study examined wintering Ross' and white-fronted geese for blood parasites. Blood smears from 16 Ross' geese (13 juveniles, 3 adults) and 46 white-fronted geese (21 juveniles, 25 adults) collected in Kleberg County, Texas, during winter 1999–2000 were examined for blood parasites. *Leucocytozoon simondi* was found in 2 juvenile white-fronted geese; density of infection was <1 parasite/2,000 erythrocytes. Additionally, 3 adult and 1 juvenile white-fronted geese had microfilaria. No blood parasites were observed in Ross' geese. We concluded that low prevalence and density of *L. simondi* gametocytes circulating in host blood precluded or greatly reduced transmission of this parasite on the wintering grounds in southern Texas.

RESUMEN—Se sabe que los parásitos de sangre causan morbilidad y mortalidad en aves acuáticas, particularmente en gansos canadienses (*Branta canadensis*). Sin embargo, se sabe muy poco sobre parásitos de sangre infectando gansos de Ross (*Chen rossii*) y gansos frente blanca (*Anser albifrons*). Este estudio examinó gansos de Ross y gansos frente blanca durante el invierno para parásitos de sangre. Muestras de sangre de 16 gansos de Ross (13 juveniles, 3 adultos) y 46 gansos frente blanca (21 juveniles, 25 adultos) colectadas en el condado de Kleberg, Texas, durante el invierno de 1999–2000 fueron examinadas para parásitos de sangre. Se encontró *Leucocytozoon simondi* en 2 gansos frente blanca juveniles; la densidad de infección fue <1 parásito/2,000 eritrocitos. Adicionalmente, 3 gansos frente blanca adultos y 1 juvenil tenían microfilarios. No se observaron parásitos de sangre en ningún ganso de Ross. Concluimos que la baja frecuencia y densidad de gametocitos *L. simondi* circulando en la sangre de los ejemplares imposibilitan o grandemente reducen la transmisión de este parásito en los terrenos invernales del sur de Texas.

Numerous studies have been conducted to determine factors affecting survival and recruitment of waterfowl in North America. One factor that might negatively affect waterfowl is blood parasites. At least 9 blood protozoan and 2 microfilarid species have been reported from Canada geese (*Branta canadensis*) and snow geese (*Chen caerulescens*) in North America (Bennett et al., 1982). Morbidity and mortality in Canada geese have occurred on the breeding grounds from *Leucocytozoon simondi* infections (Herman et al., 1975). Unfortunately, little is known about which species of blood parasites infect other goose species or their impact on host individuals and populations on the breeding or wintering grounds. For example, no published studies have examined Ross' geese (*Chen rossii*) for blood parasites, and in the only study of white-fronted geese (*Anser albifrons*), 5 birds from California were examined (Wood and Herman, 1943).

Vectors presumably are absent during winter in temperate regions, thereby breaking the infection-transmission cycle. However, Fedynich and Rhodes (1995) found high prevalence of *Leucocytozoon smithi* (100%) and *Haemoproteus meleagridis* (54%) during winter in a population of wild turkeys (*Meleagris gallopavo*) from South Carolina, suggesting continued transmission. Additionally, Atkinson et al. (1988) used sentinel turkeys to demonstrate that infections with *H. meleagridis* can occur year-round in warm climate areas, such as southern Florida. Geese wintering at the southernmost extension of their range might similarly be exposed to blood parasite infections if temperatures permit vector activity. To learn more about blood parasite infections during winter, we surveyed Ross' and white-fronted geese for

blood parasites in southern Texas during winter 1999–2000.

Geese were collected by shooting during the hunting season in Kleberg County, Texas, from 3 November 1999 through 20 January 2000. Heart blood from each bird was used to make thin smears on 2 microscope slides within 10 min of host death. Blood samples were collected between 0600 and 1100 h to increase the probability of *Leucocytozoon* detection, as at least 1 species of this genus is known to demonstrate daily cyclic activity (Noblet and Noblet, 1976). Smears were fixed in 100% methanol for 1 min, air dried, and stained with Diff-Quik. Each smear was scanned 15 min (30 min for each bird) for blood parasites with a microscope at 1,000 \times magnification. Leucocytozoids were identified using the descriptions of Bennett and Squires-Parsons (1992). To quantify density of blood protozoa, 1 slide was selected (based on stain quality and density of erythrocytes appropriate for viewing) and 2,000 erythrocytes were counted and examined in 20 replicates of 100 erythrocytes each, following the recommendations of Godfrey et al. (1987). Although the protocols of Godfrey et al. (1987) were developed for *Haemoproteus*, they have been successfully applied to *Leucocytozoon* (Fedynich and Rhodes, 1995; DeJong et al., 2001). No satisfactory techniques have been developed to estimate density of microfilaria using blood smears; consequently, only prevalence data are reported for microfilaria.

Sixteen Ross' geese (13 juveniles, 3 adults) and 46 white-fronted geese (21 juveniles, 25 adults) were collected and examined for blood parasites; *L. simondi* was found in 2 juvenile white-fronted geese (Table 1). Density in each of the 2 birds was <1 parasite/2,000 erythrocytes. Additionally, 3 adult and 1 juvenile white-

TABLE 1—Numbers of Ross' and white-fronted geese examined, numbers infected with blood parasites, and parasite prevalence (%) in heart blood during winter 1999–2000 in southern Texas.

Host species	Number examined	<i>Leucocytozoon simondi</i>	Microfilaria	Total
Ross' goose				
Adult	3	0	0	
Juvenile	13	0	0	
White-fronted goose				
Adult	25	0	3 (12)	3 (12)
Juvenile	21	2 (10)	1 (5)	3 (14)
Total	62	2 (3)	4 (6)	6 (10)

fronted geese were infected with an unknown species of microfilarid. No blood parasites were found in the 16 Ross' geese (Table 1).

Transmission of blood protozoa is dependent upon adequate numbers of susceptible hosts, vectors, and infective parasite gametocytes circulating in the blood. Although many susceptible waterfowl hosts occur in Kleberg County during winter, it is uncertain what potential vectors occur in this region or if they are active during winter due to the typically mild temperatures. Mean monthly ambient temperatures recorded in November 1999, December 1999, and January 2000 at the nearest weather station (Kingsville, Texas) to our collection sites were 19.3°C (mean monthly minimum/maximum range: 11.8 to 26.8°C), 14.9°C (7.8 to 21.9°C), and 17.4°C (10.3 to 24.4°C), respectively. These temperatures suggest that limited vector activity was possible.

We could find only 1 insect survey that was conducted during summer in nearby Victoria County (Johnson, 1982). Of the dipteran families known to transmit *Haemoproteus*, *Plasmodium*, and *Leucocytozoon*, numerous species of Culicidae, 2 species of Ceratopogonidae, and no Simuliidae were found (Johnson, 1982). However, none of these dipteran species have been implicated as vectors of waterfowl blood protozoa. Additionally, infective gametocytes were not readily observable as only 1 blood protozoan (*L. simondi*) was found, which occurred at extremely low prevalence and density in white-fronted geese, and no blood parasites

were found on smears from Ross' geese. A representative specimen of *L. simondi* was deposited in the Queensland Museum-International Reference Center for Avian Hematozoa Collection, South Brisbane, Queensland, Australia; Number G463709.

Our findings are similar to those of other studies that examined blood parasites in ducks during winter (Kocan et al., 1979; Fedynich et al., 1993). Collectively, these studies support the notion that low prevalence and density of blood parasites during winter in temperate regions likely represents an adaptive strategy by parasites to limit production of gametocytes when vector density is insufficient to maintain the transmission phase of the host-parasite-vector cycle (Greiner, 1991).

Lack of blood parasites in Ross' geese was unexpected, as snow geese are infected with several species of blood protozoa and a microfilaria (Bennett and MacInnes, 1972). Both Ross' and snow geese often co-occur on the breeding grounds (Kerbes et al., 1983), providing Ross' geese with the opportunity to be exposed to the same blood parasites as snow geese. Furthermore, our sample of Ross' geese had a high proportion of juveniles, which are thought to be more susceptible than adults (Wehr and Herman, 1954). It is possible that our sample size of Ross' geese, coupled with apparently low prevalence and density of blood parasites during winter in southern Texas, reduced the probability of detection.

It was evident in our study that the lack of gametocytes, or low densities of gametocytes circulating in host blood, precluded or greatly reduced the possibility of transmission on the wintering grounds in southern Texas. Consequently, health risks associated with protozoa inhabiting erythrocytes and leucocytes during the gametocyte phase of their life cycle were minimal for these 2 goose species wintering in southern Texas during 1999–2000.

Financial support was provided by the Caesar Kleberg Wildlife Research Institute. This is manuscript number 01-110 of the Caesar Kleberg Wildlife Research Institute. This study was approved by the Texas A&M University-Kingsville Animal Care and Use Committee, authorization number Y2K-6-3.

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- Submitted 6 March 2001. Accepted 12 July 2002.
Associate Editor was William H. Baltosser.

SUBTLE RECENT DISTRIBUTIONAL SHIFTS IN GREAT PLAINS BIRD SPECIES

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ABSTRACT—Changes in geographic distributions of 5 bird species endemic to the Great Plains of North America were examined over the last few decades based on the United States Breeding Bird Survey. Examining the mean latitude of individuals of each species, 3 species showed significant or near-significant northward shifts, and 1 a significant shift southward. Over all 5 species examined, colonization events were concentrated in the northern part of the distributions of the species; in 3 species, extinctions were concentrated in the southern part of the distributions of the species. The conclusion is that significant distributional changes have taken place, but they have been subtle, and might be associated with global climate change.

RESUMEN—Se examinaron los cambios en la distribución geográfica de 5 especies de aves en las Grandes Planicies de Norteamérica a través de las últimas décadas con base a datos del United

States Breeding Bird Survey. Examinando la latitud promedio de individuos de cada especie, 3 especies mostraron traslaciones significativas o casi-significativas hacia el norte, y 1 significativa hacia el sur. Para las 5 especies, colonizaciones se enfocaron en la parte norteña de las distribuciones de las especies, y en 3 especies las extinciones se enfocaron en la parte sureña de sus distribuciones. La conclusión es que han habido cambios distribucionales significativos, pero sutiles, posiblemente asociados al cambio de clima global.

Global climates are in the process of rapid change, apparently owing to the effects of human activities on the Earth's atmosphere (Houghton et al., 1995). Although subtle when observed over short time-spans (Hulme et al., 1999), changes are real. Indeed, significant effects on elements of biodiversity have already been documented, including distributional and phenological shifts in species and ecosystems (Parmesan, 1996; Allen and Breshears, 1998; Visser et al., 1998), among other examples. However, the magnitude and commonness of such effects are only now being appreciated. This note documents subtle northward colonization and southern extinction in bird species in North America.

The United States Breeding Bird Survey (BBS) data represent an extremely valuable source of information on bird distributions, especially in that they comprise long time series of repeated, standardized surveys. I chose for analysis bird species endemic or at least concentrated in the portion of the Great Plains well covered by the BBS (the United States and southernmost portion of Canada). I further reduced the study area to coincide with areas known to have experienced a fairly uniform increase in annual mean temperatures over the past 3 decades (Karl et al., 1996). To focus on whole-distribution effects, I eliminated Great Plains species with significant populations farther east (e.g., upland sandpiper, *Bartramia longicauda*), west (e.g., Brewer's sparrow, *Spizella breweri*), north (e.g., longspurs, *Calcarius*), or south (e.g., greater roadrunner, *Geococcyx californianus*). Hence, 6 species remained for analysis: ferruginous hawk (*Buteo regalis*), lesser prairie-chicken (*Tympanuchus pallidinuchus*), mountain plover (*Charadrius montanus*), scissor-tailed flycatcher (*Tyrannus forficatus*), lark bunting (*Calamospiza melanocorys*), and Cassin's sparrow (*Aimophila cassinii*). Although ferruginous hawk does have populations farther west, the clear concentration of its abundance is in the central and northern Great Plains, hence its inclusion. One species (lesser prairie-chick-

en) had to be eliminated because too few BBS routes had recorded it, leaving just 5 species for analysis.

BBS routes with continuous coverage over the past 25 years were chosen for analysis. This quality-control measure left 1,331 routes in the Great Plains region for analysis. Data from these routes were divided into 5 categories of 5-years for analysis: 1971–1975, 1976–1980, 1981–1985, 1986–1990, and 1991–1995. Species were tallied as present or absent, and average numbers recorded were calculated for each 5-year sample for each route.

One analytical approach was that of regressing average latitudes of occupied sites, or average latitudes of individuals of a particular species (i.e., weighting sites occupied by numbers present), on time period (Table 1). Whereas the conservative sites-occupied analysis produced no significant regressions (all $P > 0.225$), 3 species showed significant or near-significant movement of average positions of individuals: Cassin's sparrows moved northward at approximately 0.6° latitude per decade (regression slope $m = 0.328$, $R^2 = 0.865$, $P = 0.022$), mountain plovers apparently moved northward at 0.4° per decade ($m = 0.212$, $R^2 = 0.695$, $P = 0.079$, near-significant), and scissor-tailed flycatchers moved southward at 0.4° per decade ($m = -0.231$, $R^2 = 0.921$, $P = 0.010$). Regression slopes were positive (northward movement) in 4 of the 5 species analyzed. However, these analyses were confounded by nonindependence of samples in different time periods, making interpretation of statistical significance complex.

Because samples were not independent with respect to time periods, I reanalyzed the data focusing on actual colonizations and extinctions of populations. I defined colonizations as routes on which a species was absent in the first 5-year period, but present in the last, and extinctions as those routes on which a species was present in the first 5-year period, but absent in the last. I then tallied colonizations and extinctions north and south of the median latitude

TABLE 1—Summary of colonizations (newly occupied sites) and extinctions (sites no longer occupied) north and south of median latitudes for 5 bird species in the Great Plains, and regression analyses assessing trends in average latitudes of occupied sites and total individuals for each species.

Species	Colonization		Extinction		Occupied sites	Numbers
	North	South	North	South		
<i>Buteo regalis</i>	27	22	9	9	$m = 0.109$ $R^2 = 0.165$ $P = 0.497$	$m = 0.059$ $R^2 = 0.041$ $P = 0.743$
<i>Charadrius montanus</i>	3	2	1	2	$m = 0.462$ $R^2 = 0.434$ $P = 0.226$	$m = 0.212$ $R^2 = 0.695$ $P = 0.079$
<i>Tyrannus forficatus</i>	12	3	8	6	$m = 0.025$ $R^2 = 0.270$ $P = 0.369$	$m = -0.231$ $R^2 = 0.921$ $P = 0.010$
<i>Calamospiza melanocorys</i>	8	7	14	18	$m = 0.116$ $R^2 = 0.190$ $P = 0.464$	$m = 0.050$ $R^2 = 0.012$ $P = 0.859$
<i>Aimophila cassinii</i>	13	6	5	8	$m = -0.011$ $R^2 = 0.009$ $P = 0.879$	$m = 0.328$ $R^2 = 0.865$ $P = 0.022$

for the species in question (Table 1). In all 5 species, colonizations were focused in the northern half of the distribution of the species (5 of 5 species; binomial test, $P = 0.031$), whereas extinctions tended to be in the southern half of distribution of the species (3 of 5 species; binomial test, $P = 0.125$). Hence, species are moving northward via colonization, and populations at the southern extreme are either persisting or are going extinct more slowly than colonization occurs.

To provide additional detail on population processes, I examined temporal trends in different sectors of the geographic distributions of each species. Separating BBS routes into those north and south of the median latitude of sites occupied for a particular species, I calculated linear regressions between population trends and time, and summarized differences between northern and southern sectors of the distribution of the species across 25 yr (Table 1). Several species showed concentrations of declining populations in the southern part of their geographic distributions (e.g., Cassin's sparrow, mountain plover), and others showed increasing populations in northern sectors (e.g., ferruginous hawk, lark bunting, mountain plover). These patterns of population decline and increase, although preliminary in nature, are consistent with the effects of a warm-

ing climate, in which southern areas become uninhabitable and northern areas become increasingly suitable.

Climatic warming in the Great Plains has been subtle, but measurable (Karl et al., 1996); bird distributions have also been moving northward. Conclusions as to causal links, such as that climate change caused the northward distributional shifts, are difficult, perhaps requiring physiological measurements or studies of habitat distributions and resource availability not within the scope of the present study. All the same, few other plausible explanations are available; human influence on landscapes is distributed fairly evenly across the Great Plains landscape, and, hence, is not likely to have caused the shifts observed.

Especially notable regarding the shifts documented herein is their magnitude. The distribution of Cassin's sparrows is moving northward at more than half a degree of latitude per decade, which, if true, would translate into more than 5 km per year of northward shift. This rate of movement is similar to those documented in other habitat types in apparent response to climate change (Allen and Breshears, 1998). These shifts suggest that coming decades of continued climatic change will shift avian distributions in North America significantly northward. Ecological and evolutionary

ramifications of these changes (e.g., Holt, 1990) are numerous, serious, and poorly understood, thus emphasizing the importance of careful monitoring of avian populations and distributions to document such distributional changes in detail.

Many thanks to C. Parmesan for suggesting that these analyses would be fruitful. Thanks also to B. Peterjohn of the United States Fish and Wildlife Service for generously providing access to the Breeding Bird Survey data. Financial support for this work was provided by the National Science Foundation and the National Center for Ecological Analysis and Synthesis.

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*Submitted 12 March 2001. Accepted 12 July 2002.
Associate Editor was William H. Baltoser.*

EFFECTS OF TORNADO DAMAGE ON FOREST BIRD POPULATIONS IN THE ARKANSAS OZARKS

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ABSTRACT—Breeding bird populations were studied in forests recently damaged by tornados and in adjacent undamaged forests in the Ozark National Forest, Arkansas. During 1999 and 2000, surveys were undertaken at 6 points in forest moderately damaged by a tornado in 1996 and at 6 points in nearby undamaged forest. An additional 18 counts, 6 each in undamaged, moderately damaged, and heavily damaged forest, were undertaken in 2000 in an area affected by a 1999 tornado. Typical forest species, such as red-eyed vireo and ovenbird, were significantly less abundant in tornado-damaged forest than in undamaged forests, while edge species, such as indigo bunting and white-eyed vireo, were more abundant in damaged forest than in undamaged forest. Surprisingly, abundances of some species, such as black-and-white warbler, did not differ significantly between damaged and undamaged forests. Species composition differed between heavily damaged forest and moderately damaged or undamaged forest, with a number of species occurring only in the heavily damaged forest type. The congeneric summer tanager and scarlet tanager seemed to show habitat segregation in the study sites, with summer tanagers occurring in tornado-damaged forest and scarlet tanagers occurring in undamaged forest.

RESUMEN—Muestreamos poblaciones de aves reproductivas en bosques recientemente afectados por tornados y en bosques adyacentes intactos en el Ozark Nacional Forest, Arkansas. Durante

1999 y 2000, realizamos conteos en 6 lugares moderadamente afectados por un tornado en 1996 y en 6 lugares cercanos en bosque intacto. Además, en el 2000 realizamos 18 conteos adicionales, 6 en cada uno de bosque intacto, moderadamente afectado, y fuertemente afectado, en un área dañada por un tornado en 1999. Especies típicas de bosque, como *Vireo olivaceus* y *Seiurus aurocapillus*, fueron significativamente menos abundantes en áreas afectadas que en las no afectadas, mientras que especies de ecotonos, como *Passerina cyanea* y *V. griseus*, fueron más abundantes en áreas afectadas que en las no afectadas. Sorprendentemente, la abundancia de algunas especies, como *Mniotilta varia*, no difirió significativamente entre bosques dañados y no dañados. La composición de especies difirió entre bosques muy dañados y bosques moderadamente o no dañados, con varias especies encontradas solamente en los bosques fuertemente dañados. Dos especies congéneres, *Piranga olivacea* y *P. rubra* parecieron mostrar segregación de hábitat en los sitios de estudio, con *P. rubra* en bosques dañados y *P. olivacea* en bosques no dañados.

Tornados are a common disturbance in forests of the southeastern United States. In Arkansas, several hundred hectares of forest are damaged by tornados annually in the Ozark National Forest (United States Forest Service, unpublished data), and it has been suggested that tornados have been the most important historical cause of forest disturbance (Turner, 1935). Little work has focused on effects of tornados on bird populations, and there are almost no data available on bird communities in tornado-damaged forests. The few papers that have been published on the short-term effects of catastrophic wind damage on forest structure (Held and Winstead, 1976; Glitzenstein and Harcombe, 1988; Foster, 1989) indicate that tornados reduce density and total basal area of canopy trees but increase tree diversity, understory cover, and seedling recruitment. Longer-term studies show continued reduction in tree density and basal area for several years after the disturbance, probably due to delayed mortality of damaged trees (Held and Bryant, 1989); at least 20 years are generally required to recover pre-disturbance forest structure (Everham and Brokaw, 1996; Held et al., 1998). Based on these data, tornados might be expected to have significant indirect impacts on the avian community in response to disturbance-induced changes in forest structure. Altering forest structure due to changing forest management practices and regrowth of abandoned fields is likely responsible for the decline of populations of some songbirds that breed in secondary forest and successional habitats (Robbins et al., 1989; Thompson et al., 1992; Peterjohn et al., 1995). Therefore, natural disturbances that create secondary forest and edge habitat might be important in the persistence of many of these species.

On 21 April 1996, a tornado damaged approximately 2,000 ha of forest south and east of White Rock Mountain, Franklin County, Arkansas. Two 6-ha (300 m x 200 m) plots were created in forest dominated by various oaks (*Quercus*), hickories (*Carya*), and sweetgum (*Liquidambar styraciflua*) in 1998 along Salt Fork Creek just southeast of White Rock Mountain. One plot was created in moderately damaged forest (10 to 25% of trees down) and one in nearby undamaged forest on the slopes above the creek. In each plot, 6 points were established on each plot. Points were evenly spaced at 100-m intervals and 50 m from the edge of the plots. On 4 May 1999, a tornado damaged >700 ha of forest in Whitzon Hollow, a small drainage near the Arkansas-Oklahoma border in Crawford County, Arkansas. Eighteen points were established in late May 2000 along old roads through forest dominated by oaks, hickories, and sycamores (*Plantanus occidentalis*). Six points each were placed in undamaged, moderately damaged (10 to 25% of trees down), and heavily damaged (25 to 75% of trees down) forest.

To determine avian community structure in relation to forest damage, birds were surveyed at each point location using 50-m fixed-radius point counts (Hutto et al., 1986). Surveys were conducted for 10-minute periods between 0600 and 0900, and each point was surveyed twice each year. White Rock points were surveyed in late May and early June of both 1999 and 2000; Whitzon Hollow points were surveyed only in late May and early June 2000. To compare vegetation structure in relation to different levels of forest damage, percent canopy cover was measured by averaging 5 spherical densiometer readings taken at each point location (Lemmon, 1957) and at locations 30 m

TABLE 1—Relative abundance (mean \pm SD of territorial males/point) of bird species at different levels of forest damage at White Rock, Arkansas.

Species	Year	No damage	Moderate damage	ANOVA $F_{1,10}$	P -value
Acadian flycatcher	1999	0.33 \pm 0.41	0.33 \pm 0.41	0.44	0.523
	2000	0.17 \pm 0.41	0.42 \pm 0.49		
Red-eyed vireo	1999	1.91 \pm 0.55	1.33 \pm 0.41	0.09	0.771
	2000	1.50 \pm 0.55	1.58 \pm 0.37		
Black-and-white warbler	1999	0.33 \pm 0.26	0.42 \pm 0.20	0.05	0.962
	2000	0.42 \pm 0.38	0.33 \pm 0.26		
Ovenbird	1999	0.25 \pm 0.27	0.08 \pm 0.20	3.31	0.096
	2000	0.33 \pm 0.41	0.08 \pm 0.20		
Worm-eating warbler	1999	0.17 \pm 0.26	0.33 \pm 0.41	0.65	0.478
	2000	0.17 \pm 0.26	0.17 \pm 0.26		
Hooded warbler	1999	0.50 \pm 0.45	0.42 \pm 0.38	0.44	0.523
	2000	0.33 \pm 0.41	0.67 \pm 0.41		
Indigo bunting	1999	0.08 \pm 0.20	0.41 \pm 0.38	9.00	0.012
	2000	0.17 \pm 0.26	0.58 \pm 0.38		
Tanager species ^a	1999	0.50 \pm 0.42	0.17 \pm 0.26	3.39	0.087
	2000	0.33 \pm 0.41	0.17 \pm 0.20		

^a Scarlet tanager in undamaged forest; summer tanager in moderately damaged forest.

from each point in each of the cardinal directions.

We estimated relative abundance of all bird species on the study areas in territorial males per point by dividing the total number of singing males detected by the number of points surveyed. The average of the 2 surveys within a year was used as the measure of relative abundance for each point. Analysis of variance using year as a blocking factor, and points within each habitat (undamaged or moderately damaged) as subsamples, was used to determine if there were differences in the relative abundance of a given species between levels of forest damage at White Rock. Analysis of variance using points within each habitat (undamaged, moderately damaged, or heavily damaged) as subsamples was used to determine if there were differences in the relative abundance of a given species among levels of forest damage at Whitzen Hollow. Statistical analyses were only conducted on species that reached a relative abundance of 0.33 males/point in at least 1 habitat type due to problems with statistical power of the analyses for less common species. Analytical procedures were performed using JMP statistical software (SAS Institute, 1995), with significance placed at $P \leq 0.05$ for all tests.

Seven species of birds occurred at a relative

abundance sufficient for analysis at 1 of the plots at White Rock (Table 1), and 10 species of birds occurred at a relative abundance sufficient for analysis in at least 1 level of forest damage at Whitzen Hollow (Table 2). Six species, the Acadian flycatcher (*Empidonax virens*), red-eyed vireo (*Vireo olivaceus*), ovenbird (*Sieurus aurocapillus*), worm-eating warbler (*Helmitheros vermivorous*), black-and-white warbler (*Mniotilta varia*), and indigo bunting (*Passerina cyanea*) occurred at a relative abundance sufficient for analysis at both sites. One congeneric species-pair, the scarlet tanager (*Piranga olivacea*) and summer tanager (*P. rubra*), also occurred at both sites in numbers sufficient for analysis (Tables 1 and 2).

Mean canopy cover was 91 \pm 3% (range 87 to 95%) on undamaged transects, 73 \pm 8% (64 to 82%) on moderately damaged transects, and 31 \pm 6% (25 to 41%) on heavily damaged transects at Whitzen Hollow. Mean canopy cover was 94 \pm 3% (90 to 97%) on the undamaged plot and 63 \pm 15% (37 to 76%) on the moderately damaged plot at White Rock.

Based on vegetation changes typically associated with tornado damage (Held and Winstead, 1976; Glitzenstein and Harcombe, 1988), one would expect to find reduced numbers of undisturbed forest bird species and increased numbers of forest-edge species in tornado-

TABLE 2—Relative abundance (mean \pm SD of territorial males/point) of bird species at different levels of forest damage at Whitzen Hollow, Arkansas.

Species	No damage	Moderate damage	Heavy damage	ANOVA $F_{2,15}$	P -value
Acadian flycatcher	1.00 \pm 0.32	0.33 \pm 0.52	0.00	7.27	0.022
Carolina wren	0.00	0.25 \pm 0.27	0.83 \pm 0.41	8.45	0.016
Blue-gray gnatcatcher	0.50 \pm 0.35	0.42 \pm 0.49	0.25 \pm 0.27	0.92	0.496
Red-eyed vireo	2.17 \pm 0.26	1.75 \pm 0.42	0.92 \pm 0.38	19.02	<0.001
White-eyed vireo	0.00	0.33 \pm 0.41	0.67 \pm 0.41	6.43	0.030
Northern parula	0.75 \pm 0.52	0.92 \pm 0.49	0.43 \pm 0.41	2.38	0.127
Black-and-white warbler	0.42 \pm 0.49	0.58 \pm 0.38	0.67 \pm 0.40	0.53	0.599
Ovenbird	0.92 \pm 0.20	0.42 \pm 0.38	0.00	8.18	0.017
Worm-eating warbler	0.25 \pm 0.42	0.33 \pm 0.41	0.25 \pm 0.42	1.59	0.237
Indigo bunting	0.42 \pm 0.38	0.58 \pm 0.49	1.92 \pm 0.49	19.47	<0.001
Tanager species ^a	0.42 \pm 0.20	0.25 \pm 0.27	0.42 \pm 0.38	0.65	0.539

^a Scarlet tanager in undamaged forest; summer tanager in moderately and heavily damaged forests.

damaged areas. This was the case in this study, with most species responding to changes according to predictions based on habitat preferences (James, 1971; Shugart and James, 1973; Anderson and Shugart, 1974). Nevertheless, several species showed little change in density in the tornado-damaged areas. A possible explanation is that these tornados did not level all the vegetation in the study areas, often leaving many subcanopy trees and some taller canopy trees standing. The more open canopy allowed some species that use light gaps or forest edges, such as the indigo bunting, to increase in numbers, while only slightly reducing numbers of primary forest species like Acadian flycatchers.

Selective logging is probably the human treatment that most closely mimics tornado damage to forests in the eastern United States, but there are few studies of selective logging effects on bird populations in southeastern forests (Thompson et al., 1992, 1995; Dickson et al., 1995; Smith, 1998). Existing studies suggest that uneven-aged management benefits species, such as indigo buntings and hooded warblers (*Wilsonia citrina*), that are able to use small gaps in the forest, and reduces numbers of species, such as the ovenbird, that need nearly continuous forest (Thompson et al., 1992, 1995; Dickson et al., 1995). The multi-layered forest present in areas of moderate tornado damage appears to mimic that created by some types of uneven-age management. However, forest structure due to tornado damage differs from most logging practices, which tend

to create stands that are more homogenous in both age and structure (Smith, 1986; Thompson et al., 1995). These differences seem to lead to different avian community structure in forest damaged by tornados and logged areas. For example, Acadian flycatchers and red-eyed vireos are typically associated with areas of dense canopy cover and sparse understory cover in the Ozarks (Smith, 1977) and occur in significantly reduced numbers in selectively logged forest plots (Rodewald and Smith, 1998). In areas of moderate tornado damage, these species were present in only slightly reduced numbers (Tables 1 and 2). Likewise, hooded warblers, which prefer thick understory vegetation, did not increase in the tornado-damaged area at White Rock (Table 1) as they did elsewhere in the selectively logged plots of (Rodewald and Smith, 1998).

Heavy tornado damage produces quite different forest structure and avian community structure than does moderate tornado damage. In the heavily damaged areas at Whitzen Hollow, few canopy trees remained standing and many subcanopy trees were blown down. This effect was similar to that produced by a clearcut or seed-tree cut (Smith, 1986). The understory in this area had already responded to the open canopy, and a thick layer of shrubs covered most of the area at the time of the point counts. There was consequently a significant reduction or complete absence of birds associated with heavy canopy cover or light understory cover, and most of the species common in undamaged forest were in greatly re-

duced numbers in the heavily damaged forest (Table 2). By contrast, forest edge and shrub species, such as Carolina wren (*Thryothorus ludovicianus*), white-eyed vireo (*Vireo griseus*), and indigo bunting, occurred in significantly increased numbers in heavily damaged forest (Table 2), and a suite of species found rarely or not at all in forest habitats also was present. This group included blue-winged warbler (*Vermivora pinus*), prairie warbler (*Dendroica petechia*), yellow-breasted chat (*Icteria virens*), brown-headed cowbird (*Molothrus ater*), and eastern towhee (*Pipilo erythrophthalmus*). However, none of these species reached a relative abundance of 0.33 males/point.

The presence of summer tanagers rather than scarlet tanagers in the tornado-damaged areas is somewhat surprising because scarlet tanagers have been found to occur in only slightly reduced numbers in small, selectively logged plots in the Ozarks (Rodewald and Smith, 1998). Other studies in the Ozarks have shown that where scarlet and summer tanagers co-occur, summer tanagers occupy the drier and more open forest types (Shugart and James, 1973), and habitat ordinations suggest that scarlet tanagers respond positively to denser canopy cover and larger trees while summer tanagers respond positively to higher numbers of small trees (James, 1971; Anderson and Shugart, 1974). Tornado damage might have reduced canopy cover to the point where scarlet tanagers were eliminated completely, but left enough small trees to provide favorable habitat for summer tanagers. Overall tanager abundance did not differ significantly between forest damage types in either location (Tables 1 and 2), though summer tanager numbers on the moderately damaged plot at White Rock were somewhat lower than scarlet tanager numbers on the undamaged plot.

Avian populations are affected by a wide variety of natural disturbances (Rotenberry et al., 1995), many of which are still poorly studied. The data from this study suggest that tornado damage to forests might be important in determining avian community structure in forests of the southeastern United States. This natural disturbance appears to alter forest structure and avian community structure in ways that are different from logging practices. More studies are needed to determine if there are consistent responses of breeding birds to tornado

damage in forests of the eastern United States and other locations.

We thank R. Odegard of the United States Forest Service, Ozark Office, for sponsoring this project. Additional thanks go to C. Hardgrave of Oklahoma State University for help setting up plots and measuring vegetation at White Rock and to J. Briggler of the University of Arkansas for comments on the manuscript and help measuring vegetation at Whitzzen Hollow. Funding for this study was provided by a cost-share agreement between the United States Forest Service, Boston Mountain Ranger District, Ozark National Forest and the University of Arkansas.

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- Submitted 3 October 2000. Accepted 4 April 2002.
Associate Editor was William H. Baltoser.*

UNUSUAL OCCURRENCE OF WORM-EATING WARBLER (*HELMITHEROS VERMIVORUS*) IN INTENSIVE AGRICULTURAL LANDSCAPE IN NORTHEASTERN ARKANSAS

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ABSTRACT—The worm-eating warbler (*Helmitheros vermivorus*) is a ground-dwelling neotropical migrant that typically breeds within large sloping forest patches. On 7 July 2000 at 07:36 CDT, a female *H. vermivorus* was captured at an agricultural ditch in Poinsett County, Arkansas. The warbler was aged as after hatching year (ASY) and had a fully developed brood patch with complete feather loss. This occurrence was unusual because it was outside the reported breeding range for this species and the warbler was in a small (3.52 ha) forest patch in a flat landscape. Observations of worm-eating warblers in alluvial areas of Arkansas occur during the migratory period, but are relatively rare. The timing of this summer observation, presence of a brood patch, and lack of fat deposition in the furcular cavity indicated a recent breeding attempt; however, this does not preclude a dispersing individual. In either case, closer monitoring in lowland habitats during breeding and post-breeding periods might provide additional knowledge on the conservation status and habitat needs of this species.

RESUMEN—El gorjeador o cerrojillo (*Helmitheros vermivorus*) es un pájaro migrante neotropical que vive principalmente en el suelo y se reproduce típicamente en parches grandes de laderas boscosas. El 7 de julio del año 2000, a las 07:36 horas (CST) capturamos una hembra *H. vermivorus* en una acequia de riego en el condado de Poinsett, Arkansas. Se determinó que su edad era de un año

después de la primera reproducción (ASY) y presentó un parche incubador completamente desarrollado con la pérdida completa de plumaje. Este evento fue singular porque se lo encontró fuera del área de apareamiento habitual de esta especie y el pájaro se ubicó en un parche pequeño (3.52 ha) de bosque plano. Se han observado a cerrojillos en áreas fluviales de Arkansas durante el período de migración, pero en muy pocas ocasiones. La época de esta observación veraniega, la presencia de un parche incubador, y la falta de depósitos grasos en la cavidad furcular indicaron un reciente intento reproductivo; sin embargo, esto no excluye la posibilidad de que se trate de un caso aislado. De todas maneras, observaciones más minuciosas en hábitats de tierras bajas durante el período reproductivo y post-reproductivo pueden proveer conocimiento adicional sobre el estado de conservación y las necesidades ambientales de esta especie.

The worm-eating warbler (*Helmitheros vermivorus*) is a ground-nesting Neotropical migratory passerine (Hanners and Patton, 1998). Typical breeding habitat consists of large contiguous tracts of forests with steeply to moderately sloping terrain, but also mixed inland and coastal lowland forest (Wenny et al., 1993, Hanners and Patton, 1998). On 7 July 2000 at 07:36 CDT, we captured a female worm-eating warbler in a 2.6-m by 12-m, 30-mm mesh mist-net deployed at the Judd Hill Experimental Cotton Plantation (35°37.268'N, 90°31.496'W) in Poinsett County, Arkansas. This region of northeastern Arkansas contains small isolated woodlots within a landscape dominated by agriculture. The 1,620-ha plantation is planted in cotton (95%), with vegetated drainages intersecting fields. The worm-eating warbler was captured in a mist-net erected perpendicular to a drainage associated with a band of woody vegetation 32 m wide (i.e., cotton field edge to cotton field edge). The warbler was banded with a United States Fish and Wildlife Service aluminum leg band and aged as after second year (ASY) based on wear, shape, and color of tertials, primary coverts, and rectrices (Pyle, 1997). The bird had a fully developed brood patch with complete feather loss and moderate vascularization.

The occurrence of this species in an agricultural habitat of northeastern Arkansas is unusual for 2 reasons. First, the occurrence was outside the reported breeding range for this species (Hanners and Patton, 1998). Breeding populations are located to the west in the Ozark Highlands and Ouachita Mountains of Arkansas (James and Neal, 1986), to the north in the Glaciated Plains Region of Missouri (Robbins and Easterla, 1992), and to the south in northern Mississippi (Hanners and Patton, 1998); they are absent from the alluvial bottomlands of northeastern Arkansas. Second, if

breeding, the warbler was observed in atypical habitat. Worm-eating warblers are an area-sensitive species, requiring large tracts of forest for breeding and ultimately for sustaining populations (Robbins et al., 1989). In addition, they typically nest on steep forested slopes and ravines (James and Neal, 1986; Hanners and Patton, 1998). This individual was captured in a small (3.52 ha) forested habitat (32 m width, 1.1 km length) within a landscape that was flat. James and Neal (1986) reported 1 summer observation for this species in Poinsett County, Arkansas, but that occurred on Crowley's Ridge, where habitat features include forested hills.

Observations of worm-eating warblers in the alluvial areas of Arkansas do occur during the migratory period, but are relatively rare. For example, in eastern Arkansas from 1995 through 1997, J. Bednarz (pers. comm.) recorded 10 mist-net captures of this species during a fall migration sampling period of 27 August to 15 September in a large forest patch (>2,000 ha). However, the timing of our summer observation and the presence of a brood patch suggest a current or recent breeding attempt. The peak incubation activity for this species occurs during most of June, while peak activity of nests in the nestling stage can extend into early July (Hanners and Patton, 1998). This individual was lacking fat deposition in the furcular cavity, possibly indicating that it was not in migratory condition. However, this does not preclude the possibility of a dispersing individual after a recent breeding failure (L. A. Hanners, pers. comm.), because early July is nearing the end of the breeding period and fall migrants have been reported as early as the beginning of July (Dunn and Garrett, 1997).

There is little information reported on the habitat use and behavioral patterns of the spe-

TABLE 1—Habitat values from 5.0-m and 11.3-m radius point-centered vegetation plots at location of worm-eating warbler (*Helmitheros vermivorus*) capture ($n = 1$) in an agricultural drainage, 7 July 2000, Poinsett County, Arkansas. Habitat values from worm-eating warbler territories ($n = 11$) in Missouri (Wenny et al., 1993) also shown.

Habitat variable	July 2000	Wenny et al.
Total canopy coverage ^a	92.2%	95.8%
Mean litter depth (mm) ^a	16.9 mm	19.7 mm
Woody stems <2.5 cm diameter ^a	1.1/m ²	0.4/m ²
Shrub density ^b	33.6/100 m ²	—
Tree density ^b	3.0/100 m ²	—
Mean basal area of live trees ^b	7.9 m ² /100 m ²	10.7 m ² /100 m ²

^a Methods following Martin et al. (1997).

^b Point-centered quarter method following Mueller-Dombois and Ellenberg (1974).

cies during the post-breeding dispersal period. Recent work by Pagen et al. (2000) reported breeding worm-eating warblers preferred early successional forest habitats, and post-breeding worm-eating warblers were found in mature and early successional forests. Interestingly, 3 of 4 habitat values at the capture location were similar to those reported for worm-eating warbler breeding territories in Missouri (Table 1) and canopy cover was similar to that reported (79–90%) for habitats that had the greatest abundance of post-breeding individuals (Pagen et al., 2000).

It is unclear whether this report indicates an aberrant breeding attempt by a worm-eating warbler in an atypical habitat or occurrence of a dispersing individual. If the breeding interpretation is correct, this observation might suggest that suitable breeding habitat in the reported breeding range might be limiting. If the dispersal scenario is correct, then the habitats of the lowland delta might be more important for this species than previously believed. In either case, closer monitoring for worm-eating warblers in lowland habitats during the breeding and post-breeding periods might provide additional knowledge on the status and habitat needs of this species.

We thank J. C. Bednarz, L. A. Hanners, D. J. Horn, R. S. Maul, and 2 anonymous reviewers for comments that greatly improved the manuscript; M. Gibson and J. Moon for access to study sites; D. L. Feldman and H. Rauschenberger for field assistance; and E. Lombeida for the Spanish translation. We also thank J. C. Bednarz for offering information on the occurrence of migrating *H. vermivorus* in eastern Arkansas. Funding was provided by the Judd Hill Foundation and a Northeast Arkansas Environmental Association Scholarship.

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Submitted 13 February 2001. Accepted 10 July 2002.
 Associate Editor was William H. Baltosser.

INTRASPECIFIC HELPING BEHAVIOR EXHIBITED BY HATCH-YEAR HOUSE WREN

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ABSTRACT—On 1 August 2000, a house wren (*Troglodytes aedon*) was observed exhibiting intraspecific helping behavior. An unbanded individual was observed carrying a spider into and separately carrying a fecal sac from a nest box occupied by 2 color-banded adults. The feeding individual appeared to be a hatch-year bird using the aging criteria of tail-length, breast coloration, and rictus characteristics. There were no additional observations of feeding by any unbanded individuals during the 1-h sample or in 3 hourly samples over the next 3 days. It is not known if there was any genetic relationship between the house wrens occupying the nest box and the unbanded juvenile helper.

RESUMEN—El primero de agosto del año 2000 se observó un saltapared cucarachero (*Troglodytes aedon*) mostrando un comportamiento de ayuda intraespecífica. Se observó a un individuo sin banda llevando una araña al nido y en otra ocasión sacando un saco fecal del nido ocupado por dos adultos marcados con bandas de color. El individuo que llevó el alimento parecía ser un pájaro juvenil del año de su eclosión, según los criterios de longitud de cola, coloración del pecho, y las características del ensanche del rictus del pico. No hubo observaciones adicionales de eventos de alimentación por parte de individuos sin banda durante una muestra de 1 hora, ni en muestras de 3 horas en los siguientes 3 días. Se desconoce si hubo una relación genética entre los saltaparedes que ocupaban el nido y el juvenil sin banda que proporcionó la ayuda.

Male and female house wrens (*Troglodytes aedon*) are aggressively territorial during the breeding season, exhibiting agonistic behaviors toward conspecifics as well as individuals of other species (Johnson, 1998). As in most birds, house wrens rarely exhibit interspecific helping behavior, although several observations have been reported (Hills, 1924; Royall and Pillmore, 1968). Only 2 documented cases of intraspecific helping behavior have been reported in house wrens, in both instances a juvenile, or juveniles, from an earlier brood were observed feeding nestlings of a subsequent brood in the same breeding season (Skutch, 1953; Stafford, 1983). Cooperative breeding has been documented in various species of Troglodytidae (Brown, 1987; Skutch, 1987), although many remain poorly studied.

While conducting a study on breeding productivity of house wrens in Ellis County, Kansas, we documented a third instance of intraspecific helping behavior. On 1 August 2000 at 0746 h, an unbanded house wren carrying a spider was observed feeding at nest box number 88, occupied by 2 color-banded adults. It

remained in the cavity for approximately 30 s and flew away. At 0800 h, an unbanded house wren was observed carrying a fecal sac out of this nest box, suggesting a second feeding by the same bird. The unbanded individual appeared to be of hatch-year age, because its tail length was short relative to adult size and its overall appearance was characteristic of that age class. The breast was barred rather than the uniform color characteristic of adults, and the rictus was relatively large, fleshy, and yellow. There were no additional observations of feeding by the unbanded individual during the 1-h sample or in 3 hourly samples over the next 3 days. We observed this single instance of intraspecific helping behavior in 80 hours of nest observation at 20 of 115 active nests. It is unknown if there was any genetic relationship between the house wrens occupying the nest box and the unbanded juvenile helper.

We thank E. Hansen, N. Lambrecht, M. Schmidt, and D. Timson for assistance in the field, the Kansas Ornithological Society, Fort Hays State University Department of Biological Sciences, and Kansas State

University Agricultural Research Center—Hays for financial assistance, S. Fredin and C. Blanco-Montero for translating the summary, J. -C. Belles-Isles and 2 anonymous reviewers for helpful comments on the manuscript, and Alexander F. Skutch for his pioneering research on wren behavioral ecology.

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Submitted 28 October 2001. Accepted 31 May 2002.
Associate Editor was David B. Wester.

EASTERN RED BAT (*LASIURUS BOREALIS*) IMPALED BY A LOGGERHEAD SHRIKE (*LANIUS LUDOVICIANUS*)

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ABSTRACT—We describe the first record of a loggerhead shrike (*Lanius ludovicianus*) impaling a bat. The eastern red bat specimen (*Lasiurus borealis*) was found at Anahuac National Wildlife Refuge (Chambers County, Texas) and was collected the following day. We describe the habitat at the site where the bat was impaled and discuss the rarity of this event. This is the first time we have observed an impaled bat during approximately 200 h of casual observation. The prey typically impaled are orthopterans (approximately one-half of all impaled prey), frogs and toads, crayfish, birds, and, to a lesser extent, reptiles and rodents. Approximately 7% of the prey were completely consumed, 35% partially consumed, and more than half were not consumed.

RESUMEN—Describimos el primer registro del verdugo *Lanius ludovicianus* clavando a un murciélago. Un espécimen del murciélago rojo del este ó murciélago rojizo cola peluda (*Lasiurus borealis*) fue encontrado en Anahuac National Wildlife Refuge (condado de Chambers, Texas) y colectado el día siguiente. Describimos el hábitat del sitio donde fue clavado el murciélago y discutimos la rareza de este evento. Esta es la primera vez que hemos observado a un murciélago clavado en aproximadamente 200 h de observaciones casuales. Las presas típicamente clavadas son ortópteras (aproximadamente la mitad de todas las presas clavadas), ranas y sapos, cangrejos, aves, y menos frecuente, reptiles y roedores. Aproximadamente el 7% de los artículos alimenticios fueron completamente consumidos, 35% parcialmente consumidos, y más de la mitad no fueron consumidos.

An eastern red bat (*Lasiurus borealis*) was discovered impaled at Anahuac National Wildlife Refuge (ANWR), Chambers County, Texas, by DLS on 8 October 2000. A loggerhead shrike (*Lanius ludovicianus*) was flushed from the territory within minutes after discovering the im-

paled bat. The barb on which the bat was impaled also had several feathers attached, suggesting prior use by the shrike. These feathers were gray coverts with yellow tips, perhaps the feathers of a common yellowthroat (*Geothlypis trichas*), commonly associated with habitat typ-

TABLE 1—Shrike impalings recorded at Anahuac National Wildlife Refuge, Texas, on 10 January 1998. Species not determined.

Quantity	Organism	Degree consumed
3	Insects (Orthoptera)	not consumed
1	Crayfish	tail consumed, carapace remaining
7	Frogs	4 partially consumed
1	Small turtle (ca. 2 cm)	not consumed
1	Small bird	consumed within 2 h of impaling
1	Rodent	not consumed

ical of the impaling site (more detailed description below). At the time of discovery, the fence was checked for additional impalings for about 5 m in both directions with none discovered.

The 13-g bat was salvaged at dusk on 9 October 2000 by DMB and DLS (HMNS-VM 471). Measurements (mm) are as follows: total length = 96, right forearm = 41, right ear = 10. The lack of severe rigor mortis and non-decomposed condition of the bat specimen led us to believe it was impaled within 48 hr prior to discovery. The bat was found during an early cold front that began 6 October 2000. Temperatures had been cool (maximum = 10°C) with light rain. The low temperatures during this time probably kept the specimen from severely decomposing.

The bat was impaled on the top strand of barbed wire approximately 1.5 m from the ground (the bottom 2 wires of the fence were missing) and approximately 5 m from a private gravel road that is traveled less than once a week. The impaling site was approximately 100 m from East Bay in the Galveston Bay system. The immediate area of the impaling was comprised of approximately 75% *Spartina patens* on the road side of the fence, with 15% bare ground and 10% *Salicornia virginica* on the bay side of the fence. Macrohabitat around the site was a large area of brackish marsh-prairie dominated by *Spartina spartinae* with some woody invasive plants (e.g., *Celtis laevigata* and *Sapium sebiferum*).

Prior reports have documented *Lasiurus* becoming snagged on barbed wire by the

uropatagium (e.g., Iwen, 1958; Hibbard, 1963). However, the individual reported herein was impaled and twisted around the wing membrane and had other punctures indicative of multiple attempts at impalement. These data suggest that the bat was impaled by a loggerhead shrike and not an accident. Additional evidence in support of this included: 1) a shrike was flushed from the fence line about 20 m from the impaling when it was first discovered; 2) there have been other accounts describing similarly sized birds attacking red bats (e.g., Meritt, 1989); 3) the barb was wrapped with feathers, suggesting the shrike had used this barb before; 4) the fact that the bat was found in a *Spartina* dominated prairie with few potential roosts, which is atypical *Lasiurus* habitat, suggests the bat was carried there by the shrike rather than accidentally impaling itself during foraging; and 5) individuals of *L. borealis* often roost in the open, where they occasionally might be found by a shrike foraging during the day. If a crevice-roosting species of bat (e.g., *Tadarida*) were impaled, this observation would be more questionable. The shrike likely encountered the bat roosting in the open on a shrub, whereupon the bat was captured and carried to the impaling site.

A loggerhead shrike impaling a bat is apparently a rare (if ever recorded) event in nature, not cited in several references, including Bent (1965) and Oberholser (1974). Moreover, successful attacks of *L. borealis* by medium-sized birds are rare (e.g., Meritt, 1989). This is the only bat we found impaled during about 200 hours of casual field observation at ANWR and Attwater Prairie Chicken National Wildlife Refuge (APCNWR), primarily by DLS. Casual observations of prey impaled include: unidentified orthopterans (about 20), frogs and toads (8 to 10), crayfish (about 5), birds (4), and turtles, snakes, and rodents impaled less frequently (1 each). Bird species impaled (DLS, pers. observ.) include marsh wrens (*Cistothorus palustris*), which are commonly impaled (M. Whitebeck, pers. comm.), as well as individuals of Kentucky warbler (*Oporornis formosus*) and swamp sparrow (*Melospiza georgiana*).

Anecdotal notes on shrike utilization of impalings at APCNWR were collected on 10

January 1998 by DLS, walking about 0.8 km of fence line on the west side of the tour loop. A total of 14 impalings were found (Table 1), with only 1 item completely consumed (7%), 5 items partially consumed (35%), and 8 items left uneaten (57%). All organisms with the exception of a rodent were desiccated from being on the wire for an extended period of time. This suggests the primary function of impaled prey is a territorial marker (cf., Mizell, 1993) or mating lure rather than food storage.

DLS thanks M. Whitebeck (ANWR) for assistance in the field at the time the impaled bat was discovered. We thank J. Vargas-Contreras for providing the Spanish summary and are grateful to K. McDowell (ANWR) and T. Rosignol (APCNWR) for their support.

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Submitted 23 October 2000. Accepted 4 April 2002.
Associate Editor was William H. Baltosser.

RECENT OCCURRENCE OF BLACK BEARS IN THE SOUTHWESTERN GREAT PLAINS

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ABSTRACT—Historically, black bears (*Ursus americanus*) occurred throughout the southern Great Plains, including Kansas, Oklahoma, and Texas. However, due to habitat loss and human persecution, black bears were extirpated throughout this region by the early 1900s. We report on the recent range expansion of this species in the southwestern Great Plains, based on observations and records from state and federal agencies. Recent records of black bears, most of which exhibited the cinnamon color phase, suggest an established and reproducing population in Cimarron County, Oklahoma, that we estimate to consist of 20 individuals. This population apparently expanded from those in southeastern Colorado and northeastern New Mexico by the early 1980s and increased considerably during the 1990s. Today, black bears are reported annually from throughout Cimarron County. Dispersing young animals have increasingly wandered into Texas County, Oklahoma, and nearby areas of northwestern Texas and southwestern Kansas. Because black bears are protected in all 3 states and human densities are relatively low in this region, their range might continue to slowly expand.

RESUMEN—El oso negro (*Ursus americanus*) se distribuía históricamente a través del sur de las Grandes Planicies, incluyendo Kansas, Oklahoma, y Texas. Sin embargo, debido a la pérdida de hábitat y la persecución humana, el oso negro fue extirpado de esta región a principios de 1900. Reportamos la reciente expansión en la distribución de esta especie en el suroeste de las Grandes Planicies, basándonos en observaciones y registros de agencias estatales y federales. Recientes registros de osos negros, la mayoría de los cuales presentan la fase de color canela, sugieren que existe una población establecida y reproductiva en el condado de Cimarron, Oklahoma, la cual

estimamos que consiste de 20 individuos. Aparentemente esta población es una expansión proveniente del sureste de Colorado y del noreste de Nuevo México a principios de la década de 1980 y se ha incrementado considerablemente durante la siguiente década. Actualmente se registran osos negros cada año en el condado de Cimarron. Cada vez más se han dispersado animales jóvenes al condado de Texas, Oklahoma, y a áreas cercanas del noroeste de Texas y suroeste de Kansas. Debido a que el oso negro está protegido en los 3 estados y las densidades humanas son relativamente bajas en esta región, su distribución podría continuar expandiéndose lentamente.

Historically, black bears (*Ursus americanus*) occurred throughout the southern Great Plains, including Kansas, Oklahoma, and Texas. However, due to habitat loss and human persecution, black bears were extirpated throughout this region by the early 1900s. Black bears disappeared from Kansas by the 1880s (Bee et al., 1981), Oklahoma in the 1930s (Tyler and Anderson, 1990), and Texas, except for mountains in the Trans-Pecos region, by the early 1900s (Taylor and Davis, 1947). Although black bears recently have re-occupied small parts of their former range in the Ozark Mountains of northeastern Oklahoma (Pelton and Van Manen, 1994) and the Chisos and Davis mountains in southwestern Texas (Davis and Schmidly, 1994), their presence in the relatively flat terrain of the southwestern Great Plains has not been reported.

We report on the recent range expansion of this species in the southwestern Great Plains, based on the occurrence of a probable small breeding population in the western Oklahoma Panhandle and sightings in southwestern Kansas and northwestern Texas. Because black bears are now protected in all 3 states, their breeding range might continue to expand slowly in the future, despite the relative flatness of the terrain.

In Oklahoma, Glass (1951) suggested that black bears might occasionally wander into Cimarron County, the westernmost county in the Panhandle, because 1 was observed nearby in southeastern Colorado. Lee (1967) reported a specimen from a location in northeastern New Mexico, approximately 120 km west of Cimarron County. In the early 1980s, local residents reported sows with cubs in several counties adjacent to Cimarron County in southeastern Colorado and northeastern New Mexico (Caire et al., 1989). The first verified record of a black bear in the Oklahoma Panhandle occurred in May 1981, when a female was captured a few kilometers west of Boise City in

central Cimarron County and released near Black Mesa State Park after being tagged (Caire et al., 1989). The capture and release of this sow suggested a potential breeding population.

During the 1990s, one of us (LAG) investigated numerous black bear sightings and personally observed 7 individuals in Cimarron and Texas counties, Oklahoma. These observations were described as follows: 1) during spring 1990, a subadult female (ca. 50 kg) was tranquilized, photographed (photographs mentioned in this paper are in possession of the Oklahoma Department of Wildlife Conservation), and released by LAG approximately 8 km northeast of Kenton, Cimarron County; 2) during spring 1991, a cinnamon-colored bear (ca. 50 to 55 kg) was observed by LAG in a tree near a residence, approximately 9 km east of Kenton; 3) during fall 1993, an apparent adult (ca. 140 kg) was observed by LAG in a field approximately 9 km northeast of Kenton; 4) during summer 1995, a subadult male (ca. 60 kg) was electrocuted after climbing an electrical pole in Black Mesa State Park, Cimarron County (photographs taken, but specimen not recovered); 5) in August 1996, a cinnamon-colored male (ca. 130 kg) was photographed and video-taped in a cornfield approximately 7 km northwest of Griggs, Cimarron County; 6) in July 1999, a black bear was observed by LAG in a culvert approximately 16 km northwest of Keyes, Cimarron County; and 7) during spring 2000, a cinnamon-colored yearling (ca. 80 kg) with black ears and feet was photographed and video-taped in farmland near an industrial hog facility approximately 18 km north of Eva, Texas County. In addition to these observations, LAG found bear tracks approximately 35 km north of Boise City, Cimarron County, after investigating reports of an animal there in the fall of 2000. Tracks were observed by LAG on more than 50 other occasions over the past 12 years along the Cimarron River in the northern part of Cimarron County. Numerous landowners interviewed by LAG during the past 10

years also had seen bears. Black bears are now regularly seen by ranchers during annual cattle round-ups along the Cimarron River and by farmers throughout the county during corn harvests. In August 1995, several landowners watched a female bear with 2 cubs near Griggs, as did local residents near a cattle feedyard near Boise City, about 30 km to the northwest.

In addition to our observations, bear tracks were photographed (deposited in Cameron University Museum of Zoology, Lawton, Oklahoma) on 21 June 1990, approximately 13 km east of Kenton, Cimarron County. Local residents observed a young cinnamon-colored bear on 26 May 1991, approximately 13 km east of Kenton, and observed another bear on 27 May 1997, approximately 6 km south of Kenton (J. D. Tyler, Cameron University, Lawton, Oklahoma, pers. comm.).

As indicated above, in Cimarron County, Oklahoma, black bear sightings are concentrated in the northwestern part of the county, where the terrain is relatively rugged. This region is adjacent to established black bear populations in southeastern Colorado (Fitzgerald et al., 1994) and northeastern New Mexico (Findley et al., 1975). However, black bears are increasingly observed in other relatively flat parts of Cimarron County that are dominated by agricultural fields, although they tend to be concentrated along the Cimarron River and Beaver Creek, which have riparian vegetation that provides more natural and permanent cover.

In northwestern Texas, 4 documented records of black bears are known during the past 20 years. In 1984, a subadult cinnamon-colored male (ca. 115 kg) was observed by state agency personnel as it foraged in native grasslands near the Thompson Grove campground, approximately 21 km east of Texline, Dallam County (R. Burnes, Texas Parks and Wildlife Department, pers. comm.). In 1986, a subadult cinnamon-colored male (ca. 80 kg) was observed by state agency personnel near a cattle feedyard not far from Pericho, Dallam County (R. Burnes, Texas Parks and Wildlife Department, pers. comm.). In fall 1995, a cinnamon-colored cub (ca. 18 kg) was captured by local residents on a private ranch in west-central Hartley County. The cub was released by state agency personnel approximately 8 km north of Texline, Dallam County (A. Sandoval, New

Mexico Game and Fish Department, pers. comm.). In 1997, a subadult was seen near a county road approximately 6.5 km east of Texline (R. Childress, United States Forest Service, pers. comm.). In addition to these documented records, we interviewed several workers who saw a black bear in July 1999 near an industrial hog facility on Beaver Creek in northern Sherman County. That same week, several other local residents saw a bear in that same area crossing United States Highway 287 northwest of Stratford, Sherman County (D. Birkenfeld, Stratford, Texas, pers. comm.). These locations were approximately 20 km southeast of recent sightings in southeastern Cimarron County, Oklahoma.

In southwestern Kansas, 3 black bears have been documented within the past 20 years. During summer 1982, 1 was observed by state agency personnel in Cimarron National Grasslands near Elkhart, Morton County (M. Mitchner, Kansas Department of Wildlife and Parks, pers. comm.). This bear was captured, given to personnel of the Colorado Division of Wildlife, and subsequently released nearby in Colorado. In June 1997, a cinnamon-colored individual was observed along the Cimarron River in the Cimarron National Grasslands, Morton County (M. Mitchner, Kansas Department of Wildlife and Parks, pers. comm.). In May 2000, a cinnamon-colored bear (ca. 80 kg) was observed in the Cimarron National Grasslands (M. Mitchner, Kansas Department of Wildlife and Parks, pers. comm.). For the next several months, this young animal wandered north to near Ritchfield, Morton County, then followed the North Fork of the Cimarron River to near Ulysses, Grant County (Ulysses News, 1 June 2000). It subsequently wandered through Kearny, Wichita, and Greeley counties before crossing the border into Colorado (M. Mitchner, Kansas Department of Wildlife and Parks, pers. comm.). This bear was probably the same individual seen during spring 2000 by LAG in northwestern Texas County, Oklahoma.

Of all black bears reported by state and federal agency personnel in the southwestern Great Plains, most (7 of 13) exhibited the cinnamon color phase. Recent records suggest an established and reproducing population in Cimarron County, Oklahoma, that we estimate to consist of 20 individuals. This population no doubt expanded from those in southeastern

Colorado and northeastern New Mexico by the early 1980s and increased considerably during the 1990s. Today, black bears are reported annually from throughout Cimarron County. Dispersing young animals have increasingly wandered into Texas County, Oklahoma, and nearby areas of northwestern Texas and southwestern Kansas. Because black bears are protected in all 3 states and human densities are relatively low in this region, their range might continue to slowly expand.

We thank D. Birkenfeld, R. Burnes, A. Carr, M. Mitchner, J. D. Ray, and A. Sandoval for providing records of black bears. We also thank C. Jones for reviewing the manuscript, J. D. Tyler for providing his field notes on black bears and reviewing the manuscript, and R. Carrera for the Spanish translation of the summary. This is a Texas Tech University, College of Agricultural Sciences and Natural Resources Publication T-9-900.

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*Submitted 17 August 2001. Accepted 14 March 2002.
Associate Editor was Cheri A. Jones.*

EFFECTS OF A FLOOD ON RELATIVE ABUNDANCE AND DIVERSITY OF SMALL MAMMALS IN A REGENERATING BOTTOMLAND HARDWOOD FOREST

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ABSTRACT—Because effects of flooding on small mammal populations are poorly understood, we examined small mammal response, measured by relative abundance and community diversity, to a flood in a regenerating bottomland hardwood forest. We trapped small mammals immediately prior to and 5 months following retreat of floodwaters in Yazoo County, Mississippi during 1994 and 1995. Cotton rats (*Sigmodon hispidus*) and marsh rice rats (*Oryzomys palustris*) were the dominant species captured throughout the study, but abundance declined markedly after flooding and did not reach pre-flood abundance by the end of the study. Extended flood duration and depth were believed to have caused the disappearance of cotton mice (*Peromyscus gossypinus*) and a decrease in community diversity.

RESUMEN—Porque no se sabe mucho del efecto de las inundaciones sobre las poblaciones de

mamíferos pequeños, examinamos la respuesta de los mamíferos pequeños, según la abundancia relativa y la diversidad de la comunidad, a una inundación en un bosque de madera dura en proceso de regeneración de tierras bajas. Capturamos los mamíferos pequeños inmediatamente antes de la inundación y 5 meses después del retiro de las aguas en el condado de Yazoo, Mississippi, en los años 1994 y 1995. La rata algodonera (*Sigmodon hispidus*) y la rata arrocera (*Oryzomys palustris*) fueron las especies capturadas con mayor frecuencia. Su abundancia bajó sumamente después de la inundación, y hasta el fin del estudio no se había alcanzado la abundancia anterior. Se cree que las prolongadas inundaciones y la profundidad de las aguas causaron la desaparición del ratón algodonero (*Peromyscus gossypinus*) y la disminución de la diversidad de la comunidad.

Bottomland hardwood forests are recognized as being extremely productive wildlife habitat. From 1960 through 1970, substantial portions of bottomland hardwood forests in the Lower Mississippi Alluvial Valley (MAV) were destroyed and converted to agriculture (Reinecke et al., 1989; Stanturf et al., 2001). Recently, however, programs such as the Wetland Reserve Program (WRP) and Conservation Reserve Program (CRP) have resulted in large acreages being placed into hardwood regeneration areas. Landscape-level changes in the distribution and abundance of early successional habitats witnessing regeneration efforts subsequently could affect faunal communities inhabiting landscapes within the MAV. Coincidentally, government set-aside programs such as WRP and CRP frequently result in landowners enrolling marginal agricultural lands, which are often sites that receive unpredictable flood pulses during the growing season (Stanturf et al. 2001). Although early studies examined relative effects of flooding on species of small mammals (Blair, 1939), information describing effects of flooding on small mammal populations across sites selected for bottomland hardwood restoration is lacking. Therefore, we describe changes in relative abundance and diversity of small mammals following extended flooding within a regenerating bottomland hardwood forest. We used captures per effort and community diversity indices prior to and following a spring flood pulse to infer about effects of flooding on small mammal communities.

Our study area included the 3,640-ha Lake George Wildlife Management Area (LGWMA), located in Yazoo County, Mississippi. LGWMA contained 3,458 ha of early successional habitats and 182 ha of mature bottomland hardwood forest. Early successional areas were replanted in hardwood seedlings, whereas some areas remained in agriculture. Replanted hard-

woods included Nuttall oak (*Quercus nuttallii*), willow oak (*Q. phellos*), water oak (*Q. nigra*), and sweet pecan (*Carya illinoensis*). Common invader plant species included ragweed (*Ambrosia*), Johnson grass (*Sorghum halapense*), sesbania (*Sesbania exaltata*), and goldenrod (*Solidago*). Dominant agricultural crops were cotton, soybeans, and milo. Prevalent mature tree species included overcup oak (*Q. lyrata*), sugarberry (*Celtis laevigata*), and black willow (*Salix nigra*). The area was bisected by a man-made water drainage canal with 2 flood-control levees on either side. Areas between these levees were susceptible to annual flooding, with 3 to 4 month periods of inundation common. Topography was flat with 0 to 4% slopes, and soils were poorly drained, with dark-colored, clayey profiles (United States Department of Agriculture, 1959).

We trapped small mammals during February 1994 for a 3-day trapping period immediately prior to an extensive flood pulse. All areas between the levees were inundated >3 m in depth, and floodwaters did not recede until late May and early June. Outside the levees, flooding occurred for nearly 2 weeks; however, floodwaters were <0.5 m, and floodwaters had completely receded by mid-March. Trapping was reinitiated in October 1994 and continued through February 1995. Although floodwaters had receded by July, we did not trap during July through September because of a lack of vegetation on flooded sites, and because of problems with red-imported fire ants (*Solenopsis invicta*) removing bait and mutilating captured rodents. Once reinitiated, trapping occurred during one 3-day interval each month, excluding January. Transect length was either 400 m ($n = 8$) or 200 m ($n = 2$), depending on size of available habitat. Nine of 10 transects were established in regenerating forest; the remaining transect was in the sole patch of mature forest on LGWMA. On 400-m transects, 20

TABLE 1—Catch per unit effort (number captured/100 trap-nights) for 3 mammal species captured on 10 transects using live-traps in hardwood regeneration areas on Lake George Wildlife Management Area, Mississippi, 1994 through 1995.

Species	Period	Total captures	Catch per unit effort
<i>Peromyscus gossypinus</i>	Pre-flood	4	0.7
	Post-flood	0	0.0
<i>Sigmodon hispidus</i>	Pre-flood	111	20.59
	Post-flood	96	5.1
<i>Oryzomys palustris</i>	Pre-flood	36	6.7
	Post-flood	14	0.7
Total trap-nights	Pre-flood	539	
	Post-flood	1,899	

stations were located 20 m apart, whereas 200-m transects contained 10 stations placed 20 m apart. One Tomahawk live-trap (14.64 × 15.24 × 17.15 cm) was placed at each station and baited with peanut butter. Captured animals were identified to species, weighed, uniquely toe-clipped for later identification, and released at the capture site.

We determined relative abundance of small mammals using catch/unit effort (number of captures/100 trap-nights). A trap-night was defined as 1 trap set for a 24-h period. Catch/unit effort estimates were then compared for each species. Recaptures and associated trap-nights were removed from calculations of catch/unit effort. We used a binomial test for 2 proportions to compare differences in catch/unit effort and species composition between pre-flooding and post-flooding periods (Zar, 1984). Inundation length was noted and related to later sampling to evaluate differences in small mammal populations prior to and after flooding. We examined species diversity between pre-flooding and post-flooding periods by calculating Shannon-Weaver diversity indices and associated evenness values.

Catch/unit effort decreased for all species following flooding (Table 1). Total captures were composed mostly of marsh rice rats (*Oryzomys palustris*, 19%) and cotton rats (*Sigmodon hispidus*, 79%), whereas cotton mice (*Peromyscus gossypinus*) only comprised 2% of total captures during our study. Percentage composition for cotton rats increased (74 to 91%, $P =$

0.042), but decreased (11 to 0%, $P = 0.01$) for cotton mice. Likewise, diversity (0.6643 to 0.2332) and evenness (0.6048 to 0.1682) declined >0.40 after flooding.

Six transects were completely inundated from mid-February 1994 until late May 1994. After flooding, catch/unit effort decreased for cotton rats from 0.167 to 0.003 ($P = 0.01$). Similarly, catch/unit effort decreased for marsh rice rats and cotton mice from 0.047 to 0.004 ($P = 0.04$) and 0.03 to 0.00 ($P = 0.02$), respectively.

Flooding reduced relative abundance and diversity of small mammal communities on LGWMA. The decrease and near disappearance of cotton mice was likely attributable to effects of flooding. Blair (1939) reported lower numbers of white-footed mice (*P. leucopus*) after long-term flooding. A lack of recolonization by cotton mice when water receded might have been due to poor habitat quality. On LGWMA, few larger trees (>4 years) were present to provide refuge during high water levels (>4 m). Likewise, high water levels prevented mice from foraging on the ground from February through May. High flood depth and length possibly caused widespread starvation or emigration. Blair (1939) reported similar findings after prolonged flooding in Oklahoma, and Andersen et al. (2000) suggested that arboreal species, such as deer mice (*P. maniculatus*), face an increased risk as flood duration increases. Furthermore, diversity indices on LGWMA decreased after flooding because of the disappearance of cotton mice and reductions in relative abundance of other species. Species diversity increases when species captured are represented by the same number of individuals in each sample (Ludwig and Reynolds, 1988). The low diversity and evenness indices derived on LGWMA were a result of 2 species dominating captures prior to flooding and comprising all captures following flooding.

Sheppe and Osbourne (1971) reported immediate recolonization of floodplains in Zambia by small mammals as water receded. However, we did not observe such a recolonization on LGWMA. We suggest that the lack of recolonization occurred primarily because of low food and cover availability (Blair, 1939), confounded by a relatively high winter abundance of northern harriers (*Circus cyaneus*) on LGWMA (Hamel, 1992), and a lack of suitable habitat in areas surrounding LGWMA. Wiegert

(1972) found aerial predation on cotton rats as a significant factor in autumn-winter declines. Following the retreat of floodwaters, previously flooded areas on LGWMA contained little understory vegetation, resulting in decreased understory cover and available food (Blair, 1939). Animals that attempted to recolonize were likely susceptible to high stress resulting from aerial predation, weather, and food shortages. Further, where cover was present, "reservoirs" of rodents might have occurred. Schnell (1968) suggested that predation might be accelerated on these scattered populations. Lastly, areas surrounding LGWMA were primarily used for intensive agriculture, which provided little concealment cover; therefore, habitat suitability in these areas was likely low. A lack of suitable habitats in surrounding areas might have reduced the opportunity for potential source populations to recolonize depleted populations on LGWMA.

Hydrological changes in the MAV have created conditions unlike the natural floodplains studied by Sheppe and Osbourne (1971). Channelization and levee construction along the Mississippi River have reduced the natural floodplain by 90% (Fremling et al., 1989), and flood intensities have increased since 1927 (Belt, 1975). Perhaps species are unable to adapt to flood-related stresses because of the increasing severity of inundation. As channelization and levee construction continue in the MAV, species will be forced to adapt or become locally extinct. As our data indicate, even when bottomland hardwood forests are regenerated, if hydrology is drastically altered, effects on small mammals might be severe and deleterious.

This research was supported by the Mississippi Department of Wildlife, Fisheries and Parks, U.S. Army Corps of Engineers, the Forest and Wildlife Research Center at Mississippi State University, and the School of Renewable Natural Resources at Louisiana State University. We thank J. Ross and S. Stephens for field assistance.

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*Submitted 28 August 2001. Accepted 24 April 2002.
Associate Editor was Cheri A. Jones.*

FIRST RECORD OF THE YUCATAN DEER MOUSE, *PEROMYSCUS YUCATANICUS* (RODENTIA: MURIDAE) FROM GUATEMALA

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ABSTRACT—We report the first record of Yucatan deer mouse, *Peromyscus yucatanicus*, in Guatemala. The Yucatan deer mouse was restricted to Yucatán Peninsula, where it is considered a common species in deciduous to semi-evergreen tropical forests, and secondary growth forests. Two Yucatan deer mice were collected at Laguna Flor de Luna, Laguna del Tigre National Park, Guatemala, on April 1999. This new record constitutes a range extension of 116 km south of the nearest previously known locality at Escárcega, Campeche, Mexico.

RESUMEN—Presentamos el primer registro de *Peromyscus yucatanicus* para Guatemala. La distribución de esta especie era conocida sólo en la Península de Yucatán, donde es considerada común en bosque tropical caducifolio y semicaducifolio, y también en bosques de crecimiento secundario. En abril del 1999, se capturaron dos ejemplares de *Peromyscus yucatanicus* en Laguna Flor de Luna, Parque Nacional Laguna del Tigre, Guatemala. Este nuevo registro amplía el área de distribución unos 116 km al sur de la localidad más cercana registrada, o sea, Escárcega, Campeche, México.

The Yucatán deer mouse (*Peromyscus yucatanicus*) was described by Allen and Chapman (1897) on the basis of 6 specimens from Chichén-Itzá, Yucatán, Mexico. *Peromyscus yucatanicus* is a medium-sized mouse and is larger than the only sympatric species of *Peromyscus*, the white-footed mouse (*P. leucopus*). The 2 species can be distinguished easily because *P. yucatanicus* has a moderately developed supra-orbital ridge, no pectoral mammae, a glans penis with divided dorsal lappets, and a sparsely haired tail that is frequently blotched ventrally (Lawlor, 1965; Young and Jones, 1983). Lawlor (1965) considered this species to be monotypic, but Hall (1981) recognized 2 subspecies, the northern *P. y. yucatanicus* Allen and Chapman, 1897, with ochraceous buff upper parts, and the southern *P. y. badius* Osgood, 1904, with brownish upper parts (both subspecies have a mixture of dusky hairs on the upper parts). The Yucatán deer mouse is considered a Mexican endemic and has been recorded only from northern Yucatán and the north-central parts of the states of Campeche and Quintana Roo, Mexico (Hall, 1981; Young and

Jones, 1983). The southern limit of its distributional range is unknown (Huckaby, 1980). *Peromyscus yucatanicus* is considered a common species in semi-deciduous to semi-evergreen forests, banana plantations, and secondary growth forests of the Yucatán Peninsula (Birney et al., 1974; Young and Jones, 1983).

On 21 April 1999, 2 Yucatan deer mice (*P. y. badius*) were collected at Laguna Flor de Luna, Laguna del Tigre National Park, Guatemala (17°35.99'N, 90°53.84'W). The general habitat was flooded forest dominated by tinto trees (*Haematoxylon campechianum*). Transects of snap traps (Victor Rat Trap and Museum Special Mammal Trap) were established near bodies of water. Traps were baited with a mixture of peanut butter, raisins, oats, and bacon. Specimens were deposited in the Zoological Collections, Museum of Natural History, University of San Carlos, Guatemala (USAC 1410, 1411). They were identified following Huckaby (1980), Hall (1981), and Reid (1997), and compared with specimens of *P. yucatanicus* and *P. leucopus* deposited in the Colección de Mastozología, Instituto de Biología, Universidad Nacional Autó-

noma de México, México. The male (USAC 1410) was an adult and the female (USAC 1411) was a subadult based on their pelage. The female was completing the post-subadult molt; the venter was pale grayish, and the dorsum was dusky. In contrast, the venter is yellowish-white, and the dorsum is brighter ochraceous in the adult (Lawlor, 1965; Young and Jones, 1983).

External and cranial measurements (mm) of both specimens, the adult male and subadult female, respectively, are: total length 201, 203; length of tail 100, 100; length of hind foot 22, 22; length of ear 19, 19; greatest length of skull 28.14, 28.2; condylobasal length 27.72, 26.56; palatal length 4.36, 3.95; nasal length 10.64, 9.98; length of rostrum 8.72, 8.95; zygomatic breadth —, 13.28; breadth of braincase 12.19, 12.37; interorbital breadth 4.56, 4.56; mandible length 13.87, 12.91; length of mandibular tooth row 3.95, 3.90; length of maxillary tooth row 3.69, 3.83; length of incisive foramen 5.72, 5.63. Both specimens weighed 29 g. The male had testes measuring 14 mm by 9 mm.

Other species of rodents that have been recorded in association with *P. yucatanicus* include *Peromyscus leucopus*, *Heteromys gaumeri*, *Ototylomys phyllotis*, *Oryzomys alfaroi*, *O. couesi*, *O. melanotis*, *Reithrodontomys gracilis*, and *Sigmodon hispidus* (Lawlor, 1965; Dowler and Engstrom, 1988). We captured only 2 species at this locality: *P. yucatanicus* and *Ototylomys phyllotis*.

Previously, the southernmost record for *P. yucatanicus* was 7.5 km W of Escárcega, Campeche (Dowler and Engstrom, 1988). There are also 4 specimens of *P. y. badius* from the Calakmul, Campeche archaeological site in the collection of the Instituto de Biología, Universidad Nacional Autónoma de México, México (IBUNAM 37362, 37364, 37365, 37366); these records are 115 km ESE of the record in Dowler and Engstrom (1988). Our new record is a range extension of 116 km S of Escárcega and 134 km SE of Calakmul, and it represents the first record of the species in Guatemala. We speculate that *P. yucatanicus* occupies the entire Yucatán Peninsula of Mexico, northern Guatemala, and northern Belize (Fig. 1), based on the distribution and similarity of vegetation types.

We thank the team of Aqua RAP for their excellent camaraderie in the field. We are grateful to F. Cervantes and Y. Hortelano (Instituto de Biología, Universidad Nacional Autónoma de México) who

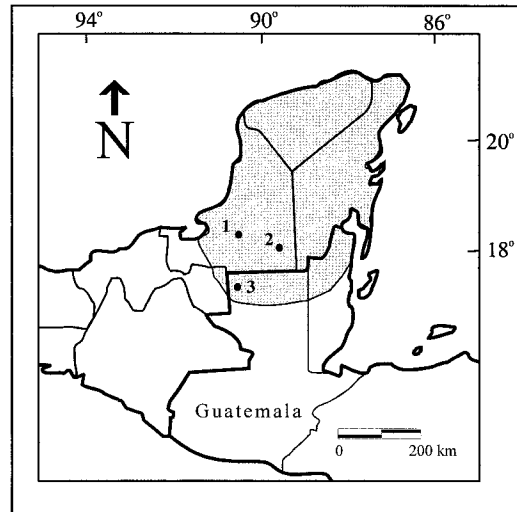


FIG. 1.—Hypothesized distribution of *Peromyscus yucatanicus* on the Yucatán Peninsula and in adjacent Guatemala and Belize. Localities discussed in text. MEXICO, CAMPECHE: 1) 7.5 km W Escárcega; 2) Calakmul archeological site; GUATEMALA, EL PETÉN: 3) Laguna Flor de Luna.

permitted access to the Colección de Mastozoología and who identified the specimens, and to 2 anonymous reviewers for critiquing an earlier draft of the manuscript. We also thank the Consejo Nacional de Areas Protegidas for permission to work in the Laguna del Tigre National Park and the following institutions: Centro de Estudios Conservacionistas, Conservation International—ProPetén, and Conservation International's Rapid Assessment Program. Special thanks to L. Alonso for support in the development of this project. This research was supported by funding provided by the United States Agency for International Development and The Rufford Foundation.

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- Submitted 21 December 2000. Accepted 8 March 2002.
Associate Editor was Loren K. Ammerman.

NOTEWORTHY RECORDS OF TWO RARE MAMMALS IN SIERRA NORTE DE OAXACA, MEXICO

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ABSTRACT—We made an intensive survey of small mammals in the Sierra Norte de Oaxaca region, in Oaxaca, México. We recorded the presence of 2 interesting species: water mouse (*Rheomys mexicanus*) and river otter (*Lontra longicaudis annectens*). The locality for the river otter represented the highest altitude recorded in México for this species. The locality for the water mouse is the fourth known in all of its range and the second record in Sierra Norte.

RESUMEN—A partir de un muestreo de campo intensivo de mamíferos pequeños en la región conocida como Sierra Norte, en el estado de Oaxaca, México, se registró la presencia de 2 especies interesantes: el ratón acuático (*Rheomys mexicanus*) y la nutria de río (*Lontra longicaudis annectens*). La localidad en que ocurrió la nutria de río representa la mayor altitud para esta especie en México, mientras que la localidad del ratón acuático representa la cuarta localidad conocida en toda su distribución y la segunda en la Sierra Norte.

We report 2 species of interest from our field survey of mammals in the Sierra Norte de Oaxaca, Oaxaca, México: water mouse (*Rheomys mexicanus*) and river otter (*Lontra longicaudis annectens*).

The water mouse is endemic to the state of Oaxaca, México (Goodwin, 1969; Hall, 1981; Musser and Carleton, 1993), and it previously has been recorded from only 3 localities. Goodwin (1959) described it from 4 specimens collected at San José Lachiguirí (16°22'40"N, 96°20'7"W), Distrito de Miahuatlán, 4,000 feet elevation, in Sierra Madre del Sur. Goodwin (1969) later reported 7 additional specimens from this locality. Other records are from Unión Hidalgo (16°28'30"N, 94°49'47"W), Dis-

trito de Juchitán, in the Istmo de Tehuantepec (Goodwin, 1969) and 3.6 miles W Totontepec (17°13'N, 96°3'W) (Voss, 1988). This species is considered "rare" by the Mexican government (SEDESOL, 1994).

On 5 April 2001, we found an adult male water mouse clogging a hosepipe used to collect water from the Río Yavesía (17°13'16"N; 96°25'07"W), 4 km SE of Santa María Yavesía, Municipio de Santa María Yavesía, Distrito de Ixtlán, 2,000 m elevation, and 67 km NE of Oaxaca. The main vegetation in this area is pine-oak forest, with common subcanopy trees of the genus *Alnus*.

The standard external and select cranial and mandibular measurements (in mm) of the spec-

imen are total length, 289; length of vertebral tail, 143; hind foot length, 32; ear length, 7; mass, 88 g; condylobasal length, 29.8; zygomatic breadth, 15.7; interorbital constriction, 5.3; alveolar length of upper molar tooththrow, 4.9; alveolar length of lower molar tooththrow, 4.9; and mandibular length, 13.7. The specimen was preserved as a skin with skeleton in the Colección de Mamíferos, Centro Interdisciplinario de Investigación para el Desarrollo Integral Regional, Unidad Oaxaca, Instituto Politécnico Nacional (OAXMA 2077). Along with the specimen collected west of Totontepec, this is the second record of *R. mexicanus* for Sierra Norte de Oaxaca and represents only the fourth locality known in this entire range.

The river otter inhabits aquatic habitats in tropical and temperate forests. The species has been recorded from sea level to 1,700 m in elevation. The presence or absence of this species indicates habitat quality (Gallo, 1986, 1997).

In Oaxaca, the river otter has been recorded in several localities in the regions of the Istmo de Tehuantepec, Costa, and Sierra Mixteca (Goodwin, 1969; Hall, 1981; Gallo, 1997). In March 1998 we saw 1 otter, and in February 2001, we collected scats (M. Aranda, pers. comm.) in the Soyaltepec River at Laa-dú, 10.6 km NW of Los Reyes Yagalaxi, Municipio de Ixtlán de Juárez, Distrito de Ixtlán (17°37'30"N; 96°15'55"W; 360 m elevation). The dominant vegetation in this area is subtropical broadleaf forest. The nearest record of this species is 130 km to the north in the Río Blanco, 20 km W of Piedras Negras, Municipio de Piedras Negras, Estado de Veracruz (18°46'02"N; 96°21'15"W; 165 m elevation) (Hall and Dalquest, 1963).

In April 2001, we collected several more scats along the Río Yavesia that were identified as those of *L. l. annectens* (M. Aranda, pers. comm.). Between March and May 2001, local villagers from Yavesia corroborated the presence of otters in the river, although such encounters were infrequent. These records represent the highest reported elevation (2,000 m) for this otter in México (Gallo, 1997). The next highest elevation reported was 1,550 m at a locality 157 km NW in the Río del Oro (also known as Río Tamazulapan), close to the hydroelectric station plant of Tamazulapan, Municipio de Villa de Tamazulapan del Progreso, at 18°40'N; 96°36'W; (Gallo, 1997).

World Wildlife Fund (WWF, Project S-055), Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO, Project R-104), and Dirección de Estudios de Posgrado e Investigación, Instituto Politécnico Nacional (DGEPI, Project 970549) provided financial support for the field survey. A. Trujano, A. Marín, J. García, J. Calderón, L. Velázquez, and M. Peralta gave invaluable assistance in the field work. M. Aquino, E. Ramírez, and M. Cruz, Ixtlán de Juárez and Santa María Yavesia, respectively, authorized access to their lands. The young promoters of ecotourism of both communities collaborated in diverse phases of the field work. A. Pérez Hernández provided invaluable information about the occurrence of the water mouse in Yavesia, as well as field support. E. Martínez-Meyer, V. Sánchez-Cordero, and 2 anonymous reviewers made valuable suggestions to earlier drafts of the manuscript. M. Aranda identified otter scats.

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Submitted 15 March 2002. Accepted 26 June 2002.
Associate Editor was Cheri A. Jones.