EFFECTS OF CATTLE GRAZING ON ANURAN SPECIES DIVERSITY AND COMPOSITION

By

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Bachelor of Science

University of California

Davis, California

1996

Submitted to the Faculty of the
Graduate College of the
Oklahoma State University
In partial fulfillment of
The requirements for
The Degree of
MASTER OF SCIENCE
December, 2002

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Thesis approved:

Thesis Advisor

Dean of the Graduate College

ACKNOWLEDGEMENTS

I wish to express my gratitude to my major advisor, Dr. Charles C. Peterson, for his encouragement, support, and guidance. I would also like to extend my appreciation to my other committee members, Dr. David Duvall and especially Dr. Eric Hellgren for their support, assistance, and encouragement. I would like to thank the Department of Zoology for providing me with this educational opportunity.

I would also like to express my appreciation for those who provided assistance and suggestions for this study, including Dr. Mark Payton, Dr. Mahesh Rao, and Dr. David Engle.

Thanks are also due to the U. S. Fish and Wildlife Service and U.S. Army, Fort Sill Military Reservation, for granting me special use permits to explore frogs and toads on these federal lands. Special thanks to the Styron family of Kiowa county for their generosity and kindness in granting me permission to conduct this study on their land. I owe a special debt to the staff of Wichita Mountains Wildlife Refuge for their assistance with the portion of the study that was conducted on U.S.F.W.S. land, and for providing many materials and supplies.

Finally, I would like to thank my family for their unwavering support of my mind-boggling choice of careers, and especially Tammy James, whose emotional support sustained me during these last three years.

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INTRODUCTION

Livestock grazing is the predominant land use practice in the western United States, occurring across approximately 70 percent of land resources, both public and private (Fleischner 1994). The introduction of a large, non-native herbivore has obvious ecological implications. Cattle impose different selection pressures on plant species than native grazers, changing plant species composition and often facilitating the establishment of exotic species (Fleischner 1994; Dobkin *et al.* 1998; Milchunas *et al.* 1998; Yates *et al.* 2000). Despite the ubiquity of livestock grazing across this landscape, relatively few studies have explored the effects of grazing-induced habitat alteration on native wildlife (Fleischner 1994, Scott 1996).

Researchers have recently begun to examine possible effects of livestock grazing on some species. Milchunas *et al.* (1998) found a high degree of variability in the responses of invertebrates, mammals and birds to grazing pressure. Interestingly, the degree of response did not necessarily relate to the degree of change in plant composition, suggesting that some species are very sensitive to a small amount of grazing pressure. Although general wildlife response to grazing is not predictable, it is clear that the response of an animal species to grazing will depend on its natural history characteristics and habitat requirements.

Species may be dependent on the level and type of vegetation with which they have evolved. Reptiles and amphibians of North American grasslands show characteristics and trends that imply a co-evolved herpetofauna in this region (Scott 1996). Changes in vegetation due to grazing may be beneficial to species that thrive in open patches of grassland, but negatively affect those that are dependent on dense vegetation for cover. For example, the riparian garter snake (*Thamnophis elegans* *vagrans*) was four times more abundant in areas excluded from grazing, presumably because such areas provided thermal cover and protection from predators and because grazing reduced the snake's invertebrate prey population (Szaro *et al.* 1985). Over 10 times as many *Sceloporous scalaris* were found in ungrazed land as on grazed land, presumably because this lizard requires a bunchgrass habitat as protection from predators (Bock *et al.* 1990). Conversely, grazing did not affect the demography or growth of the Texas tortoise (*Gopherus berlandieri*; Kazmaier *et al.* 2001*a*). Grazing may have influenced habitat selection in the Texas tortoise (Kazmaier *et al.* 2001*b*), but not in the Texas horned lizard (*Phrynosoma cornutum*; Burrow *et al.* 2001).

Studies on the responses of amphibians to livestock grazing are few, and information on anuran natural history indicates that amphibians may be especially sensitive to direct effects of livestock grazing (Scott 1996). Heavily grazed sites can have altered water chemistry and damaged or completely lost riparian vegetation (Yates *et al.* 2000). Because amphibians are dependent upon water for reproduction, changes in water chemistry are likely to affect their survival and reproduction. Similarly, the loss of riparian and emergent vegetation, which provides necessary cover from predators, can be detrimental (Scott 1996). Conversely, Natterjack toads (*Bufo calamita*) in Britain were unaffected by seasonal (May-September), low-intensity (0.33 AU/ha) grazing on heathland (Denton & Beebee 1996). However, Natterjack toads may be especially compatible with grazing, as they showed a preference for open areas with nearly no vegetation (Denton & Beebee 1996), a habitat presumed to be historically uncommon in the Great Plains of North America. In the United States, livestock grazing is considered a threat to the survival of the Arroyo southwestern toad (*Bufo microscaphus californicus*).

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The elimination of grazing in an area was considered a primary factor in the toad's expansion into that area (USFWS 1999), suggesting that livestock, when present, rendered that area unsuitable for habitation. Changes in soil compaction and soil moisture availability caused by livestock (Yates *et al.* 2000) may directly affect burrowing anuran species (Duellman & Trueb 1986) and indirectly affect the anuran community by altering conditions suitable for plant establishment.

Whereas no major North American studies have documented specific effects of livestock grazing on anurans, other types of habitat change affect their diversity and distribution. Clearcutting in a northern Florida flatwoods reduced amphibian abundance by a factor of 10; breeding success was reduced because logging efforts reduced standing water habitat (Enge & Marion 1986). Perison *et al.* (1997) found that although diversity indices remained similar, amphibians showed differential responses to timber harvest. Some species showed much higher numbers in clearcuts (over 1500% increase in individuals caught), whereas other species showed decreased abundance in clearcuts (over 90% decline in individuals caught). Cole *et al.* (1997) found similar differential effects of clearcutting and burning in Oregon. Amphibians were more abundant in burned than unburned areas in an Appalachian forest (Kirkland *et al.* 1996). Some amphibian species clearly show a positive response to some land-use practices while others are negatively affected.

Ecological interactions among anuran species are density dependent (Duellman & Trueb 1986). Increased competition or predation pressure could result if environmental disturbances give a few species a selective advantage. Conversely, disturbances such as grazing may result in the reduction of a highly competitive species. Intermediate levels of disturbance may prevent or delay competitive exclusion of species that would be out-competed under undisturbed conditions, maintaining higher species diversity under the intermediate disturbance hypothesis (Connell 1978). Because North American grasslands evolved in the presence of native grazers and were subject to moderate levels of grazing disturbance, it is plausible that some degree of grazing pressure is necessary to maintain historical levels of anuran diversity.

I investigated anuran species diversity and species composition in southwestern Oklahoma. My primary objective was to determine the effects of grazing on anuran diversity and species composition in this region by comparing data between areas with three different levels of grazing pressure.

STUDY AREA

The study areas were located in Comanche and Kiowa counties in southwestern Oklahoma (Fig. 1). The region is ecologically diverse, encompassing 15 land-cover classifications (Pogue and Schnell 2001) and 806 known plant species (Buck & Crockett 1963). Summers are hot, with average monthly temperatures in July and August frequently exceeding 38°C (N.O.A.A. 2001). Winters are dry and relatively mild, and spring is characterized by warm temperatures and thunderstorms. The 30-year average annual precipitation is 79.04 cm, most of which falls in May and June (N.O.A.A. 2001). Three 2.5-km^2 study areas representing different grazing levels were located in the southwest portion of Wichita Mountains National Wildlife Refuge (WMNWR), the northwest portion of the Quanah range of Fort Sill Military Reservation, and privately owned land to the west of WMNWR (Fig. 1).

The WMNWR study area was contained within the Public Use Area and was open to the public for hiking and fishing, although it was not a popular location for either and no members of the public were encountered during sampling. Open grassland areas at the WMNWR study area were primarily mixed-grass and tallgrass prairie with smaller riparian and oak savanna regions (Pogue and Schnell 2001). They were characterized by big bluestem (*Andropogon gerardii* var. *gerardii*), little bluestem (*Schizachyrium scoparium*), blue grama (*Bouteloua gracilis*), hairy grama (*Bouteloua hirsuta*), Indian grass (*Sorghastrum nutans*) and switchgrass (*Panicum virgatum*). Cross timbers habitat (oak savanna) was dominated by post oak (*Quercus stellata*), blackjack oak (*Quercus marilandica*) and eastern red cedar (*Juniperus virginiana*). Streams were both permanent and seasonal. Riparian plant species included black willow (*Salix nigra*) and buttonbush (*Cephalanthus occidentalis*). Ponds at WMNWR were man-made, but temporary pools were not, although the water levels of some temporary pools may be influenced by the proximity to and positioning of these pools relative to the larger ponds.

Quanah range of Fort Sill Military Reservation provided habitat and dominant species similar to WMNWR. There were large, man-made, permanent ponds and temporary pools both associated with and independent of the larger ponds. Quanah range was secured from general public access, but members of the public may have obtained permission seasonally to enter the range (when not in use for military training) to fish and hunt.

The private rangeland study area included similar habitat types to the aforementioned study areas. Cross timbers habitat was lacking, although cross timbers species occurred in riparian zones. Permanent ponds were created to water cattle.

Temporary pools existed independently of ponds and did not appear to receive any runoff from them. Grasslands consisted of exotic species including Johnsongrass (*Sorghum halepense*) and yellow bluestem (*Bothriochloa ischaemum*). Mesquite (*Prosopis glandulosa*) was locally abundant and appeared to be encroaching on the rangeland.

Burning and Grazing History

Fire has historically played a key role in the maintenance and species composition of grasslands (Vankat 1979), including the mixed-grass prairies of southwestern Oklahoma. Prescribed burning is currently used as a management tool on WMNWR and Fort Sill Military Reserve, where two of the three study areas were located. Considering the natural role of fire in this region, the possible interactive effects of fire with grazing cannot be overlooked. Potential confounding effects of different burning treatments were minimized because all three areas were last burned before 1993.

Wichita Mountains NWR comprised three pastures: Quanah (6,572 ha), Mount Scott (3,097 ha), and Pinchot (13,920 ha). The WMNWR study area was located in the Quanah pasture, which receives a low level of cattle and bison grazing year-round (0.021 animal units per hectare per year). Although prescribed burning is employed on fourteen fire management units on an approximately 5-year burn cycle, exact fire patterns are unpredictable and are influenced by weather conditions at the time of the burn. Although the burn unit encompassing the WMNWR study area has burned more recently, the WMNWR study area itself has not burned since 1992.

The Fort Sill Military Reservation study area has received no grazing pressure since at least 1957, when the Quanah range was added to the reservation (G.E. Wampler, Fort Sill Military Reservation, personal communication). Prescribed burning is scheduled on a 3-6 year burn cycle on most of the reservation, although impact areas may burn more frequently. The Fort Sill Military Reservation study area has not burned since 1991 (G.E. Wampler, Fort Sill Military Reservation, personal communication), probably due to variations in fire pattern or other management priorities.

The private rangeland study area is located on private land approximately 8 km to the northwest of the other two areas. This area is heavily grazed periodically (0.15 AU ha^{-1} y^{-1}) and has not been burned since at least 1990.

METHODS

Data were collected from 19 March 2001 to 29 July 2001. Sampling sites were classified according to habitat type (permanent ponds, temporary pools, and streams/riparian). The WMNWR study area contained nine temporary pools, three permanent ponds, and three stream locations. The Fort Sill Military Reservation study area contained seven temporary pools, three permanent ponds, and two stream locations. The private rangeland study area contained four temporary pool habitats (two of which consisted of groups of 10-15 small, shallow pools), two permanent ponds, and one stream habitat.

Most sampling methods show a bias in the type of species that are most effectively sampled, sampling efficiency between habitat types, or both (Heyer *et al.* 1994). Accordingly, three different sampling methods were used to increase the probability of locating all species present in an area. These methods (described in more detail below) were: visual encounter surveys, pitfall trap-drift fences, and audio recording of calling males.

Visual encounter surveys

Visual encounter surveys (Crump & Scott 1994) have been among the most effective means of surveying amphibians at the Ouachita Mountains in Arkansas (Crosswhite *et al.* 1999) and at Fort Sill Military Refuge, especially when high-intensity searches of shoreline and water were conducted (Caldwell *et al.* 1992). For all visual encounter surveys, I identified and recorded numbers of individuals of each species, time, weather, habitat type, and location.

Two types of visual encounter surveys were conducted. The first was time-constrained searches of the banks of streams, permanent ponds, and temporary ponds, particularly breeding sites (Scott & Woodward 1994). I conducted intermediate-intensity searches (Crump & Scott 1994) by identifying individuals in the open and by turning over surface objects such as rocks and small logs. Each sampling site was searched for 30-45 min on 2-3 separate dates 4-6 weeks apart. Searches were primarily conducted between 2000 and 2300 hrs. To reduce sampling bias due to calling activity, only individuals that were visually located were counted. Individuals were identified to species and tallied. Because sampling effort differed among study sites, I standardized search data by dividing the number of individuals by the sampling time to obtain the number of each species recorded per sampling hour.

The second type of visual encounter survey was road cruising, conducted weekly between 2200 and 0400 hrs. Road cruising effort comprised 65.3 km over 17.5 hr in private rangeland, 53.4 km over 15.1 hr at Ft. Sill, and 81.1 km over 24.5 hr at WMNWR. Although attempts were made to sample after rains, roads within the study areas were not paved and heavy rains made some roads impassable for several days.

Many roads on Fort Sill Military Reservation's Quanah range were impassable for the duration of the study, and the paved road outside the range substituted a portion of the distance. The dirt road on WMNWR was not maintained and was not open to the public, whereas the road surrounding the private rangeland area was a frequently-traveled gravel county road. Road-cruising surveys were conducted over a constant distance on a set route. Distance, time, and all individuals present on the road were recorded. Individuals outside the boundary of the road were not counted to minimize bias caused by the presence of concealing cover.

Pitfall trap-drift fences

Drift fences and pitfall traps can often capture species that are not easily found through visual encounter surveys. They are particularly effective for sampling terrestrial toads (*Bufo* spp., *Scaphiopus* spp.) that are not skilled jumpers or climbers (Corn 1994). Two drift fences were placed in each study area to capture individuals of species not encountered using other search methods. One drift fence was placed on one side of a temporary pool (Dodd & Scott 1994), and the other was placed in an open area. Each drift fence was approximately 16 m in length and 30 cm in height. Eight-liter pitfall traps were placed 4 m apart along the drift fence, with one pair of traps (one on each side of the fence) at the center, 1 pair of traps between the center and each end, and a single trap on each end, for a total of 8 traps per drift fence. Traps had small holes in the bottom to prevent traps from becoming flooded, and saturated sponges were placed in the traps when open to help prevent desiccation of animals. When traps became flooded, they were kept closed until they had drained, which reduced the number of available traps per night. A board was raised 3-5 cm above each trap to provide some protection from predators and direct sunlight. When not in use, traps were closed by placing the board directly on top of the trap and weighing it down with a large rock (Corn 1994). Traps were opened in the evening and were checked and closed the following morning. Pitfall trapping was conducted twice weekly and amounted to 175 trapnights in private rangeland, 194 trapnights at WMNWR, and 202 trapnights at Ft. Sill. Anurans were identified to species and tallied.

Audio recording of anuran calls

Five automated recording systems, modified from Peterson & Dorcas (1994), were placed simultaneously at 1-2 ponds in each study area to determine presence or absence of species and to compare relative abundance between areas. A voice clock created an audio time and temperature stamp for each sampling period. The timers were set to record 60 seconds every 62 minutes for the duration of one 30-minute side of a 60-minute tape. The recording systems were used to sample one habitat type (e.g., temporary pool) in each study area on the same night. They were moved among ponds within each study area to more fully sample that habitat type for the entire study area. Anuran calls were identified to species for each sampling interval, and species-specific call intensity was quantified as follows: 0 = no individuals calling; 1 = a single individual calling; 2 = a few individuals calling (little or no call overlap); 3 = many individuals calling with overlap (Mohr & Dorcas 1999, Bridges & Dorcas 2000). Only tapes containing call samples from a species were used to calculate averages for that species; however, time intervals for which call intensity values were 0 were included in the calculations. Call intensities of each species from different sites of the same habitat type within a study area were averaged for comparison across study areas (Mohr & Dorcas 1999). Each site was sampled 4-10 nights, 1-4 weeks apart. Audio recording effort totaled 48 nights in private rangeland, 85 nights at WMNWR, and 56 nights at Ft. Sill. Because this method does not allow for exact numbers of amphibians to be counted, they represent ordinal measures of data (Stevens 1946) and I did not compare call intensity values statistically. However, call intensities can be compared among areas for preliminary interpretation of relative abundance (Bridges & Dorcas 2000).

GIS Data

Ecological systems are influenced by a large number of variables. Although study areas were in close proximity, many variables besides grazing level could differ among areas. Geographical Information System data characterize environmental characteristics of an area without the need for time-consuming and costly field sampling. By overlaying spatial environmental data with the study areas, I identified environmental parameters for use in further analyses.

I classified study sites according to habitat type (permanent ponds, temporary pools, and streams/riparian), and sampling sites were entered as a point coverage in a Geographical Information System using a combination of ArcView 3.2 and ArcGIS 8.1. Attributes were added to identify each sampling location and to indicate whether the sampling point was representative of a permanent pond, temporary pool, or stream.

I used road, stream, and county and federal lands boundary data provided by the U.S. Geological Survey's Digital Atlas of Oklahoma. I obtained 30 m Digital Elevation Models (DEMs) from GIS Data Depot (www.gisdatadepot.com) in SDTS format and merged the four quadrants into one DEM grid. I used this grid to compare elevation, slope, and aspect among areas. I obtained STATSGO soils data (USDA 2002) and

Oklahoma GAP data (M. Gregory, Oklahoma State University), which I used to further identify local environmental characteristics.

Individual sampling sites were represented by points, which, by definition, contain no area. To obtain a representative sample of the area around the sampling location, I created 50-meter buffers around each point. I used Spatial Analyst in ArcGIS to generate zonal statistics for soils, land cover, elevation, slope, and aspect. These statistics identified a mean value of elevation, slope, and aspect for each buffer zone. I calculated a variety statistic to determine the number of landcover types contained within the buffer zone, and majority statistics to identify the primary landcover and soil type of each buffer zone.

Available spatial GIS data may vary in resolution. The DEMs used had a spatial resolution of 30 m, whereas the GAP data had a resolution of about 25 m. Because of resolution limitations and the small areas (circles of 50 m radius) used to characterize them, caution is warranted when interpreting analyses that use these data.

Calculation of species diversity

Species diversity was calculated from search data. Because species richness among all three locations is similar (WMNWR = 6 species, Ft. Sill = 7 species, and private land = 7 species), a diversity index that is sensitive to minor differences would best facilitate comparison among sites. I calculated diversity using the Shannon-Wiener function because it weights rare species more heavily than do other diversity indices (Krebs 1999). I calculated the Shannon-Wiener function according to Krebs (1999) as follows:

$$H' = -\Sigma p_i \log_2 p_i$$

where p_i = relative abundance of the i^{th} species. I calculated H' for each habitat type within each study area, and compared similar habitat types across study areas (Jones 1988).

Ordination

I used CANOCO for Windows 4.0 (ter Braak & Smilauer 1998) to perform two ordination procedures, detrended correspondence analysis (DCA; Gauch 1982) and canonical correspondence analysis (CCA; ter Braak 1986) to reveal gradients in species composition. I used standardized search data to examine species composition patterns among habitat types within study areas and among study areas within habitat types.

DCA is an indirect gradient analysis technique that reveals gradients in species composition independent of measured environmental variables. Therefore, it is useful for examining the data for potential environmental patterns, regardless of whether the gradient displayed represents a measured environmental variable (Gauch 1982). The DCA axes are measured in units of constant beta diversity, which is a measure of how far apart the sample scores are on species composition gradients. I performed DCA using no downweighting of rare species and accepted all other program defaults. I used DCA to compare species composition among habitat types within study areas and among study areas for temporary pool and permanent pond habitats.

CCA is a direct gradient analysis technique in which species composition patterns are related to measured environmental variables. Environmental variables that I used for the analysis included grazing level, landcover type, landcover variety, soil type, mean elevation, elevation range, mean slope, and mean aspect.

RESULTS

Visual Encounter Surveys

Searches of temporary pools, permanent ponds, and streams amounted to 71.5 search hours and yielded nine species of anurans (Table 1). Analysis of variance indicated that Shannon-Wiener indices (Table 2) did not differ ($P \geq 0.41$ for all comparisons) among the three study locations for temporary pools, permanent ponds, or streams (Table 4). Shannon-Wiener indices differed significantly among habitat types ($P = 0.0002$, Table 3).

The sum of all unconstrained eigenvalues for Ft. Sill was 0.648, and DCA Axis 1 explained 51.2% of the variance in the data. The sum of all unconstrained eigenvalues for WMNWR was 0.660, and DCA Axis 1 explained 52.8% of the variance in the data. For private rangeland, the sum of all unconstrained eigenvalues was 1.175, and DCA Axis 1 explained 72.5% of the variance in the data. DCA Axis 1 likely represented a similar gradient in all three study areas (less permanent to more permanent water), but the trend was most pronounced at the private rangeland study area. Species composition showed consistent differences among habitat types, even when the sampling locations were in very close proximity within the same study area (Fig. 2a-c). In all cases, there was a clear separation of habitat types along the ecological gradient with temporary pool habitat the lowest on the first DCA axis.

I eliminated the effects of habitat on species composition by using only data from one habitat type at a time to compare the three study areas using DCA. Although the Fort Sill (no grazing) and Wichita Mountains NWR (0.021 AU ha^{-1} y^{-1}) areas did not show clear differences in species composition, the private rangeland location (0.15 AU ha^{-1} y^{-1})

was clearly set apart from the others along DCA Axis 1, showing strong differences in species composition (Fig. 3). DCA Axis 1 had an eigenvalue of 0.731 and explained 47.2% of the explainable variance. I was unable to determine with certainty what DCA Axis 1 represented; however, it may represent a gradient due to grazing, soil type, or local precipitation patterns. Permanent pond habitat showed no trends (Fig. 4). There were insufficient stream habitat observations to perform the analysis.

Canonical correspondence analysis revealed that grazing level and soil type, which had the longest arrows on the CCA plot, were the most significant contributors to species composition (Figure 5). The ratio of the sum of all canonical eigenvalues (0.903) to the sum of all unconstrained eigenvalues (1.500) was 0.602, suggesting that the variables used in the analysis explain 60.2% of the total variance in the data. Monte Carlo analysis identified grazing as the most significant environmental variable ($P = 0.02$ with 999 permutations), explaining 62% of the variance explained by all environmental variables in the model. On CCA axis 1, which had an eigenvalue of 0.673, grazing had the highest species-environment correlation (0.9574) of the environmental variables.

The spotted chorus frog (*Pseudacris clarkii*), Texas toad (*Bufo speciosis*), and Couch's spadefoot toad (*Scaphiopus couchii*) were all positively associated with grazing and were found either primarily or exclusively on the private rangeland (heavily grazed) area. Other species, such as Strecker's chorus frog (*Pseudacris streckeri*) and Woodhouse's toad (*Bufo woodhousii*) were negatively associated with soil type and grazing and were found only in the Fort Sill (no grazing) and Wichita Mountains (low grazing) areas.

Road cruising detected densities as high as 1.4 *Bufo speciosis* and 0.5 *Scaphiopus couchii* per km at the private rangeland area. However, because *B. speciosis* was found on only one occasion on the Ft. Sill area and no anurans were found on WMNWR during road cruising surveys, no comparisons of road-cruising results were made among study areas.

Pitfall trap-drift fences

Drift fence/pitfall traps located at breeding ponds at Fort Sill and WMNWR were effective in capturing newly emerging individuals of *Pseudacris streckeri*, *Acris crepitans*, and *Rana blairi*, but resulted in few adult captures; juveniles were rarely captured on the private rangeland study area (Table 5). No juveniles and few adult anurans were captured by traps placed in open areas away from breeding ponds (Table 6).

Call intensity

Audio recordings of anuran calls accounted for all species found by other methods, and increased confidence that all species present in each study area were identified. Call intensities for many species were similar across study areas, but differences in species composition were reflected by the data. *P. streckeri* was not heard on the private rangelands site, and *B. woodhousii* was heard calling only on Fort Sill. Conversely, *S. couchii* and *P. clarkii* were heard calling only on the private rangelands study area (Table 7).

Weather

A climate-data gathering site for the refuge was located on the west side of WMNWR (N.O.A.A. 2002). Average monthly precipitation for March 2001 through July 2001 was 0-27% of the five-year (1996 - 2000) average for those months, with the exception of May (Fig. 6a). The N.O.A.A. station recorded 22.02 cm of precipitation in May, a 350% increase over the five-year average of 6.29 cm. Over half of this precipitation fell on 18-19 May (H. Hubbs, USFWS, personal communication). Average daily maximum temperature for March 2001 through July 2001 was 0-6 °C higher than the five-year average for those months (Fig. 6b). Average daily minimum temperature March 2001 through July 2001 was similar to the five-year average for those months (Fig. 6c).

DISCUSSION

Comparison of Survey Methods

Time-constrained searches and audio recording of anuran calls detected all nine species of anurans that were found using all four survey methods during the study period. Road cruising detected two species and pitfall trapping detected seven.

Searches: Because all identified species were detected and because the data could be used to compare relative abundance among study areas, time-constrained searching was the most effective survey method used in this study. This finding is consistent with those of workers in the Ouachita Mountains of Arkansas (Crosswhite *et al.* 1999) and in Fort Sill, Oklahoma (Caldwell *et al.* 1992). Although searches may not detect the most secretive and least active species (Parris 1999), I detected all species that were detected using the other survey methods.

Audio recording: Audio recording of calling males also detected all species found during the study, and is considered to be more effective at detecting secretive species than is time-constrained searching (Peterson & Dorcas 1994). Relative abundance data obtained from recording of calling males are less reliable than data obtained from searches because only males are heard and because it is difficult to accurately count individuals when many calling males are present; estimates of abundance are nearly always underestimates (Parris 1999).

Average call intensities of species across study areas and habitat types (Table 7) were concordant with search data (Table 2). Species were consistently detected by both methods at each habitat type and within each study area, although the relative calling intensity was not always proportional to the relative search data. A species that had higher search encounter frequency at a habitat area or within a study area did not necessarily have higher average call intensity. Calling intensity for each species was averaged over a taping night. If many individuals were present but called for a short period, average call intensity could be lower than if fewer individuals were present and called for a longer period. Automatic audio recording survey would be useful for detecting species not detected by other methods, but less useful for relative abundance comparisons.

Road cruising: Road cruising is effective for detecting large, active species of reptiles and amphibians, particularly at night and after rains (Parris 1999). I found road cruising to be effective for locating *Bufo speciosis* and *Scaphiopus couchii* on the private rangeland study area, but less effective on the other two study areas. The three study areas differed considerably in road quality and the amount of concealing cover. The road at the WMNWR study site was an employee access road not intended for regular travel; it was rocky and overgrown compared to the private rangeland road, which may have made detection of anurans more difficult. It was also impassable after heavy rains, when driving surveys are most efficient. Fort Sill roads on the Quanah range were not maintained for use by private citizens and were often impassable. I substituted the paved highway bordering the region, which had recently been resurfaced. Although there was no concealing cover present, I located few anurans. Across all survey methods, *B. speciosis* and *S. couchii* were found infrequently or not at all on the Fort Sill and WMNWR study sites. Although the road cruises were not directly comparable due to differences in concealing cover, results of the other survey methods agree that there were many more of these two species present at the private rangeland study area than at the other two areas.

Pitfall trapping: Although pitfall trapping is often very effective at sampling amphibians (Crosswhite *et al.* 1999, Parris 1999), it is not effective at Fort Sill (Caldwell *et al.* 1992). Drift fences with pitfall traps were effective at capturing newly metamorphosed individuals when placed near breeding sites at Fort Sill and WMNWR, but were not very effective when placed away from breeding sites. Anurans typically travel the most after heavy rains (Duellman & Trueb 1986). With the exception of two days in May, precipitation was below average (Fig. 6a), which could account for low trapping success of drift fence-pitfall traps in open areas (Table 6). The low juvenile capture rate at breeding pond habitat in the private rangeland study site may be explained by chance, as the traps were not consistently open and the individual pools may not have been used equally by anurans.

Species Diversity

Habitat types: Species diversity differed significantly among habitat types (Table 3), with temporary pool habitats showing the highest level of species diversity (Table 2). Temporary pools are smaller than permanent ponds, so would be expected to have lower species diversity than permanent ponds on the basis of size alone (but see Snodgrass *et al.* 2000). The enhanced species diversity of temporary pool habitats may be attributable to temporal effects that were not explored in this study. Species that use temporary pool habitats to breed have different temperature requirements and require a shorter development period (Duellman & Trueb 1986). Because the breeding requirements of each species are not met at all times of the year, different species occupy the site at different times in the season. Accordingly, calculation of *H'* across the sampling period was high.

Study areas: Although species diversity (*H'*) from search data was generally (but not significantly) higher at the private rangeland site, this may be due to site-specific, rather than treatment, effects. Two temporary pool sites at the private rangeland study area consisted of 10-15 small pools that were in very close proximity to one another. Although these small pools cover an overall area that is larger than the other temporary pool sites (which consisted of one, larger temporary pool), larger wetland areas are not necessarily correlated with higher species richness (Snodgrass *et al.* 2000). However, individual breeding ponds tend to become dominated by one or two species. Individuals are attracted to the breeding pond by calling males (Duellman & Trueb 1986); it follows that whichever species arrives at a pond first and calls the loudest may dominate that pond. Because anurans often show site fidelity (Semlitsch 2002, Marsh & Trenham 2001), species associations may be established before the breeding season begins. Dayton & Fitzgerald (2001) found that among desert anurans breeding in ephemeral habitats, only 12% of sites were occupied by more than one species. When several separate pools exist in close proximity, they may provide habitat for several different species that would not otherwise coexist. In the private rangeland study area, I observed loud choruses of *Pseudacris clarkii* and *B. speciosis* simultaneously in separate pools and at opposite ends of the study site. The Shannon-Weiner index is a measure of heterogeneity (Krebs 1999), but does not consider species composition differences. Although heterogeneity was similar among study areas and even slightly higher at the private rangelands area, species composition showed strong differences.

Species Composition

Habitat types: Habitat type played a strong role in species composition, even when the sampling locations were in close proximity to one another within the same study area. This relationship may hold because many species are specialized for the hydroperiod of the site they occupy. This trend was especially apparent at the private rangeland location, which also represented the highest level of grazing. Differences in species composition among habitat types are likely due to differences in hydroperiod (Snodgrass *et al.* 2000). Temporary pools undergo changes in temperature, depth, and other parameters as the season progresses (Snodgrass *et al.* 2000). Because temporary pools contain less water than do permanent ponds, fluctuations in precipitation are more likely to affect their physical parameters and their longevity. Species that require temporary pools to breed have adapted to their ephemeral nature and typically develop faster than species that utilize permanent water (Duellman & Trueb 1986, Jones 1988). *S. couchii*, a desert anuran found in the private rangeland study area, is an explosive breeder that has a pre-metamorphic period of only 10 days (Tinsley & Tocque 1995). Species that use permanent water for breeding, such as *Rana catesbeiana*, require mechanisms of dealing with increased predation pressure from both aquatic and terrestrial vertebrates that depend on permanent water sources (Duellman & Trueb 1986).

Study areas: Species composition of temporary pool habitat clearly differed among the three grazing levels. Although grazing level was the primary difference among areas, temporary pool habitat on the private rangeland site was shallower and had a shorter hydroperiod than temporary pool habitat on the Ft. Sill and WMNWR areas. Even among habitat types within study areas, the difference in species composition of temporary pool habitats was more pronounced in the grazed area. These differences in species composition could be due to natural differences in the temporary pool habitat at the grazed area. However, temporary pool habitat may be more severely affected by grazing than permanent pond or stream habitat. Small pools are shallower and contain more edge area relative to their surface area than larger ponds do; the entire pool is subject to trampling. When cattle move into an area with small pools, they can more significantly reduce the water level (causing a shorter hydroperiod) and trample aquatic vegetation, as well as amphibian eggs and tadpoles.

Great Plains anurans possess adaptations to microhabitats within grasslands, and have evolved with historical levels of grazing (Scott 1996). Grazing pressure that is much higher than historical levels, however, can seriously alter habitat structure and plant species composition (Fleischner 1994). That Couch's spadefoot toad (*S. couchii*) was found only on the private rangeland area was suggestive of how grazing alters the habitat for amphibians. Couch's spadefoot is distributed from the Sonoran desert in southern California through southern Arizona, New Mexico, and the Texas panhandle, reaching the northeast limit of its distribution in southwestern Oklahoma (Conant & Collins 1998).

Because Couch's spadefoot toad is a desert species, it is not surprising that it is not negatively affected by heavy grazing on these mixed-grass and tallgrass sites and may even be positively affected by some habitat alterations that grazing provides. The Texas toad (*B. speciosis*), which was also more abundant in the grazed site, shares these benefits. The Texas toad is distributed from northern Mexico to southern Oklahoma, and prefers open grasslands, mesquite savannas, and loose, sandy soils (Conant & Collins 1998). Reduced vegetation from grazing may be beneficial for the Texas toad and Couch's spadefoot because it facilitates locomotion and burrowing, and the resulting mesquite encroachment may be beneficial because fallen logs can provide shelter.

Woodhouse's toad (*B. woodhousii*) was found only on the Fort Sill (ungrazed) study area. Although this toad is a wide-ranging species found in a variety of habitats, it may prefer ungrazed sites. *B. woodhousii* may use dense vegetation and existing burrows more often than creating burrows of its own (Conant & Collins 1998). Areas not subject to a high level of grazing are likely to provide dense vegetation that can provide cover without the energy demands of burrowing in an area without adequate cover. The plains leopard frog (*Rana blairi*) and Blanchard's cricket frog (*Acris crepitans blanchardii*) are generalist species that were commonly found in all aquatic habitats. These species were not frequently found in temporary pool habitat on the private rangelands area, which may be due to grazing effects. However, they were abundant at permanent pond and stream habitats within the private rangelands area.

Two chorus frogs were found during the study period. Although both species were found exclusively in temporary pool habitat, Strecker's chorus frog (*Pseudacris streckeri*) was not found on the heavily grazed study area, whereas the spotted chorus frog (*P. clarkii*) was found exclusively on the heavily grazed area. *P. streckeri* calls earlier in the season than do many frogs (Duellman & Trueb 1986) and was not heard after April 2001 during this study. Its absence from the private rangeland study area may have been due to low water level in the temporary pool habitat at that time. However, differences between the two species may affect their habitat preferences. *P. streckeri* is more associated with wetland habitats, whereas *P. clarkii* prefers grassland habitats (Conant & Collins 1998), possibly for reasons similar to those of *S. couchii* and *B. speciosis*.

Conclusions

My results suggest that heavy grazing may alter anuran species composition in temporary pool habitats, although this study was unreplicated and more work needs to be done in this area. Recent concerns about global amphibian declines (Carey *et al.* 2001) have prompted the investigation of land use practices that could impact amphibian survival and development (Enge & Marion 1986, Denton & Beebee 1996, Kirkland *et al.* 1996, Cole *et al.* 1997, Perison *et al.* 1997). Cattle grazing can alter many characteristics of terrestrial anuran habitat, such as vegetation density and soil moisture, and aquatic habitat, such as water chemistry and density of riparian vegetation (Yates *et al.* 2000). These alterations may be beneficial to some species and detrimental to others, effecting a change in species composition that may not be readily reversible, even after grazing is removed (Curtin 2002).

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Table 1. Comparison of numbers of individuals of nine species encountered per search hour from three study areas at Fort Sill Military Reservation (no cattle grazing), Wichita Mountains National Wildlife Refuge (low cattle grazing), and private rangeland (high cattle grazing) in southwest Oklahoma, March 2001 – July 2001.

Common name	Scientific name	Fort Sill MR	Wichita Moutains	Private Rangeland
Woodhouse's Toad	*Bufo woodhousii woodhousii*	0.36	0	0
Texas Toad	*Bufo speciosus*	0.18	0.27	3.56
Couch's Spadefoot	*Scaphiopus couchii*	0	0	1.88
Great Plains Narrowmouth Toad	*Gastrophryne olivacea*	2.35	1.75	5.06
Blanchard's Cricket Frog	*Acris crepitans blanchardi*	18.50	23.60	23.44
Spotted Chorus Frog	*Pseudacris clarkii*	0	0	9.56
Strecker's Chorus Frog	*Pseudacris streckeri streckeri*	1.53	3.50	0
Bullfrog	*Rana catesbeiana*	0.54	1.08	2.44
Plains Leopard Frog	*Rana blairi*	7.40	8.49	7.13

Table 2. Shannon-Weiner (H') diversity indices from anuran search data. Minimum, maximum, and mean H' for sites within each habitat type in each of three study areas at Fort Sill Military Reservation (no cattle grazing), Wichita Mountains National Wildlife Refuge (low cattle grazing), and private rangeland (high cattle grazing) in southwest Oklahoma, March 2001 – July 2001.

Habitat Type		*n*	Minimum H'	Maximum H'	Mean H'
Temporary Pools					
	Fort Sill MR	7	0.684	1.892	1.523
	Wichita Mountains	9	0.723	2.213	1.384
	Private Rangeland	4	1.206	2.074	1.762
Permanent Ponds					
	Fort Sill MR	3	0.935	1.167	1.081
	Wichita Mountains	3	0.734	1.419	0.994
	Private Rangeland	2	1.058	1.238	1.149
Streams					
	Fort Sill MR	2	0.677	0.840	0.759
	Wichita Mountains	3	0.722	0.918	0.817
	Private Rangeland	1	0.706	0.706	0.706

Table 3. Analysis of variance examining the effect of habitat type on Shannon-Wiener diversity (*H'*) of anurans calculated from search data in three study areas at Fort Sill Military Reservation (no cattle grazing), Wichita Mountains National Wildlife Refuge (low cattle grazing), and private rangeland (high cattle grazing) in southwest Oklahoma, March 2001 – July 2001.

Source	df	SS	MS	*F*	*P*
Habitat type	2	2.99	1.50	16.54	0.0002
Grazing * Habitat Type	6	0.44	0.07	0.46	0.8329
Error	25	4.00	0.16		
Total	33	7.35			

Table 4. Analysis of variance examining the effect of grazing on Shannon-Wiener diversity (H') of anurans calculated from search data in three study areas at Fort Sill Military Reservation (no cattle grazing), Wichita Mountains National Wildlife Refuge (low cattle grazing), and private rangeland (high cattle grazing) in southwest Oklahoma, March 2001 – July 2001.

Habitat Type	Source	df	SS	MS	*F*	*P*
Temporary pool	Grazing	2	0.40	0.20	0.93	0.41
	Error	17	3.64	0.21		
	Total	19	4.04			
Permanent pond	Grazing	2	0.03	0.01	0.23	0.80
	Error	5	0.32	0.06		
	Total	7	0.35			
Streams	Grazing	2	0.01	0.01	0.48	0.66
	Error	3	0.03	0.01		
	Total	3	0.04			

Table 5. Drift fence-pitfall trap captures of adult and juvenile (juv.) anurans near breeding ponds at Ft. Sill, WMNWR, and private rangeland study areas in southwestern Oklahoma in 2001.

Location	*B. woodhousii*		*B. speciosis*		*S. couchii*		*G. olivacea*		*A. crepitans*		*P. clarkii*		*P. streckeri*		*R. catesbeiana*		*R. blairi*	
	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.
Fort Sill MR (202 trapnights)	0	0	0	0	0	0	0	0	16	97	0	0	0	107	0	0	1	36
WMNWR (194 trapnights)	0	0	0	0	0	0	2	0	23	74	0	0	1	81	0	0	0	43
Private Rangeland (175 trapnights)	0	0	0	16	0	1	0	0	1	0	6	0	0	0	0	0	0	0

Table 6. Drift fence-pitfall trap captures of adult and juvenile (juv.) anurans in open areas at Ft. Sill, WMNWR, and private rangeland study areas in southwestern Oklahoma in 2001.

Location	*B. woodhousii*		*B. speciosis*		*S. couchii*		*G. olivacea*		*A. crepitans*		*P. clarkii*		*P. streckeri*		*R. catesbeiana*		*R. blairi*	
	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.	Adults	Juv.
Fort Sill MR	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
WMNWR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Private Rangeland	0	0	7	0	2	0	1	0	0	0	0	0	0	0	0	0	0	0

Table 7. Average calling intensity (Mohr & Dorcas 1999, Bridges & Dorcas 2000) of male anurans in temporary pools (temp), permanent ponds (perm), and streams in Fort Sill, WMNWR, and private rangeland study areas in southwestern Oklahoma in 2001.

	Fort Sill MR			Wichita Mountains			Private Rangeland		
Species	Temp	Perm	Stream	Temp	Perm	Stream	Temp	Perm	Stream
Bufo woodhousii woodhousii	0.36	0	0	0	0	0	0	0	0
Bufo speciosis	0.2	0	0	0.15	0	0	1.4	0	0
Scaphiopus couchii	0	0	0	0	0	0	0.33	0	0
Gastrophryne olivacea	0.21	0	0	0.27	0	0	0.14	0	0
Acris crepitans blanchardi	0.31	0.67	0.59	0.25	0.64	0.59	0.16	0.78	0.58
Pseudacris clarkii	0	0	0	0	0	0	0.27	0	0
Pseudacris streckeri streckeri	0.42	0	0	0.48	0	0	0	0	0
Rana catesbeiana	0	0.28	0	0.09	0.42	0	0	0.42	0
Rana blairi	0.33	0.42	0.33	0.26	0.26	0.23	0.18	0.49	0.32

Figure 1. Location of three study areas in relation to Wichita Mountains National Wildlife Refuge and Fort Sill Military Reservation in Oklahoma.

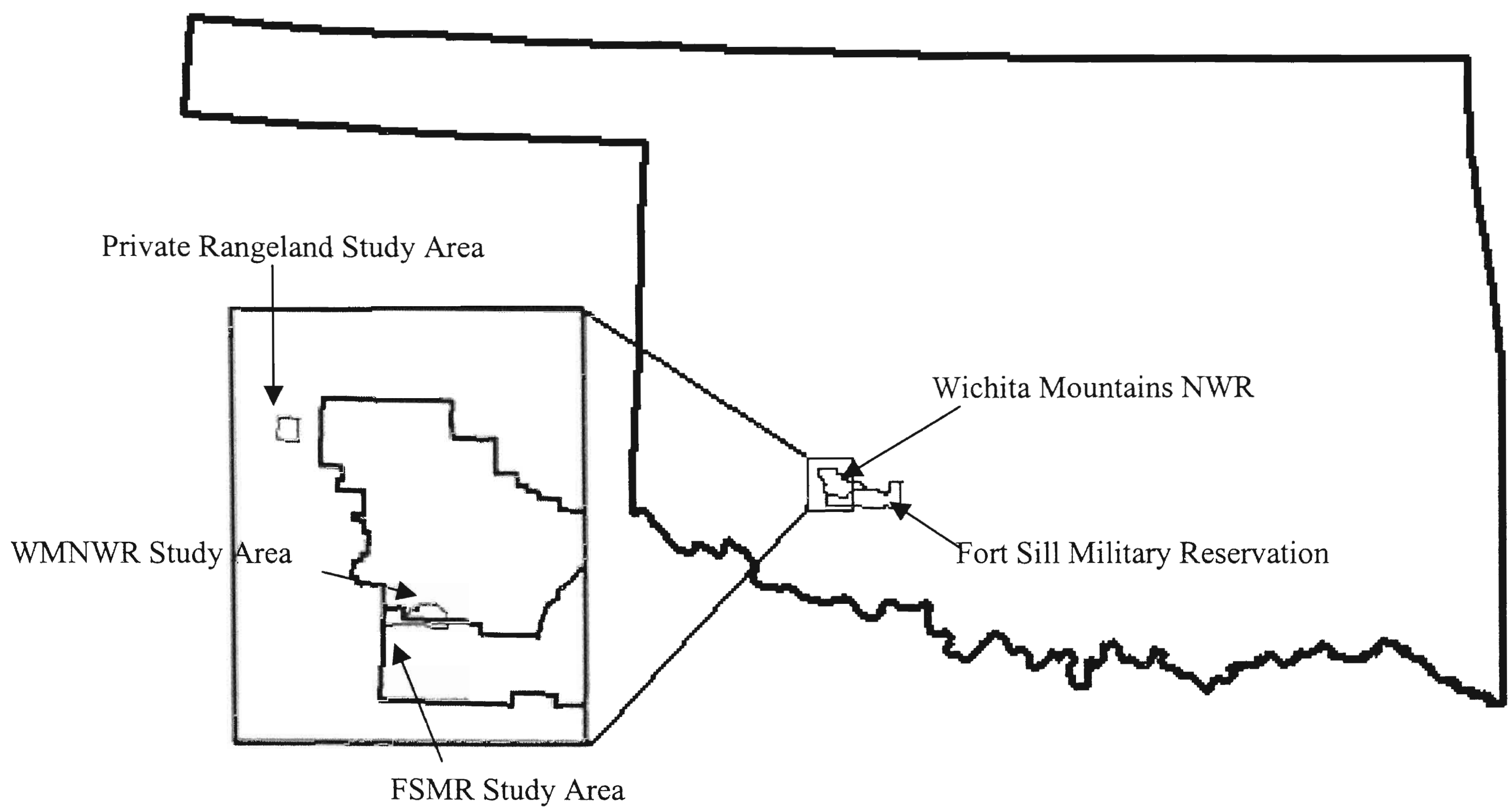


Figure 2. Detrended correspondence analysis of anuran species composition in southwestern Oklahoma, 2002; a) Ft. Sill, b) WMNWR, and c) private rangeland. Circles = temporary pools, squares = permanent ponds, triangles = streams.

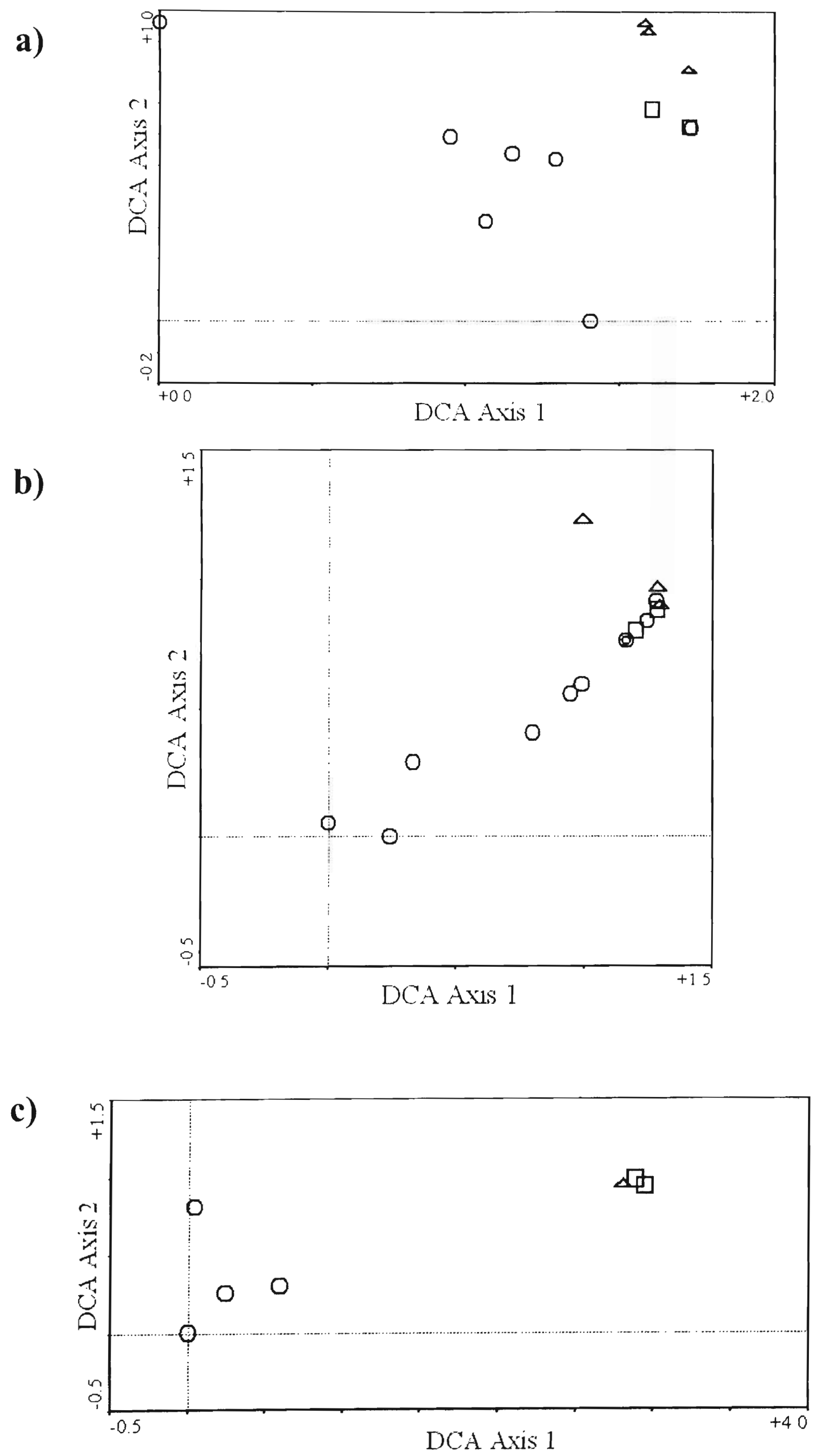


Figure 3. Detrended correspondence analysis of anuran species composition in temporary pool habitat in three study areas in southwestern Oklahoma, 2001. Circles = WMNWR, squares = Ft. Sill, triangles = private rangeland.

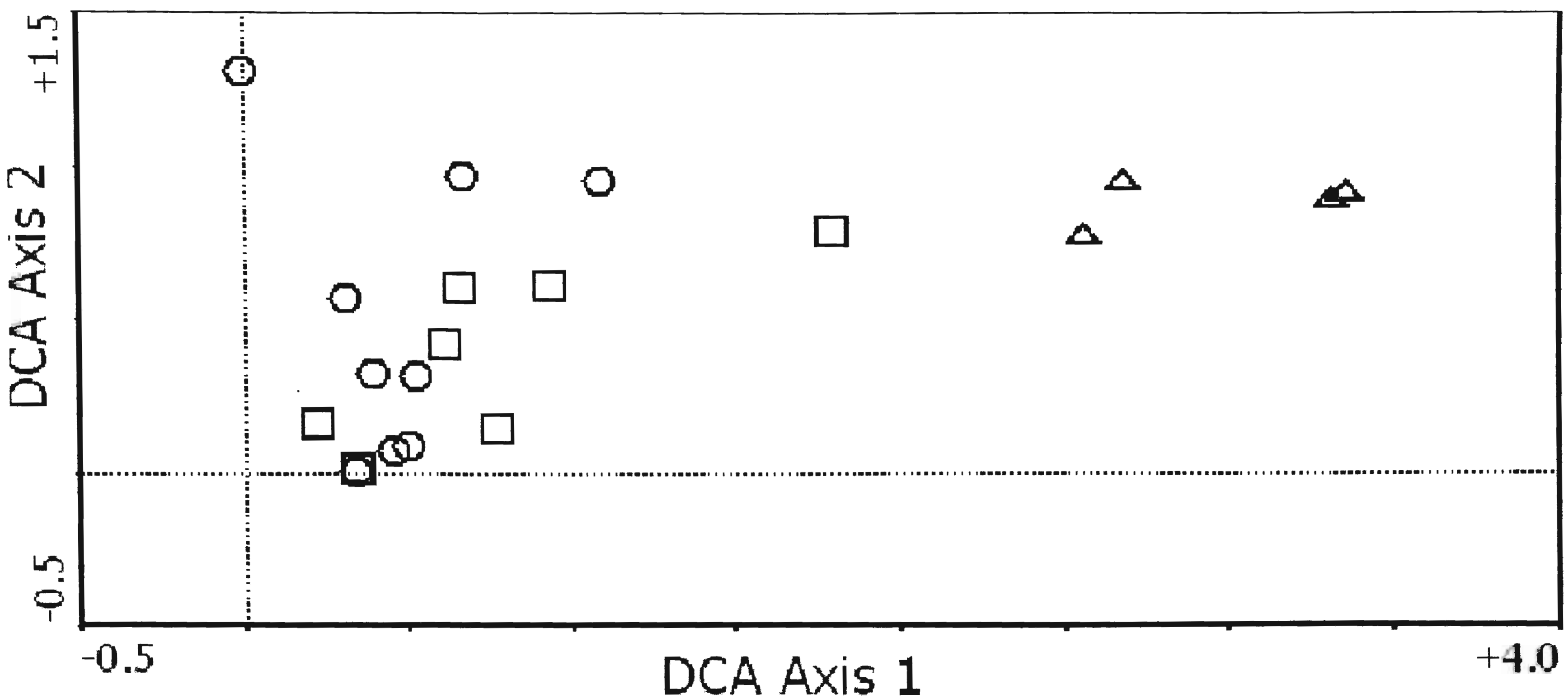


Figure 4. Detrended correspondence analysis of anuran species composition in permanent pond habitat in three study areas in southwestern Oklahoma, 2001. Circles = WMNWR, squares = Ft. Sill, triangles = private rangeland.

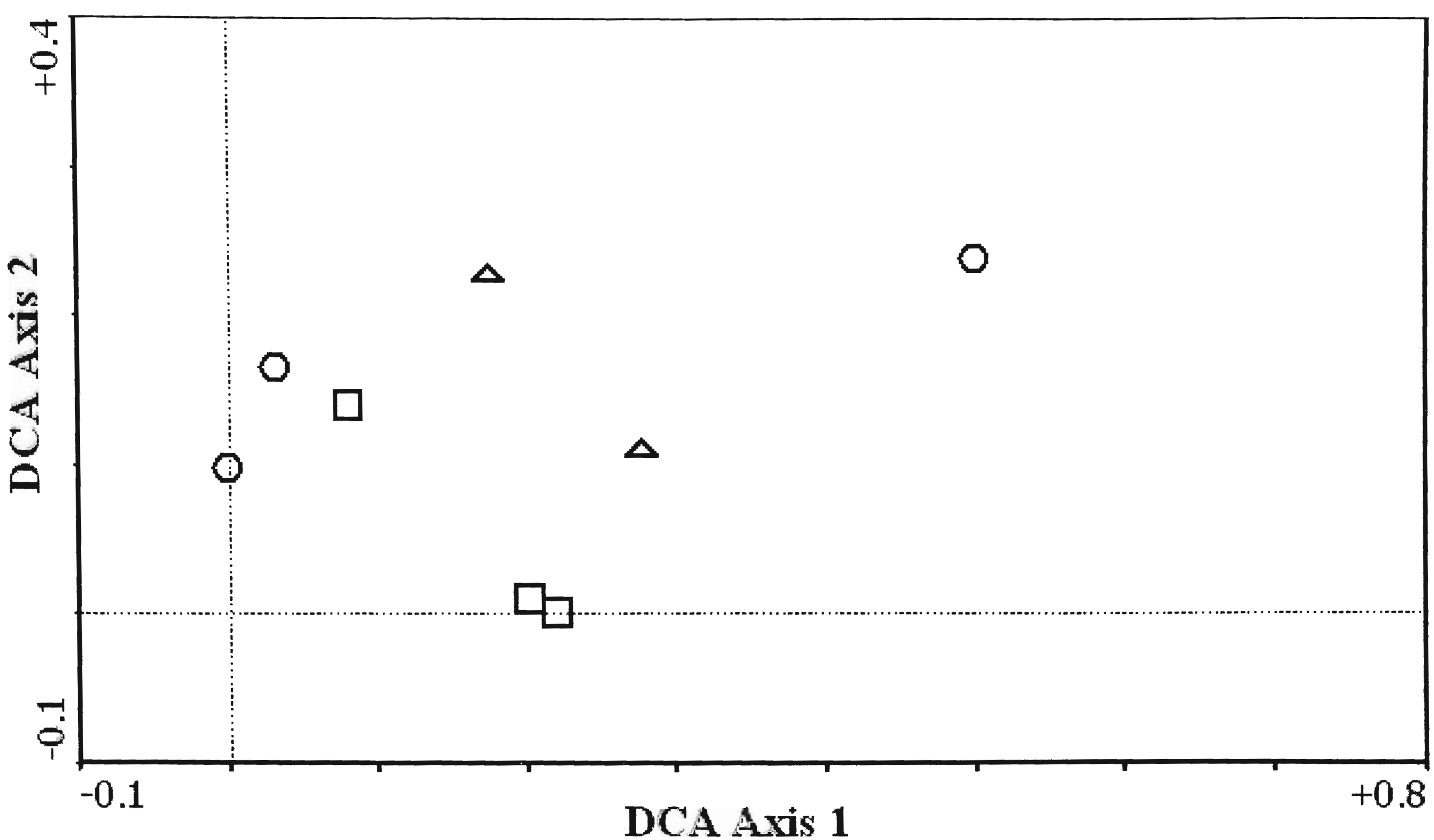


Figure 5. Canonical Correspondence Analysis of search data for anuran species in temporary ponds from Ft. Sill, WMNWR, and private rangeland in southwestern Oklahoma with eight environmental variables.

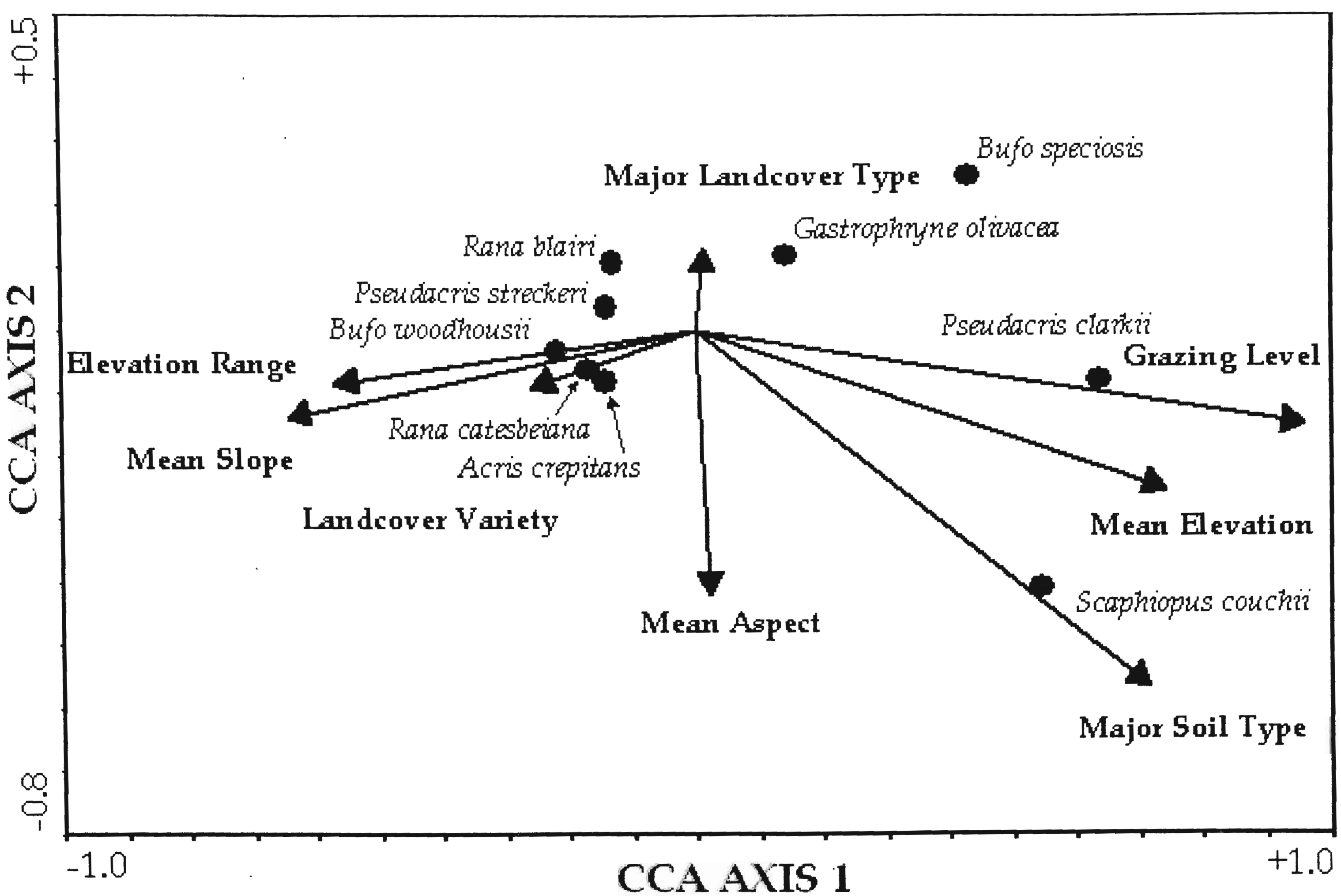
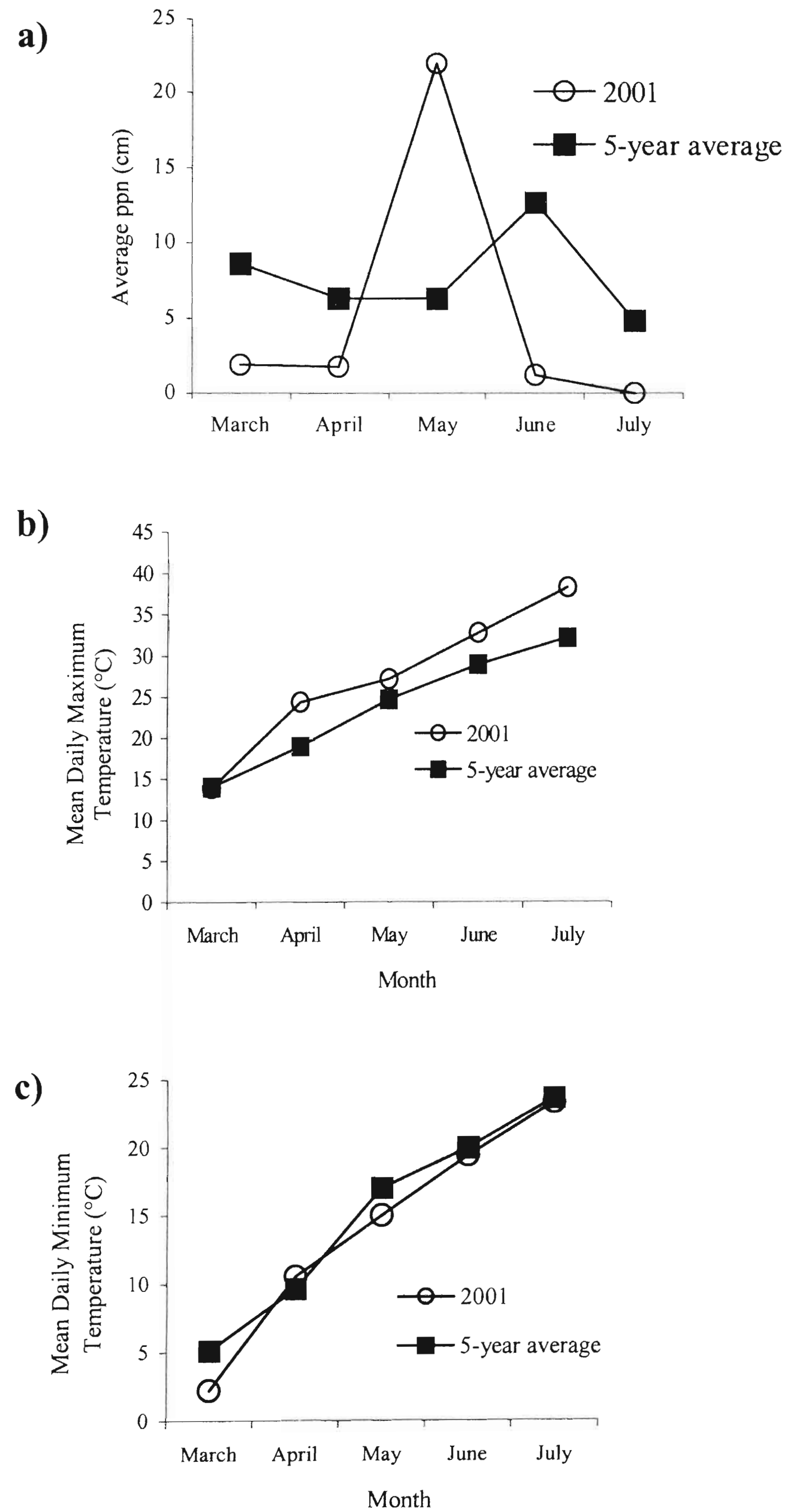


Figure 6. Weather data at N.O.A.A. Cooperative Station # 349629, WMNWR, from March 2001 to July 2001 compared to the 5-year average; a) mean precipitation, b) mean daily maximum, c) mean daily minimum (N.O.A.A., 2001).



VITA 2

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Master of Science

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