MANAGING WILDLIFE IN THE SOUTHWEST:
New Challenges for the 21st Century

Proceedings of the Symposium
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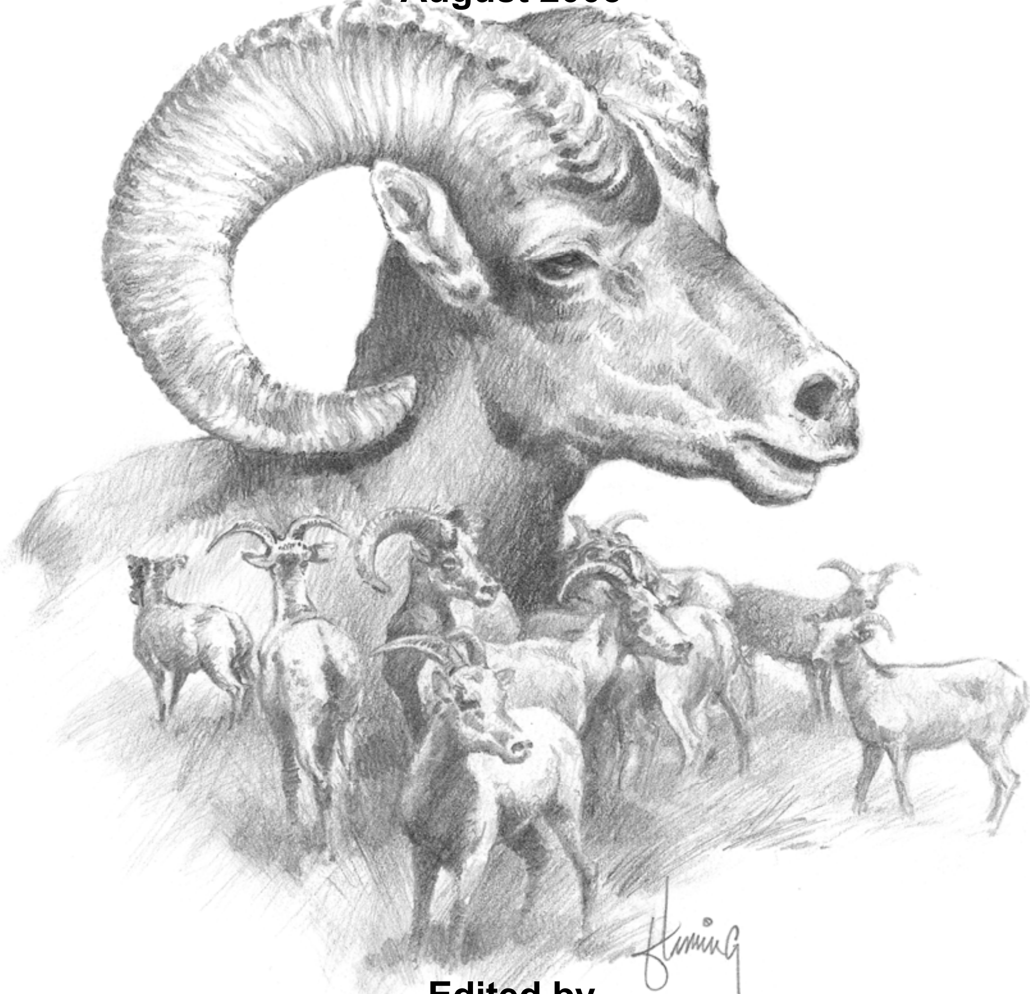

Edited by

James W. Cain III and Paul R. Krausman

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MANAGING WILDLIFE IN THE SOUTHWEST:
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Papers

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PREFACE

In 1990, The Southwest Section of The Wildlife Society published Managing Wildlife in the Southwest, which included papers presented at a symposium in Tucson, Arizona. Biologists and managers thought the symposium was successful for several reasons but an important key to success was the cooperative efforts of wildlifers from Arizona, New Mexico, Texas, and Mexico. Although these states and Mexico make up the Southwest Section of The Wildlife Society, biologists from the different areas rarely meet. Wildlifers from Arizona and New Mexico meet annually with occasional representation from Mexico but Texas holds their own meetings. The interactions that we envision Sections should have do not happen often enough in the Southwest. Thus, the officers of the Section (Paul R. Krausman and Jim Heffelfinger) decided another symposium with all states and countries was due. Over the past 30 years there have been few opportunities for wildlifers from the Section to interact; an unfortunate situation because of commonalities and unique offerings each area brings. Just as important as the papers is the interaction at the socials, dinners, and coffee breaks. Like the other symposiums, workshops, and meetings in the past this one was equally successful and we hope stimulated additional interest so officers (and members) in the future encourage more and additional ways for the Section to interact.

The Southwest Section has tremendous biodiversity, concerns, politics, and wildlife related issues and isolation hampers our ability to be productive. We have the potential to be the best section in the society. Bringing our talents together will ensure that we are.

PRK
JWC

FOREWORD

ACKNOWLEDGEMENTS

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GENETIC SUBSPECIES IDENTIFICATION OF A RECENTLY COLONIZED BIGHORN SHEEP POPULATION IN CENTRAL ARIZONA

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Abstract: Two subspecies of bighorn sheep currently occur in Arizona: the desert bighorn sheep (*Ovis canadensis mexicana, O. c. nelsoni*) and Rocky Mountain bighorn sheep (*O. c. canadensis*). In central Arizona (Game Management Unit 23 [GMU 23]), bighorn sheep colonized an area along the Salt River; however, the source of this population was enigmatic. Although the nearest desert bighorn sheep herd is <30 km to the southwest of the herd in Unit 23, no obvious movement corridors were evident between them. Rocky Mountain bighorn sheep from an earlier translocation occur about 160 km east of the herd in Unit 23, and these animals could have used the Salt River drainage as a movement corridor to colonize this new area. In an effort to clarify the subspecies affinity of bighorn sheep in the colonized area, we obtained mitochondrial DNA sequences (473-bp of the control region) from bighorn sheep in GMU 23 (n = 5), Rocky Mountain bighorn sheep as a reference (n = 8), and desert bighorn sheep references (n = 58). Our data provided strong support for the hypothesis that bighorn sheep in GMU 23 were of Rocky Mountain origin, suggesting that these sheep have moved about 160 km west along the Salt River drainage over the last 25 years. These data will facilitate effective management of this herd to minimize its impact on neighboring native desert bighorn sheep populations. The future growth of this population could jeopardize the integrity of subspecific classifications in central Arizona. Given documented long-distance movements of males, sheep populations (including translocation programs) should be managed to maintain subspecific separation.

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Key words: Arizona, bighorn sheep, colonization, mitochondrial DNA, *Ovis canadensis*, population genetics, subspecies.

Although the concept and application of subspecies is controversial (Mayr 1982, Ryder 1986, Moritz 1994, Paetkau 1999), there is no argument that different geographic forms of the same species exist as a result of adaptation to local environmental conditions. Bighorn sheep are no exception and were historically classified into 7 subspecies (Cowan 1940). Historically, subspecies descriptions were sometimes based on vague morphological characters measured for a few individuals. These subspecies names are then perpetuated for decades because of a

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lack of clarifying analyses. In recent years, more extensive morphological analyses and the advent of high-resolution genetic markers has led to a fuller understanding of phylogeographic differentiation in many species of large mammals (Cronin 1992, Lee et al. 1994, Cronin and Bleich 1995, Cronin et al. 1995, Lou 1998, Hundertmark et al. 2002, Williams et al. 2004, Stephen et al. 2005). In bighorn sheep, a more sophisticated analysis of skull morphology combined with genetic techniques revealed subspecies classifications different from Cowan's (1940) analysis (*Ovis canadensis auduboni* is extinct and thus excluded from analysis; Ramey 1993). Ramey's (1993) analyses did not support the recognition of separate desert bighorn sheep subspecies in the Southwest (i.e., *O. c. nelsoni, O. c. cremnobates, O. c. mexicana, and O. c. weemsi*; Ramey 1993), and only weakly supported differentiation between this desert complex and *O. c. californiana* in the Sierra Nevada. However, this same analysis found pronounced differences between desert bighorn sheep subspecies (collectively) and the Rocky Mountain bighorn sheep (Ramey 1993).

A common management tool for bighorn sheep and other game species is translocation, either for restoration or augmentation of populations or for increased variety in hunting opportunities. Often, these translocations have mixed subspecies that traditionally were separated, creating the opportunity for hybridization to occur. Biologically, hybridization may result in the loss of unique genetic, morphological, behavioral, or ecological characteristics that have evolved in local populations over time. Groups of genes that have evolved to work together (i.e., locally adapted gene complexes) may be disrupted, leaving hybrid populations poorly adapted to local environments (Dobzhansky 1970), and potentially leading to extinction of naturally-occurring types (Rhymer and Simberloff 1996). The administrative implications of hybridization also are critical, particularly when dealing with game species. Management recommendations, hunting regulations and record-keeping, and hunter enthusiasm are often subspecies-specific, and will be seriously confounded if populations are composed of hybrid individuals or individuals of unknown subspecies affinity.

Two subspecies of bighorn sheep currently reside in Arizona: the desert bighorn sheep and the Rocky Mountain bighorn sheep (Cowan 1940). Rocky Mountain bighorn sheep from Alberta were released in New Mexico near Arizona in 1971 and currently occupy areas in east-central Arizona (Hoffmeister 1986, Heffelfinger et al. 1995). Desert bighorn sheep occur in scattered populations throughout the southern and western halves of Arizona (Fig. 1). Over the last decade, groups of bighorn sheep have been reported periodically along the Salt River Canyon in the southern portion of GMU 23 (Fig. 1). Some observers concluded that these sheep appear more like Rocky Mountain bighorn than desert bighorn with heavier musculature, larger bodies, and darker pelage (Fig. 2). However, if the sheep in this newly colonized area were of Rocky Mountain origin, individuals would have had to utilize the Salt River drainage as a movement corridor from the nearest source population over 160 km to the east. The nearest desert bighorn sheep herd is <30 km to the southwest, however, no evident movement corridor exists between these herds.

The mixing of genetic stock from Alberta with endemic Arizona desert bighorn sheep has legal and administrative repercussions, and could have negative biological consequences for the resulting population. Our objective was to use available genetic tools to determine subspecific affinity of the sheep in the southern part of GMU 23. With such information, we can minimize potentially detrimental interbreeding between distinct subspecies of bighorn sheep in Arizona, and design management strategies to maximize hunting opportunities in the state.

METHODS

In December 2004, we captured and radiocollared 4 bighorn sheep in the Black Mesa area of southern GMU 23 to monitor their movements, survival, and habitat use. We took blood samples from these 4 individuals and collected muscle tissue from a fifth sheep that died during attempts to capture it. These 5 sheep were compared to bighorn sheep of known subspecies affiliation from throughout Arizona. The reference collection (58 desert and 8 Rocky Mountain bighorn sheep) came from known subspecies from checking stations operated by the

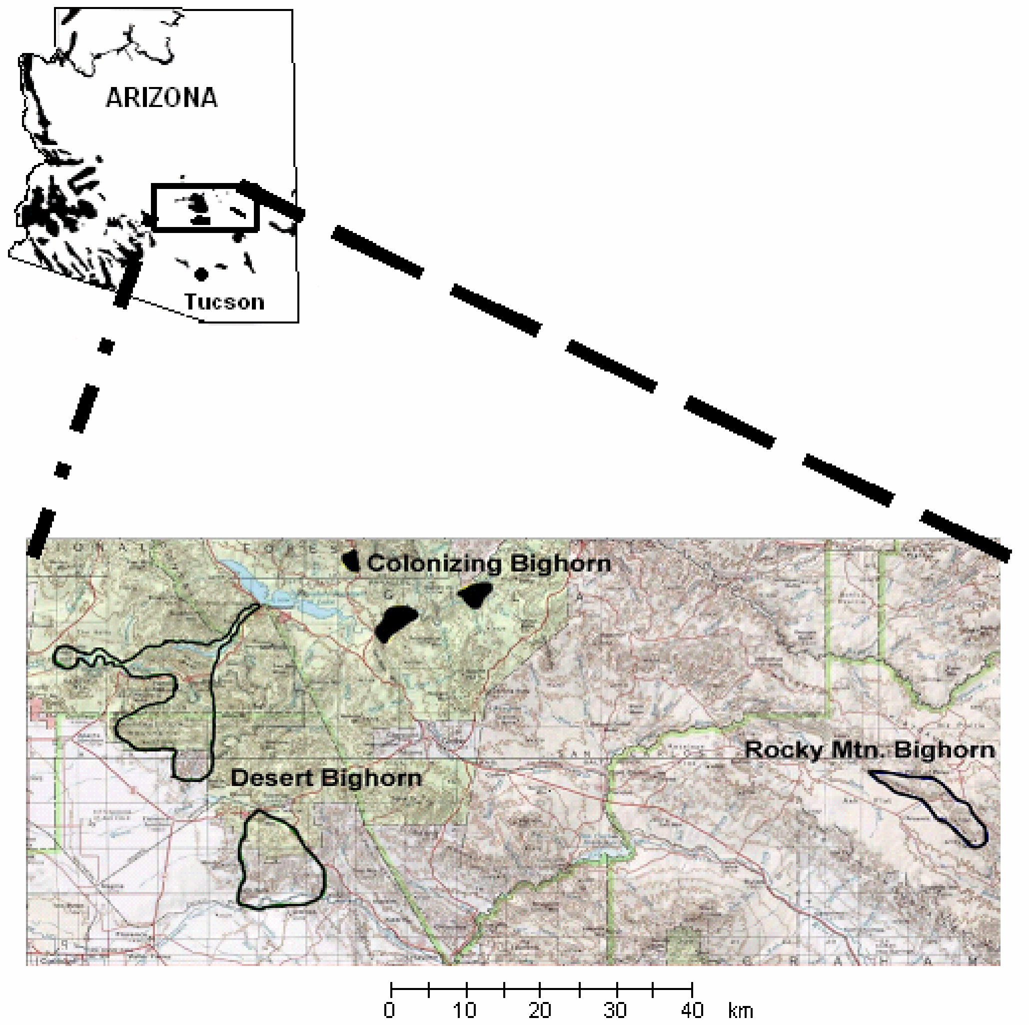

Fig.1 Location of desert bighorn and Rocky Mountain bighorn sheep populations in relation to the recently colonized area in the southern portion of Game Management Unit 23, central Arizona, 2004

Arizona Game and Fish Department. The reference desert bighorn sheep samples represented most populations in the western and southern half of Arizona (Fig. 1). Rocky Mountain bighorn samples were collected in the same manner from GMU 27 and 28 in east central Arizona.

To prepare blood samples for DNA extraction, we added 900 µL of 20 mM Tris-HCl to each sample, mixed by vortexing, allowed to sit at room temperature for 10 minutes, then centrifuged at 14,000 rpm for 20 seconds. We repeated this procedure 2 additional times using the pellet from the previous spin to ensure removal of most of the red blood cells, which in mammals do not contain DNA. For tissue samples and prepared blood samples, we extracted DNA using a modified sodium acetate precipitation protocol (modified from the PUREGENE kit; Gentra Systems, Minneapolis, Minnesota). We assessed the quantity and quality of extracted DNA via electrophoresis through an agarose gel stained with ethidium bromide, and diluted each sample to approximately 10 ng/µL in TLE (10 mM Tris-HCl, 0.1 mM EDTA).

We amplified a 473 base pair portion of the mitochondrial control region using PCR primers from Epps et al. (2005*a*, *b*). We gener-

Fig. 2. Ram showing phenotypic resemblance to Rocky Mountain bighorn sheep (right) seen in a desert bighorn sheep population (desert bighorn ram on left; GMU 22) <30 km southwest of the colonizing Rocky Mountain bighorn sheep in central Arizona, October 2004.

ated amplicons using the following PCR thermocycler profile: an initial denaturation step of 5 minutes at 94° C, followed by 35 cycles of 94° C for 60 seconds, 61° C for 70 seconds, and 72° C for 90 seconds, and a final extension step at 72° C for 5 minutes. We estimated the quality and relative quantity of PCR products by electrophoresis through agarose gels stained with ethidium bromide. We cleaned PCR products using a low sodium precipitation protocol, in which we precipitated the DNA with a sodium acetate solution (0.12 mM NaOAc in 100% ethanol), centrifuged to form a pellet, washed with 70% ethanol, and resuspended in water.

Ten microliter sequencing reactions contained approximately 30 ng PCR product (as determined by agarose gel electrophoresis), 5 pmol forward or reverse primer, and 1 µL ABI Big Dye Terminator version 3.1 cut with 3 µL 5X buffer (Applied Biosystems, Foster City, California, USA). Sequencing reactions were carried out as follows: 98° C for 5 minutes, followed by 26 cycles of 98° C for 30 seconds, 50° C for 15 seconds, and 60° C for 2 minutes. We cleaned sequenced products using the low sodium precipitation protocol described above, and the Purdue University Core Genomics Center ran these products on an ABI 3730 automated DNA sequencer. We edited sequences using Sequencher version 4.1 software (Gene Codes Corp., Ann Arbor, Michigan, USA). We sequenced all individuals in the forward and reverse directions to ensure consistency.

We combined forward and reverse sequences together to make a consensus sequence for each individual using Sequencher version 4.1 and exported these consensus sequences into PAUP* version 4.0b10 software (Swofford 2000). We performed a heuristic search for the most parsimonious phylogenetic tree that best described the relationships among our sequences. We then computed a consensus tree to collapse any nonsignificant branch nodes, and generated confidence values for branch nodes using 100 bootstrap replicates. Thus, our final bootstrapped consensus tree describes the relationships among individual control region sequences in our dataset. Each individual is represented by a horizontal 'branch,' all of which are the same length. Individuals that are connected by a 'node' (represented by a vertical line) are genetically more similar to each other than they are to other individuals to which they are not connected. Bootstrap numbers represent the confidence in that particular branching pattern; higher values indicate that the data provide stronger support for the given branching pattern than lower values.

RESULTS

We aligned 473 bases of mitochondrial control region sequence across 71 individual bighorn sheep (58 desert, 8 Rocky Mountain, and 5 unknown). Forty nucleotide sites were variable within this portion of the mtDNA, resulting in the detection of 18 distinct haplotypes. We identified 14 haplotypes in desert bighorn sheep and 4 haplotypes in Rocky Mountain sheep (Fig. 3). The discrepancy in the numbers of haplotypes may not reflect a lack of genetic diversity in Rocky Mountain bighorn sheep, but may be an artifact of the small sample size for this subspecies. None of the 18 haplotypes were shared between the 2 subspecies (Fig. 3). We identified 6 bases within the mtDNA sequence that were diagnostic between subspecies; in other words, these sites did not vary within subspecies, only between them. All colonizing sheep had haplotypes that were identical to one of the 4 haplotypes found in Rocky Mountain bighorn sheep (Fig. 3), suggesting that the colonizing sheep were of Rocky Mountain origin.

The phylogenetic tree provided additional support for the hypothesis that the colonizing sheep in Unit 23 were of Rocky Mountain origin. We found that Rocky Mountain sheep and the colonizing sheep clustered together with strong bootstrap support, and that these sheep were genetically differentiated from all desert bighorn sheep (Fig. 4). We found evidence for genetic structuring within subspecies, as indicated by bootstrap-supported branching within subspecies; however, in nearly every instance this was uncorrelated to geographic location (Fig. 4).

DISCUSSION

Given the geographic distribution of subspecies in Arizona and the results of our analysis, Rocky Mountain bighorn sheep apparently moved westward along the Salt River drainage and into the southern part of GMU 23 during the last 25 years. Perhaps these results should not be surprising in light of the history of Rocky Mountain bighorn sheep in Arizona. The Rocky Mountain subspecies colonized eastern Arizona by movements west from a translocated bighorn sheep population in New Mexico (Heffelfinger et al. 1995). This Rocky Mountain bighorn sheep population in New Mexico was established near the Arizona border with a 1964 translocation of animals from Banff National Park, Alberta and a supplemental release of sheep that previously originated from Banff (Larsen 1971, Ogren 1957). As animals moved west from this population along the San Francisco River they entered Arizona as early as 1971 (Apache County Independent News 1971). The Arizona Game and Fish Department supplemented Rocky Mountain bighorn sheep with 1979 and 1980 translocations into Bush Creek in east-central Arizona (Heffelfinger et al. 1995). The sheep in Bush Creek came from Rocky Mountain National Park (2M:6F) and near Tarryall (5M:7F), Colorado.

The Rocky Mountain bighorn sheep now occupying southern GMU 23 are geographically close to native desert bighorn herds. Marked desert bighorn sheep in Arizona may travel distances of several hundred km; 1 bighorn sheep traveled 110 km from the Superstition Mountains east of Phoenix to the Catalinas near Tucson (Arizona Game and Fish Department,

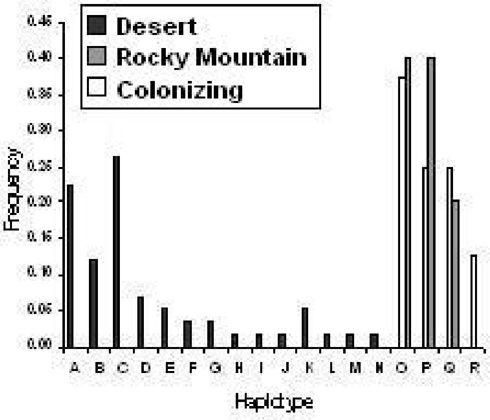

Figure 3. Mitochondrial DNA haplotype frequency distribution for desert ($n = 58$), Rocky Mountain ($n = 8$), and colonizing Game Management Unit 23 ($n = 5$) bighorn sheep, based on 473 bases of control region sequence.

unpublished data). Currently the Rocky Mountain sheep in the southern portion of GMU 23 are < 30 km to the northeast of the nearest native bighorn sheep population. The landscape between these 2 subspecies is not conducive to sheep movements, but the 110 km-movement mentioned above occurred through similarly inhospitable terrain.

Our data illustrate the potential for intermixing of these 2 subspecies in central Arizona. Because our data are mitochondrial in origin, at this point we know only that each of the colonizing bighorn sheep we sampled had a Rocky Mountain sheep mother. Since we found no desert sheep haplotypes in our unknown sample, it seems unlikely that there are desert bighorn sheep (or Rocky Mountain bighorn male x desert bighorn female hybrids) in GMU 23 at this time. If desert bighorn males are in GMU 23 but were unsampled, it is possible that they are hybridizing with Rocky Mountain females; such hybrids would not be detectable using our methods (they would have a Rocky Mountain haplotype). Although it seems unlikely that introgression is occurring within the newly colonized population in GMU 23 at this time,

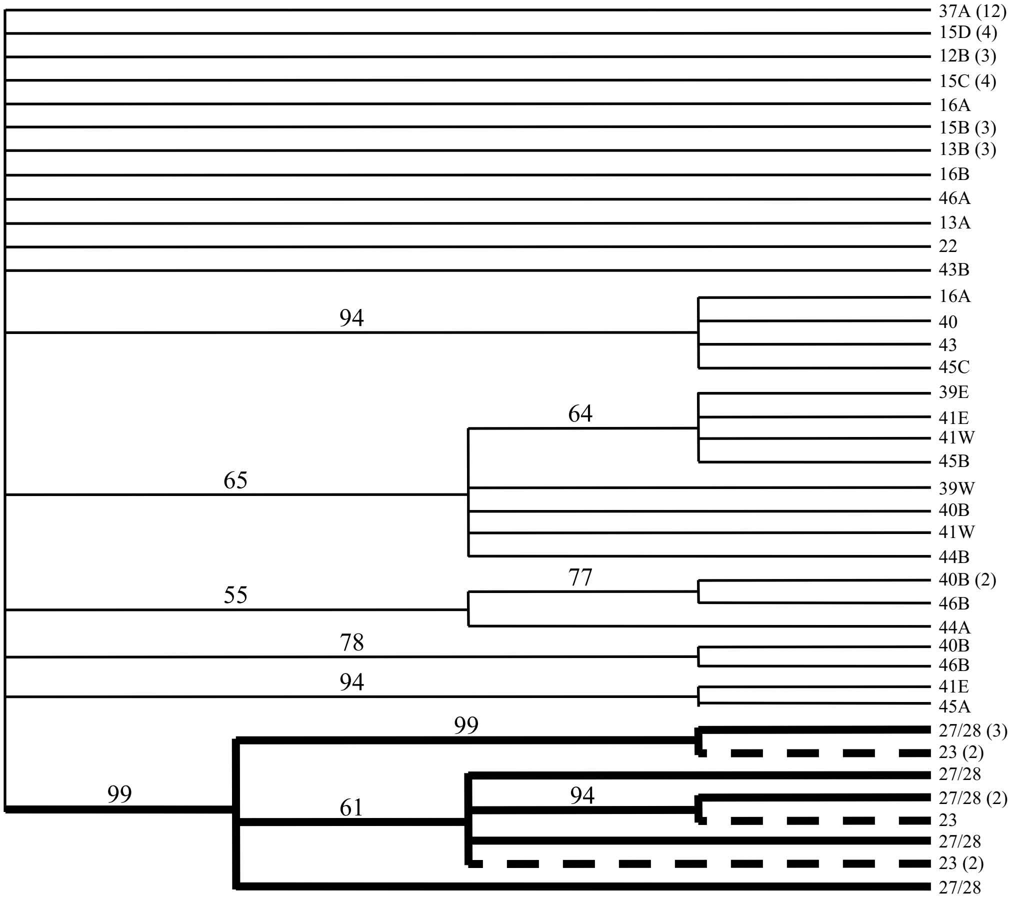

Fig. 4. Maximum parsimony consensus tree depicting genetic relationships among desert (thin lines), Rocky Mountain (bold lines), and colonizing (dashed lines) bighorn sheep. Individuals are identified by the Arizona Game Management Unit from which they were sampled. Individuals possessing the same haplotype and sampled in the same management unit were collapsed into a single branch; the number of individuals represented by each branch is indicated in parentheses. Bootstrap values, based on 100 replicates as implemented in PAUP*, are indicated at nodes.

several observations of a phenotypically Rocky Mountain ram in nearby GMU 22 suggest that Rocky Mountain bighorn sheep may be moving beyond the boundaries of the newly colonized population. This ram has darker pelage and heavier musculature than any of the other rams observed in this population (J. Heffelfinger, personal observation; Fig. 2). Additionally, a few of the sheep radio-collared on Black Mesa north of the Salt River have crossed to the south side of the river near Klondike Butte, showing that the river may not completely prohibit movements. Previous to this study, bighorn sheep have been reported periodically on Klondike Butte (J. Heffelfinger, personal observation).

MANAGEMENT IMPLICATIONS

Diagnosing an introgressed population of mixed subspecies may not be straight-forward. For example, because males are more prone to long-distance, exploratory movements (Monson and Sumner 1980), they are the most likely to move to a nearby population of a different subspecies. Offspring of a Rocky Mountain bighorn male in an otherwise desert bighorn population will all carry desert bighorn sheep mtDNA and would not be detectable with the methods used here. There are microsatellite loci available for bighorn sheep that would be informative and allow managers to diagnose mixed populations by looking at nuclear DNA (Epps et al. 2005*b*).

Given the documented movements of bighorn sheep in the past, the colonization of Rocky Mountain bighorn sheep we documented could jeopardize the subspecific integrity of bighorn sheep in central Arizona. This is potentially problematic from a biological and administrative perspective. From a biological standpoint, the size difference between the 2 subspecies (Rocky Mountain sheep can be 20-25% larger than desert sheep; J. Heffelfinger, personal observation) could cause reproductive problems such as dystocia, as has been documented in white-tailed deer (*Odocoileus virginianus*, Galindo-Leal and Weber 1994). It is possible that larger Rocky Mountain bighorn sheep males impregnating smaller desert females could result in prepartum lambs that are too large for successful birthing.

There also are several administrative issues. First, hunters in Arizona are allowed to harvest only 1 Rocky Mountain and 1 desert bighorn sheep in a lifetime. A population of sheep that is known or suspected to be a mixture of these 2 subspecies obviously presents an administrative problem for managers and hunters. In a mixed population, it would be necessary to administratively designate which subspecies was represented by the animals harvested from that population. Additionally, some organizations, such as the Boone and Crockett Club, keep records of hunter-harvested animals and have different record-keeping categories for desert and Rocky Mountain bighorn sheep. A mixed population renders any animals taken from that population ineligible for entry or would have to be entered in the larger Rocky Mountain category regardless of outward appearance.

There is considerable interest in the sheep hunting community in collecting a mature specimen from each of the 4 major categories of mountain sheep: Dall's sheep (*Ovis dalli*), Stone's sheep (*O. d. stonei*), desert bighorn, and Rocky Mountain bighorn. A population of compromised subspecific integrity obviously has social and biological implications. Desert bighorn sheep are not as widely distributed or abundant as Rocky Mountain bighorns. Hunting opportunities for desert sheep are quite limited as compared to the other 3 recognized forms of wild mountain sheep. Managers must keep this in mind when dealing with Rocky Mountain bighorn sheep expanding beyond their natural range and into historic desert sheep range.

Once a population becomes a mixture of subspecies, the situation cannot be reversed without depopulation and re-establishment. Because of this, it is imperative that managers consider the consequences of natural movements and use translocation to lessen, rather than hasten, the occurrence of intermingling. Managers should retain geographic buffers between bighorn sheep subspecies.

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HABITAT MODELS: DESERT BIGHORN SHEEP IN THE SILVER BELL MOUNTAINS REVISITED

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Abstract: The population of desert bighorn sheep (*Ovis canadensis*) in the Silver Bell Mountains, Arizona, was studied from 1993 to 1995 to determine habitat use, habitat quality, and potential effect of mine expansion at the Silver Bell Mine (Asarco Limited Liability Corporation and Mitsui and Company Limited). We returned to the Silver Bell Mountains to study current bighorn sheep habitat use, reevaluate the previous habitat models, and create new habitat models for comparison. We radiocollared 16 bighorn sheep from 2003 to 2005 and used locations from those animals to test the previous habitat quality models. Past habitat quality models did not perform the same with new location data as they had with past data, but some models did accurately identify areas used greater than available as being good or excellent habitat. We created Classification and Regression Tree (CART) models for female and male bighorn sheep that correctly identified 80 and 90% of locations, respectively. The CART model required fewer variables and less time to create and evaluate and was less subjective, more accurate, and easier to update than the previous ratings-based model. We recommend that if animal location data become available in an area where a habitat rating system was used previously, the habitat quality model should be reevaluated and revised.

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Key words: Arizona, CART, Classification and Regression Tree, desert bighorn sheep, mining, *Ovis canadensis*.

Many habitat quality rating models have been created for desert bighorn sheep to evaluate habitat, select areas for translocation, and determine impact of changes in land use (Hansen 1990, McCarty and Bailey 1994, Dunn 1996). The population of desert bighorn sheep residing in the Silver Bell Mountains, Arizona, was studied from 1993 to 1995 when Bristow et al. (1996) evaluated habitat use, rated habitat quality with a scoring model, and evaluated potential effect of mine expansion at the Silver Bell Mine. Since that study, the mine has re-opened and another open-pit entered production. As the last endemic population of bighorn sheep in south-central Arizona, the bighorn sheep population in the Silver Bell Mountains is an important population for bighorn management (Krausman and Leopold 1986). As development continues around the Silver Bell Mountains, the Ironwood Forest National Monument has formed, and mine operations proceed, it is important that we accurately identify areas used by bighorn sheep to aid in management decisions. We returned to the Silver Bell Mountains to study current bighorn sheep habitat use, re-evaluate the previous habitat model, and create new habitat models for comparison with models created in 1996.

STUDY AREA

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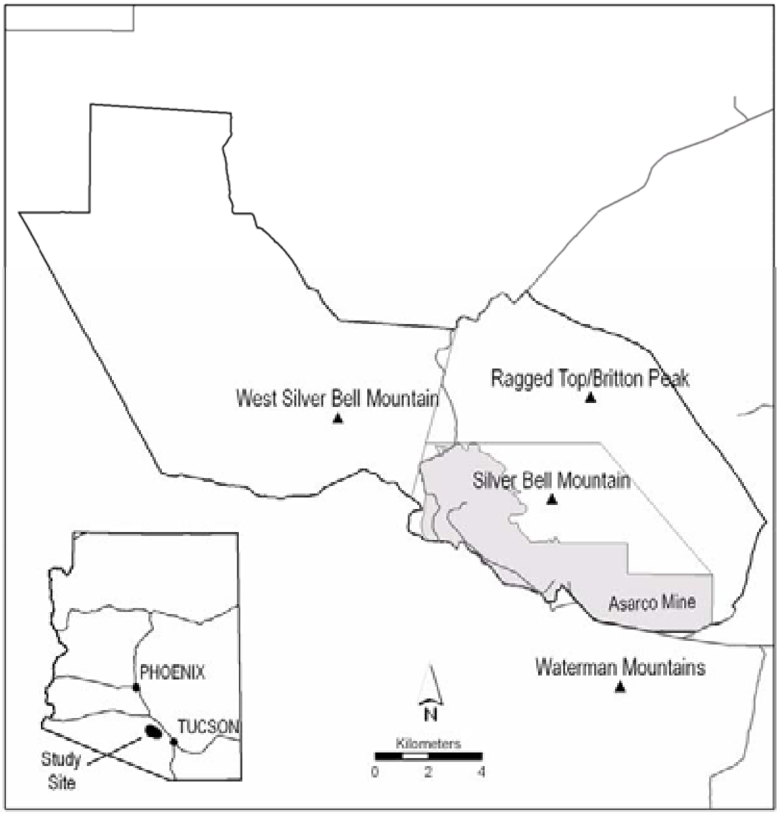

Fig. 1. Silver Bell Mountains study area, Arizona, USA, including the 3 subunits defined by Bristow et al. (1996): West Silver Bells, Ragged Top/Britton Peak, and Silver Bell Peak) and the Waterman Mountains. The majority of the Asarco Silver Bell Mine occurred in the Silver Bell Peak subunit.

The Silver Bell Mountains (111º 30' W, 32º 24' N) are located 64 km northwest of Tucson, Arizona. The distribution of bighorn sheep included the Silver Bell Mine and portions of Ironwood Forest National Monument and the Tohono O'odham Indian Reservation. Our study area included the 3 subunits identified by Bristow et al. (1996): Silver Bell Peak, Ragged Top/Britton Peak, and West Silver Bells and the Waterman Mountains (Fig. 1). Average minimum temperature (Tucson 17 NW station; Western Regional Climate Center 1982–2004) ranged from 4.4°C in December to 23.8°C in July. Average maximum temperature ranged from 18.6°C in December to 38.4°C in July. Average annual precipitation was 30.0 cm/year (range = 14.0–57.3 cm).

The Silver Bell Mountains are within the Sonoran Desert section of the Basin and Range physiographic province, an area characterized by block-faulted mountains and wide alluvial valleys (Chronic 1983). The dominant biome was the Arizona Upland subdivision of Sonoran desertscrub (Turner and Brown 1994). Common tree species in the Arizona Upland subdivision included palo verde (*Parkinsonia* spp.), ironwood (*Olneya tesota*), and catclaw acacia (*Acacia greggii*). Shrub species included bursage (*Ambrosia* spp.) and creosotebush (*Larrea tridentata*). Numerous species of cacti were present, including buckhorn cholla (*Opuntia acanthocarpa*), teddy bear cholla (*O. bigelovii*), saguaro (*Carnegiea gigantea*), and barrel cactus (*Ferocactus* spp.; Turner and Brown 1994). Plant nomenclature follows the Integrated Taxonomic Information System (2005).

Water developments were present on

Silver Bell Peak and Ragged Top. Additional water could often be found on Silver Bell Mine in ponds meant to prevent water from running off the mine. Mining has occurred in the Silver Bell Mountains for >100 years; the Silver Bell Mine has been present since 1954. Active cattle grazing allotments abut the mine.

METHODS

Bighorn Sheep Locations

We captured and radiocollared bighorn sheep in the Silver Bell Mountains through separate efforts in 2003 and 2004 with a net-gun fired from a helicopter (Krausman et al. 1985). We fitted bighorn sheep with radiocollars (Telonics, Mesa, Arizona, USA) that had store-on-board Global Positioning System (GPS) units or GPS units with ARGOS (Service Argos, Largo, Maryland, USA) satellite uplink. All radiocollars attempted to collect a GPS fix every 5 hours. We retrieved radiocollars after mortality and recaptured remaining study animals in February 2005 to remove radiocollars. We downloaded data from radiocollars into ARC/INFO® (Environmental Systems Research Institute, Redlands, California, USA) for all Geographic Information System (GIS) analyses.

Evaluation of Previous Habitat Quality Model

Bristow et al. (1996) created 6 different models, 1 including a human use factor (Cunningham 1989; low, medium, or high density human use or economic potential) and 1 without for males and females, in 3 different subunits of the Silver Bell Mountains: West Silver Bells, Ragged Top/Britton Peak, and the Silver Bell Mine. The models created by Bristow et al. (1996) were based on a technique developed by Cunningham (1989) and modified by Ebert and Douglas (1994). Bristow et al. (1996) assigned scores for vegetation, precipitation, water availability, human use, and slope class to 8,100 m^2 cells on the study area and score totals were used to classify habitat into 4 classes: poor, fair, good, and excellent. We evaluated the fit of models created by Bristow et al. (1996) in each subunit with bighorn sheep locations collected during the current study. For all bighorn sheep locations located within the study area defined by Bristow et al. (1996), we used GIS coverages of the habitat quality models created by Bristow et al. (1996) to classify bighorn sheep locations in 1 of 4 habitat quality classes (i.e., excellent, good, fair, and poor) and ran chi-square goodness of fit tests (Neu et al. 1974, Byers et al. 1984) to determine if use of the habitat quality classes differed from availability in each subunit. When we found evidence of use different than availability ($P < 0.05$), we calculated Bonferroni confidence intervals to determine if areas were selected (i.e., use > availability), avoided (i.e., use < availability), or used neutrally (i.e., use = availability).

We further evaluated models from Bristow et al. (1996) by delineating the 4 habitat quality classes into habitat and non-habitat categories and determining the number of bighorn sheep locations collected by Bristow et al. (1996) and during our study that were located in areas identified as habitat by the models. For the first evaluation, we delineated "excellent" and "good" categories as habitat and "fair" and "poor" as non-habitat (similar to what has been done when evaluating logistic regression models). For the second evaluation, we delineated "excellent," "good," and "fair" categories as habitat and "poor" as non-habitat.

New Model Development, Validation, and Application

Our modeling involved model creation, validation, application, and further testing. First, we delineated used and unused areas from bighorn sheep locations collected during our study, applied values of explanatory variables from GIS coverages to used and unused points, randomly separated our dataset into 2 halves, and ran CART (Breiman et al. 1984) analyses to model used and unused areas with half of the data. Second, we validated models with the second half of the used and unused points. Third, we mapped results to predict potential habitat on the evaluation area. Fourth, we tested our model with bighorn sheep locations collected from 1993 to 1995 by Bristow et al. (1996).

Delineation of Used Areas.–We used 2- and 3-dimensional GPS bighorn sheep locations in our analyses. We removed locations from the first week post-capture because some animals were observed to make temporary (<1 week) movements away from areas where they were captured (B. D. Jansen, University of Arizona, personal communication). We also removed mortality locations because the ultimate location of a radiocollar could have been altered by attempted escape during predation and subsequent scavenging and, thus, may not have reflected habitat use. During the course of our study, the population of bighorn sheep experienced an outbreak of keratoconjunctivitis that temporarily (<6 weeks) blinded 5 (2 F, 3 M) of the radiocollared bighorn sheep. For analyses, we removed all locations from periods when bighorn were blind, and 1 week before and after blindness. We used all remaining locations to create separate 100% minimum convex polygons (MCP) for males and females, which we considered to be the areas available to our study animals. We used the 100% MCP because we were studying 1 intermingling population of sheep, not disjunct populations, and we wanted to identify the overall area available to the bighorn sheep. Because the 100% MCP used points at the outer edge of the home range to delineate the area used, did not leave any bighorn sheep locations out of the area, and contained no areas that we believed to be otherwise inaccessible to bighorn sheep, we used it to identify the available area. The MCP was superior to the fixed kernel for this purpose because it was less stringent and thus encompassed greater variations in the explanatory variables for the area identified as available.

Our objective was to evaluate population-level habitat use, not use by individual animals, so we pooled locations from all radiocollared bighorn sheep together for analyses (Manly et. al 2002). However, combination of locations from multiple individuals, particularly of a herding species, can result in pseudoreplication due to lack of independence between locations of individuals (Hurlbert 1984). We did not test the spatio-temporal association of individual radiocollared sheep and concede that some sheep, particularly females, could have occurred within the same subgroup.

Delineation of Unused Areas.–We buffered bighorn sheep locations by 50 m to ensure that the "used" areas included all actual locations but limited inclusion of areas potentially unused by bighorn sheep. We assumed all areas within the 100% MCP but outside buffered used points to be unused by bighorn sheep. We generated random points within the unused areas equal in number to bighorn sheep locations for model development for males and females. Additionally, we used the random unused points to test the habitat quality model created by Bristow et al. (1996).

Explanatory Variable Identification and Assignment.–We used available GIS coverages to determine values of 7 explanatory variables (i.e., elevation, slope, aspect, land cover vegetation, soil association, mine presence, and presence of escape terrain) at bighorn sheep locations and unused points (Table 1). We included slope as a continuous explanatory variable because bighorn sheep in the Silver Bell Mountains previously used slopes >20% (Bristow et al. 1996) and other studies of bighorn sheep have found preferential use of higher slopes (Cunningham 1982, Berger 1991, Dunn 1996, Alvarez-Cárdenas et al. 2001). Though elevation is a highly site-specific variable, we included it as a continuous explanatory variable because we were developing a model specific to the Silver Bell Mountains and elevation is directly related to local topography. We considered aspect to be biologically meaningful because of rainshadow effects (Ingram 2000), topographic thermal cover, and aspect-specific frost deposition that can influence vegetation growth (Turner and Brown 1994). We coded aspect into 9 different categories: north (336.6–21.5°), northeast (21.6–66.5°), east (66.6-111.5°), southeast (111.6–156.5°), south (156.6–201.5°), southwest (201.6–246.5°), west (246.6–291.5°), northwest (291.6–336.5°), and flat (0 slope). We included land cover vegetation as a categorical variable because vegetation is important as forage, affects visibility and thermal cover, and has been used in other modeling efforts (Etchberger et al. 1989, Hansen 1990, Dunn 1996). The land cover vegetation map we used, which was based on ground-truthed remotely sensed data, may have had some inaccuracies in our study area. Particularly, we noted Madrean Pinyon (*Pinus* spp.)-Juniper (*Juniperus* spp.) Woodland and Mogollon chaparral that may have been misidentified as being present in the Silver Bell

Table 1. GIS coverages used in development of habitat models for bighorn sheep (2003–2005), Silver Bell Mountains, Arizona, USA.

GIS coverage	Source, scale	Derived attribute(s)
30-m Digital Elevation Model (DEM)	United States Geological Survey 1:100,000	Slope, (%), elevation (m) and aspect (N, NW, W, SW, S, SE, E, NE, and flat)
STATSGO State Soil Geographic Database	USDA Natural Resources Conservation Service, 1:250,000	General soil association, identified by map unit ID (muid)
Southwest Regional GAP Land cover	Southwest Regional Gap Analysis Project, 1:100,000[a]	Land cover vegetation type
Escape Terrain	Arizona Game and Fish Department[b], derived from 30-m DEM, 1:100,000	Escape terrain binary vari able indicating if it was or was not escape terrain if it was or was not escape terrain
Mine Boundary	Silver Bell Mine, 1:100,000	Binary variable indicating if a location was on or off the mine

[a] United States Geological Survey National Gap Analysis Program 2004.

[b] McKinney et al. 2003.

Mountains. We accepted these possible inaccuracies into our modeling process because they affected very small portions of the total study area (0.17% of total area). We included soil association in our variable set because soil attributes can affect moisture retention and vegetation growth (McAuliffe 2000) that could influence habitat use by bighorn sheep. We used mine presence as a model variable because open-pit mining can be beneficial to bighorn sheep (MacCullum and Geist 1992) and female bighorn have been found to regularly use the Silver Bell Mine (Bristow et al. 1996). We coded mine presence as a binary variable with bighorn sheep classified as being on or off mine property. Open-pits constituted only a small portion of the total mine property; however, we used mine boundary to code our mine presence variable because there are many activities throughout the mine (e.g., traffic, leach dumps, administrative buildings, and repair shops) that could affect habitat use. McKinney et al. (2003) previously modeled presence of escape terrain (slope ≥60% plus any area within a 150-m contiguous buffer that had slope of 40–60%), which we coded as a binary presence or absence variable. Due to regular mining activities and resulting changes in topography and land cover, some areas on mine property were not accurately portrayed by the land cover or digital evaluation model (DEM). Since daily mine changes would be difficult to track, we accepted that the assignment of variable cases for some bighorn sheep locations on the mine would represent past landscape features (e.g., presence of a particular land cover) rather than current landscape features (e.g., open, cleared area).

To reduce the number of variables and simplify modeling, we sorted the 10 land vegetation cover types into 3 use categories. We used chi-square goodness-of-fit tests (Neu et al. 1974, Byers et al. 1984) to compare observed versus expected use of 10 land cover vegetation types available in the MCP of males and females. We compared proportions of bighorn sheep observations among vegetation types with percent of total area of each vegetation types within male and female bighorn sheep 100% MCPs. We calculated simultaneous 95% Bonferroni confidence intervals to infer selection or avoidance when we found a statistically significant difference ($P < 0.05$) between expected and observed use of individual vegetation types. Based upon results of chi-square goodness-of-fit tests and Bonferroni confidence intervals, we placed the 10 vegetation types into selected, avoided, or neutral categories. We replaced vegetation type with vegetation category as a variable for model development.

Modeling.–We used Microsoft Excel (Microsoft Incorporated, Redmond, Washington, USA) to assign random binary numbers to bighorn sheep locations and unused points. We used these random numbers to split bighorn sheep locations and unused points into approximately equal groups, creating learning and test subsets for model development. We used the Statistica 6.1 (StatSoft Incorporated, Tulsa, Oklahoma) implementation of CART (Breiman et al. 1984) to create habitat models.

We used CART because it is relatively free of assumptions, has been used successully in habitat modeling (Andersen et al. 2000, De' ath and Fabricius 2000, Debeljak et al. 2001, McGrath et al. 2003), and produces decision trees that are readily applied in a management context. The CART models consist of a decision tree with binary (i.e., yes-no) splits based upon specific values of predictor variables. Decision pathways originate from a starting node that contains all observations and end at multiple terminal nodes containing unique subsets of observations. Terminal nodes are assigned a final outcome based on group membership of the majority of observations (i.e., either "used" or "unused"). Our analysis used the Gini goodness-of-fit measure, estimated prior probabilities of group membership from proportions in the learning dataset, and specified equal misclassification costs for used and unused predictions. We used P-value = 0.05 for selection of the variables used to create binary splits. We pruned candidate trees using the FACT-style direct stopping method (Loh and Vanichestakul 1988), which required tree growth to end when nodes contained ≥5% of the learning sample. We cross-validated our CART models with the remaining test subset of locations. Additionally, we further tested models using bighorn sheep locations collected by Bristow et al. (1996). We used outputs from CART models to create GIS maps of habitat suitability (mapping unit = 900 m^2). We delineated model maps into habitat and non-habitat.

RESULTS

We radiocollared 16 bighorn sheep (6 F, 10 M) in the Silver Bell Mountain range with store-on-board Global Positioning System (GPS) units (3 F and 10 M) or GPS/ARGOS collars (3 F). We collected 9,321 locations of females and 14,180 locations of males from May 2003 to February 2005, after removing locations post-capture and during the disease outbreak.

We did not have sufficient locations to test habitat classifications in the West Silver Bell subunit. Female bighorn sheep used habitat quality classes identified by Bristow et al. (1996) disproportionately to availability in the Silver Bell Peak (χ_3^2 = 714.52, $P < 0.001$ for model including human use score; χ_3^2 = 1,069.23, $P < 0.001$ for model excluding human use score) and Ragged Top/Britton Peak (χ_3^2 = 18,649.11, $P < 0.001$ for model including human use score; χ_3^2 = 24,925.43, $P < 0.001$ for model excluding human use score) subunits. The habitat classification model excluding human use score fit the data best in the Ragged Top/Britton Peak subunit, whereas the model with the human use score fit best in the Silver Bell Peak subunit (Table 2).

Male bighorn sheep also did not use habitat quality classes in proportion to availability in the Silver Bell Peak (χ_3^2 = 5,497.45, $P < 0.001$ for model including the human use score; χ_3^2 = 5,335.98, $P < 0.001$ for model excluding the human use score) and Ragged Top/Britton Peak (χ_3^2 = 11,690.82, $P < 0.001$ for model including the human use score; χ_3^2 = 11,702.55, $P < 0.001$ for model excluding the human use score) subunits. Models with and without human use scores fit the data equally well for male bighorn habitat use in the Ragged Top/ Britton and Silver Bell Peak subunits (Table 3).

Bighorn sheep did not use land cover vegetation types in proportion to availability (F: χ_3^2 = 1,728.13, $P < 0.001$; M: χ_3^2 = 7,162.71, $P < 0.001$). Two vegetation types were selected, 3 were avoided, and 5 were used as expected by

Table 2. Bonferroni confidence intervals for use versus availability of habitat classifications by female bighorn sheep in the Ragged Top/Britton Peak and Silver Bell Peak subunits of the Silver Bell Mountains, Arizona, USA, 2003–2005.

Model Habitat classification[a]	% available	% use	95% CI
Ragged Top/Britton Peak			
With human use score			
Excellent	4.82	3.00	0.0236–0.0364 (–)[b]
Good	20.61	7.36	0.0637–0.0835 (–)
Fair	63.42	65.25	0.6345–0.6704 (+)
Poor	11.16	24.39	0.2277–0.2601 (+)
Without human use score			
Excellent	1.74	27.37	0.2569–0.2905 (+)
Good	10.64	49.45	0.4757–0.5134 (+)
Fair	78.23	23.18	0.2158–0.2477 (–)
Poor	9.39	0.00[c]	(–)
Silver Bell Peak			
With human use score			
Excellent	8.44	46.15	0.4437–0.4792 (+)
Good	12.23	40.29	0.3855–0.4204 (+)
Fair	46.99	13.54	0.1232–0.1475 (–)
Poor	32.34	<0.01[c]	0.0000–0.0008 (–)[d]
Without human use score			
Excellent	5.00	2.39	0.0185–0.0294 (–)
Good	27.01	45.80	0.4403–0.4758 (+)
Fair	53.31	34.37	0.3267–0.3606 (–)
Poor	14.68	17.44	0.1609–0.1879 (+)

[a] classifications from Bristow et al. (1996).
[b] (+) = selection and (–) = avoidance.
[c] <5 observations in the habitat classification.
[d] 0.0000 was used in place of a negative lower limit.

female bighorn sheep (Table 4). Five vegetation types were selected, 4 were avoided, and 1 was used as expected by male bighorn sheep (Table 5). We grouped vegetation types into 3 vegetation categories based upon the 3 levels of preference and used these categories in subsequent modeling. We included results for subunits with <5 bighorn sheep locations in 1 habitat classification to indicate trends, but results are more speculative than those with larger samples (Manly et al. 2002).

The final CART model for male bighorn sheep had 3 splits and 4 terminal nodes, 2 of which described areas used by bighorn sheep (Fig. 2 and 3). The model used 2 explanatory variables: slope and elevation. The model had an overall classification accuracy of 90%, with classification accuracy of 96% and 84% for used and unused points, respectively. Cross-validation with test data yielded an overall correct classification rate (CCR) of 90%, with classification accuracy of 95% for used locations and 84% for unused points. Seventy-seven percent of male bighorn sheep locations from Bristow et al. (1996; *n* = 503) were located in areas designated as habitat by the CART model.

The final CART model for female bighorn sheep had 10 splits and 11 terminal nodes, 6 of which described areas used by bighorn sheep (Fig. 4 and 5). The model used 4 explanatory variables: slope, elevation, aspect category, and land cover category. This model had an overall classification accuracy of 81%, with classification accuracy of 83% and 80% for used and unused points, respectively. Cross-validation with test data yielded overall correct classification of 80%, with classification ac-

Table 3. Bonferroni confidence intervals for use versus availability of habitat classifications by male bighorn sheep in the Ragged Top/Britton Peak and Silver Bell Peak subunits of the Silver Bell Mountain range, Arizona, USA, 2003–2005.

Model Habitat classification[a]	% available	% use	95% CI
Ragged Top/Britton Peak			
With human use score			
Excellent	4.82	52.18	0.4955–0.5483 (+)[b]
Good	20.61	33.08	0.3060–0.3557 (+)
Fair	63.42	14.73	0.1286–0.1660 (–)
Poor	11.16	0.00[c]	(–)
Without human use score			
Excellent	1.74	25.85	0.2354–0.2816 (+)
Good	10.64	49.46	0.4682–0.5211 (+)
Fair	78.23	24.69	0.2241–0.2697 (–)
Poor	9.39	0.00[c]	(–)
Silver Bell Peak			
With human use score			
Excellent	8.44	25.54	0.2437–0.2670 (+)
Good	12.23	24.24	0.2310–0.2538 (+)
Fair	46.99	40.05	0.3874–0.4136 (–)
Poor	32.34	10.17	0.0936–0.1098 (–)
Without human use score			
Excellent	5.00	11.95	0.1108–0.1281 (+)
Good	27.01	56.31	0.5499–0.5763 (+)
Fair	53.31	24.99	0.2284–0.2513 (–)
Poor	14.68	7.75	0.0704–0.0847 (–)

[a] classifications from Bristow et al. (1996).
[b] (+) = selection and (–) = avoidance.
[c] <5 observations in the habitat classification.

curacy of 82% for used locations and 78% for unused points. Testing with female bighorn sheep locations from Bristow et al. (1996; *n* = 614) yielded a CCR of 79%.

To further evaluate the habitat quality model created by Bristow et al. (1996), we first considered areas scored as "excellent" or "good" to be habitat and areas scored "fair" and "poor" to be non-habitat. With this delineation, the habitat model correctly identified 68% of the bighorn sheep locations collected by Bristow et al. (1996) and 52% of the locations collected in 2003–2005 as habitat, and 76% of the randomly generated unused locations as non-habitat (Table 6). When we considered areas scored as "excellent," "good," and "fair" to be habitat, the model correctly identified 96% of the locations collected by Bristow et al. (1996) and 90% of the locations collected in 2003–2005 as habitat, but only 15% of the unused locations as non-habitat.

DISCUSSION

Models we created do not imply causal inference between explanatory variables and bighorn sheep habitat use. It is possible that some explanatory variables we used were strongly correlated with other, unmeasured variables that were related to bighorn sheep habitat use. Regardless, CART models we developed had good classification rates of used and unused areas for male and female bighorn

Table 4. Bonferroni confidence intervals for use versus availability of land cover types by female bighorn sheep (n = 9,321 locations) within the 100% MCP in the Silver Bell Mountain range, Arizona, USA, 2003–2005.

Land cover	% available	% use	95% CI	Use category[a]
Madrean Pinyon-Juniper Woodland	0.08	0.05	0.000–0.001	3
Mesquite Upland Scrub	0.04	0.58	0.004–0.008	1
Mogollon Chaparral	0.09	0.08	0.000–0.002[b]	3
North American Warm Desert Riparian Mesquite Bosque	0.01	0.00[c]	–	2
Open Water (likely dry)	0.18	0.34	0.002-0.005	3
Recently Mined or Quarried	17.30	29.67	0.283–0.310	1
Sonoran-Mojave Creosote bush-White Bursage Desertscrub	0.85	0.87	0.006–0.011	3
Sonora-Mojave Desert Mixed Salt Desertscrub	0.01	0.00[c]	–	2
Sonoran Mid-Elevation Desertscrub	3.43	3.75	0.032–0.043	3
Sonoran Palo Verde-Mixed Cacti Desertscrub	77.99	64.65	0.633–0.660	2

[a] 1 = use > availability, 2 = use < availability, and 3 = neutral use.

[b] 0.0000 was used in place of a negative lower limit.

[c] <5 observations in the habitat classification.

sheep.

Our CART model suggested slope and elevation were the 2 variables that best described habitat use by male bighorn sheep in the Silver Bell Mountains. The CART model of female habitat use in the Silver Bell Mountains included aspect and land cover vegetation type along with slope and elevation. Both models can be mapped using statewide GIS coverages that are available at no cost, which makes it easy for managers to apply and update them as needed for the Silver Bell Mountains.

Bristow et al. (1996) found their habitat quality models excluding the human use scores fit better with bighorn locations they collected. We found that, based upon our new location data, this was no longer true for female bighorn sheep that used the Silver Bell Peak subunit or male bighorn sheep that used the Silver Bell Peak or Ragged Top/Britton Peak subunits. These changes may have occurred because human use scores now more appropriately reflect human use in the Silver Bell Peak subunit, where mining has increased since 1995. Human use is not known to have increased in the Ragged Top/Britton Peak subunit, which might likewise explain the superiority of the model excluding the human use score for female bighorn sheep.

When we sorted classifications of the habitat quality model created by Bristow et al. al. (1996) into habitat and non-habitat, the habitat quality model had a lower CCR than the CART models. Identification of areas rated "excellent" and "good" as habitat and areas rated "fair" and "poor" as non-habitat yielded the best CCRs of the habitat quality model. Though a majority (60%) of the total area was classified as "fair," the high inclusion rate of new bighorn sheep locations and unused locations into this classification suggests that the habitat quality model did not include sufficient variables or had

Table 5. Bonferroni confidence intervals for use versus availability of land cover types by male bighorn sheep (n = 14,180 locations) within the 100% MCP in the Silver Bell Mountain range, Arizona, USA, 2003–2005.

Land cover	% available	% use	95% CI	Use category[a]
Madrean Pinyon-Juniper Woodland	0.04	0.16	0.0009–0.0023	1
Mesquite Upland Scrub	0.17	0.10	0.0004–0.0016	2
Mogollon Chaparral	0.04	0.23	0.0015–0.0032	1
North American Warm Desert Riparian Mesquite Bosque	0.01	0.00[b]	–	2
Open Water (likely dry)	0.07	0.09	0.0004–0.0015	3
Recently Mined or Quarried	8.82	14.32	0.1369–0.1496	1
Sonoran-Mojave Creosote bush-White Bursage Desertscrub	0.79	1.40	0.0118–0.0161	1
Sonora-Mojave Desert Mixed Salt Desertscrub	<0.01	0.00[b]	–	2
Sonoran Mid-Elevation Desertscrub	1.53	9.63	0.0909–0.1016	1
Sonoran Palo Verde-Mixed Cacti Desertscrub	88.53	74.07	0.7328–0.7486	2

[a] 1 = use > availability, 2 = use < availability, and 3 = neutral use.

[b] <5 observations in the habitat classification.

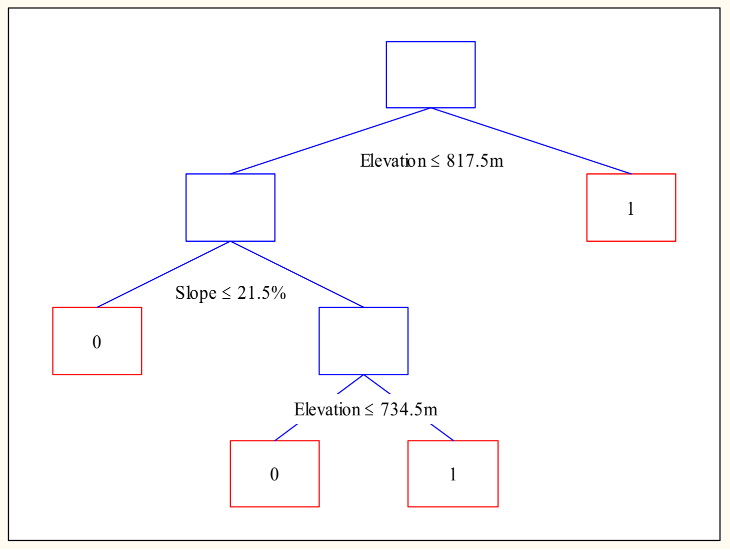

Fig. 2. The CART model of habitat use by male bighorn sheep in the Silver Bell range, Arizona, USA. Decision rules at splits apply to the left branch, while the opposite rule applies to the right branch. Numbers (1 and 0) inside nodes indicate majority classification of each node: 1 = bighorn sheep locations (habitat) and 0 = unused locations (non-habitat).

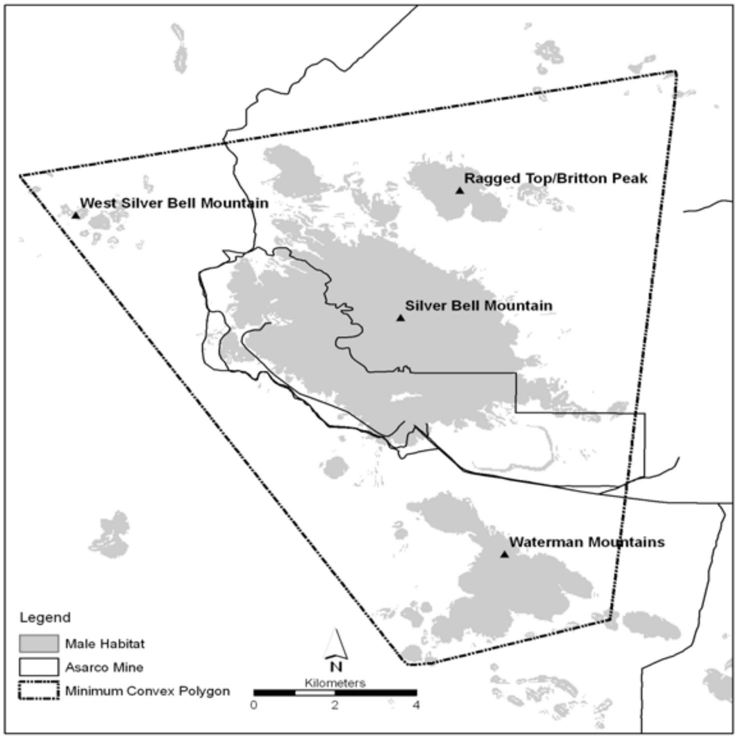

Fig. 3. Map of male bighorn sheep habitat in the Silver Bell range, Arizona, USA, identified by CART model.

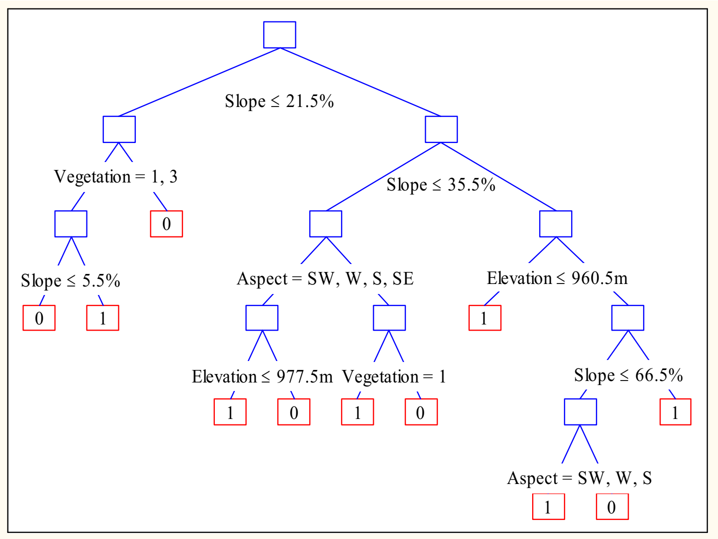

Fig. 4. The CART model of habitat use by female bighorn sheep in the Silver Bell range, Arizona, USA. Decision rules at splits apply to the left branch, while the opposite rule applies to the right branch. Numbers (1 and 0) inside nodes indicate majority classification of each node: 1 = bighorn sheep locations (habitat) and 0 = unused locations (non-habitat).

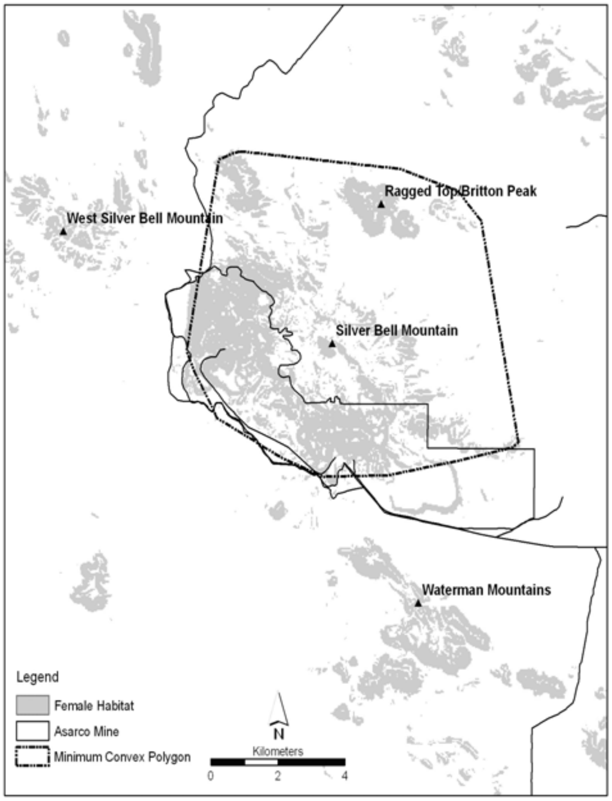

Fig. 5. Map of female bighorn sheep habitat in the Silver Bell range, Arizona, USA, identified by CART model.

insufficient definition between variable classifications to determine the difference between habitat and non-habitat in this category.

Bristow et al. (1996) applied their habitat classification model to an area they delineated to be the Silver Bell study area. This delineated study area included 99% of female bighorn sheep locations but only 62% of male bighorn sheep locations collected from 2003 to 2005. To rate additional areas used by male bighorn sheep with their habitat classification model, considerable time and money would need to be spent to extend the model. In comparison, the CART model can be extended to the Silver Bell Mountains and surrounding areas as long as variables used in the model are within the range of variables observed in areas where the model was developed.

MANAGEMENT IMPLICATIONS

Habitat rating systems such as that used by Bristow et al. (1996) require intensive ground-based data collection and mapping that can be expensive and time-consuming, incorporate scoring matrices that can be subjective and dependent upon proper observer training, and may need to have some or all components reevaluated regularly to keep the model current. Use of multivariate statistics, such as CART modeling, does not require ground-based assessment, can be done with pre-existing GIS coverages, determines a set of objective rules that do not rely upon a human observer, can use a smaller mapping and evaluation scale than that used by many habitat quality models (Douglas and Leslie 1999), and results in a model that can be easily reevaluated as GIS coverages are updated by respective agencies, but requires pre-existing habitat use data for the area of interest. In our case, the multivariate model was more accurate at identifying areas used by bighorn sheep. This suggests that managers should reevaluate past habitat classification models if additional animal location data

Table 6. Percent of locations classified in each of 4 habitat classifications by models from Bristow et al. (1996) in the Silver Bell Mountains, Arizona, USA. Old locations are from Bristow et al. (1996), new locations are from 2003–2005, and unused locations are random locations generated in areas unused by female (F) and male (M) radiocollared bighorn sheep 2003–2005.

Model	% old locations[a]	% new locations[a]	% unused locations[a]
Habitat classification			
All with human use score			
Excellent	41.1	27.44	5.68
Good	26.9	24.60	18.21
Fair	28.3	37.66	61.04
Poor	3.7	10.30	15.07
All without human use score			
Excellent	NR[b]	14.48	2.41
Good	NR	51.53	18.52
Fair	NR	26.42	71.80
Poor	NR	7.57	7.27
F with human use score			
Excellent	NR	23.32	8.85
Good	NR	22.86	20.42
Fair	NR	40.90	59.61
Poor	NR	12.92	11.12
F without human use score			
Excellent	NR	14.15	3.60
Good	NR	47.52	24.96
Fair	NR	29.10	65.89
Poor	NR	9.23	5.54
M with human use score			
Excellent	NR	30.93	2.31
Good	NR	26.06	15.87
Fair	NR	34.92	62.55
Poor	NR	8.09	19.26
M without human use score			
Excellent	NR	14.76	1.16
Good	NR	54.91	11.67
Fair	NR	24.15	78.08
Poor	NR	6.17	9.09

[a] Old locations *n* = 774; new locations Female *n* = 9,320, Male *n* = 11,021; unused locations Female *n* = 9,253, Male *n* = 8,721.

[b] Not reported.

become available, especially if management decisions regarding habitat preservation or destruction are to be made.

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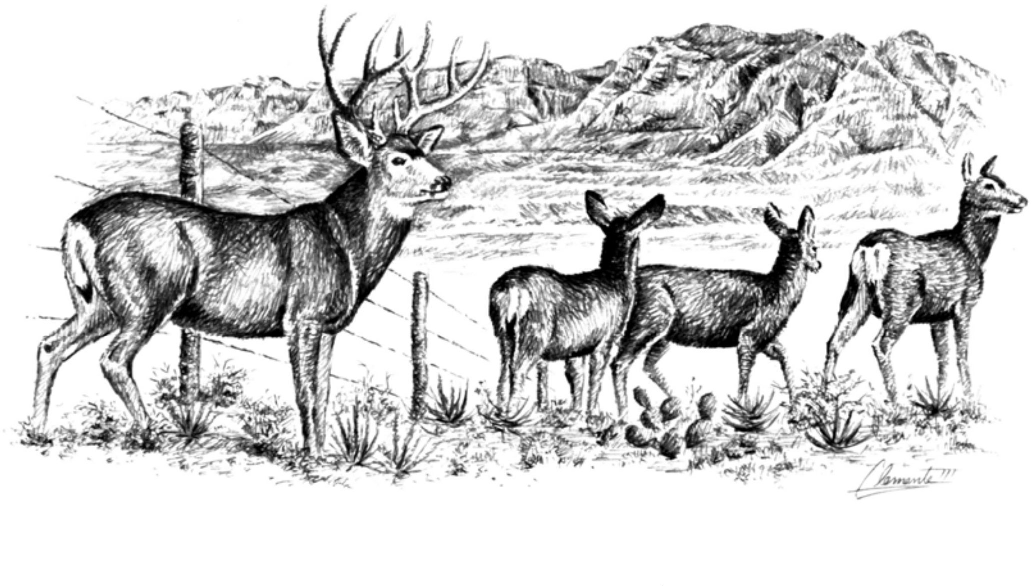

ARIZONA BISON GENETICS: VERIFYING ORIGINS

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***Abstract*:** Recent issues concerning genetic purity of bison (*Bison bison*) in Arizona prompted investigations to determine if cattle genes were present within Arizona bison herds. I obtained 33 blood samples from hunter-harvested bison from 2 herds managed by the Arizona Game and Fish Department at House Rock Valley (House Rock herd) and the Raymond Wildlife Area (Raymond herd). Additionally, 7 tail hair samples were opportunistically collected from th90e House Rock herd. I examined mitochondrial DNA haplotypes and 12 nuclear microsatellite loci for 40 bison samples from the House Rock (n = 22) and Raymond (n = 18) herds. Thirty-nine of the 40 animals had domestic cattle mitochondrial DNA haplotypes and 15 of 40 animals had nuclear markers (5 markers of 12 tested; no individual animal had >2 markers) consistent with those observed from cattle. Although bison in North America number over 300,000, fewer than 10,000 animals comprise herds lacking domestic cattle genetics. Compared with other public and private hybrid bison herds, the levels of cattle genes detected in these Arizona bison herds are high. Nevertheless, phenotypically and behaviorally, the animals cannot be distinguished from pure bison.

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Key words: Arizona, bison, cattle, DNA, hybrid, genetics.

The earliest written record of bison in Arizona is the sighting of a small herd in northern Arizona by a Spanish conquistador in the 1500s, with archaeological evidence from the 1200s (Reid 1952). Native Americans left pictographs of bison in Kanab Creek just north of the Arizona-Utah border and at 18 other locations in Utah. The sighting by conquistadors and the pictographs suggest bison have occurred at least occasionally in northern Arizona. There is no evidence bison were historically abundant in Arizona. However, apart from the representations depicted in those very early records, bison have been a component of the wildlife resources of Arizona since 1905 when Charles J. "Buffalo" Jones brought the species to the North Kaibab Plateau (Hoffmeister 1986).

The United States Congress listed bison as 1 of the wildlife species that should be maintained on the Kaibab Plateau when they established the Grand Canyon Game Preserve. The North Kaibab was described as "ideal for buffalo, deer and other wild game" (Unpublished report to U. S. Congress, Protection of Wild Animals in the Grand Canyon Forest Preserve, 1906). The Preserve was created on 28 November 1906 by President Theodore Roosevelt and is still in effect. It predates the establishment of the National Forest, Grand Canyon National Park (GCNP), and the Arizona Game and Fish Commission. The Preserve originally included all lands in the current North Kaibab Ranger District of the U. S. Forest Service and the GCNP, although the GCNP was later excluded.

Shortly before the establishment of the Preserve in 1906, C. J. Jones acquired a small herd of bison and brought it to the Kaibab Plateau. He formed a partnership with 4 local men whom he persuaded to invest in a "cattalo" experiment. The buffalo and cattle were pastured near Bright Angel Point (now part of GCNP) and crossbreeding began. A few hybrids were produced, but the venture failed due to high birth mortality and sterility (Dary 1974). In 1926, 98 bison were sold to the State of Arizona, and the bison were eventually moved to House Rock Valley (Hoffmeister 1986) from the Kaibab Plateau. On 8 August 1950, the U. S. Forest Service, Bureau of Land Management, Arizona Game and Fish Commission, and livestock

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grazing permittees entered into an agreement that effectively established the House Rock Wildlife Area for use by wildlife. Currently, about 200 bison occupy the North Kaibab Plateau (Arizona Game and Fish Department, unpublished data), although this population has numbered over 300 historically (Hoffmeister 1986).

Raymond Wildlife Area, 80 km east of Flagstaff, was established in 1942 to manage pronghorn (*Antilocapra americana*) habitat. Bison were introduced in 1945 from House Rock. Bison were also maintained in a separate herd on Fort Huachuca from 1949 until 1956. When the army reactivated Fort Huachuca during the Korean conflict, the bison were removed from Fort Huachuca to Raymond Wildlife Area. The Department still manages bison on Raymond in a semi-wild state, with no regular supplemental feeding or gathering. The Raymond herd has numbered as high as 358 animals (Hoffmeister 1986), although currently <100 occupy the area (Arizona Game and Fish Department, unpublished data).

Both of Arizona's bison herds have received supplemental transplants. Twelve bulls were moved from Wichita Wildlife Refuge in Oklahoma to House Rock in 1942, and 6 more were added in 1946. The Raymond herd received supplemental transplants from the Wichita Wildlife Refuge in 1956 and 1962 (Hoffmesiter 1986). In 1980, 3 bison were added to the House Rock herd from the Montana Bison Refuge. In 2000, 5 females and 1 male were added to the House Rock herd and 5 females and 1 male were added to the Raymond herd from the Henry Mountains in Utah (Arizona Game and Fish Department, unpublished data). Because of the background of Arizona’s bison herds, questions regarding their genetic disposition and purity surfaced. About 300,000 bison currently inhabit North America, and based on DNA screenings and testing, <10,000 are genetically pure (J. N. Derr, Texas A&M University, personal communication). My objective was to determine the genetic purity of bison in Arizona.

METHODS

Because Arizona’s bison herds are hunted, hunters can provide genetic material for analysis. We contacted hunters who voluntarily provided blood samples from 15 bison from the House Rock herd and 18 from the Raymond herd. About 1 ml of whole blood was applied to FTA cards (Whatman, Newton Center, Massachusetts, USA). In addition, I collected 7 tail hair samples (1 hair with root follicle/sample) from bison from House Rock to supplement blood samples. All samples were sent to the Texas Veterinary Medical Center, Texas A&M University for genetic analysis. Mitochondrial DNA sequences (Ward et al. 1999) and fragment lengths for nuclear microsatellite markers (Schnabel et al. 2000) were determined as described by Halbert et al. (2004).

RESULTS

Mitochondrial DNA haplotypes and 12 nuclear microsatellite loci were examined for 40 bison samples from the House Rock (n = 22) and Raymond (n = 18) herds. Thirty-nine of the 40 animals had domestic cattle mitochondrial DNA haplotypes (Ward et al. 1999) and 15 of 40 animals had nuclear microsatellite alleles (5 markers of 12 tested; no individual had >2 markers) consistent with those observed from cattle (Schnabel et al. 2000, Halbert et al 2004) (Table 1). Only 1 individual from the Raymond herd was lacking any cattle genetic material based on the markers analyzed in my study.

DISCUSSION

Management of bison on the Kaibab Plateau is complex. Bison are managed cooperatively under the 1950 agreement between the U. S. Forest Service and the Arizona Game and Fish Department that limits their numbers to about 100, specifically around the House Rock Wildlife Area in House Rock Valley. Bison have been increasingly moving onto GCNP, an area that has federal statutory limitations on hunting and introduction of new species, such as bison. Expansion of bison range onto the GCNP requires a lengthy public and legal process whereby GCNP mandates would be modified to allow a species on the fringe of its native range to reside within its boundaries. This process is further complicated because this herd has cattle genetic material and cannot be considered pure bison. This process appears somewhat capricious, because Merriam's turkey (*Meleagris gallopavo merriami*) occur in GCNP, and they

Table 1. Designation of bison (B) or cattle (C) classification based on mitochondrial DNA haplotype or nuclear microsatellite marker length for blood sampled from the Raymond herd (R) and House Rock herd (HR) and tail hair sampled from the House Rock herd (HRT), 2003. Bison and cattle allele ranges for each loci (loci 1 = 15 for bison, 214, 216-218 for cattle; 2 = 137 B, 141-167 C; 3 = 132, 134 B, 40-160 C; 4 = 108, 110 B, 116-118 C; 5 = 176 B, 182-188 C; 6 = 66, 68, 70 B, 80-98 C; 7 = 75 B, 84-110 C; 8 = 158 B, 162-182 C; 9 = 167, 169, 171 B, 175-219 C; 10 = 123 B, 125-13 C; 11 = 128, 130, 132 B; 134-154 C; 12 = 73 B, 76-102 C.

Bison no.	mtDNA	Nuclear loci 1	Nuclear loci 2	Nuclear loci 3	Nuclear loci 4	Nuclear loci 5	Nuclear loci 6	Nuclear loci 7	Nuclear loci 8	Nuclear loci 9	Nuclear loci 10	Nuclear loci 11	Nuclear loci 12
R 1	C	B	C	B	B	B	B	C	B	B	B	B	B
R 2	B	B	B	B	B	B	B	B	B	B	B	B	B
R 3	C	B	B	B	B	B	B	B	B	B	B	B	B
R 4	C	B	B	B	B	B	B	C	B	B	B	B	B
R 5	C	B	B	B	B	B	B	B	B	B	B	B	B
R 6	C	B	B	B	B	B	B	C	B	B	B	B	B
R 18	C	B	B	B	B	B	B	B	B	B	B	B	B
R 19	C	B	C	B	B	B	B	B	B	B	B	B	B
R 20	C	B	B	B	B	B	B	B	B	B	B	B	B
R 21	C	B	B	B	B	B	B	B	B	B	B	B	B
R 22	C	B	B	B	B	B	B	B	B	B	B	C	B
R 23	C	B	B	B	C	B	B	B	B	B	B	B	B
R 24	C	B	B	B	B	B	B	B	B	B	B	B	B
R 25	C	B	B	B	B	B	B	B	B	B	B	B	B
R 26	C	B	C	B	B	B	B	B	B	B	B	B	B
R 27	C	B	C	B	B	B	B	B	B	B	B	C	B

Table 1. Continued

Bison no.	mtDNA	Nuclear loci 1	Nuclear loci 2	Nuclear loci 3	Nuclear loci 4	Nuclear loci 5	Nuclear loci 6	Nuclear loci 7	Nuclear loci 8	Nuclear loci 9	Nuclear loci 10	Nuclear loci 11	Nuclear loci 12
R 28	C	B	C	B	B	B	B	B	B	B	B	B	B
R 29	C	B	B	B	B	B	B	B	B	B	B	C	B
HR 7	C	B	C	B	B	B	B	B	B	B	B	B	B
HR 8	C	B	C	B	B	B	B	B	B	B	B	B	B
HR 9	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 10	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 11	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 12	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 13	C	B	B	B	B	B	B	B	B	B	B	C	B
HR 14	C	B	C	B	B	B	B	B	B	B	B	B	B
HR 15	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 16	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 17	C	B	C	B	B	B	B	B	B	B	B	B	B
HR 30	C	B	B	B	B	B	B	B	C	B	B	B	B
HR 31	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 32	C	B	B	B	B	B	B	B	B	B	B	B	B
HR 33	C	B	B	B	B	B	B	B	B	B	B	B	B
HRT 1	C	B	B	B	B	B	B	B	B	B	B	B	B
HRT 2	C	B	B	B	B	B	B	B	B	B	B	B	B
HRT 3	C	B	B	B	B	B	B	B	B	B	B	B	B

Table 1. Continued

Bison no.	mtDNA	Nuclear loci 1	Nuclear loci 2	Nuclear loci 3	Nuclear loci 4	Nuclear loci 5	Nuclear loci 6	Nuclear loci 7	Nuclear loci 8	Nuclear loci 9	Nuclear loci 10	Nuclear loci 11	Nuclear loci 12
HRT 4	C	B	B	B	B	B	B	B	B	B	B	B	B
HRT 5	C	B	B	B	B	B	B	B	B	B	B	B	B
HRT 6	C	B	B	B	B	B	B	B	B	B	B	B	B
HRT 7	C	B	B	B	B	B	B	B	B	B	B	B	B

are not native to habitats in Arizona north of the Colorado River; however their populations were established in GCNP prior to passage of the National Environmental Policy Act. Although bison were once located within the boundaries of the current GCNP, they were removed prior to establishment of GCNP and many environmental statutes.

The Raymond herd is not affected administratively by National Park Service lands, however bison trespass onto neighboring private ranches and competition for forage with other wildlife needs must be considered when determining population objectives. Pronghorn on neighboring Anderson Mesa have received substantial herd and land management focus recently because their numbers are below management objectives (Arizona Game and Fish Department, unpublished data).

Herd management for these bison is challenging, especially at House Rock. This herd is not restricted by fences, and limiting population size can be challenging. Hunters have had relatively little success in harvesting animals from the House Rock herd because the animals frequently take refuge on the GCNP and on a nearby wilderness area that is heavily forested and steep. Recent hunt success has been ≤3% (Arizona Game and Fish Department, unpublished data). Hunting is not allowed on the GCNP, and removing even the annual increase from recruitment can be difficult. Trap and transplant activities are costly, and removal options are somewhat limited. Recent efforts to trap bison on GCNP have been unsuccessful.

Genetic material from cattle is relatively common in most bison herds in the USA (J. N. Derr, Texas A&M University, personal communication). Evidence of cattle gene introgression is present in the Texas State Bison Herd (Ward et al. 1999), however the Texas herd is apparently suffering low recruitment and high calf mortality a result of small population of founders, multiple bottlenecks, chronically small population size, and genetic drift rather than cattle gene introgression (Halbert et al. 2004). Relatively pure bison herds in Yellowstone National Park and Theodore Roosevelt National Park that originated from a small number of founders (30-50 and 29, respectively), but grew in population size rapidly, have not experienced similar demographic performance issues as has the Texas herd (Halbert et al. 2004). To date, demographic performance has not been negatively impacted in either the House Rock or Raymond herds in Arizona.

Options for managing the breeding segments of these populations range in difficulty. First, no change in the management is a viable option. No depressed genetic health of the herds has been noted and routine disease screenings have not detected exposure to brucellosis, common in some pure strains of bison like those from Yellowstone. Second, intensive, selective breeding could be used to manage towards individuals with increasingly smaller proportions of cattle genes. This would be an expensive and intensive undertaking, requiring substantial use of domestic animal husbandry techniques and knowledge of individual animal genetic composition. This approach may be possible with the smaller, more intensively managed Raymond herd, although lack of genetic diversity within populations held at low numbers for long periods of time can be more detrimental to population demographics and increasing genetic diversity through supplemental translocations of additional bison may be important (Senner 1980, Halbert et al 2004). Selective breeding might prove impossible on the House Rock population, as this is a fully free-ranging population that would be difficult, if not impossible, to manage using domestic animal husbandry. Finally, these herds could be removed and replaced by animals that were genetically pure. Again, this would be expensive, intensive, and difficult at House Rock. Introductions of animals known to be disease free and genetically pure would be critical to this option.

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NUTRITIONAL CONTENT OF MULE DEER FORAGE IN BURNED AND UNBURNED INTERIOR CHAPARRAL

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Abstract: A wildfire burned 159 ha of the Walnut Canyon Enclosure, central Arizona in 1996. We evaluated the effect of the fire on crude protein, calcium (Ca), and phosphorus (P) contents of forage of mule deer (*Odocoileus hemionus*). We collected a sample of 10 forage species, prickly pear (*Opuntia* spp.), a grass mix, and a forb mix quarterly in burned and unburned interior chaparral during 1998 and 2000. Burning did not appear to improve forage content of crude protein, Ca, or P. Forage species in unburned chaparral generally contained higher levels of Ca and P when differences between burned and unburned chaparral were significant. Forage increased in Ca and P content between 1998 and 2000, possibly due to limited rainfall and growth in 2000. Forage in burned and unburned areas provided adequate protein levels for mule deer. Phosphorus levels failed to meet mule deer requirements in 1998, but increased above the required level in 2000. Calcium levels were high in the forage tested. Fire did not increase crude protein, Ca, or P levels in most deer forage tested 2 and 4 years after the fire.

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The bimodal rainfall pattern in the Sonoran Desert results in floristically distinct summer and winter annuals (Shreve 1964, Rogers and Vint 1987, Pase and Brown 1994). High winter precipitation, particularly in consecutive years, can result in extensive growth of forbs and grasses that can provide highly flammable fine fuels to carry summer fires (McLaughlin and Bowers 1982, Rogers and Vint 1987). The intensity and frequency of fires in the Sonoran Desert have increased due to fire suppression, the proliferation of exotic annuals, overgrazing of

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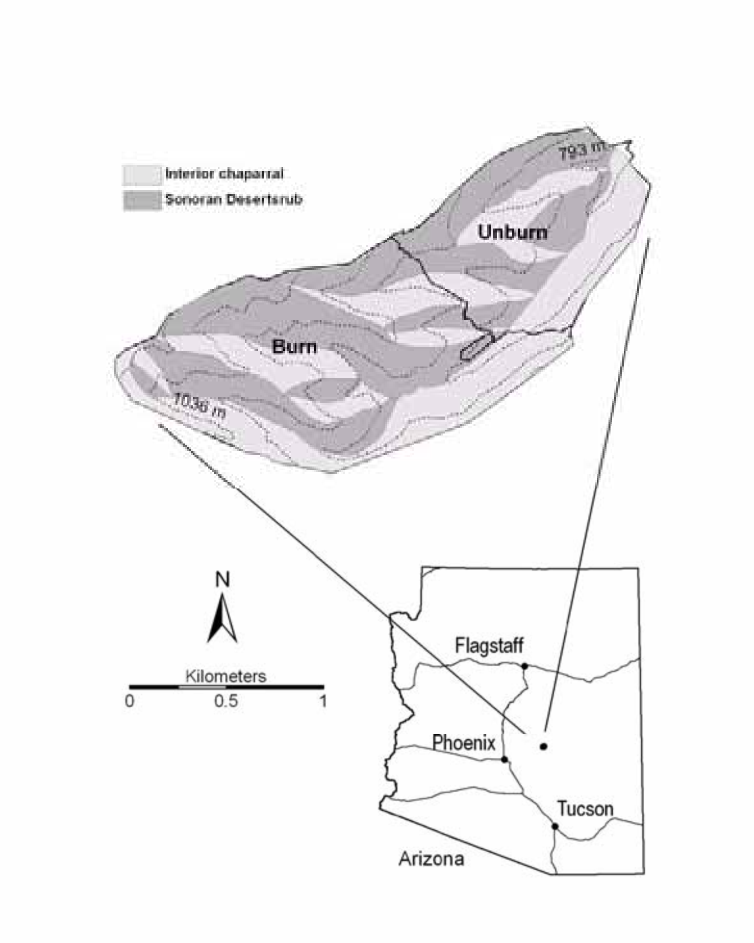

Fig. 1. The Walnut Canyon Enclosure, Three Bar Wildlife Area, Arizona with fence, burn boundary, and contour lines, 1998 and 2000.

livestock, changing weather patterns, and increasing human populations (Rogers and Steele 1980, Schmid and Rogers 1988, McAuliffe 1995, Swantek 1997).

The effect of fire on the nutritional qualities of forage plants and ungulate diets has not been well-studied in the transition zone between lower desert and chaparral and results from other biomes are varied. Some studies reported improved forage quality (i.e., increased nutrient or mineral content) in burned areas (DeWitt and Derby 1955, Springer 1977, Willms et al. 1981, Hobbs and Spowart 1984), whereas others reported no difference in forage quality between burned and unburned areas (Dills 1970, Seip and Bunnell 1985). As managers in the southwestern United States consider decisions to begin prescribed burning and face landscapes that have experienced recent catastrophic wildfires, it is important to understand the effect these fires may have on the nutritional content of forage for local fauna. Our objective was to compare crude protein, Ca, and P content of some forage of desert mule deer available in burned and unburned interior chaparral 2 and 4 years after a wildfire.

STUDY AREA

The Walnut Canyon Enclosure (33° 41' N 111° 13' W) was a fenced area (246 ha; Fig. 1) on the Three Bar Wildlife Area, located 60 km northeast of Phoenix in central Arizona on the east slope of the Mazatzal Mountains in Tonto National Forest. Domestic livestock had not grazed the area since 1947 (Smith and LeCount 1976). The Arizona Game and Fish Department (AGFD) built the predator-resistant enclosure in 1970 to study mule deer and collared peccary (*Pecari tajacu*). Elevations in the enclosure ranged from 790 to 1,130 m. South-facing slopes were characterized by Sonoran desertscrub vegetation including saguaro (*Carnegiea gigantea*), jojoba (*Simmondsia chinensis*), catclaw acacia (*Acacia greggii*), and prickly pear (Turner and Brown 1994). North-facing slopes were more mesic and characterized by interior chaparral vegetation including shrub live oak (*Quercus turbinella*), mountain mahogany (*Cercocarpus montanus*), buckwheat *(Eriogonum* spp.) and false mesquite (*Calliandra eriophylla*; Horejsi and Smith 1983, Pase and Brown 1994). Plant nomenclature follows the Integrated Taxonomic Information System (2005).

Hot, dry summers and mild winters typified the climate of the study area. The average annual rainfall (1976 – 2000) for the area was 44.4 cm (range = 25.5 – 90.63 cm; Western Regional Climate Center [WRCC] 1976 – 2000). Summer monsoons (July – September) produced an average of 11.7 cm of precipitation (WRCC 1976 – 2000), resulting in a brief growing season. From October through April (1976 – 2000), the area received an average of 31.3 cm of precipitation. Average (1976 – 2000) monthly maximum temperatures ranged from 39 (July) to 15 °C (December). Temperatures were below freezing an average of 15 days/year with occasional snow (<3cm; WRCC 1976 – 2000). Monthly rainfall during October 1997 through December 1998 and October 1999 through December 2000 ranged from 0 to >14 cm (WRCC 1976 – 2000).

From 28 April to 14 May 1996, the Lone Fire burned about 24,280 ha of Sonoran Desert, chaparral, and ponderosa pine (*Pinus ponderosa*) forest, including most fuels in the western 159 ha of the enclosure. Most thermal cover (i.e., vegetation >75 cm in height that could shelter an adult deer from the sun; Tull et al. 2001) in desertscrub was destroyed. During wet periods post-fire, forbs and grasses were abundant but there was minimal woody vegetation taller than 40 cm. Standing dead trees and shrubs were rare, and most cacti were dead or damaged after the fire. The trunks and large branches of most woody species in chaparral were charred, but otherwise intact. There were no leaves on burned branches, but most trees regenerated at the base and provided good thermal cover and browse.

METHODS

We collected all plant samples from north-facing slopes in burned and unburned chaparral. We collected a sample (≥110 g) of catclaw acacia, false mesquite, desert ceanothus (*Ceanothus greggii*), mountain mahogany, buckwheat, range ratany (*Krameria erecta*), prickly pear, mesquite (*Prosopis* spp.), shrub live oak, jojoba, globemallow (*Sphaeralcea* spp.), mixed forbs, and mixed grasses in February, May, August, and November of 1998 and 2000. All samples were pooled from ≥20 individuals of each species and represented portions of the plant commonly selected by foraging deer (Hill 1956). Samples consisted of flowers, leaves, new shoots, and branch tips for shrub and tree species, cladophylls for prickly pears, and entire above-ground portions of herbaceous forbs and grasses. The grass mix included a mix of grasses present. The forb mix included any herbaceous forbs present except filaree (*Erodium cicutarium*), weakleaf burr ragweed (*Ambrosia confertiflora*), spurges (*Euphorbia* spp.), snakeweed (*Gutierrezia sarothrae*), menodora (*Menodora scabra*), and wormwood (*Artemisia ludoviciana*). Though many of the forbs listed above that were not included in our collections were deer forage, we omitted them because initial collections did not include them, though they were present in the study area, and we wanted consistency between samples for comparative analyses. We collected prickly pear fruits, shrub live oak acorns, jojoba nuts, and catclaw acacia and mesquite legumes as samples separate from vegetative samples when available during August.

We weighed samples to the nearest 5 g with a 1,000 g scale (Pesola, Switzerland) and stored samples immediately after collection. We stored all samples other than prickly pear as they were collected in ventilated (open top) paper bags at ambient air temperature for ≤10 days. Due to the tendency of prickly pear to mold, we cut the cladophylls and fruits into small pieces to facilitate air-drying, stored them in ventilated paper bags, and transported them within 2 days to the University of Arizona to complete drying. We completed drying all vegetation samples to a steady weight in a convection oven set at 50 – 65 °C for ≥72 hours for grass, forbs, and browse, and ≥120 hours for cacti.

We ground dried samples through a 2 mm screen with a Thomas-Wiley Laboratory Mill (A. H. Thomas Company, Philadelphia, Pennsylvania, USA). We determined percents of crude protein, Ca, and P on a dry weight basis. We calculated crude protein by multiplying Kjeldahl nitrogen (measured using a micro-Kjeldahl H_2S0_4 digestion, NaOH neutralization, and an autoanalyzer) by 6.25 (Goering and Van Soest 1970, Robbins 1993). Mineral analyses were conducted after acid digestion using the spectrophotometric atomic absorption technique for Ca and calorimetric technique for P. We diluted samples to fit into standard measurement curves (Fox 1997). The Department of Animal and Range Sciences, New Mexico State University, Las Cruces, New Mexico conducted laboratory analyses following the proximate system of analysis (Robbins 1993, Van Soest 1994) and procedures approved by the Association of Official Agricultural Chemists (1980). Each nutritional value was measured once per sample. We calculated ratios of Ca to P from laboratory results.

We tested for differences between samples from burned and unburned areas within each year and differences between samples from each year within each area. Using season as a replicate, we used 2-tailed paired *t*-tests. Because our small sample sizes increased the likelihood of a Type II error and decreased statistical power, we chose an α-level of 0.10 for all *t*-tests.

RESULTS

We analyzed 11 plant species and grass and forb mixes quarterly in 1998 and 2000 (Tables 1, 2). Shrub live-oak acorns were not available in the burned chaparral in 1998. Acorns and catclaw acacia and mesquite legumes were not available for collection in 2000.

Percent crude protein was greater in samples of catclaw acacia, desert ceanothus, mesquite, and shrub live oak from the burn and lower in samples of false mesquite and range ratany from the burn as compared to unburned samples in 1998 (Table 3). Calcium was greater in mountain mahogany and jojoba collected in unburned versus burned chaparral in 1998 (Table 3). Phosphorus was greater in unburned false mesquite in 1998 (Table 3).

Percent crude protein in samples of false mesquite, mountain mahogany, buckwheat, range ratany, and jojoba was greater from unburned than burned chaparral in 2000 (Table 3). Desert ceanothus, mountain mahogany, the forb mix, range ratany, and shrub live oak contained more Ca in samples from unburned than burned chaparral in 2000 (Table 3). Catclaw acacia, shrub live oak, jojoba, and mountain mahogany contained more P in unburned chaparral than burned chaparral in 2000 (Table 3).

Calcium was higher in 2000 than 1998 in burned and unburned areas in desert ceanothus, mountain mahogany, range ratany, prickly pear, mesquite, and jojoba (Table 4). False mesquite, buckwheat, and globemallow contained more Ca in the burned chaparral in 1998 than 2000 while shrub live oak and catclaw acacia contained more Ca in the unburned chaparral in 1998 than 2000 (Table 4). Catclaw acacia, false mesquite, desert ceanothus, mountain mahogany, range ratany, prickly pear, mesquite, jojoba, globemallow, and shrub live oak had more P in both burned and unburned areas in 2000 than 1998 (Table 4). Buckwheat contained more P in burned chaparral in 2000 than 1998 (Table 4). The ratio of Ca to P was generally higher across all forage species in 1998 than 2000 (Table 2).

DISCUSSION

In most cases where we found a significant difference between burned and unburned chaparral, the unburned chaparral supported plants higher in crude protein, calcium, and phosphorus. If burning improved the nutritional content of forage in our study area, the changes did not persist 2 years after the burn. Springer (1977) reported that increases in crude protein and phosphorus

Table 1. Crude protein content of deer forage species sampled seasonally in burned (B) and unburned (U) interior chaparral in the Walnut Canyon Enclosure, Arizona, 1998 and 2000.

	% crude protein			
	B		U	
Forage	1998	2000	1998	2000
Acacia greggii				
February	14.69	9.20	12.20	9.30
May	25.11	15.50	19.73	14.60
August	17.19	16.80	10.59	18.70
August (fruits)	17.53	-	17.47	-
November	11.98	14.50	11.41	16.20
Calliandra eriophylla				
February	12.85	5.60	14.07	6.40
May	17.63	12.00	18.15	12.80
August	10.45	20.20	12.63	20.50
November	11.82	12.10	14.01	12.70
Ceanothus greggii				
February	9.24	6.40	8.65	7.60
May	13.00	5.20	10.84	6.60
August	7.05	6.60	5.26	6.70
November	8.15	8.80	7.53	9.10
Cercocarpus spp.				
February	14.68	9.20	9.78	9.10
May	13.50	8.60	11.33	9.80
August	8.11	6.60	8.26	8.70
November	8.57	11.50	9.70	12.60
Eriogonum spp.				
February	14.75	5.70	15.49	5.8
May	10.36	7.70	6.81	8.90
August	6.06	8.00	5.52	8.30
November	7.08	14.40	8.64	15.10
Forb mix[a]				
February	24.71	6.00	27.01	7.10
May	9.86	8.50	8.35	12.60
August	3.33	33.70	3.03	20.20
November	16.13	20.50	12.26	24.40
Grass mix[a]				
February	18.01	3.40	18.88	2.80
May	7.60	4.60	7.32	4.00
August	3.20	12.50	2.93	9.60
November	21.02	11.30	10.99	12.10
Krameria erecta				
February	7.72	5.10	8.61	6.10
May	11.78	7.60	13.26	8.50
August	5.81	5.60	5.83	6.50
November	6.45	6.80	7.68	7.30

Table 1. Continued

	% crude protein			
	B		U	
Forage	1998	2000	1998	2000
Opuntia spp.				
February	8.05	4.80	7.51	4.20
May	5.38	4.60	5.87	5.00
August	8.03	4.90	9.06	4.40
August (fruits)	7.44	7.40	8.50	7.80
November	4.50	5.40	3.90	5.40
Prosopis velutina				
February	17.38	12.40	14.24	11.20
May	26.13	17.40	20.95	17.30
August	18.00	17.80	15.21	17.50
August (fruits)	17.72	-	19.35	-
November	16.11	15.90	16.11	19.10
Simmondsia chinensis				
February	12.28	7.00	10.37	8.70
May	21.13	6.90	11.93	8.80
August	9.72	7.00	9.66	11.00
August (fruits)	14.64	9.20	14.63	9.20
November	8.58	8.80	10.90	10.60
Sphaeralcea spp.				
February	26.37	13.00	26.56	11.20
May	16.63	13.40	13.50	15.40
August	9.18	20.50	6.30	23.00
November	19.55	22.70	18.80	25.10
Quercus turbinella				
February	10.48	3.90	8.77	7.10
May	10.17	7.90	9.36	8.30
August	7.78	7.70	7.13	8.90
August (fruits)	-	-	8.00	-
November	8.34	9.10	8.21	9.60

[a] Composite of available species.

following prescribed burning lasted only 1 year.

The growth of most desert plants is limited by water availability (MacMahon 1985) and seasonal rains affect nutritional content of forage (Short 1981, Urness 1981). Calcium levels were highest during periods of senescence in forage tested by Jones and Weeks (1985). The increase in Ca and P in the forage we tested may be attributable to limited rainfall or vegetation senescence in 2000. Rainfall during 1998 (52.1 cm without June rainfall that was not measured) was 117% of average (44.4 cm), while rainfall in 2000 (33.6 cm, 76% of average) bordered on drought conditions (WRCC 1976 – 2000). A drought year has <75% normal precipitation (Anthony 1976). Even more striking, rainfall October through April of 1999 – 2000 (9.27 cm) was only 34% of average while 1997 – 1998 rainfall (37.62 cm) was 120% of average (WRCC 1976 – 2000).

It is difficult to predict dietary mineral levels and detect mineral deficiencies without results from stomach content or fecal analyses (Urness and McCulloch 1973). Elk (*Cervus elaphus*) in a burned area maintained similar diet quality through the year despite large fluctuations in forage quality (Rowland et al. 1983). The relatively small mouth size of deer allows them

Table 2. Calcium and P content and ratios of deer forage species sampled seasonally in burned (B) and unburned (U) interior chaparral in the Walnut Canyon Enclosure, Arizona, 1998 and 2000.

	%Ca				%P				Ca:P			
	B		U		B		U		B		U	
Forage	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000
Acacia greggii												
February	2.88	2.62	3.22	4.15	0.02	0.27	0.03	0.35	144	10	107	12
May	0.71	2.91	0.99	3.78	0.35	0.37	0.30	0.43	2	8	3	9
August	1.78	5.15	2.78	2.85	0.00	0.48	0.01	0.63	593	11	348	5
August (fruits)	0.41	-	0.42	-	0.01	-	0.01	-	59	-	42	-
November	2.37	3.54	2.34	4.24	0.00	0.33	0.01	0.37	2370	11	390	11
Calliandra eriophylla												
February	1.21	1.17	0.89	1.07	0.04	0.27	0.05	0.21	30	4	18	5
May	0.98	1.77	2.35	2.13	0.07	0.26	0.09	0.29	14	7	26	7
August	1.08	1.65	1.20	2.05	0.00	0.56	0.01	0.44	540	3	120	5
November	0.76	1.94	1.37	2.13	0.00	0.29	0.01	0.25	380	7	274	9
Ceanothus greggii												
February	1.41	1.55	1.88	2.08	0.01	0.20	0.07	0.26	141	8	27	8
May	1.06	2.17	1.58	2.93	0.08	0.23	0.09	0.22	13	9	18	13
August	1.04	1.94	0.81	3.04	0.00	0.18	0.00	0.20	260	11	405	15
November	0.95	2.09	1.00	2.64	0.00	0.20	0.00	0.27	-	10	500	10

Table 2. Continued

	%Ca				%P				Ca:P			
	B		U		B		U		B		U	
Forage	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000
Cercocarpus spp.												
February	1.76	3.04	1.95	3.77	0.02	0.21	0.10	0.37	88	14	20	10
May	1.48	3.04	2.14	4.22	0.06	0.23	0.09	0.33	25	13	24	13
August	1.41	3.24	1.67	3.82	0.00	0.22	0.00	0.36	1410	15	557	11
November	1.12	2.76	1.44	3.45	0.00	0.25	0.01	0.35	-	11	206	10
Eriogonum spp.												
February	1.49	2.00	1.54	2.43	0.17	0.18	0.20	0.21	9	11	8	12
May	0.84	1.61	1.29	1.67	0.15	0.31	0.11	0.31	6	5	12	5
August	0.61	1.25	0.66	0.65	0.01	0.30	0.01	0.14	55	4	66	5
November	1.11	1.86	0.91	2.16	0.01	0.41	0.01	0.47	159	5	114	5
Forb mix[a]												
February	1.45	1.81	2.71	1.89	0.54	0.30	0.52	0.30	3	6	5	6
May	1.71	2.38	2.64	3.24	0.26	0.38	0.20	0.42	7	6	13	8
August	1.20	1.15	1.37	2.17	0.01	1.08	0.00	0.64	133	1	457	3
November	1.36	3.67	1.16	4.82	0.01	0.85	0.01	0.84	113	4	105	6
Grass mix[a]												
February	0.82	0.67	1.26	0.52	0.28	0.24	0.25	0.14	3	3	5	4
May	0.50	0.41	0.46	0.52	0.03	0.18	0.11	0.20	17	2	4	3
August	0.27	0.70	0.21	0.63	0.01	0.39	0.01	0.24	54	2	21	3

Table 2. Continued

	%Ca				%P				Ca:P			
	B		U		B		U		B		U	
Forage	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000
November	0.59	0.72	0.38	0.88	0.04	0.43	0.01	0.49	14	2	29	2
Krameria erecta												
February	0.86	1.57	1.23	1.68	0.03	0.33	0.02	0.39	29	5	62	4
May	0.95	1.71	0.90	2.17	0.20	0.40	0.21	0.39	5	4	4	6
August	0.85	1.73	1.12	2.06	0.01	0.28	0.01	0.38	142	6	224	5
November	0.96	1.58	1.16	1.63	0.00	0.27	0.01	0.31	480	6	193	5
Opuntia spp.												
February	2.60	10.65	3.53	10.92	0.07	0.31	0.05	0.23	37	34	71	47
May	4.07	8.76	4.40	8.21	0.05	0.27	0.07	0.21	81	32	63	39
August	2.65	8.73	4.96	10.16	0.00	0.29	0.00	0.18	2650	30	1240	56
August (fruits)	1.13	7.21	1.52	8.56	0.01	0.35	0.00	0.39	226	21	1520	22
November	3.43	9.52	1.29	11.11	0.01	0.33	0.00	0.37	686	29	1290	30
Prosopis velutina												
February	2.51	3.84	2.03	4.35	0.03	0.37	0.01	0.41	84	10	203	11
May	1.16	2.73	1.08	3.13	0.31	0.39	0.19	0.40	4	7	6	8
August	1.26	4.45	1.78	3.95	0.01	0.29	0.01	0.31	252	15	356	13
August (fruits)	0.61	-	0.45	-	0.01	-	0.01	-	68	-	41	-
November	1.69	3.78	1.67	3.31	0.00	0.31	0.00	0.33	563	12	1670	10

Table 2. Continued

	%Ca				%P				Ca:P			
	B		U		B		U		B		U	
Forage	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000	1998	2000
Simmondsia chinensis												
February	1.14	2.43	1.78	2.28	0.00	0.14	0.00	0.19	-	17	-	12
May	0.92	2.03	1.31	2.68	0.16	0.14	0.06	0.14	6	15	22	19
August	0.72	3.39	1.09	3.04	0.00	0.13	0.01	0.17	360	26	182	18
August (fruits)	0.11	0.14	0.15	0.18	0.01	0.26	0.01	0.28	18	1	17	1
November	0.81	2.23	0.89	2.88	0.00	0.14	0.00	0.17	-	16	-	17
Sphaeralcea spp.												
February	3.10	3.63	3.62	3.59	0.43	0.51	0.38	0.45	7	7	10	8
May	2.13	3.52	1.95	3.51	0.31	0.54	0.22	0.58	7	7	9	6
August	2.53	3.43	1.92	2.89	0.01	0.92	0.00	0.75	506	4	960	4
November	1.79	4.98	2.06	5.11	0.02	0.61	0.02	0.92	105	8	108	6
Quercus turbinella												
February	1.49	1.07	1.69	1.86	0.07	0.25	0.07	0.39	21	4	24	5
May	0.83	1.11	1.18	2.20	0.05	0.32	0.09	0.38	17	3	13	6
August	0.52	1.34	0.90	2.17	0.01	0.29	0.01	0.35	87	5	150	6
August (fruits)	-	-	0.30	-	-	-	0.01	-	-	-	60	-
November	1.85	1.16	0.85	1.37	0.00	0.33	0.01	0.37	463	4	121	4

[a] Composite of available species.

Table 3. Mean differences (*d*) in crude protein, Ca, and P content (burned minus unburned) of deer forage samples collected seasonally ($n = 4$) in burned and unburned interior chaparral, Walnut Canyon Enclosure, Arizona, 1998 and 2000.

		% crude protein			% Ca			% P		
Forage	Year	*d*	SE	*P*	*d*	SE	*P*	*d*	SE	*P*
Acacia greggii	1998	3.76	1.37	0.071	-0.40	0.20	0.164	0.01	0.01	0.642
	2000	-0.70	0.67	0.372	-0.20	0.85	0.830	0.08	0.02	0.041
Calliandra eriophylla	1998	-1.53	0.41	0.033	-0.45	0.36	0.307	-0.01	0.00	0.072
	2000	-0.63	0.12	0.013	-0.21	0.11	0.158	0.05	0.03	0.222
Ceanothus greggii	1998	1.29	0.40	0.049	-0.20	0.18	0.339	-0.02	0.01	0.311
	2000	-0.75	0.32	0.103	-0.74	0.13	0.011	-0.04	0.02	0.155
Cercocarpus spp.	1998	1.45	1.34	0.360	-0.36	0.10	0.042	-0.03	0.02	0.192
	2000	-1.08	0.45	0.098	-0.80	0.13	0.009	-0.13	0.02	0.004
Eriogonum spp.	1998	0.45	1.12	0.716	-0.09	0.13	0.562	0.00	0.01	0.874
	2000	-0.58	0.24	0.099	-0.05	0.23	0.849	0.02	0.05	0.745
Forb mix[a]	1998	0.85	1.28	0.557	-0.54	0.34	0.206	0.02	0.01	0.198
	2000	1.10	4.19	0.810	-0.78	0.24	0.048	0.10	0.11	0.431
Grass mix[a]	1998	2.43	2.55	0.411	-0.03	0.14	0.833	-0.01	0.03	0.815
	2000	0.83	0.77	0.361	-0.01	0.07	0.875	0.04	0.05	0.453
Krameria erecta	1998	-0.91	0.32	0.066	-0.20	0.09	0.115	0.00	0.00	0.852
	2000	-0.83	0.11	0.005	-0.24	0.10	0.088	-0.05	0.02	0.129
Opuntia spp.	1998	-0.10	0.40	0.827	-0.36	0.93	0.726	0.00	0.01	0.967
	2000	0.18	0.23	0.506	-0.69	0.51	0.269	0.05	0.03	0.205

Table 3. Continued

Forage	Year	% crude protein			% Ca			% P		
		d	SE	*P*	*d*	SE	*P*	*d*	SE	*P*
Prosopis velutina	1998	2.78	1.07	0.080	0.02	0.21	0.946	0.04	0.03	0.301
	2000	-0.40	0.96	0.706	0.02	0.27	0.960	-0.02	0.01	0.037
Simmondsia chinensis	1998	2.21	2.49	0.439	-0.37	0.11	0.048	0.02	0.03	0.411
	2000	-2.35	0.55	0.024	-0.20	0.26	0.502	-0.03	0.01	0.069
Sphaeralcea spp.	1998	1.64	0.81	0.136	0.00	0.25	1.000	0.04	0.02	0.201
	2000	-1.28	1.03	0.304	0.12	0.15	0.490	-0.03	0.10	0.789
Quercus turbinella	1998	0.85	0.33	0.080	0.02	0.33	0.961	-0.01	0.01	0.359
	2000	-1.33	0.65	0.134	-0.73	0.19	0.029	-0.08	0.02	0.043

[a] Composite of available species

to be selective when browsing, while the small rumino-reticular volume to body size of deer suggests they are adapted to a diet of young grass, forbs, and browse (Hanley 1982). Due to the selectivity of deer, the quality of deer diets may improve far more than any improvement observed in forage (Hobbs and Spowart 1984). Even in ruminants that are less selective, such as steers, esophageal samples have been higher in crude protein and ash content and lower in crude fiber, N-free extract, and acid detergent fiber than hand-clipped samples (Rao et al. 1973). Although deer are selective foragers, nutritional contents of forage can reveal potential deficiencies or other problems on which to focus future diet research.

Phosphorus and protein levels are important on western ranges (Dietz 1965). Protein content decreased in forage and in the diet of elk on a burned area compared to elk in an unburned area (Rowland et al. 1983). Forage from a burned ponderosa pine forest in Arizona was higher in crude protein for the first growing season after the fire (Pearson et al. 1972), and vegetation in a 5-year-old burn had higher protein content than unburned areas in the Prescott, Arizona area, but not on an 8-year-old burn (Swank 1956). However, our results suggest that in most cases protein content was neither higher nor lower on the burned area within 2 years of the fire and actually may have been lower on the burned area 4 years after the fire. The 4 forage species that contained significantly more protein in the burned area in 1998 (i.e., catclaw acacia, desert ceanothus, mesquite, and shrub live oak) did not maintain that difference 4 years after the fire.

Most forage tested contained enough protein to meet maintenance requirements (i.e., 10%; Short 1981) for deer. The protein requirement for optimal growth of young deer is 13 – 16% of the diet (French et al. 1956). Several forage species have high enough protein content in the Three Bar Wildlife Area that deer should be able to select a diet to meet the protein requirements for antlerogenesis, growth, and fawn production. The seasonal variations we observed in the forage tested were consistent with previous research that has shown protein content is usually higher in

Table 4. Mean differences (*d*) in crude protein, Ca, and P content between years (1998 minus 2000) of deer forage samples collected seasonally (*n* = 4) in burned and unburned interior chaparral, Walnut Canyon Enclosure, Arizona, 1998 and 2000.

		% crude protein			% Ca			% P		
Forage	Burn status	*d*	SE	*P*	*d*	SE	*P*	*d*	SE	*P*
Acacia greggii	Burned	3.24	2.69	0.315	-1.62	0.77	0.127	-0.27	0.01	0.067
	Unburned	-1.22	3.13	0.723	-1.42	0.59	0.095	-0.36	0.01	0.038
Calliandra eriophylla	Burned	0.71	3.84	0.865	-0.63	0.26	0.092	-0.32	0.08	0.032
	Unburned	1.62	3.42	0.669	-0.39	0.25	0.218	-0.26	0.06	0.023
Ceanothus greggii	Burned	2.61	1.88	0.259	-0.82	0.23	0.040	-0.18	0.01	0.001
	Unburned	0.57	1.36	0.704	-1.36	0.43	0.050	-0.20	0.03	0.006
Cercocarpus spp.	Burned	2.24	1.93	0.330	-1.58	0.11	0.001	-0.21	0.02	0.001
	Unburned	-0.28	0.96	0.788	-2.02	0.07	<0.001	-0.30	0.03	0.002
Eriogonum spp.	Burned	0.61	3.47	0.871	-0.67	0.06	0.002	-0.22	0.08	0.084
	Unburned	-0.41	3.50	0.914	-0.63	0.28	0.109	-0.22	0.10	0.127
Forb mix[a]	Burned	-3.67	10.16	0.742	-0.82	0.52	0.210	-0.45	0.31	0.240
	Unburned	-3.41	8.22	0.706	-1.06	0.94	0.341	-0.37	0.23	0.214
Grass mix[a]	Burned	4.51	5.18	0.448	-0.08	0.13	0.585	-0.22	0.10	0.122
	Unburned	2.91	4.84	0.591	-0.06	0.28	0.846	-0.17	0.12	0.258
Krameria erecta	Burned	1.67	1.06	0.213	-0.74	0.05	0.001	-0.26	0.02	0.001
	Unburned	1.75	1.20	0.243	-0.78	0.20	0.029	-0.31	0.05	0.007
Opuntia spp.	Burned	1.57	1.00	0.215	-6.23	0.69	0.003	-0.27	0.02	0.002
	Unburned	1.84	1.36	0.270	-6.56	1.31	0.016	-0.22	0.05	0.025

Table 4. Continued.

Forage	Burn status	% crude protein *d*	SE	*P*	% Ca *d*	SE	*P*	% P *d*	SE	*P*
Prosopis velutina	Burned	3.53	2.07	0.186	-2.05	0.41	0.016	-0.25	0.06	0.023
	Unburned	0.35	1.74	0.852	-2.05	0.15	0.001	0.31	0.04	0.004
Simmondsia chinensis	Burned	5.50	3.12	0.176	-1.62	0.35	0.020	-0.10	0.04	0.089
	Unburned	0.94	0.95	0.397	-1.45	0.35	0.025	-0.15	0.02	0.008
Sphaeralcea spp.	Burned	0.53	5.21	0.925	-1.50	0.59	0.084	-0.45	0.19	0.094
	Unburned	-2.39	6.68	0.745	-1.39	0.64	0.120	-0.52	0.19	0.070
Quercus turbinella	Burned	2.07	1.63	0.295	0.00	0.34	0.995	-0.27	0.03	0.003
	Unburned	-0.11	0.86	0.909	-0.75	0.25	0.057	-0.33	0.02	<0.001

[a] Composite of available species

vegetation during growth periods (Swank 1956). Browse species contained more crude protein than grasses (Carpenter et al. 1979). Our results indicate that protein content in grass was less than most forage during dry seasons, but increased during wet seasons and surpassed several of the browse species tested.

Phosphorus deficiencies may contribute to low fawn production in western rangelands (Dietz 1965). The seasonal diet of mule deer on semidesert grass-shrub habitat on the Santa Rita Experimental Range south of Tucson, Arizona appeared to be deficient in phosphorus (Short 1977). Foods of white-tailed deer (*O. virginianus*) in Missouri were low in phosphorus content for most of the year (Torgerson and Pfander 1971). Most forage tested in 1998 failed to meet the phosphorus requirement of mule deer (i.e., 0.20 – 0.25%; Short 1981), though the increase in P observed in 2000 brought the P content of most forage above the minimum requirements. The only phosphorus values high enough to meet mule deer requirements in 1998 occurred during periods of growth in the spring. This is consistent with pre-vious studies that have demonstrated higher P in new growth than senescent forage (Swank 1956, Jones and Weeks 1985, Sowell et al. 1985). Growing forage in the Prescott, Arizona area contained high P content with an average of 0.32% (Swank 1956). Our results for 2000 indicate a similar level of P in forage, though the extreme difference observed between 1998 and 2000 suggests a high level of annual variability.

Adequate amounts of Ca are provided by most western rangelands (Dietz 1965). Most forage tested contained high levels of Ca. A high Ca diet (i.e., 0.62%) has previously yielded abnormal cartilaginous formation of the coccyx vertebrae (Ullrey et al. 1973). Excess dietary Ca is not detrimental when P levels are adequate, though it might heighten deficiencies in elements such as P, Mg, Fe, I, Zn, and Mn (National Research Council 1980).

Urness and McCulloch (1973) reported high ratios of Ca:P (i.e., >5:1) on the Three Bar Wildlife Area and speculated that if high Ca levels inhibited P metabolism, fawn mortality could increase. Our results indicate that Ca:P ratios continue to be high on this range and none of the forage tested provided high P content with low Ca content that could be consumed to offset high Ca intake. However, we did not test all available forage and may have missed important sources of macroelements and trace elements. For example, deer will eat mushrooms (Longhurst et al. 1979, Wood and Tanner 1985), which have been reported to be high in P (e.g., 2.14%) and low in Ca (Jones and Weeks 1985, Scrivner et al. 1988). Mushrooms are available in the enclosure seasonally but we did not collect them for nutritional testing. In addition, geophagy can contribute a significant portion of trace elements to ungulate diets (Arthur and Gates 1988). We did not test the local soils for mineral content and recognize that deer could depend upon soils to augment the mineral content of their diet.

Though we did not document many changes in the nutritive quality of forage following the Lone Fire, the actual diet quality of mule deer still may have increased. Hobbs and Spowart (1984) documented significant increases in the diet quality of mule deer and mountain sheep (*Ovis canadensis*) following a wildfire despite minimal changes in forage nutritional quality. Fire related improvements in ungulate diets are not limited to nutrient levels in specific species as ungulates may shift their diets to include plants such as forbs and grasses that can increase after fire (Hobbs and Spowart 1984).

Important deer browse on the Three Bar Wildlife Area included false mesquite, jojoba, mountain mahogany, and shrub fruits (e.g., acorns, jojoba nuts, acacia legumes; McCulloch 1973). Acorns are a nutritious and generally sought-after supplement to poor winter diets in Michigan for white-tailed deer (Duvendeck 1962). In 1998, legumes from catclaw acacia and mesquite and jojoba nuts were more abundant in unburned chaparral. During 1998 and 2000, the jojoba nuts produced in the burn were smaller than those produced in unburned chaparral, and individual plants produced fewer nuts in the burn. We were unable to find acorns in the burned chaparral in 1998. During 2000, we were unable to locate legumes or acorns in either the burned or unburned chaparral. Tall shrubs in the burn appeared to invest more energy in regeneration and those in unburned chaparral appeared to invest more energy in reproduction. However, insufficient precipitation appeared to have severely limited reproduction in shrub live oak, mesquite, and catclaw acacia in 2000.

MANAGEMENT IMPLICATIONS

Fire does not appear to have improved long-term crude protein, Ca, or P content of forage on the Three Bar Wildlife Area. Our sampling suggests P content may be highly variable annually and Ca levels are high. We recommend that future research should try to determine the source of variability in P content of forage, assess seasonal diets and diet quality of mule deer on this range, evaluate the possible effects of consistently high Ca levels on deer health and survival, and determine nutritional content of regenerating plants ≤2 years post-fire.

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A COMPARISON OF PRONGHORN HORN SIZE IN RELATION TO ENVIRONMENTAL FACTORS

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Abstract: Pronghorn (*Antilocapra americana*) horn size may be related to the animal's age and climate factors. We found that northern states and provinces produced proportionately fewer trophy pronghorn than states farther south ($P < 0.02$). This same phenomenon also occurred from east to west; the opportunity for obtaining a trophy pronghorn is statistically greater west of the 100th Meridian. These relationships appear to be related to climate, especially winter temperatures ($r^2 = 0.35$; $P < 0.01$). Moreover, a relationship between horn growth and environment may vary in time and space as we found a significant relationship between the number of days below freezing and annual horn growth measurements of pronghorn on a ranch in southwestern New Mexico ($r^2 = 0.33$; $P < 0.06$). This relationship was much improved when summer precipitation amounts were also considered ($r^2 = 0.64$; $P < 0.02$), indicating that age and environmental factors may influence horn size more than genetics.

MANAGING WILDLIFE IN THE SOUTHWEST 2006: 49–54

Key words: Allen's Rule, *Antilocapra americana,* Arizona, climate, horn growth, New Mexico, pronghorn, temperature, trophy.

Arizona habitats are known for producing pronghorn with large horns (Einarsen 1948, Seton 1953, Hoffmeister 1984). Seven of the top 10 pronghorn currently recorded as trophies by the Boone and Crockett Club (B&C) are from Arizona, which consistently produces more trophy animals than states having larger pronghorn populations (O'Gara and Morrison 2004). The conventional explanation for this phenomenon has been that pronghorn live longer in Arizona due to the state's mild winters, and that the larger males are older animals (O'Connor 1961). Recent studies, however, have shown that males with the longest horns and greatest BC scores are 2 to 6-year old animals (Mitchell and Maher 2001, 2004; Brown et al. 2002).

Another explanation for this phenomenon is suggested by Allen's Rule (Allen 1887), which states that the limbs and extremities of closely related vertebrates are shorter and more compact in individuals living in cold, northern climates than those living in hot, dry regions (i.e., the ears, noses, and legs of closely related mammals in polar areas tend to be shorter than those living in warm, arid environments). One hypothesis for this phenomenon is that it is easier to warm and maintain a heat balance in attenuated extremities than in lengthy appendages. Conversely, larger surface areas tend to facilitate heat dissipation, and a commonly given example of Allen's Rule are hares (*Lepus* spp.) that range from the relatively short-eared, compact arctic hare (*L. arcticus*) of the far north to the long, lanky antelope-jack rabbit (*L. alleni*) of northwest Mexico and southern Arizona (Brown and Lomolino 1998).

We hypothesized that Allen's Rule may also apply to horns, which have been documented as a major source of heat loss when growing (Picard et al. 1994). Unlike cervids, which grow antlers during the warm season, pronghorn horn sheaths grow mostly during cold winter months (O'Gara 2004). In cold, northern areas, long horn sheaths, like ears, should come at a higher metabolic cost than shorter ones. Although there is some debate as to whether heat conservation is more

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important to northern mammals than heat dissipation is to desert ones (Brown and Lomolino 1998), we suspect that geographic variations in pronghorn horn growth are more a function of heat conservation than dissipation as most of the horn sheath's development is during the winter months. It is also reproductively advantageous for a male pronghorn to grow horns as fast and large as he can (Byers 1997), with horns attaining maximum length during his second or third year (Mitchell and Maher 2001, Brown et al. 2002).

Pronghorn horn size may also vary with time and space. Pronghorn entries in the record books have increased since 2000 and 3 additional pronghorn trophies from Arizona have been added to the B&C list of 10 largest entries; 2 of these vying for a new world record (W. Keebler, Boone and Crockett Club, personal communication.). This recent apparent increase in pronghorn with large horns prompted us to compare mean annual changes in horn scores of pronghorn harvested on the Armendaris Ranch in southern New Mexico, where "green" BC scores have been recorded since 1994 (Brown et al. 2002), with annual climate data.

STUDY AREA

A former land grant, the privately owned Armendaris Ranch, New Mexico is now managed by Turner Enterprises. Pronghorn occurred at elevations from 1,375 to 1,525 m, and the mean annual precipitation was <250 mm. The climate was warm-temperate with an average of 213 frost-free days/year (Truth or Consequences, New Mexico). Approximately 105,220 ha of the ranch were classified as pronghorn habitat, in which the primary vegetation was semidesert grassland (Brown 1994) characterized by such grasses and shrubs as black grama (*Bouteloua eriopoda*), palmilla (*Yucca elata*), and Mexican tea (*Ephedra torreyana*). The latter species, along with mesquite (*Prosopis torryeana*) and sand sage (*Artemisia filiforma*), were the only available browse plants other than cacti. Most of the ranch's remaining vegetation was Chihuahuan desertscrub. Wildfires were allowed to burn and no coyote (*Canis latrans*) control was prescribed.

Bison (*Bison bison*) were the only permitted grazing animals. Other large herbivores included an increasing number of gemsbok (*Oryx gazella)* and small populations of mule deer (*Odocoileus hemionus*) and desert bighorn sheep (*Ovis canadensis*). The pronghorn population was subject to climate-induced variations but was estimated to number between 800 to 1,000 animals after winter surveys were conducted in 2000. Limited archery hunting for male pronghorn was permitted in late August and a rifle hunt was conducted in September. Relatively few permits were issued each year and the pronghorn harvest never exceeded 10% of the available males. Monthly precipitation data have been collected on the ranch since 1994 and both precipitation and temperature data were available through NOAA from the nearby (<20 km) Aleman Ranch and Truth or Consequences climatic stations.

METHODS

To test the applicability of Allen's Rule, we compared the number of pronghorn trophies/ estimated 1,000 male pronghorn harvested in each U. S. state and Canadian province by latitude and longitude. We then compared the number of trophies/1,000 males harvested in each state and province with the mean January temperature of climatic stations representative of that state or province's pronghorn habitat (Table 1).

To test whether annual climatic variations might affect pronhorn horn size, we compared annual temperature and precipitation data with horn measurements of pronghorn taken on the Armendaris Ranch the following autumn. We wanted to test if selected pronghorn horn sizes varied annually and to determine if horn growth might be related to winter temperatures the previous winter. Because we deemed it reasonable to assume that variations in nutritional condition and other factors might also play a role in horn growth, we also considered the effects of the summer's precipitation previous to horn shedding and growth (April through August), winter precipitation during horn growth (October through March), and the summer's precipitation prior to the breeding season and harvest (April through August) as possibly affecting horn growth. An important

Table 1. Pronghorn trophies /1,000 males harvested by province and state, 1935 – 2004 in the United States and Canada.

Province/ state	Climatic station	Lat.	Long.	Mean January temp (C)[a]	Estimated harvest 1935 - 2004	Total male harvest	No. Boone & Crockett entries[c]	Trophies/ 1,000 males	No. Safari Club International entries	Trophies/ 1,000 males
SK	Moose Jaw	50.4	105.6	-13.7	75,224	45,135	14	0.3	7	0.2
AB	Medicine Hat	50.0	110.7	-10.2	76,903	46,142	31	0.7	10	0.2
MT	Fort Peck	48.0	106.4	-8.1	1,130,088	678,050	133	0.2	244	0.4
ND	Dickinson	46.9	102.8	-11.1	73,846	37,661	13	0.4	0	0.0
SD	Belle Fourche	44.7	103.9	-4.8	309,982	204,588	22	0.1	1	0.0
ID	Idaho Falls	43.5	112.1	-6.1	86,019	55,912	24	0.4	23	0.4
OR	Malhuer	43.3	118.8	-3.1	48,158	41,175	110	2.7	26	0.6
WY	Casper	42.9	106.5	-5.9	2,367,572	1,430,013	681	0.5	904	0.6
NE	Alliance	42.1	102.9	-4.3	38,742	34,480	20	0.6	6	0.2
CA	Alturas	41.5	120.5	-0.7	15,865	14,596	38	2.6	21	1.4
UT	Dugway	40.2	112.9	-2.4	22,292	17,834	60	3.4	24	1.3
NV	Battle Mtn.	40.6	116.9	-1.2	25,696	24,257	164	6.8	32	4.7
CO	Fort Morgan	40.2	103.5	-4.0	301,234	165,679	80	0.5	76	0.5
KS	Garden City	38.0	100.6	-1.9	4,882	4,223	4	0.9	5	1.2
OK	Goodwell	36.6	101.6	0.7	902	541	1	1.9	0	0.0
NM	Fort Sumner	34.5	104.3	2.8	126,848	116,700	373	3.2	320	2.7
AZ	Chino Valley	34.8	112.5	3.3	32,392	29,285	264	9.0	113	3.9
TX	Big Springs	32.3	101.5	5.9	43,839	42,962	84	2.0	74	1.7

[a] Estimates from O'Gara and Morrison (2004).

[b] Extrapolated from harvest data in Pronghorn Workshop Proceedings. As of 8/17/2004 courtesy of B. Keebler, Boone & Crockett Club.

assumption of our comparisons was that hunters generally select the largest male pronghorn available.

Annual pronghorn harvests are limited to between 10 and 25 males, and hunters are encouraged to take the largest male they and their outfitters can find. Each animal harvested is checked before leaving the ranch, and ranch personnel measure the animal's horns according to the scoring procedures described by B&C.

RESULTS

The number of B&C pronghorn trophies/1,000 males harvested in North America significantly increased from north to south ($r^2 = 0.28$; $P < 0.03$), and from east to west ($r^2 = 0.25$; $P < 0.04$; Table 1). We also found significant correlations between the number of pronghorn trophies/1,000 males harvested in the B&C ($r^2 = 0.35$; $P < 0.01$) and Safari Club International(SCI) ($r^2 =0.36$; $P < 0.01$) record books and the mean January temperatures of weather stations located near each state and province's center of pronghorn distribution (Western Regional Climate Center 2005, Table 1). These data indicated that pronghorn horn size was affected by temperatures and that pronghorn horn growth was greater in the warmer, more southern states than farther north.

Although the annual variation in horn size on the Armendaris Ranch was not particularly large, mean "green scores" for all pronghorn age classes negatively correlated with the numbers of days having temperatures $\leq 0^\circ$ C during the previous winter ($r^2 = 0.33$; $P \leq 0.06$; Table 2). Although the amount of winter precipitation appeared to have no significant effect on horn growth, adding either of the April through August precipitation totals to the mean number of days <0º C significantly improved the probability value in a multiple regression equation ($r^2 = 0.64$; $P \leq 0.02$; Table 2). Our hypothesis that pronghorn horn growth is

Table 2. Climatic and pronghorn horn measurement information from Armendaris Ranch, New Mexico, 1994 - 2005.

Year	April – Aug. precipitation (cm) prior to period of horn growth	Previous Oct. – Mar. precipitation (cm)	April – Aug. precipitation (cm) prior to the hunting season	No. days < 0 C[a]	Mean Boone & Crockett Club score[b]
1994	0.76	0.0	4.1	99	74.75
1995	4.1	9.9	10.9	65	76.15
1996	10.9	1.5	19.6	63	77.77
1997[c]	19.6	5.3	24.4	55	81.93
1998	24.4	2.0	13.9	102	78.20
1999	13.9	5.8	20.3	66	80.42
2000[d]	20.3	2.8	11.2	80	81.56
2001[e]	11.2	9.4	14.5	91	74.17
2002	14.5	2.5	19.8	94	76.70
2003	19.8	7.6	12.7	81	75.52
2004	12.7	9.9	12.5	100	75.29
2005	11.9	18.3	9.1	61	75.97

[a] Truth or Consequences climatic station.
[b] "Green" scores.
[c] November 1997 temperature data from Aleman Ranch.
[d] February 2000 temperature data from Aleman Ranch.
[e] January 2001 temperature data extrapolated from mean.

determined at least in part to environmental conditions therefore appears valid.

DISCUSSION

We were unable to substantiate an annual relationship between winter temperatures and horn growth in other populations. The only other published horn growth data that we were able to locate were 4 years of horn length data from Middle Park, Colorado (Minn 1997) and 3 years of horn length and base measurement data collected on the Fort Belknap Indian Reservation in Montana (Mitchell and Maher 2001, 2004). Both of these areas are subject to extremely cold winters in comparison to southern New Mexico. Whatever the reason, these more northern measurements showed little relationship between horn size and either the numbers of days ≤0º C the previous winter or summer precipitation ($P < 0.10$).

Horn size may also be influenced by genetics. The largest pronghorn trophy of record was a male killed in 1878 north of the Colorado River on the "Arizona Strip" (Ely 1939). This specimen remains extraordinary even after it was discovered that the base of the horns had been built-up by a taxidermist, making the trophy ineligible for entry into the current B&C record book. This endemic population north of the Colorado River in Arizona was extirpated between 1890 and 1912, however, and pronghorn from Arizona, Montana, Wyoming, Colorado, and Utah were translocated to the "Strip" beginning in 1951 (Munig 2004). Nonetheless, no pronghorn from Arizona north of the Colorado River were entered into either the state or national record books for 40 years after hunts were resumed in this area in 1962 (Lewis 2000; Keebler 2004; Quimby 1997,1999; Comus 2001; Hack and Menzel 2002). Then, in 2003 a male was harvested on the Arizona Strip that tied for a new world record. This individual animal had been translocated to the "Strip" in 2001 from west-central Arizona, the site of numerous trophy animals (Keebler 2004).

Biological rules, including Allen's Rule, are rarely absolute (Geist 1987). One anomaly with the progressions in Table 1 is that pronghorn from Mexico are not included, that country not having a legal hunting season on pronghorn after 1922. According to Allen's Rule, pronghorn from Coahuila and Chihuahua should have horns at least as large as those from New Mexico and Texas, and males from Sonora and Lower California should have the largest horns. But although pronghorn taken in Mexico prior to 1922 are included in some record books (Ely 1939, Ward 2002), evidence that Mexican pronghorn have larger horns than those in the American Southwest is lacking. Hence, the harsh, environmental conditions experienced in Mexico's arid environments may hinder maximum horn development as suggested by O'Connor (1961).

MANAGEMENT IMPLICATIONS

It thus appears that age and environmental factors, especially winter temperatures, influence pronghorn horn size with warm, mesic conditions promoting large pronghorn horns. If age and environmental conditions are the primary determinants in pronghorn horn development, wildlife managers need not be concerned by hunters harvesting a large percentage of a population's trophy animals as genetics are probably of lesser importance in the production of trophy animals. Indeed, the recent increase in trophy entries in the B&C record book (Keebler 2004) is likely due more to an increased harvest of younger animals and "global-warming" than reduced hunt pressure.

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CORRELATION BETWEEN FREE WATER AVAILABILITY AND PRONGHORN RECRUITMENT

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Abstract: Several studies have implied that availability and quality of free water are important to pronghorn (*Antelocapra americana*) populations. However, a relationship between pronghorn fawn recruitment and water availability has not been established. We surveyed water availability and measured water quality during summer across 8 study areas in Arizona, representing different levels of pronghorn population density and fawn recruitment. We regressed estimates of water availability against pronghorn fawn:female ratios, estimated from annual aerial surveys, to determine if availability of free water was associated with fawn recruitment. Pronghorn have been documented to avoid water sources with high pH and dissolved solids (pH ≥9.2 and total dissolved solids ≥5000 ppm). While we found few water sites with dissolved solids levels ≥ 5,000 ppm, 30% of water sources surveyed during June had pH values > 9.2. Availability of free water of acceptable quality was lowest in June and related to previous winter (October-April) precipitation measurements ($r^2 = 0.26$, $P = 0.02$). Estimates of availability of free water and pronghorn fawn:female ratios were related ($r^2 = 0.27$, $P = 0.01$). The relationship between water availability and pronghorn fawn:female ratios should be most profound during dry years, however we found the correlation to be highest during the wettest precipitation year, indicating a possible link to forage availability. The relationship between previous winter precipitation and fawn:female ratios ($r^2 = 0.38$, $P = 0.01$) supports the contention that forage availability is more important than is free water to fawn recruitment.

MANAGING WILDLIFE IN THE SOUTHWEST 2006:55–62

Key Words: *Antilocapra americana*, antelope, Arizona, chemistry, fawns, pH, recruitment, pronghorn.

Water is essential for wildlife to sustain metabolic processes, control body temperature, lubricate joints, and excrete wastes (Robbins 1993). In addition, milk of many ungulates at mid-lactation is comprised of between 70 and 85% water (Robbins 1993). Water can be acquired in 3 ways: through drinking free water, through metabolic processes, or oxidation of hydrogen-containing compounds, and through food items, or preformed water.

Pronghorn in the western United States have several morphologic and behavioral adaptations to conserve water, including decreased water content of urine, decreased respiratory rate, and cessation of panting (Yoakum 1994). Despite these adaptations, pronghorn densities were highest in areas where free water was available (Yoakum 1994). Pronghorn densities in dry desert environments were <1/2.6 km^2, whereas densities in Wyoming and Montana, where free water was available, averaged between 5 and 10 pronghorn/2.6 km^2.

Several studies have investigated the importance of free water for pronghorn, but results are equivocal (Wesley et al. 1970). Radiomarked pronghorn in the Chihuahuan Desert in southern New Mexico stayed within 3 km of livestock drinking tanks (Clemente et al. 1995). Fox (1997) hypothesized that Sonoran

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pronghorn (*A. a. sonorensis*) in Arizona were able to acquire adequate water through succulent forage during most of the year, but females may not acquire adequate preformed water during lactation. Moreover, Ockenfels et al. (1992) located pronghorn fawns <1 km from an identified water source during the first 6 months, potentially due to the increased water requirements for females during lactation.

Apparently pronghorn in Arizona drink free water when it is available, especially during periods when succulent forage, particularly forbs, are not available (Fox 1997). Pronghorn have been shown to avoid water sources with high dissolved solids and high pH, thus water quality affects use (Sundstrom 1968, O'Gara and Yoakum 1992). Because lactating females have an increased water requirement, and may require free water to meet demands, pronghorn populations could remain healthier and have higher fawn recruitment when they have access to free water.

Over the last 15 years, pronghorn have declined throughout most areas in Arizona. In 1987, the statewide population of pronghorn was estimated at 12,000 individuals but declined to <8,000 by 2000 (Arizona Game and Fish Department [AZGFD] 2001). The AZGFD (2001) identified low fawn recruitment as a major contributor to the pronghorn decline, and Lee et al. (1998) identified availability of free water as a factor potentially influencing fawn recruitment. Although studies have found that water availability is associated with pronghorn population densities and habitat selection, the impact of water quality and availability on pronghorn recruitment has not been investigated. We surveyed water quality and availability across several areas of pronghorn habitat in Arizona during the summer lactation period. We then related water quality and availability to fawn recruitment estimates to establish a relationship between water availability and pronghorn fawn recruitment.

STUDY AREA

We conducted this study at 8 sites in 5 grassland regions in Arizona (Fig. 1). We selected sites to reflect a range of pronghorn fawn recruitment based on long term averages of aerial survey estimates in these areas; because some sites were adjacent to one another there was a potential lack of independence. However, these sites represented isolated distinct habitats separated by forested and mountainous areas or highways acting as effective barriers to pronghorn exchange (Ockenfels et al. 1996).

Sites 1 and 2 encompassed approximately 423 and 432 km^2, respectively, in northeastern Arizona in the White Mountain Grassland Wildlife Area, 7 km west of Springerville, AZ (34° 11' N, 109° 18' W) at an elevation of 2,070 m. Four sites were located in north-central Arizona; site 3 encompassed approximately 128 km^2 in Garland Prairie, 7.2 km south of Parks (35° 12' N, 111° 57' W) at an elevation of 2,072 m; site 4 encompassed approximately 86 km^2 on Anderson Mesa, 11 km east of Mormon Lake (34° 58' N, 111° 22' W) at an elevation of 2,194 m; site 5 encompassed approximately 344 km^2 in Lonesome Valley 15 km east of Prescott (34° 44' N, 112° 18' W) at an elevation of 1,550 m; and site 6 encompassed approximately 159 km^2 in the Fain Ranch area, 5 km east of Prescott Valley (34° 37' N, 112° 15' W) at an elevation of 1,550 m.

The remaining 2 sites were located in southeastern Arizona; Empire Ranch (site 7), 8.2 km east of Greaterville (31° 46' N, 110° 139' W) encompassed approximately 215 km^2 at an elevation of 1,462 m, and the Buenos Aires National Wildlife Refuge (site 8), 97 km southwest of Tucson (31° 33' N, 111° 29' W), encompassed approximately 128 km^2 at an elevation of 1,096 m.

Predominate biomes in sites 1 and 2 were Great Basin grasslands with sections of Petran montane conifer forest and Great Basin conifer woodland (Brown 1994). Temperatures ranged from below zero in the winter to 18°C in summer, with mean annual temperature of 8°C, and precipitation of 30.0 cm (National Oceanic and Atmospheric Administration 2003). The mean pronghorn fawn:female ratio (1993-2002) was greater in site 1 (25.9 fawns/100 F) than in site 2 (15.7 fawns/100 F).

Predominate biomes in study sites 3-6 were Great Basin grasslands, with some inclusions of Great Basin conifer woodland (Brown 1994). Temperatures ranged from

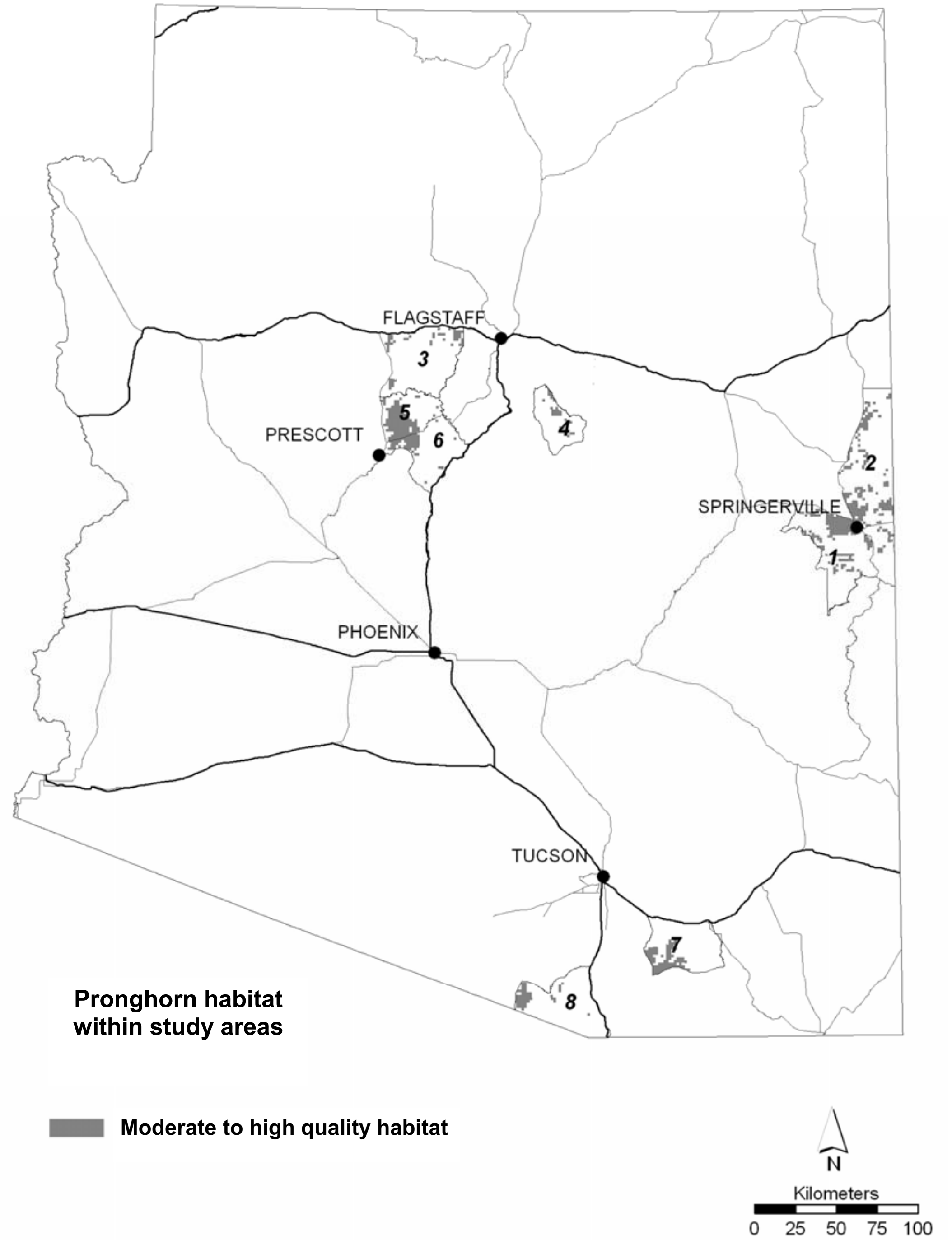

Fig. 1. Arizona study sites (1-8) showing areas of moderate to high quality pronghorn habitat where estimates of pronghorn fawn:female ratios and water quality/availability estimates were collected, 2002 - 2004.

below zero in the winter to 18°C in summer with an annual average of 9°C and an annual average precipitation of 55.0 cm (National Oceanic and Atmospheric Administration 2003). The mean pronghorn fawn:female ratio (1993-2002) was 37.1 (fawns/100 F) at Garland Prairie, 11.7 at Anderson Mesa, 37.7 at Lonesome Valley, and 41.8 at Fain Ranch.

Predominate biomes in study sites 7 and 8 were semidesert grasslands, with some remnants of Sonoran savannah grassland in Buenos Aires (Brown 1994). Temperatures ranged from 9°C in the winter to 26°C in summer, and the mean annual temperature was 18°C in both sites. Mean precipitation is greater at Empire Ranch (50.0 cm) than in Buenos Aires (44.0 cm, National Oceanic and Atmospheric Administration 2003). The mean pronghorn fawn:female ratio (1993-2002) was greater in Empire Ranch (23.2 fawns/100 F) than in Buenos Aires (12.9 fawns/100 F).

METHODS

We identified sources of free water using data, maps, and records from the United States Forest Service (USFS), United States Fish and Wildlife Service (USFWS), AZGFD, and State of Arizona Land Department (SLD). We used 3 available GIS cover layers to initially identify sources of free water in each study site. The SLD digitized all naturally occurring springs in Arizona, and we combined that information with locations of all man-made developments constructed by AZGFD, with a layer identifying all perennial water sources (i.e., lakes, streams, cattle tanks) to create a potential water availability map for each site. Water availability for each study site was verified through ground surveys. Additional sources encountered during ground surveys were also recorded and included in the map.

We visited ≥15 sources in the core of each study site to evaluate pronghorn access to the water at each site (i.e., do fences enclose the source?). We monitored all sites with adequate access monthly from May through August (late gestation through conception for pronghorn in Arizona) for presence of water. We used portable water quality measuring devises (Hanna Instruments, Woonsocket, RI) to determine total dissolved solids (TDS) (mg/L), acidity (pH), and conductivity (salinity) (mS/cm) at each site that contained accessible water.

We estimated water availability within each study site as the number of sources with accessible free water of adequate quality (pH ≤ 9.2 and total dissolved solids ≤ 5,000 ppm; Sundstrom 1968, O'Gara and Yoakum 1992)/ km^2 of pronghorn habitat. We used historic pronghorn survey maps and information from Ockenfels et al. (1996) to delineate the area of pronghorn habitat within each study site. We calculated relative water availability by month to determine seasonal trends in water availability and water quality by site to compare water availability with the identified critical periods for female pronghorn. We estimated annual (2002-2004) winter precipitation for each study site by summing monthly totals recorded during the previous October through April at the nearest weather station (National Oceanic and Atmospheric Administration 2003).

We used a Kolmogorov-Smirnov 1-sample test to determine if frequency dist-ributions of each water quality and availability, and pronghorn recruitment data set differed from a normal distribution (Zar 1999). We used nonparametric tests for all data sets that were not normally distributed. We considered all statistical tests to be significant if $\alpha \leq 0.10$. We compared annual water availability during months where free water was most limited among all study sites using 1-way analysis of variance (ANOVA). We compared pronghorn recruitment estimates and water quality measurements among years and study sites using 1-way ANOVA for fawn:female ratios and pH. We used Kruskal-Wallis ANOVA for TDS and salinity measurements. To describe water quality by study site, we averaged pH, TDS, and salinity for months when water was most limited each year. To determine if availability of free water is associated with pronghorn recruitment estimates, we regressed annual fawn:female ratios on water availability estimates for months when water was most limited.

RESULTS

Water availability across all study sites was most limited during June each year (Fig. 2), and varied more by years than study site (F = 1.37, P = 0.29). Therefore, we restricted analysis of water quality and availability to June each year. Fawn:female ratios varied more by year than by study site (F = 0.71, P = 0.67). Average water quality measurements varied more by study site than by year (Table 1). Although TDS and salinity were greater at site 2 than any other site, average water quality measurements during June usually fell within limits suggested for pronghorn habitats (Table 1). Only 1 site contained water with >5,000 mg/L TDS, while pH was > 9.2 at 37% of sites measured. Across all study sites winter precipitation was greatest in 2004 (Table 2). We found a positive relationship between fawn:female ratios and water availability estimates (Fig. 3). Relationships between annual fawn:female ratios and water availability were strongest in 2004 when average precipitation and water availability were highest (Table 2). We found a positive relationship between fawn:female ratios and previous winter precipitation estimates (Fig. 4).

DISCUSSION

Most areas supporting pronghorn in Arizona are arid habitats where naturally occurring free water is limited. Livestock producers have installed spring improvements, water catchments, troughs, and windmills in most of these sites to provide water for livestock. These water sources, and developments created specifically for wildlife, are readily used by pronghorn and have greatly enhanced availability of free water across our study sites. Pronghorn drink freely when water is available, and water deprivation may affect their health and reproduction (Beale and Smith 1970). Our study indicates a potential connection between availability of free water in summer and pronghorn recruitment in Arizona. However, published literature is equivocal relative to whether pronghorn require free water to sustain viable populations (Fox 1997, O'Gara and Yoakum 2004).

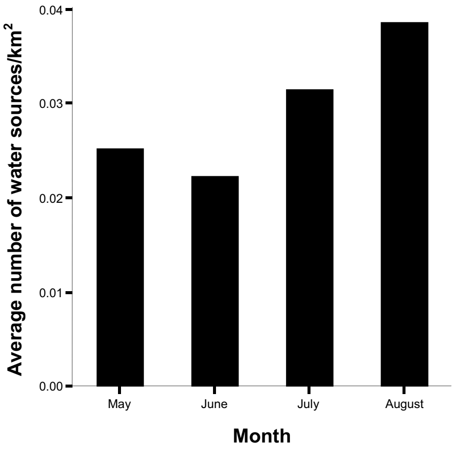

Fig. 2. Monthly availability of free water (average no. of water sources/km^2) at 6-8 sites in Arizona, 2002 - 2004.

We found that pH was the water quality factor that may be most limiting to availability of free water for pronghorn in Arizona. Sundstrom (1968) found pronghorn in Wyoming avoided free water when pH >9.25. This situation occurred >30% of the sources we measured during June. However, pronghorn in Arizona may be less particular about quality of water consumed, especially in drought years when forage production is low and other sources of water (i.e., preformed and metabolic) are less available. Water quality of site 2 was different from the other sites; likely associated with a coal-fired power plant site. The most consistent water source at site 2 was downstream of tailings piles and was the only source where TDS exceeded 5,000 mg/L.

Small livestock ponds in treeless habitats, ideal water sources for pronghorn, are subject to constant mixing from winds and runoff, which tends to increase suspension of particulate matter concentrating TDS (Mortimer 1941). But many catchments and livestock ponds in our study sites dry up each year and thus may resist accumulating TDS that could occur over several years. During June, often the only available water sources at some sites were ponds and troughs that were fed with ground water. Water quality at sites fed by ground water may be more consistent among sources and perhaps more attractive to pronghorn than ponds where precipitation runoff has accumulated and TDS concentrations and salinity have increased with evaporation.

We found water availability and quality was most limited during June across our study sites. This corresponds with critical fawning and

Table 1. Average water quality measurements during June for free water sources in pronghorn habitat at 6-8 sites in Arizona, 2002 - 2004.

		Total dissolved solids (mg/L)[a]		pH [b]		Salinity (mS/cm)[c]	
Site	*n*	x̄	SD	x̄	SD	x̄	SD
1	27	237	237	9.20	0.69	0.48	0.47
2	28	1,011	1,655	8.52	0.87	1.53	1.49
3	18	270	258	8.96	0.95	0.32	0.29
4	11	129	71	8.67	0.37	0.16	0.08
5	9	207	98	9.41	0.86	0.42	0.20
6	33	201	83	9.55	1.02	0.41	0.17
7	43	336	276	9.20	0.86	0.54	0.44
8	16	163	82	9.45	0.85	0.26	0.13

[a] Difference determined by Kruskal-Wallis ANOVA ($\chi^2 = 42.7$, $P < 0.01$).
[b] Difference determined by one-way ANOVA ($F = 4.21$, $P < 0.01$).
[c] Difference determined by Kruskal-Wallis ANOVA ($\chi^2 = 60.6$, $P < 0.01$).

weaning periods of pronghorn populations in Arizona, when water requirements would be highest. Beale and Smith (1970) measured forage use, water consumption, and fawn production of pronghorn in western Utah and found that pronghorn drank from sources of free water only when succulent forage species, particularly forbs, were not available. As a result, they concluded that pronghorn were able to acquire enough water through food sources to meet their needs during part of the year. Availability of preformed water in forage would likely be more limited in the relatively arid pronghorn habitats of Arizona.

Deblinger and Alldredge (1991) found that pronghorn densities in the Red Desert of Wyoming were higher in areas where free water was available in drinking troughs than in areas without free water. Although, once water in the troughs was turned off, distribution of pronghorn did not change. Given that water content of forage was high throughout the experimental period, preformed water may have been adequate to fulfill water requirements (Deblinger and Alldredge 1991). Sundstrom (1968) found that pronghorn densities were much higher in areas that contained free water than in areas without water; 85% of the pronghorn in the study area were located in areas that contained 90% of the free water on the site.

MANAGEMENT IMPLICATIONS

We found a weak relationship between our estimates of availability of free water and pronghorn fawn to female ratios estimated from standard aerial surveys. If water availability was affecting pronghorn fawn recruitment, the effect should be most profound during dry years. However, we found the relationship was most significant during the wettest precipitation year. Most water sites throughout our study area consisted of earthen-dammed livestock ponds,

Table 2. Average (± SD) availability of free water sources during June, previous winter precipitation, and relationship (r^2) to annual pronghorn fawn:female ratios at 6-8 pronghorn habitat sites in Arizona 2002 – 2004.

Year	*n*	No. free water sources/ km² in June	October-April Precipitation (cm)	r^2	*P*
2002	6	0.01 ± 0.007	2.70	0.05	0.67
2003	8	0.02 ± 0.017	6.52	0.03	0.67
2004	8	0.03 ±.0.019	7.04	0.37	0.11

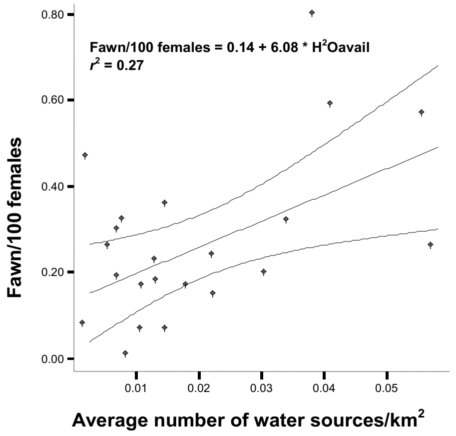

Fig. 3. Relationship of free water availability (average no. of water sources/km^2) during June to annual pronghorn fawn:female ratios at 6-8 sites in Arizona, 2002 - 2004.

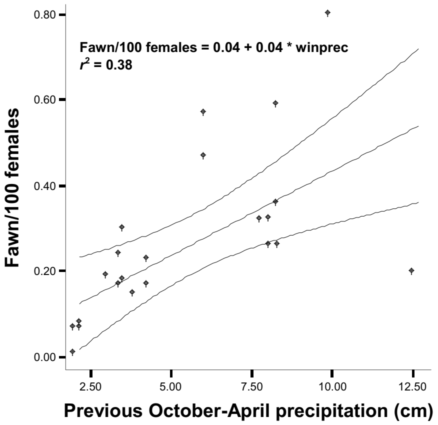

Fig. 4. Relationship of previous winter precipitation to annual pronghorn fawn:female ratios at 6-8 sites in Arizona, 2002 - 2004.

which often dry up during summer. Areas with more ponds available were usually in wetter habitats. During wet years, these habitats would have greatly increased water and green forage availability.

Beale and Smith (1970) found that fawn production was positively correlated with precipitation received during the previous summer, and suggested a link to forage availability. The relationship between winter rainfall and fawn:female ratio that we found was stronger than that for free water availability, supporting a connection to forage availability. If a threshold level of free water availability were exceeded across all study sites then availability of green forage may affect fawn recruitment more than would free water. This threshold level of free water availability would likely vary with availability of preformed water in forage. Models constructed to predict the free water needs of pronghorn based on the amount of preformed water in the forage might be useful in identifying these thresholds (Fox 1997). If a threshold level of free water availability was established, then land management efforts in Arizona could be focused on improving or creating water developments when necessary, and improving vegetative conditions where possible.

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PRONGHORN USE OF EPHEDRA DURING A DROUGHT IN SOUTHWEST NEW MEXICO

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Abstract: Few dietary studies of pronghorn (*Antilocapra americana*) have been conducted in semidesert grasslands and little mention has been made of Mexican-tea or ephedra (*Ephedra* spp.) as a browse species. During summer 2003, we estimated the mortality rate of a pronghorn population in southwestern New Mexico to be nearly 75%. As the severity of the summer drought progressed, pronghorn used ephedra (*Ephedra torreyana*) until many plants showed signs of heavy browsing. Perennial browse is limited in this area, and we suspect this normally winter forage species was used as an emergency food. If pronghorn carrying capacity in semidesert areas is limited by insufficient forage during dry years, browse occurrence and condition is of increased importance.

MANAGING WILDLIFE IN THE SOUTHWEST 2006:63–66

Key words: *Antilocapra americana*, browse, cacti, diet, drought, *Ephedra torreyana*, Mexican-tea, pronghorn, semidesert grassland.

Mexican-tea or ephedra are evergreen, dioecious, mostly spikey, gymnosperms commonly found on sandy and gravel plains in the North American southwest between 600 and 1,825 m (Cutler 1939). In the western United States and Mexico, the genus was a member of the Miocene-Pliocene flora, evolving under a winter rainfall regime concomitant with pronghorn (Benson and Darrow 1981).

All species of *Ephedra*, with the possible exception of *E. trifurca,* are valuable browse plants during winter months when better livestock and big game forage is lacking (Kearney and Peebles 1960). Two species of particular importance in the Southwest are *E. torreyana* and *E. aspera* (= *E. nevadensis*, var. *aspera*), the latter having a life span of about 100 years with most reproduction from adventitious roots and off-shoots (Turner et al. 1995). These shrubs, which are rarely more than 1.5 m tall, are drought adapted with scale-like leaves to conserve water; phototsynthesis takes place mostly through the spike-like stems (Cutler 1939).

Few dietary studies of pronghorn have been conducted in the Chihuahuan Desert and semidesert habitats, and little mention has been made of *Ephedra* as pronghorn food (Miller and Drake 2004, Yoakum 2004). Büechner (1950) rated this plant as only "fair" winter browse. Russell (1964) collected a series of pronghorn rumens from 4 areas in New Mexico and only found *E. torreyana* in samples from Chavez County, the only area representative of semidesert grassland (Brown 1994), where *E. torreyana* was used primarily in winter when it comprised about 2% of the total food consumed.

Smith and Beale (1982), however, found pronghorn to feed on Nevada Mexican-tea (*E. nevadensis*) throughout the year in Utah's southwest desert. They noted that frequency of use was highest in spring (41-70%) and that the species constituted 1 to 4% of the animal's diet although the plant was $\leq$ 5% of the mass of the rumen contents. *E. nevadensis* was an important dietary item for short periods during the summer when it was taken with other browse species such as brickellia (*Brickellia oblongifolia*) and horse-brush *(Tetradymia nuttalli*; Smith and Beale 1982).

While working on the Armendaris Ranch, southwestern New Mexico, we noted heavy use of Mexican-tea during the summer of 2003, and speculated that this use was by pronghorn as bison (*Bison bison*), mule deer (*Odocoileus*

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hemionus), and gemsbok (*Oryx gazella*) were absent or infequent in these pastures. We also assumed that this use was in response to drought as the monthly Palmer Drought Severity Index (PDSI) for southwestern New Mexico progressively went from –1.42 in April 2003 to – 4.29 in September 2003, an indication of "extreme drought" (Palmer 1965). Our objective was to verify pronghorn use of ephedra during drought.

STUDY AREA

A former land grant, the privately owned Armendaris Ranch is now managed by Turner Enterprises. Pronghorn habitat elevations ranged from 1,375 to 1,525 m, and the mean annual precipitation was <250 mm. The climate was warm-temperate with an average of 213 frost-free days/year. Approximately 105,220 ha of the ranch were classified as pronghorn habitat, in which the primary vegetation was semidesert grassland characterized by black grama (*Bouteloua eriopoda*), palmilla (*Yucca elata*), and Mexican-tea. The latter species, along with mesquite (*Prosopis torreyana*) and sand sage (*Artemisia filiforma*) represent the only significant browse plants in pronghorn habitat. Most of the ranch's remaining vegetation was Chihuahuan desertscrub.

Bison were the only livestock and other large herbivores included an increasing number of gemsbock and small populations of mule deer and desert bighorn sheep (*Ovis canadensis*). The pronghorn population was subject to climate-induced variations but was estimated to number between 800 to 1,000 animals after winter surveys conducted in 2000 (Brown et al. 2006). Limited archery hunting for male pronghorn was permitted in late August and a rifle hunt was conducted in September. Relatively few permits were issued each year and the pronghorn harvest never exceeded 10% of the available males. Most wildfires were allowed to burn and no prescribed coyote (*Canis latrans*) control was conducted.

METHODS

Pronghorn herd composition and trend surveys have been conducted each September since 1994 according to procedures described by Lee et al. (1998). In 1999 we began conducting studies on the effects of precipitation and temperature on pronghorn fawn recruitment, horn development, and female mortality (Brown et al. 2002*a*,*b*, 2006). To determine the dietary habits of pronghorn during the 2003 drought, we attempted to collect approximately 100 pronghorn pellets from 6 fresh pellet groups each month from February through July, 2003. These samples of approximately 600 pellets were then sent to Cascabel Range Consultants, Benson, Arizona, for microhistological analysis (Sparks and Malechek 1968).

RESULTS

During the extremely dry summer of 2003, 3 dead females were found with bone marrow conditions indicating animals in a starving condition (Cheatum 1949, Ransom 1965). Fawns tired easily when pursued and appeared small and underweight. Pronghorn harvested on the Armendaris Ranch during autumn 2003 were diagnosed for the first time with epizootic hemorrhagic disease (EHD) and the September survey indicated an annual female mortality rate of nearly 75% (Brown et al. 2006).

As the drought progressed, the monthly progression of plant composition in collected pronghorn pellets reflected the declining availability of such leafy forbs as spurges (*Euphorbia* spp.) and borages (*Boraginaceae*), and an increasing use of Mexican-tea (Table 1). This decline in leafy forage may have even been greater had some of the unidentified food items been known. Nonetheless, we regarded the increased use of snakeweed (*Gutierrezia sarathroe*), cacti (*Opuntia* spp), mesquite, and other less preferred plants as indicative of animals in a stressed condition (Büechner 1950).

DISCUSSION

These diet changes, coupled with a significant relationship between July PDSI values and female mortality (Brown et al. 2006), suggested that a scarcity of winter-spring precipitation reduces the forb diet of pronghorn in semidesert grasslands. A dependence on annual forbs in the summer diets of pronghorn has also been recorded for other Southwest locales (Koerth et al. 1984, Stephenson et al. 1985), and we suspect that the production of winter-spring forbs during drought years can be in-

Table 1. Pronghorn diets on Armendaris Ranch, New Mexico February 2003 – July 2003.

	Percent composition					
Forage	February	March	April	May	June	July
Ephedra	6	2	2	3	10	14
Miscelaneous Forbs	64	61	67	59	27	15
Mesquite	0	0	0	3	15	28
Cacti	8	3	8	11	30	36
Unidentified	25	34	23	22	20	10

adequate to sustain lactating females through the spring-summer drought period. We observed that during the driest summers, such as 2002 and 2003 when the July PDSI was –3.24 and –2.80, respectively, leafy green forbs were lacking, forcing the pronghorn to feed on browse species. These arid conditions also resulted in an increase in the use of cacti and mesquite, probably in response to increased demands for moisture and protein.

We concluded that the reason ephedra was consumed by pronghorn was that it was the most nutritious available forage. Seasonally available herbaceous forbs such as, filaree (*Erodium cicutarium*), the dry mass of which may contain ≤74% protein in February, can contain <10% protein in June, and the plant may be totally unavailable by July (Morgart et al. 1986). Even in normal years, herbaceous browse plants normally preferred by pronghorn, such as winterfat (*Krascheninnikoria lanata*), globe mallow (*Sphaerolcea* sp), Wright's buckwheat (*Eriogonum wrightii*), and blue sage (*Artemisia ludoviciana*), while containing >25% protein during winter, typically retain >10% protein by June or July (Mogart et al. 1986, Rautenstrauch et al. 1988, Krausman et al. 1990, Seegmiller et al. 1990). Other woody perennials such as mesquite and catclaw (*Acacia constrica)* maintain >25% protein throughout the year, but often pose a visual barrier to pronghorn (Ockenfels et al. 1994), and are high in cellulose (Rautenstrauch et al. 1988, Krausman et al. 1990, Lee et al. 1998).

The protein content of ephedra makes this plant an important emergency food for pronghorn throughout the year. Although protein percentages, even during winter, may only range between 14 and 20%, these plants may retain 10-15% protein into July (Morgart et al. 1986, Krausman et al. 1990, Seegmiller et al. 1990). When drought decreases abundance of more nutritious plants, pronghorn use of ephdera increases.

MANAGEMENT IMPLICATIONS

Ephedra and other browse plants normally consumed during winter months should be protected from overuse as these same species may be needed as summer browse during drought years. As drought results in increased interspecific and intraspecific competition for these plants, pronghorn numbers become increasingly density dependent (Kohlmann et al. 1998) and populations decline. The availability of Ephedra and other emergency use plants such as cacti, while not providing highly nutritious forage, allow at least some percentage of a pronghorn population to survive drought periods (Hervert et al. 2005).

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RECOVERY EFFORTS FOR THE SONORAN PRONGHORN IN THE UNITED STATES

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Abstract: Sonoran pronghorn (*Antilocapra americana sonoriensis*) have been listed as endangered longer than most species, yet, until recent recovery efforts they were likely more imperiled than when they were originally listed in 1967 (6 years before the enactment of the Endangered Species Act [ESA]). We reviewed the history of Sonoran pronghorn as a federally listed endangered species and the recovery efforts initiated for the subspecies. The first recovery plan was published in 1982 and revealed that little was known about basic Sonoran pronghorn life history characteristics. By 1998 there was an increase in the knowledge of basic life history parameters, but even as late as 1992, the status of the Sonoran pronghorn population was not clear. In 1992, a systematic effort to estimate their numbers biennially was initiated. A captive breeding program was mentioned in both recovery plans for Sonoran pronghorn (1982 and 1998) and was established in 2004 after nearly 80% of the population perished during a severe drought in 2002. Habitat manipulation for Sonoran pronghorn recovery was not initiated until 2002, when forage enhancement plots (i.e., watering desert vegetation) and emergency water sources were established on portions of their range. Since then, more management has been initiated including land-use restrictions and the creation of additional forage enhancement plots. Currently, these management actions are experimental, but if they are successful they may serve as a model for the conservation of other species in arid environments.

MANAGING WILDLIFE IN THE SOUTHWEST, 2006:67–77

Key words: *Antilocapra americana sonoriensis*, Arizona, conservation, Endangered Species Act, Sonoran pronghorn.

The goal of listing a species as endangered under the ESA is to recover the species from the threat of extinction (Yoakum 2004*a*). An amendment to the ESA in 1978 requires a recovery plan be developed for all endangered species (Clark 1994), outlining the steps that are required for the recovery of the species and designating criteria for delisting (Scott et al. 1996). Recovery plans and efforts made to recover endangered species alone do not always make a difference, as many species with revised recovery plans are more imperiled than they were when their original recovery plan was written (Tear et al. 1995). Reviewing recovery efforts for a species is important to determine what has worked and what has not, and to provide insight into endangered species recovery that may improve the recovery process for other species (Clark et al. 1994).

The Sonoran pronghorn was on the first list of endangered species in 1967 (United States Fish and Wildlife Service 1967), 6 years before the enactment of the ESA. The subspecies is still listed as endangered and until 2003 was probably more imperiled than when originally listed. Reviewing recovery efforts for Sonoran pronghorn is appropriate now because the Final Revised Recovery Plan for Sonoran Pronghorn (United States Fish and Wildlife Service 1998) stated that if actions in the plan were completed successfully, then downlisting of Sonoran pronghorn to threatened was

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anticipated by 2005; an action that did not happen as planned.

We reviewed the history of the Sonoran pronghorn as an endangered species and outline the conservation and recovery efforts initiated for the subspecies before and after its listing in 1967. We limited our review of Sonoran pronghorn recovery efforts to those initiated for the United States subpopulation (2 other subpopulations occur in Sonora, Mexico and are functionally separated by a highway, and agricultural developments; Arizona Game and Fish Department 1981).

REASONS FOR ENDANGERMENT

Sonoran pronghorn were historically distributed in the United States from the Imperial Valley, California, east to the Altar Valley, Arizona, and from near the Gila River in the north to the international boundary with Mexico in the south (Wright and deVos 1986). The current distribution of Sonoran pronghorn in the United States is almost entirely limited to Cabeza Prieta National Wildlife Refuge (CPNWR), Organ Pipe Cactus National Monument (OPCNM), and the Barry M. Goldwater Range (BMGR) (Hervert et al. 2000).

Sonoran pronghorn have existed at densities lower than those found for non desert-dwelling pronghorn subspecies because deserts are marginal pronghorn habitat (Yoakum 2004*a*). Both endangered subspecies of pronghorn (i.e., Sonoran pronghorn and peninsular pronghorn [*A. a. peninsularis*]) inhabit deserts (Yoakum 2004*a*). The Sonoran pronghorn declined from levels during the late 1800s due to over-harvest and loss of habitat (Yoakum 2004*b*). This reduction in numbers occurred rapidly; pronghorn (likely Sonoran pronghorn) numbered in the thousands and by 1907, pronghorn were already rare along the USA/Mexico border (Mearns 1907). By 1924, there were an estimated 105 Sonoran pronghorn in Arizona, USA (Nelson 1925). Unlike other populations of pronghorn in North America, Sonoran pronghorn did not increase in numbers during the 1900s (Yoakum 1968). Estimates (better classified as "guesstimates" because the populations were not systematically sampled) of Sonoran pronghorn numbers in the United States during the 1900s varied between 100-200 and never exceeded 250 (United States Fish and Wildlife Service 1998). The continued low numbers of Sonoran pronghorn in the United States led to their listing as an endangered species in 1967 (United States Fish and Wildlife Service 1967)

Conservation measures were enacted to limit or ban the harvest of pronghorn throughout their range that in turn allowed for recovery of all but the desert-dwelling subspecies (O'Gara and McCabe 2004). Therefore, other factors must play a role in the continued low numbers of Sonoran pronghorn since they have been protected from hunting for nearly 60 years (United States Fish and Wildlife Service 1998). The most commonly cited suggestion for the ultimate cause of endangerment of the Sonoran pronghorn is loss of habitat due to the creation of roads and other barriers to movement and over-grazing by livestock (United States Fish and Wildlife Service 1982, 1998; Wright and deVos 1986). The drying of the Gila and Sonoyta rivers in Arizona and Sonora, respectively, may also have contributed to the decline in numbers of Sonoran pronghorn (Carr 1972). Sonoran pronghorn may have used these areas during dry periods as sources of succulent and nutritious forage and drinking water (Arizona Game and Fish Department 1981).

RECOVERY AND CONSERVATION EFFORTS

Pre-listing

The first conservation effort initiated to protect Sonoran pronghorn occurred in 1923, when a special game warden was appointed to patrol the international border with Mexico to protect pronghorn and bighorn sheep (*Ovis canadensis*) from poaching (Leopold 1959). However, this appointment only lasted a few years (Leopold 1959). This conservation action may have reduced the illegal killing of Sonoran pronghorn but to what extent it benefited the continued existence of the subspecies is unknown. Sonoran pronghorn have been protected from hunting for over 60 years in the United States and the subspecies is still imperiled (United States Fish and Wildlife Service 1982).

An important conservation action for

Sonoran pronghorn was the creation of OPCNM in 1937, the creation of the Cabeza Prieta Game Range (now CPNWR) in 1939, and the creation of the BMGR in 1941 (Phelps 1978). These areas are protected from development and encompass approximately 1.6 million ha (Wright and deVos 1986); Sonoran pronghorn use approximately 610,000 ha of this area (Hervert et al. 2000).

Post-listing

Sonoran pronghorn were 1 of 14 mammals listed by the United States Fish and Wildlife Service (USFWS) as being threatened with extinction under the Endangered Species Preservation Act (United States Fish and Wildlife Service 1967). Following the listing of Sonoran pronghorn in 1967, the Arizona Game and Fish Department initiated a study to collect biological information on the subspecies (Arizona Game and Fish Department 1981). Prior to 1967, no research had been conducted to describe basic life history traits of the subspecies.

The Sonoran Pronghorn Recovery Team first met in 1975 (United States Fish and Wildlife Service 1998). The recovery team produced the first recovery plan for Sonoran pronghorn in 1982 (United States Fish and Wildlife Service 1982). The recovery team set a recovery goal of maintaining an average population of 300 Sonoran pronghorn over a 5-year period in the United States. When this goal was met and the recovery team believed that major threats to the subspecies were eliminated, the USFWS would consider delisting Sonoran pronghorn (United States Fish and Wildlife Service 1982). However, little was known about basic life history characteristics (i.e., survival and mortality rates, home range size, seasonal movements, habitat selection, and productivity, and recruitment estimates) of Sonoran pronghorn (United States Fish and Wildlife Service 1982).

One of the problems the recovery team cited inhibiting the recovery of Sonoran pronghorn was inadequate knowledge of methods to increase the numbers or range of Sonoran pronghorn. The recovery team also stated that while it could be possible to transplant Sonoran pronghorn to other areas as a means of increasing the overall population, at that time there was inadequate knowledge of suitable transplant sites, capture methods, or numbers of animals required to successfully establish a new population (United States Fish and Wildlife Service 1982). The 1982 recovery plan did not outline a proposed method for reaching the recovery goal. Therefore, the objective set forth in the plan was to maintain Sonoran pronghorn numbers until techniques were developed to reach the recovery goal.

Actions proposed in the 1982 recovery plan to maintain Sonoran pronghorn numbers included: population surveys, maximize public ownership of habitat, preserve existing habitat (i.e., minimizing human disturbance and cattle trespass), determine life history, modify limiting factors (e.g., predation, forage quantity and quality, and water) when they are determined, establish a captive breeding population for transplant stock, and reestablish Sonoran pronghorn in historic habitat.

The first conservation action with the potential to increase Sonoran pronghorn numbers was the removal of cattle on most of the current Sonoran pronghorn range in the late 1970s and early 1980s (1978 on OPCNM, 1983 on CPNWR, and 1986 on BMGR; O'Gara and McCabe 2004). On ranges in good ecological condition, cattle and pronghorn do not normally compete for forage (Yoakum et al. 1996), however, on marginal pronghorn habitat (Yoakum 2004*a*) cattle may compete with pronghorn (Ellis 1970). Cattle may also change the vegetation associations so the landscape supports fewer pronghorn (Wagner 1978). Removing livestock from the current range of Sonoran pronghorn may have benefited pronghorn, however, reverting the areas to better habitat for native ungulates may take decades (Valone et al. 2002) or may even be impossible (Van Auken 2000).

Between the mid-1980s and 1990s, 3 studies on life history characteristics of Sonoran pronghorn were conducted (Wright and deVos 1986, Hughes 1991, Hervert et al. 2000). In addition, all fences were removed from guzzlers and drinkers on CPNWR to facilitate their use by pronghorn, OPCNM modified their boundary fences with CPNWR to facilitate pronghorn movements, and the first full-time ecologist was employed at CPNWR (United States Fish and Wildlife Service 1998). Various studies were also conducted to determine what effects military operations on BMGR might have on pronghorn behavior and survival (see Krausman et al. 2005 for a review).

In 1992, a systematic population monitoring program was initiated to conduct biennial population surveys (Snow 1994). At the time, Sonoran pronghorn were the only endangered mammal in Arizona that had not been intensively surveyed, and prior to 1992, there had not been a range-wide population survey (Snow 1994). Therefore, as late as 1992 the population status of Sonoran pronghorn in the United States was not known. Prior to 1992, there had been periodic attempts to estimate pronghorn numbers in the United States, but they were not true estimates and therefore their reliability is unknown. Since 1992, the entire range of Sonoran pronghorn in the United States has been surveyed biennially to obtain population estimates.

In 1996, a population viability analysis (PVA) was used to model the probability of Sonoran pronghorn becoming extinct given population status and conditions present in 1996 (Hosack et al. 2002). The PVA also examined the sensitivity of the remaining Sonoran pronghorn population to varying estimates of population parameters and frequency of severe droughts. Using an es-timate of 100 animals in the population at the start of the modeling exercise, the probability of extinction in the next 50 years was 12%. Results of the PVA also revealed that populations with numbers <100 have a 10 – 65% increased risk of extinction (Hosack et al. 2002). An increase in the frequency of catastrophic droughts (i.e., severe enough to cause >50 % mortality of the population) caused greater population fluctuations, an increase in loss of genetic variation, and a decreased population growth rate. More importantly, the PVA revealed that reduced fawn survival (i.e., <25%) might affect the population more than reduced adult survival (i.e., <78% for males and <90% for females; Hosack et al. 2002).

The second Sonoran Pronghorn Recovery Plan was written in 1998 (United States Fish and Wildlife Service 1998) and updated the recovery criteria based the results of the PVA (Hosack et al. 2002) and the 3 studies on Sonoran pronghorn life history (Wright and deVos 1986, Hughes 1991, Hervert et al. 2000). The new recovery criteria states that Sonoran pronghorn will be considered for downlisting when there are 300 Sonoran pronghorn in 1 United States population, and a second population is established in the United States that remains stable over 5 years, or when numbers are determined to be adequate to sustain a viable population (United States Fish and Wildlife Service 1998). The 1998 recovery plan also states that if actions presented in the plan are successfully completed, Sonoran pronghorn were anticipated to be downlisted to threatened by 2005. The plan also acknowledged that significant aspects of Sonoran pronghorn life history were not known and that this hampered the ability to estimate a delisting date and possibly to develop effective recovery actions.

The 1998 recovery plan, like the 1982 recovery plan, mentioned that captive breeding and the possibility of reintroductions to areas of historic range should be further investigated. The 1998 recovery plan also called for the investigation of habitat modification (i.e., food plots, water catchments, chain fruit cholla [*Opuntia fuligida*] establishment), land-use restrictions in areas of high pronghorn use, and further research on limiting factors.

By the beginning of 2002, none of the actions that were to be investigated in the 1998 recovery plan (i.e., forage plots, captive breeding, reintroductions, land-use restrictions) had been implemented. However, by the end of the year, many of those proposed recovery actions were implemented or were being implemented because nearly 80% of the Sonoran pronghorn population in the United States perished after a severe drought in 2002 (Bright and Hervert 2003).

Hervert et al. (2001) suggested the creation of forage enhancement plots in key areas of Sonoran pronghorn habitat to increase fawn survival by providing lactating females and foraging fawns access to more succulent and nutritious forage during times of the year with limited rainfall. Since 2002, 4 forage enhancement plots have been established (1 in 2002, 3 in 2005). Each of the forage enhancement plots also provides a source of free-standing water for Sonoran pronghorn. Additionally, the 2002 drought spurred the creation of 6 emergency water catchments for Sonoran pronghorn between 2003 and 2004 (Morgart et al. 2005).

Following the 2002 drought, plans were made to implement a captive-breeding program for Sonoran pronghorn (Arizona Game and Fish Department 2003). The plans for a captive-breeding facility for Sonoran pronghorn were modeled after a facility developed for captive-breeding of peninsular pronghorn in Mexico (Cancino et al. 2005). The Sonoran pronghorn captive-breeding facility (enclosure) was built in 2003 and is located on CPNWR (Fig. 1). The enclosure encompasses 260 ha, although only half is currently available to Sonoran pronghorn to manage genetic diversity when more animals are captured. Forage enhancement plots (Hervert et al. 2001) and drinkers were created in the enclosure (Fig. 1) to enhance the natural forage available to captive pronghorn and provide water throughout the year.

Captive breeding began in early 2004 when 2 females from Sonora, Mexico (January capture) and 1 male from the United States subpopulation (April capture) were captured and transported to the enclosure. Four additional females from the United States subpopulation were captured and released into the enclosure in December 2004. At the time of capture, ultrasound revealed that all 4 females were pregnant; most with twins. By mid-March 2005, all 6 females gave birth increasing the total captive population to 17 animals. However, in July 2005, 4 fawns (3 female, 1 male) died from unknown causes, and in November 2005, 1 adult female died (cause of death has not yet been determined). There are plans to capture additional adult females and ≥1 adult male in December 2005.

One of the goals of the Sonoran pronghorn captive breeding program is to produce healthy individuals so a second population of Sonoran pronghorn can be established in the United States (United States Fish and Wildlife Service 1998). To determine where a future reintroduction might occur, a habitat evaluation study was conducted (O'Brien et al. 2005). Six areas outside of the current distribution of Sonoran pronghorn were identified as potential habitat for a reintroduced population (O'Brien et al. 2005). However, the models used in the study only contained coarse vegetation and landscape features (i.e., slope, aspect, biome, distance to wash, and soil type) so future ground-based studies should be conducted to further evaluate the identified areas (O'Brien et al. 2005).

Another conservation effort, enacted in 2002 in response to the catastrophic drought and mentioned in the 1998 recovery plan, was a temporary land-use closure on CPNWR, portions of OPCNM, and surrounding Bureau of Land Management (BLM) lands from 15 March until 15 July each year to limit disturbance to Sonoran pronghorn from recreationists during fawning. While disturbance of Sonoran pronghorn during fawning could be detrimental to individual productivity (Phillips and Alldredge 2000), the effectiveness of this conservation measure is likely reduced because of the increase in numbers of illegal immigrants and the subsequent increase in border law enforcement activity (Goodwin 2000).

The most recent action to benefit Sonoran pronghorn was the retirement of the Cameron Grazing Allotment on BLM land south of Ajo, Arizona in September 2004 (T. Hughes, BLM, personal communication). This allotment is Sonoran pronghorn habitat and removal of cattle, and the subsequent removal of fences may allow more pronghorn to use the area. This action may increase the number of Sonoran pronghorn that can be supported on their current range by increasing access to available habitat and allowing more flexibility in responding to seasonal rainfall events.

DISCUSSION

Recovery efforts for the Sonoran pronghorn over the last 3 decades have focused on studying the subspecies' natural history and potential impacts of military operations (Krausman et al. 2005; Krausman et al. 2004) while little habitat manipulation to benefit Sonoran pronghorn occurred until recently (United States Fish and Wildlife Service 1998). O'Gara and McCabe (2004) suggested that listing as endangered under the ESA has not hastened the recovery of the Sonoran pronghorn. To effectively conserve an endangered species, reasons

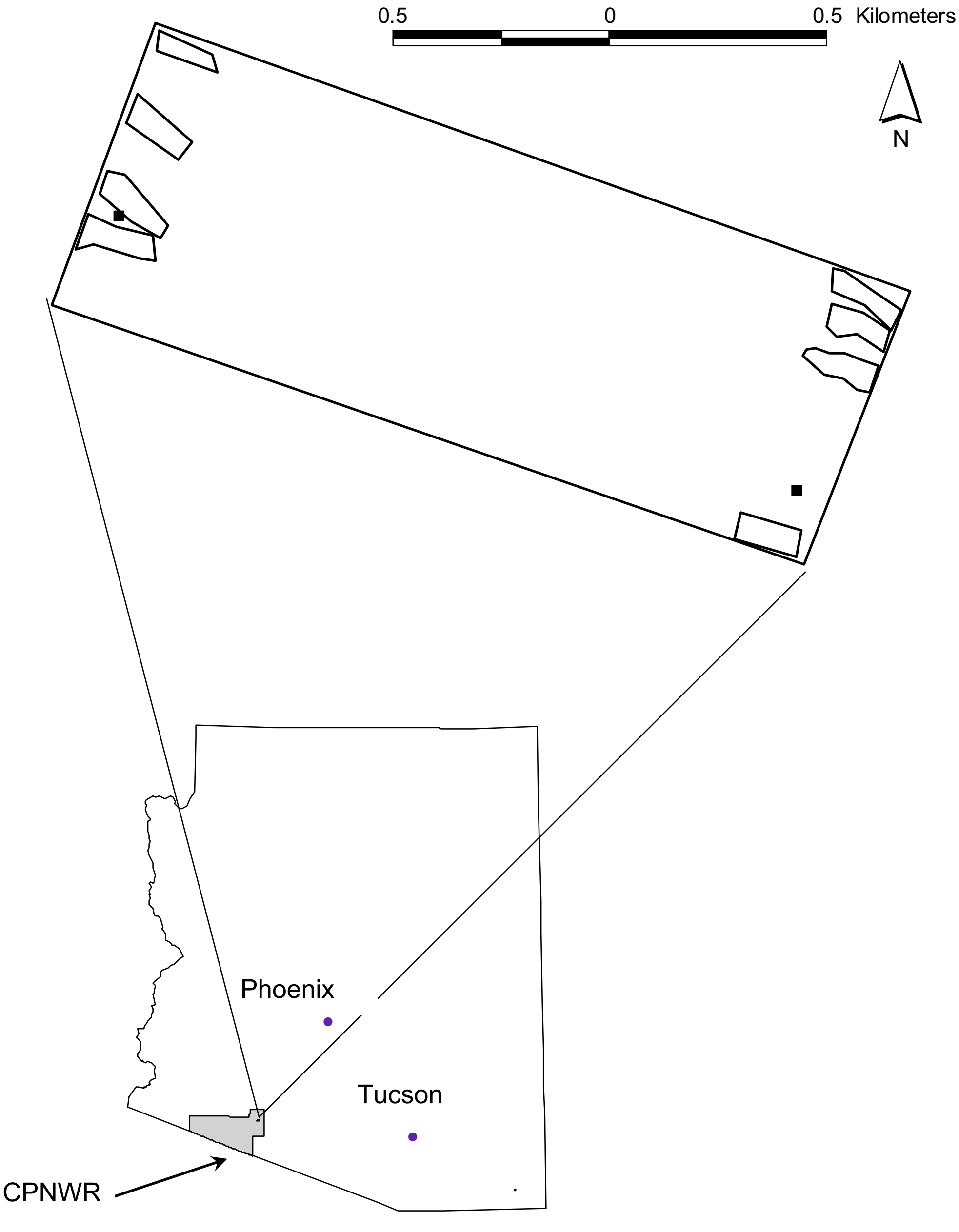

Fig. 1. Location of forage enhancement plots (□) and waters (■) in the currently occupied portion of the Sonoran pronghorn captive breeding enclosure on Cabeza Prieta National Wildlife Refuge (CPNWR), Arizona, 2005.

why the species is imperiled and what factors contribute to this imperilment must first be determined (Scott et al. 1996), which requires knowledge of a species' natural history.

Biologists and managers lacked information on basic life history characteristics of Sonoran pronghorn until 3 studies (Wright and deVos 1986, Hughes 1991, Hervert et al. 2000) were completed between the mid-1980s and 1990s. Estimates of survival and mortality rates and of productivity and recruitment are important for endangered species management because they allow biologists to determine potential factors limiting population growth. Biologists can then develop strategies to increase survival and recruitment to stimulate population growth even if the limiting factors are only proximate causes of the species' endangered status (Mills et al. 2005). Knowledge of home range size, seasonal movements, and habitat use are also needed for effective management of endangered species because they identify the minimum area needed to maintain an individual, habitat requirements, and important areas that need to be protected for survival of the species (Hervert et al. 2005). This information can then be used to more effectively implement habitat management by considering habitat preferences of the species (Hervert et al. 2005), and to find potential habitat for future reintroductions.

However, knowledge of a species' natural history will not facilitate recovery unless concomitant recovery actions can minimize or eliminate limiting factors. Implementation of recovery actions is probably the most challenging part of the recovery process (Culbert and Blaire 1989). Until these basic life history data were known, efforts to manage the proximate factors of Sonoran pronghorn endangerment were not suggested and implemented. Both the 1982 and 1998 Sonoran pronghorn recovery plans discussed further research into implementing habitat management actions and captive breeding (United States Fish and Wildlife Service 1982, 1998), however, these actions were not initiated until the middle of a severe drought, during which there was an 80% reduction of an already small population (Bright and Hervert 2003).

Prior to 2002, much had been said about potential negative impacts of a severe drought on the remaining Sonoran pronghorn in the United States. The results of the 1996 PVA (Hosack et al. 2002) suggested that an increased frequency of catastrophic droughts increased the probability of extinction over the next 100 years by 46%. It was therefore recommended that management actions that reduce the impacts of drought on a population be implemented (i.e., provisioning of food and water) to reduce the chances of the population going extinct (Hosack et al. 2002).

Hosack et al. (2002) noted that it may also be beneficial to establish a captive population to guard against the extinction of the remaining United States subpopulation of Sonoran pronghorn. The 1998 recovery plan stated that "actions that result in a decrease in mortality rates for adults and juveniles would be expected to provide the most drastic benefits for Sonoran pronghorn" (United States Fish and Wildlife Service 1998:26). An extreme drought provided the impetus for the initiation of recovery efforts mentioned in 1982, 1998, 2001, and 2002 (United States Fish and Wildlife Service 1982, 1998; Hervert et al. 2001; Hosack et al. 2002).

Forage enhancement plots and captive breeding may provide the best tools for protecting the remaining Sonoran pronghorn in the United States from extinction. Vegetation manipulation is common management technique for increasing the number of pronghorn that can be supported on an area (Yoakum et al. 1996), but this is the first time it has been implemented to help increase Sonoran pronghorn numbers. Because one goal of Sonoran pronghorn recovery is to increase the population size, it is important to initiate management actions that assure adequate forage is available (Yoakum 2004*a*). One of 2 situations requiring the manipulation of habitat to increase pronghorn numbers occurs when either food, water, or cover are limiting factors (i.e., forage and water in the case of Sonoran pronghorn; Fox et al. 2000) and the possibility exists for improvement of those factors (Yoakum and O'Gara 2000). Forage enhancement plots (Hervert et al. 2001) will hopefully increase survival and recruitment by allowing individuals to meet their nutritional demands, especially during periods of drought, pregnancy, and lactation (Fox et al. 2000, Koerth et al. 1984, Hervert et al. 2001).

Forage enhancement plots are still an experimental management tool as there have been no studies that show the plots are increasing survival and recruitment of Sonoran pronghorn. However, there are plans to radiocollar 5 free-ranging Sonoran pronghorn in December 2005 to determine use of forage enhancement plots and any benefits the plots may confer (J. J. Hervert, AGFD, personal communication). The size and number of forage enhancement plots that will be adequate to enhance forage for the population is unknown and a study to quantify the increase in forage quality and quantity has not been conducted.

Some studies have indicated that supplemental feeding of wild ungulates is either ineffective or detrimental to the management of those populations. In a supplementally fed population of white-tailed deer (*Odocoileus virginianus*), as density increased so did neonatal mortality of fawns born to 2 and 3 year old females (Ozaga and Verme 1982). Also, in other studies of deer, when limited food is provided to starving individuals in a patchy environment, males usually dominate other deer in obtaining forage (Ozaga 1972, Grenier et al. 1999). Supplemental feeding of elk (*Cervus elaphus*) did not increase fecundity, but may have influenced sex ratios at birth in favor of males (Smith 2001). These studies present possible implications of forage enhancement plots for Sonoran pronghorn. While the potential exists for forage enhancement plots to be positive, they may not be effective. Therefore, a study should be conducted to determine the effects of forage enhancement plots on forage quantity, quality, and water content. Until such a study is conducted, forage enhancement plots should continue to be operated.

The Sonoran pronghorn captive breeding program has potential to aid in the conservation and recovery of the subspecies in the United States. In addition to serving as a source of stock for supplementing the existing wild subpopulation, the program will be able to provide a source of animals for translocations to portions of historic range. The 1998 recovery plan (United States Fish and Wildlife Service 1998) suggested that the most effective recovery effort for Sonoran pronghorn may be expanding the current range of Sonoran pronghorn and the 1982 recovery plan (United States Fish and Wildlife Service 1982) also discussed translocation of Sonoran pronghorn as a way to increase their numbers.

Other endangered species recovery programs have been successful in rearing individuals in captivity for translocations to reestablish populations in historic habitat (Stüwe and Nievergelt 1991). The captive breeding facility for peninsular pronghorn (Cancino et al. 2005) has been successful in rearing large numbers of individuals for eventual release into historic habitat, but the release of animals into habitat has not occurred. Raising Sonoran pronghorn in a large enclosure in their habitat likely increases the chances that they will exhibit natural behaviors once released and, therefore, will increase the chance of successful future reintroductions, as has been demonstrated with black-footed ferrets (*Mustela nigripes*) reared in a naturalistic captive environment (Vargas et al. 1999). The Sonoran pronghorn captive breeding facility could also be a useful tool for increasing genetic diversity, especially after nearly 80% of the United States population perished in 2002. This will be accomplished by capturing and transferring Sonoran pronghorn from Mexico into the enclosure (Arizona Game and Fish Department 2003).

The ability to save an endangered species becomes more limited when fewer animals exist. Tear et al. (1995) recommended that aggressive and proactive efforts need to be initiated sooner than later for the conservation of endangered species. In the case of Sonoran pronghorn, funding for their conservation has recently increased, likely due to the near extinction of the United States subpopulation. Additionally, the amount of research on Sonoran pronghorn has increased as there were more peer-reviewed publications on Sonoran pronghorn from 1996-2005 (*n* = 17) than from 1926-1995 (*n* = 10) (Krausman et al. 2005). It is important to review past recovery efforts for Sonoran pronghorn to determine past successes and shortcomings of the recovery program. Managers should then focus on maximizing the effectiveness of the current recovery efforts by investigating their efficacy (Jarman and Brock

1996) and by implementing future recovery efforts experimentally (Sinclair 1991). More effective recovery efforts will aid in reaching the eventual goal of recovery and serve as a model for the recovery of other threatened and endangered species.

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REDUCING FERAL HOG ACTIVITY NEAR DEER FEEDERS: COMPARING COTTONSEED AND PELLETED SUPPLEMENT

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Abstract: Feral hogs (*Sus scrofa*) are an agricultural pest causing damage to ranch infrastructure, predation of game bird nests and young animals, and possibly spreading disease to livestock. Numbers of feral hogs are increasing, partially due to use of supplemental feed provided for deer (*Odocoileus* spp.). I investigated whether replacing pelleted deer feed with EasiFlo™ cottonseed, which is inedible to hogs, will reduce hog foraging activity around deer feeders. I measured hog activity through predation rates on artificial quail nests set on 100-m radial transects around 6 deer feeders. After 28 days only 1 out of 120 nests near the 3 feeders filled with pelleted supplement remained intact, yet at the 3 cottonseed feeders 28 nests (23.3%) were untouched. Based on nest predation characteristics, and photographic evidence, feral hogs were probably responsible for predation of 58% of nests near the pelleted supplement feeders and 38% of nests around cottonseed feeders. Remaining predated nests were attributed mainly to armadillos (*Dasypus novemcinctus*), and were not related to the product in the deer feeders. Hog activity around deer feeders can be reduced by using a supplement such as cottonseed, which is palatable to ruminants but toxic to monogastric animals.

MANAGING WILDLIFE IN THE SOUTHWEST 2006:78–84

Key words: artificial nests, armadillo, cottonseed, feral hogs, game birds, supplemental feed, Texas, white-tailed deer.

Year-round supplemental feeding of white-tailed deer (*Odocoileus virginianus*) is becoming an increasingly common management practice in south Texas rangelands, where the production of quality white-tailed males is a multi-million dollar industry (Crozier and Anderson 2002). The most common supplemental feeds are 16-20% protein alfalfa- or grain-based pellets. These feeds are also eaten by a wide variety of non-target wildlife species (Rollins 1996, Lambert and Demarais 2001). Feral hogs are one of the most problematic non-target species due to the damage they cause to ranch infrastructure, predation on other wildlife and capacity to transfer disease to livestock (Mapston 2004). Populations of feral hogs are expanding and access to supplemental feed is an important factor causing the increase in numbers (Taylor 1991, Rollins and Carroll 2001).

Non-target animals attracted to deer feeders do more harm than just eating feed designated for commercially valuable wildlife. Survival of simulated wild turkey (*Melagris gallopavo*) nests is greatly reduced near deer feeders due to predation by non-target wildlife species attracted to the feeders (Cooper and Ginnett 2000). Feral hogs are major predators of game bird eggs (Rollins and Carroll 2001). Quail hunting is an important and profitable wildlife industry. A survey of Quail Unlimited members revealed that in 1999 quail hunters spent $1.33 billion on their sport (Rollins 2002). Given the value of quail hunting, and an overall decline in northern bobwhite (*Colinus virginianus*) and scaled quail (*Callipepla squamata*) populations throughout their range (Sauer et al. 2000), any wildlife management practice that further imperils the survival of quail populations needs to be altered. Development of a supplemental feed that benefits deer but does not attract problem species is necessary.

Whole cottonseed is a high-protein supplement (>20% crude protein) used extensively for cattle, although gossypol within cottonseed limits the use of this product for swine and poultry. Gossypol is a phenolic substance toxic to

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monogastric animals even at low levels, but tolerated by ruminants due to deactivation by the rumen microflora (Reiser and Fu 1962, Morgan 1989). In captivity feral hogs avoid eating cottonseed, and when forced to eat cottonseed they show weight loss and signs of toxicosis (Huston and Rollins 2002). EasiFlo™ (Cotton Incorporated, Cary, NC) cottonseed is whole cottonseed with a starch coating that is designed to flow through conventional deer feeders. My objective was to determine whether replacing high-protein pelleted supplement for deer with cottonseed would reduce the activity of feral hogs around deer feeders due to the low attraction of cottonseed for monogastric animals.

STUDY AREA

The study site was a 3,237-ha private ranch, in Zavala County, Texas (N 28° 58', W 99° 47'). Mean annual precipitation was 56 cm (Texas Agricultural Experiment Station Weather Station website). The ranch was primarily managed for sport hunting of trophy white-tailed deer, elk (*Cervus elaphus*), northern bobwhite, and for cattle production. The natural rangeland vegetation is dominated by honey mesquite (*Prosopis glandulosa*) with an under-storey of whitebrush (*Aloysia gratissima*) and other shrubs. The ranch has an active wildlife management program and contained numerous brush control areas, irrigated fields, forage plots for deer and quail, water sources and gravity feeders providing pelleted supplemental feed for deer. The ranch feeds locally manufactured 16% protein pelleted feed (Lyssey and Eckle, Poth, Texas) to white-tailed deer and elk year-round. Prior to initiation of the study, the ranch conducted an intensive program to reduce raccoon (*Procyon lotor*) populations by trapping around the deer feeders. Feral hogs provide sport hunting and were not removed. As a result, feral hogs were the primary large non-target species using deer supplements for the duration of the project.

METHODS

I matched 3 replicate pairs of feeders on the basis of landscape and vegetation characteristics, and distance from water, high fences, and agricultural fields. All sites were in natural vegetation and were ≥0.5 km from attraction points such as other feeders, water, irrigated fields or food plots. At each pair of feeders I randomly selected 1 to remain stocked with pelleted supplement while the other was gradually converted to EasiFlo™ cottonseed throughout March 2002. This gave the deer a transition to the new food over 1 month. Starting on 1 April 2002 only EasiFlo™ cottonseed was provided in these 3 feeders.

I measured feral hog activity by recording predation rates on simulated quail nests. Nests were distributed at a higher density than is natural for wild quail to increase sample size. I used nest predation rates as an index of predator activity, not to quantify the natural risk of nest predation for wild quail. At each site I laid 4 100-m x 40-m belt transects radiating out from the feeder. Orientation of the initial transect at each site was at random, and subsequent transects were set at 90° to the first transect, but with the provision that no transects followed roads, fence lines or drainage lines. I placed 10 artificial quail nests, consisting of 4 commercial bobwhite quail eggs, on each transect line (i.e., 40 nests/site). Nest points were distributed at 10-m intervals along the line at random distances between zero and 20-m either side of the line. To decrease the chance that predators would associate location markers with the nests, I flagged bushes near nest sites 2 weeks before the eggs were set out. Natural cover for the eggs was poor due to inadequate rainfall in the previous season. So I placed eggs in a simulated nest to protect them from aerial predators, which do not eat deer feed and would confound the results. Quail typically fold the leaves of overhead grasses to create a canopy over their eggs (Lehmann 1984). However, nests constructed of hay were eaten by elk and cattle; therefore, I constructed the artificial nest from half a brown paper sandwich bag laid on its side. The top half of the bag was shredded and used as nest material for the eggs. A silver foil marker hidden under the nest helped to locate nest sites where the entire nest was destroyed by predators. Artificial nests were placed in the nearest suitable cover to the random nest point to imitate nest site selection of wild quail, ideally a grass clump >20-cm tall and about 30-cm wide (Lehmann 1984), but often under alternative vegetation. Furthermore, to ensure independence of nest sites, any nest site visible from any previously established nest site was re-randomized.

The study was conducted in late May to coincide with the peak breeding season of northern bobwhites in south Texas (Rosene 1984, Guthrey 1986). I monitored nests 2 times/week. I did not replace predated eggs but returned eggs that had been rolled from the nest. After 3 weeks I replaced all uneaten eggs with fresh eggs to prevent spoiled eggs from altering the attraction of the nests to predators. The study was terminated prematurely at 28 days, when access to the study site was blocked by flooding following torrential rains (45 cm) over south Texas.

I placed Trailmaster™ automatic cameras at the feeders to monitor the presence of potential nest predators. To identify species responsible for nest predation I recorded patterns of egg disappearance and shell breakage and placed hair-traps, consisting of short metal stake wrapped in double-sided sticky-tape, beside 1 randomly selected nest on each transect. Hairs adhering to the traps were identified by use of a forensic hair key (Yates 1999). After the main study, I placed automatic cameras over additional artificial nests to identify characteristic nest predation patterns of animals.

I classified a nest as predated as soon as ≥1 egg was eaten, because once a nest has been discovered by predators it rarely survives (Cooper and Ginnett 2000). I compared nest predation at feeders stocked with pelleted supplement or EasiFlo™ cottonseed using survival analysis (SAS Procedure LIFETEST). This is a nonparametric estimation of the survival distribution function. I tested significant differences in nest predation by the Wilcoxon test. I also used SAS Procedure CATMOD for categorical data analysis to assess effects of distance from the feeder on predation rates.

RESULTS

Nest Predation Patterns

Predation of artificial quail nests around feeders stocked with pelleted supplement was greater than around feeders containing EasiFlo™ cottonseed (Wilcoxon test, $\chi^2 = 74.01$, $P < 0.0001$) (Fig. 1). Distance from the feeder had no effect on nest survival. At 2 sites with pelleted feed 42.5% of the nests were destroyed after only 1 week on the ground, and none of the nests survived >18 days. The third site had more grass cover, and nest predation rates were initially slower, but after the second week, predation was heavy. After 28 days only 1 of the original 120 nests remained intact. At the cottonseed feeders, there was a steady slow attrition of nests. Predation rates varied somewhat among sites but ≥50% of nests were still intact on all the cottonseed sites after 14 days. After 28 days, nest survival rates at the cottonseed sites varied from 5 to 55%, with an overall mean survival rate of 23.3% (± 22.5%).

Predator Identification

Repeated damage to the cameras by elk preclude accurate quantification of frequency of visitation to feeders by non-target animals, however, photographic records can still be used to give some indication of which animals frequented the feeders. At the pelleted supplement feeders, the most commonly photographed non-target animals were feral hogs ($n = 10$), rabbits (*Lepus californicus* and *Sylvilagus floridianus*) ($n = 10$) and wild turkey ($n = 5$). Only the hogs are predators of bird nests (Rollins and Carroll 2001). A single raccoon was photographed. Several small animals including Mexican ground squirrels (*Spermophilus mexicanus*) and birds also visited the pelleted supplement feeders. At the cottonseed feeders the only non-target animal photographed was a single raccoon. Despite evidence from hair caught on barbed wire around the feeders before their conversion to cottonseed, no hogs were detected at the feeders after they were filled with cottonseed.

Cameras set over additional artificial quail nests revealed that armadillos ($n = 11$) and feral hogs ($n = 8$) were the most common predators at the nests. Coyotes (*Canis latrans*) ($n = 4$) and raccoons ($n = 2$) were less common, and no skunks (*Memphitis memphitis*), opossums (*Didelphis virginiana*), or avian predators were recorded. Ground squirrels ($n = 8$) investigated the nests but did not appear to break any eggs. Hair-traps set at nests became contaminated with grass awns and sand. Other than 1 record of a feral hog, the hair-traps were of little use in identifying nest predators.

Nest predators also can be identified, but

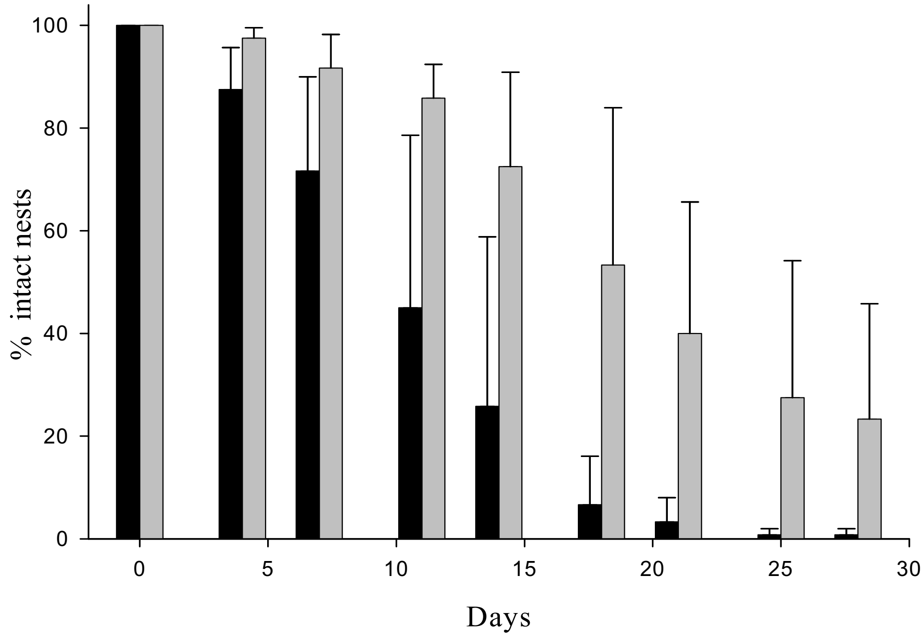

Fig. 1. Mean survival ±1 SD of artificial quail nests set within 100m of deer feeders (*n* = 3) filled with alfalfa-based pelleted supplement (black) or EasiFlo^{TM} cottonseed (grey), June 2002, Zavala County, Texas.

with less confidence, by patterns of predation on the nests. Partial predation of nests occurred on 38.0% of the nests; generally, these nests were completely consumed by the next monitoring visit to the site. Quail eggs are small, so partial consumption of nests indicates predation by small animals. A hole had been bitten through the side or end of 57% of the eggshells found, suggesting they were eaten by small animals. The number of partially eaten nests was similar at all 6 sites. Full predation of nests, often with no eggshell remnants found, was attributed to larger mammals. Based on photographic evidence, feral hogs were the most likely predators. I confirmed that hogs consume the entire clutch of eggs, including shells, by feeding quail eggs to a group of feral hogs penned at the ranch headquarters. Complete predation of nests was 30% lower at the cottonseed feeders than at the pelleted supplement feeders ($\chi^2 = 4.96$, $P < 0.05$).

DISCUSSION

The rise in numbers and distribution of feral hogs in the southern U.S. has occurred at a time when rangeland is increasingly being managed for wildlife production (Mapston 2004). Improved distribution of water and year round supplementation of deer with high protein feed have been implicated in the rise in the feral hog population (Taylor 1991, Rollins and Carroll 2001). Hogs readily eat from deer feeders and that leads to improved survival and greater reproductive output (Clark et al. 1996). Hogs are difficult to eradicate. They are nocturnal, take cover in dense vegetation, and are highly fecund (Mapston 2004). Extensive predator removal programs are rarely cost effective (Rollins and Carroll 2001).

Deer feeders can be focal points for animal foraging activity. Deer with access to supplemental feed continue to use native browse (Doenier et al. 1977), and examination of vegetation indicates that their browsing tends to be concentrated around the deer feeders (Cooper et al 2006). Raccoons also show a tendency to forage near deer feeders (Cooper and Ginnett 2000). Hogs are likely to show similar behavior patterns. It is not economically feasible to cease all supplementation of deer marketed for the trophy hunting industry, but replacing the supplement provided for deer with a feed that is less attractive to non-target species may be a possible way to reduce local densities of feral hogs. Cottonseed may be a suitable replacement feed. It has been used as a high protein supplement in the cattle industry for many years but the gossypol content renders it toxic and unpalatable to non-ruminants (Huston and Rollins 2002). Concerns of deer over eating cottonseed and suffering toxicity are unfounded (Cooper 2004); given free choice deer consume <0.5% body mass of cottonseed/day which is the recommended level for ruminants (Arieli 1998).

Feral hogs are major predators of commercially important game birds such as quail and wild turkey (Rollins and Carroll 2001), thus predation rates on artificial nests can be used as an index of hog foraging activity around the feeders. This method was not designed to quantify true predation risk for natural quail nests. Predation rates for artificial nests are generally higher than those of natural nests (Major and Kendal 1996). Artificial nests differ from real bird nests in many ways (Reitsma 1992, Major and Kendal 1996, Butler and Rotella 1998, Pärt and Wretenberg 2002). Predation rates may be more severe on artificial nests due to higher nest density, poor camouflage, human scent trails, and lack of protection by the female. Conversely, predation rates may be lower because predators cannot follow the bird to the nest, artificial nests typically have fewer eggs, and the associated human scent may deter some predators.

Identification of the species responsible for nest predation was inferred from patterns of nest damage, eggshell breakage, animal signs, hair traps, and use of automatic infrared cameras. Except for photography, none of these techniques are particularly accurate (Hernandez et al. 1997, Marini and Melo 1998), but backed up by photographic evidence they suffice to give useful information on which animals are active in the area. At the study ranch feral hogs were major non-target species and large nest predator due to intensive trapping of raccoons prior to the project. Two major suites of nest predators were identified. Small animals, including many armadillos, accounted for the background level of nest predation. Armadillos have previously been reported as predators of quail nests (Hernandez et al 1997, Staller et al 2002), but since they are primarily insectivores their activity was independent of the supplements in the feeders. Near feeders stocked with pelleted supplement, nest predation by large animals was 58% while around feeders filled with cottonseed nest predation was 38%. Hogs were identified as the primary source of this predation. Thus, feeding cottonseed instead of pelleted feed locally reduced hog activity by a third. This means that, the foraging activity of feral hogs can be manipulated by altering the type of feed provided for the deer. Cottonseed may be a useful supplement for this purpose.

MANAGEMENT IMPLICATIONS

One problem with providing supplemental feed to deer, or free-ranging livestock, is that undesirable animals, such as feral hogs, also benefit from the feed. This may lead to problems such as increased predation on the nests of game birds. This study shows that foraging activity of non-target wildlife at and around deer feeders can be reduced by replacing traditional high protein pelleted supplement with a feed that is less palatable to non-target species. Cottonseed fulfills this criterion. It is a source of protein and energy that can only be consumed by ruminant animals. Monogastric animals lack the symbiotic rumen microflora to breakdown the phenolic compounds present in cottonseed. Although used as a high protein supplement in the cattle industry for many years, cottonseed has not found favor as a supplement for deer. The main problem is that fluffy whole cottonseed will not flow through conventional gravity deer feeders. Coated cottonseeds, such as EasiFlo^{TM}, are designed to have better flow characteristics for use in hopper style feeders and should lessen this problem.

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HARVEST METHODS AND HUNTER SELECTIVITY OF MOUNTAIN LIONS IN ARIZONA

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Abstract: Mountain lion (*Puma concolor*) hunting, particularly with hounds, has come under increasing scrutiny by non-hunting publics and has been banned in Oregon and Washington. We compared mountain lion harvest data from Arizona with findings from other western states regarding hunting methods and hunter selectivity. We summarized and analyzed data collected through the Arizona Game and Fish Department Harvest Questionnaire Program and mandatory mountain lion harvest reporting to assess sex and age of mountain lion harvest by various methods. Legal hunter harvest was the leading known human-caused mortality source for mountain lions in Arizona from 1982 to 2002. Mountain lion seasons were open year-round in Arizona, although few lions were harvested during warmer months (i.e., May-September). The sex composition of the harvest did not differ greatly from 1:1. Hunters employed 2 main methods to harvest mountain lions in Arizona: with (i.e., selective hunters) and without (i.e., nonselective hunters) the aid of hounds. Mountain lions taken without hounds are typically harvested by calling, glassing-stalking, or incidental to hunting other game. Selective hunters accounted for 65% of annual lion harvest. Sex ratios of harvested mountain lions differed by hunter method ($\chi^2 = 216.08$, $P < 0.001$). Selective hunters harvested more males (~60%) than females. Seventy-nine percent of these were >2 years of age. Conversely, females comprised >60% of the harvest by nonselective hunters, which is representative of sex ratios in lion populations. Additionally, a much larger proportion of nonselective harvest appeared to be subadults ≤2 years of age (43%). Nonselective harvest accounted for 35% of total annual harvest, and 47% of annual female harvest. Managers can reduce female harvest rate by encouraging selective harvest of mountain lions.

MANAGING WILDLIFE IN THE SOUTHWEST 2006:85–89

Key words: Arizona, hunting methods, hunter selectivity, mountain lion, *Puma concolor*.

Mountain lion populations are regulated by intra- and inter-specific strife, disease, starvation, and human-caused mortality (Anderson 1983). In Arizona, human-caused mortality occurs from legal sport harvest, depredation harvest (i.e., by livestock operators or their agents), and road kills, other accidents, public safety removals, and illegal kills.

Legal sport harvest contributes the most to known human-caused mountain lion mortality in Arizona (Arizona Game and Fish Department, unpublished data). However, no analysis had been conducted in Arizona regarding the effects that harvest method may have on mountain lion sex and age structure.

In Arizona, mountain lion seasons were open year-round. Mountain lion permits were available over the counter, and hunters could legally harvest 1 mountain lion/year throughout the majority of the state. Hunters could legally exceed the 1 lion/year limit if they harvested lions in a Game Management Unit that had an established mountain lion harvest management objective. These harvest management objectives were established if mountain lions were negatively influencing achievement of manage-

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ment objectives for other species (e.g., bighorn sheep [*Ovis canadensis*]).

In Arizona, sport hunters harvest mountain lions with and without the use of hounds. Much of Arizona lion hunting occurs in dry ground conditions (e.g., no snow cover), in difficult terrain and vegetation (e.g., cholla cactus and prickly pear [*Opuntia* spp], dense shrubs, and trees), and extreme temperatures. These all create additional challenges for mountain lion hunters, particularly those using hounds. Mountain lions taken without hounds are typically harvested by calling, glassing-stalking, or incidental to hunting other game.

Hunting mountain lions with hounds is considered the most selective form of harvest (Cougar Management Guidelines Working Group 2005). Shifts in age and sex of harvested mountain lions following a ban of hound hunting was documented in Washington (Martorello and Beausoliel 2003). Nonselective hunters harvested more females, and the average age of harvest declined. We examined mountain lion harvest data in Arizona to determine whether it was consistent with the results reported by Martorello and Beausoliel (2003) and to assess the effects method of take may have on mountain lion population sex and age structure.

METHODS

We compiled harvest data of mountain lions from 1982 to 2002. Reported sport harvest of mountain lions included method of take, sex, and age of harvest. Data were collected through the Arizona Game and Fish Department's mountain lion hunter questionnaire (1982-1988) and the mandatory reporting system (1989-2002). The mountain lion hunter questionnaire was sent to hunters in possession of a mountain lion non-permit tag. The mandatory reporting system required successful mountain lion hunters to report their kill in person or by phone within 10 days of harvest in accordance with Arizona Game and Fish Commission rule. Hunters were not required to physically present a harvested lion for inspection. Successful hunters were asked during reporting about tooth wear and other external characteristics (e.g., presence or absence of spots or leg barring, lactation; Shaw 1983, Anderson and Lindzey 2000) which allowed personnel to classify lions as a subadult (≤2 years) or adult (>2 years of age). Using these data, we were able to obtain method of take, sex of harvest, and age category estimates of mountain lions harvested.

We classified method of take data into selective and nonselective hunters. We tested differences in harvest between the methods of take using chi square contingency tables and *t*-tests (Zar 1984).

RESULTS

More male than female mountain lions are harvested during most years in Arizona by all methods combined ($t = 3.55$, $P = 0.002$). While significant, the difference between male and female annual harvest is not large (1982 to 2002 $\bar{x}$ = 109 and 96, respectively; Table 1). Female mountain lions represent 46% of total annual harvest from 1982 to 2002 in Arizona.

Most (65%) sport harvest of mountain lions in Arizona is accomplished with the aid of hounds (Fig. 1), yet accounts for only 53% of female harvest. Selective hunters harvest more males than nonselective hunters ($\bar{x}$ proportion = 63%; CV = 7.7); females ($\bar{x}$ = 37%; CV = 13.0) are harvested at a lower rate ($t = 10.47$; $P < 0.001$; Table 1). Nonselective hunters did not select males ($\bar{x}$ = 38%; CV = 22.6), and data suggested selectivity toward females ($t = -6.81$; $P < 0.001$; $\bar{x}$ = 62%; CV = 13.9; Table 1).

Adult lions made up an average of 79% (CV = 7.5) of reported harvest by selective hunters from 1982 to 2002. Nonselective hunters harvested subadults from 1982 to 2002 at a higher rate ($\bar{x}$ = 43%; CV = 32.1) than did selective hunters. Hunters using calls to harvest mountain lions were more likely to harvest a subadult ($\bar{x}$ = 49%) than with other nonselective methods.

DISCUSSION

Houndsmen in Arizona harvested adult male mountain lions at a higher rate than other lion hunters. This likely occurred because greater daily movements by males increases the likelihood that their trails are encountered by houndsmen (Anderson 2003), houndsmen avoided releasing their hounds on multiple lion tracks (i.e., females with kittens), or the hunter chose not to harvest a treed mountain lion due to its sex or size. Conversations with hounds-

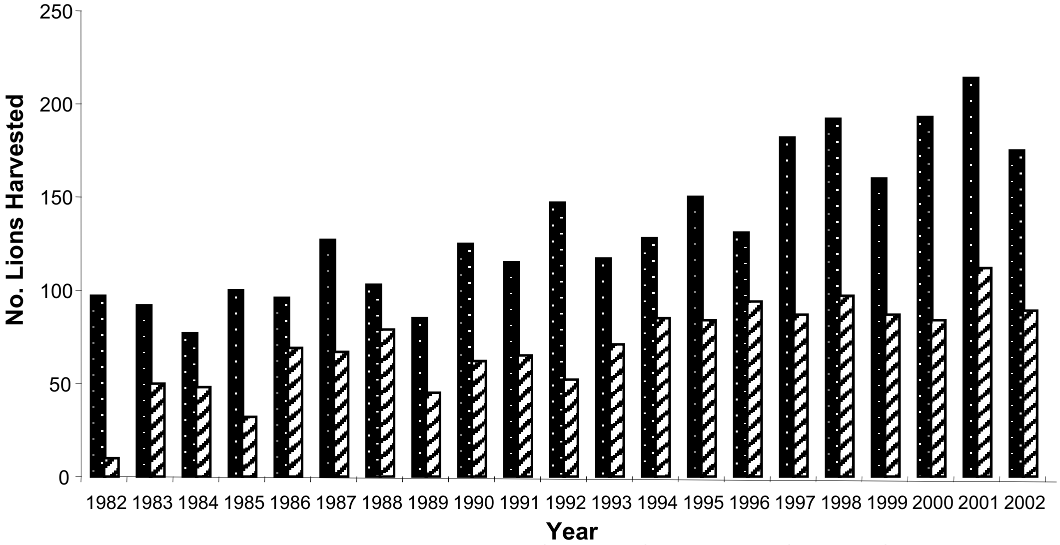

Figure 1. Arizona mountain lion sport harvest with (dark bars) and without (light bars) hounds, 1982 – 2002.

men suggested any of these could have occurred depending on the individual hunter.

Our data suggest a selectivity of females and younger animals by nonselective hunters. However, we do not believe selectivity is occurring. It is far more likely females and younger animals were harvested at the rate they were encountered by hunters and occurred in the population, and no explicit selectivity occurred on the part of these hunters. Hunters without hounds typically harvested lions at longer distances, were less likely to be capable of determining the sex or size of the animal because of the distance, and were more likely to harvest the first lion they saw, regardless of sex or size. Average sex composition data from long-term studies suggest lion populations consist of around 66% females and 33% males (Ross and Jakotzy 1992, Lindzey et al. 1994, Logan and Sweanor 2001, Anderson 2003), which is consistent with observed harvest proportions among the nonselective group.

Female mountain lions make up a larger percentage of annual harvest in Arizona than in neighboring states. The mean proportion of females harvested (46%) is higher than in New Mexico (1987 – 2002 = 39%; Winslow 2003) or Utah (1989 – 2001 = 41%; McLaughlin 2003). New Mexico and Utah use quotas to regulate harvest, which influences harvest proportion.

Managers rely on varying levels of female harvest to manage hunted wildlife populations, particularly with ungulates and large carnivores (Connolly 1981, Taber et. al 1982, Creed et al. 1984, Timmerman and Buss 1997, O'Gara and Morrison 2004, Cougar Management Guidelines Working Group 2005). The greater the annual female harvest, the greater the potential impact harvest may have on population trends. Reducing or increasing harvest intensity on the female portion of a population tends to increase or decrease the population, respectively. States typically use some form of permit or quota system to regulate sport take, through limited entry, total quota, female quota or subquota, or a combination of ≥2 forms (Cougar Management Guidelines Working Group 2005). However, in Arizona, no quotas have been established for mountain lion harvest, with the exception of the southwestern portion of the state where lion abundance is very low. If managers want to reduce the probability of female harvest by hunters, they should encourage the use of hounds in mountain lion hunting.

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Table 1. Mountain lion sport harvest by sex with and without hounds, 1982 - 2002, Arizona.

	Total sport harvest[a]			Harvest with hounds			Harvest without hounds		
Year	M	F	% F	M	F	% F	M	F	% F
1982	63	75	52	49	45	46	1	9	90
1983	72	73	50	54	38	41	16	33	66
1984	65	71	51	44	32	42	17	29	60
1985	82	87	50	57	42	42	13	19	59
1986	85	78	46	63	32	33	21	43	62
1987	109	89	46	76	47	37	29	37	55
1988	82	99	54	61	40	39	20	59	75
1989	77	51	39	58	26	31	19	25	56
1990	108	74	40	82	41	33	26	32	52
1991	107	71	39	79	36	31	28	36	55
1992	113	83	42	91	52	35	22	30	58
1993	106	81	43	76	41	35	31	40	56
1994	120	93	44	87	41	32	32	51	60
1995	126	103	44	97	50	33	29	53	63
1996	119	106	47	86	45	34	33	61	65
1997	134	134	50	98	84	46	36	50	57
1998	150	136	47	117	74	39	33	62	64
1999	126	120	49	93	66	41	33	54	62
2000	133	141	51	112	80	41	22	61	73
2001	176	144	44	137	76	36	39	68	61
2002	144	116	44	108	65	37	36	51	57
Mean	109	96	46.3	80	48	37.4	27	42	62.2

[a] Total sport harvest includes known and unknown method of take.

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MANAGEMENT OF SHRUBLAND SONGBIRDS IN CENTRAL TEXAS: ISSUES AND CHALLENGES

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Abstract: Many shrubland (i.e., low, woody vegetation) -dependent songbirds are species of concern. Except for the endangered black-capped vireo (*Vireo atricapilla*), there has been limited study and management of shrubland-dependent songbirds in central Texas. Based on a review of the literature, I identify topics related to the management of the black-capped vireo and other shrubland-dependent songbirds that need additional research. For example, although prescribed fire and mechanical disturbance are commonly used to create or maintain shrubland habitat, there has been limited assessment of how these treatments affect shrubland structure and avian response (e.g., colonization rates and nest success). Spatially replicated and controlled experiments are needed to evaluate the responses of black-capped vireos and other shrubland-dependent songbirds to management. Additionally, management efforts have been applied primarily on public lands. A full recovery of the black-capped vireo will be dependent on the establishment of viable populations on private lands. However, shrublands on private lands might be too small, or might lack suitable habitat structure for shrubland-dependent species due to over-browsing by domestic livestock and deer (*Odocoileus* spp.) and afforestation resulting from altered disturbance regimes (i.e., the absence of fire). A better accounting of the availability and suitability of black-capped vireo habitat and of the viability of black-capped vireo populations on private lands is needed. Coordinated, landscape-level management efforts to benefit black-capped vireo populations on private lands are also needed. The success of such efforts will likely be contingent on additional landowner incentives and partnerships. Finally, there is a need for a community-level approach to the management of shrubland-dependent songbirds. The response to shrubland management may vary across species. Further, the needs of the shrubland bird community likely differ between the breeding and wintering seasons. Therefore, there is a need to obtain data on species other than the black-capped vireo, and a need to assess whether the black-capped vireo can be used as an umbrella species for the conservation of multiple species of shrubland songbirds during all seasons. Data on the aforementioned topics would further the conservation of the black-capped vireo and other shrubland-dependent songbirds.

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Key words: avian community, black-capped vireo, central Texas, shrublands, *Vireo atricapilla*.

Conservationists have largely ignored native shrublands, which have been severely reduced (Noss et al. 1995). Native shrublands continue to be threatened by development and suppression of natural disturbances such as fire (Askins 2001). In some instances, conservation of more imperiled systems (e.g., grasslands) has taken precedence over shrubland conservation (Herkert 1995). In other instances, society's perceptions of shrublands, which have often been negative (Gobster 2001), have hindered shrubland conservation. Not surprisingly, populations of shrubland-dependent wildlife have declined as their habitat has been degraded and lost (Herkert 1995).

Concern about population declines of shrubland-dependent songbirds has generated research and management in some parts of the U.S., primarily in the east (e.g., Askins 2000, Hunter et al. 2001) and in western sagebrush (*Artemisia* spp.) ecosystems (Knick et al. 2003). In the Southwest (i.e., Arizona, New Mexico, Texas, and Mexico) there has yet to be much focus on shrubland-dependent songbirds. Indeed, the federally endangered black-capped vireo, which breeds in Texas and northeast

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Mexico (Grzybowski 1995), may be the only shrubland-dependent songbird to have received substantial conservation attention in the Southwest.

I reviewed the literature on shrubland-dependent songbirds in central Texas. My objective was to identify issues related to the conservation and management of shrubland songbirds that need to be addressed. Because the conservation of shrubland-dependent songbirds in central Texas has focused primarily on the black-capped vireo, much of my discussion will focus on the black-capped vireo. However, my discussion is pertinent to the management of other shrubland-dependent songbird species. In particular, I comment on the need for additional data on the response of songbirds to shrubland management. I also make a case for expanding research and management efforts. Finally, I comment on the need for a community-level approach to the management of shrubland-dependent songbirds in central Texas.

BLACK-CAPPED VIREO MANAGEMENT NEEDS

The black-capped vireo is a shrubland-dependent songbird with a breeding range restricted to Oklahoma, Texas, and northeast Mexico (Grzybowski 1995). Federally endangered status was conferred on the species because of population declines, high levels of brood parasitism by brown-headed cowbirds (*Molothrus ater*), and habitat degradation and loss (U.S. Fish and Wildlife Service 1991). The listing of the black-capped vireo focused conservation attention on the species, resulting in several research and management programs. The program at Fort Hood Military Reservation, Texas has been the best-funded, and arguably, the most widely known (Eckrich et al. 1999, Hayden et al. 2000, Kostecke et al. 2005, Cimprich and Kostecke 2006). However, research and management have been conducted on other public lands (e.g., Kerr Wildlife Management Area—O'Neal et al. 1996, Dufault 2004), and private lands owned by individuals and non-profit organizations (e.g., The Nature Conservancy). Cowbird removal and habitat management (e.g., mechanical disturbance and prescribed fire) have led to increases in some black-capped vireo populations (e.g., Fort Hood; Eckrich et al. 1999, Hayden et al. 2000, Kostecke et al. 2005). Recent discussion of downlisting or delisting the black-capped vireo has likely been due, in part, to such local recoveries (R. M. Kostecke, personal observation).

The current existence of viable black-capped vireo populations in some locations is an optimistic sign (e.g., Fort Hood; Cimprich and Kostecke 2006). The species may no longer be in imminent danger of extinction. However, despite being listed as endangered since 1987 (Ratzlaff 1987) and, subsequently, receiving substantial conservation attention, there is still much to learn about black-capped vireo conservation (U.S. Fish and Wildlife Service 1991, 1996). Research and management efforts have focused almost exclusively on cowbird management and population monitoring (Eckrich et al. 1999, Hayden et al. 2000, Kostecke et al. 2005, Cimprich and Kostecke 2006, Summers et. al. 2006*a*). Outside of cowbird management, there has been little direct study of black-capped vireo response to management actions.

Therefore, I emphasize the need for well-designed studies (i.e., studies with controls and replication of experimental units) to evaluate black-capped vireo response to management, particularly habitat management. Prescribed fire and mechanical disturbance are useful tools for creating and maintaining shrubland habitat that is suitable for the black-capped vireo (Grzybowski et al. 1994). Indeed, publicly available management guidelines even promote the use of these tools for black-capped vireo management (Campbell 1995). Yet, there has been little rigorous assessment of black-capped vireo response to habitat management. To date, O'Neal et al. (1996) have produced the only peer-reviewed publication evaluating black-capped vireo response to habitat management. Perhaps this lack of rigorous assessment is related to anecdotes, gray literature, and comments published in the peer-reviewed literature (Graber 1961, Grzybowski et al. 1994) which suggest that the black-capped vireo should respond positively to management. Hence, there seems to be little urgency to confirm what is assumed to be fact. However, there are several reasons managers should evaluate the response of black-capped vireos to management.

First, managers need robust data to assess the general applicability of management actions. The need for assessing the generality of management is highlighted by O'Neal et al. (1996), who reported a variable response by the black-capped vireo to prescribed fire. They observed that the number of black-capped vireo territories increased following prescribed burning in most instances, but there were exceptions. Additionally, to truly assess the generality of management (cowbird or habitat management), the response by black-capped vireos needs to be assessed throughout the species' range. Factors such as vegetative conditions and land use vary across the range of the black-capped vireo and such variability could affect the applicability and success of different management techniques (Grzybowski et al. 1994). Peer-reviewed, published data from other sites are needed to confirm the generality of cowbird (Eckrich et al. 1999, Hayden et al. 2000, Kostecke et al. 2005, Summers et. al. 2006*a*) and habitat (O'Neal et al. 1996) management.

Second, studies that evaluate different methods of reaching habitat goals for the black-capped vireo are needed. Evidence suggests that the black-capped vireo may respond differently to different types of disturbance (Bailey 2005, Noa 2005). Further, response to the same type of disturbance may be variable and may not even differ from controls (O'Neal et al. 1996). To fully evaluate black-capped vireo response, managers need to collect data on occupancy (e.g., colonization and re-colonization rates, and territory density) and factors related to habitat quality (e.g., frequency of nest depredation and parasitism, and nest success), especially because occupancy does not necessarily correlate with habitat quality or productivity (Van Horne 1983, Vickery et al. 1992, Bailey 2005). These data then need to be compared among sites disturbed naturally (e.g., wildfire), by prescribed fire, and by mechanical means (e.g., mulching). Such comparisons are needed if managers are to make informed management decisions.

For example, black-capped vireos respond positively to fire (Graber 1961, Grzybowski et al. 1994, O'Neal et al. 1996, Bailey 2005). Additionally, prescribed fire is often the most cost-effective means of maintaining the disturbance-dependent early successional shrublands used by the black-capped vireo (T. A. Greene, The Nature Conservancy, personal communication). However, the use of fire might not always be possible for liability or safety reasons, or due to lack of fuel because of overgrazing. Thus, do black-capped vireos respond equally well to mechanical disturbance? Further, more data are needed on when to apply management like prescribed fire. Hot fires would likely mimic historical fire regimes more closely, but cooler burns may be safer to implement (Grzybowski et al. 1994). Do vireos respond differently to hot versus cool burns? Vegetation response to disturbance should also be compared among disturbance types. Data suggest that vegetative features, such as degree of nest concealment, may be correlated to songbird nest success (Budnik et al. 2002, Bailey 2005, Noa 2005). Yet, there has been little study of how different types of disturbance affect habitat characteristics (e.g., greater cover in lower height intervals and habitat heterogeneity) important to black-capped vireos.

EXPANDING BLACK-CAPPED VIREO MANAGEMENT

The black-capped vireo recovery plan recommends implementation of cowbird and habitat management (U.S. Fish and Wildlife Service 1991). In 1995, Grzybowski (1995:20) commented that broad-scale management for the black-capped vireo was lacking. Over a decade later, broad-scale management for the black-capped vireo is still lacking. Cowbird and habitat management have been implemented, but typically only locally and particularly on public lands (e.g., Fort Hood; Eckrich et. al. 1999, Kostecke et al. 2005). Indeed, the fact that 75% of the known breeding population of black-capped vireos occurs on 4 intensely managed, public properties (i.e., Fort Hood Military Reservation [Texas], Kerr Wildlife Management Area [Texas], Wichita Mountains National Wildlife Refuge [Oklahoma], and Fort Sill Military Reservation [Oklahoma]) underscores the need to more fully assess the status of the species at other locations and to expand management for the species, particularly on private lands (Wilkins et al. 2006).

Granted, cowbird traps have been deployed throughout central Texas and the decrease in the number of cowbirds "dispatched" from these traps over the past several years has been interpreted as an indication of reduced cowbird populations at local and regional scales (Texas Parks and Wildlife Department 2005*b*, 2005*c*). Lower numbers of cowbirds could result in reduced frequency and intensity of parasitism on host species (Jensen and Cully 2005). However, because the majority of trapped cowbirds are migrants that would not have bred locally or regionally (DeCapita 2000, Summers et al. 2006*b*), it is difficult to accurately assess the effects of cowbird management on host populations. Demographic data (e.g., nest success and parasitism frequency) collected from host species suggests that cowbird control has occurred at some locations (e.g., Fort Hood; Eckrich et. al. 1999, Hayden et al. 2000, Kostecke et al. 2005), but such data are lacking for most locations where cowbird management occurs. Further, traps are not uniformly distributed across central Texas. Although some counties have many (≥10) traps, the majority of counties have few (<8) traps (Texas Parks and Wildlife 2005*c*). More uniformly distributed and more intensive trapping efforts may be needed to affect the regional cowbird population, as well as the viability of the regional black-capped vireo population.

Regardless of whether cowbird management is being implemented in a manner that will affect viability of black-capped vireos at a regional scale, habitat (e.g., creation, maintenance, and restoration) needs to be addressed. Decreasing brood parasitism by cowbirds is of limited utility if suitable habitat does not exist within which to maintain and grow populations of endangered songbirds (e.g., Southwestern willow flycatcher [*Empidonax traillii extimus*]; Rothstein et al. 2003). Managers have created black-capped vireo source populations on public lands such as Fort Hood and Kerr Wildlife Management Area (Kostecke et al. 2005, Texas Parks and Wildlife Department 2005*a*, Cimprich and Kostecke 2006). However, with few exceptions (e.g., The Nature Conservancy's Dolan Falls Preserve), there seem to be few viable black-capped vireo populations on private lands. Although extensive shrublands have been identified on private lands throughout the range of the black-capped vireo (U.S. Fish and Wildlife Service 2004), there is limited evidence to suggest that these shrublands are actually suitable for the black-capped vireo (i.e., groundtruthing has been limited). The black-capped vireo needs low, scrubby, and typically deciduous growth with vegetative cover to ground level (Grzybowski 1995), vegetation characteristics that are often missing from private lands where overbrowsing by goats and white-tailed deer (*Odocoileus virginianus*) occurs, or where natural disturbances such as fire have been suppressed, allowing the shrublands to mature into woodlands (Archer 1994, Fuhlendorf et al. 1996). Until the suitability of these private shrublands can be improved, techniques such as con-specific attraction are of little use (Ward and Schlossberg 2004).

COMMUNITY-LEVEL MANAGEMENT

Management approaches that focus on communities and the integrity of ecological systems and processes at landscape levels have recently gained favor over traditional single-species management approaches (Grumbine 1994). The need for an ecosystem management approach for central Texas has been recognized (The Nature Conservancy 2004). Further, data on black-capped vireo associations and interactions with other species are also needed (Partners in Flight 2004). Particularly, do other species of conservation concern (e.g., Bell's vireo [*Vireo bellii*]) co-occur with the black-capped vireo and how are their densities and reproductive success affected by management for the black-capped vireo?

Despite criticisms of single-species management approaches and at least some awareness of the need for data with which to assess the effects of our management at the level of the community, managers have been slow to implement community-level management approaches. Managers' hesitation to implement community-level approaches is likely related to the limited availability of conservation funds. For example, the black-capped vireo has been the only shrubland-dependent songbird to receive any substantial conservation attention in central Texas, primarily because of its status as a federally endangered species. Unfortunately, there are often restrictions on such funding that limit a manager's ability to implement community- or ecosystem-level research and management. For example, management (i.e., cowbird removal and relatively large scale habitat manipulations) for the black-capped vireo at Fort Hood has likely had substantial impacts on avian community structure. However, it has been difficult to justify research, management, and monitoring that focuses on species other than the black-capped vireo or the golden-cheeked warbler (*Dendroica chrysoparia*), another federally endangered songbird (Ladd and Gass 1999). For community- and ecosystem management to work, managers will need to find ways to justify research, management, and monitoring efforts for multiple species and to expand such efforts across administrative and political boundaries.

Therefore, I emphasize the need for a broader approach to the management of central Texas shrublands. In particular, there is a need to expand our information base to include data on populations of shrubland species other than the black-capped vireo. The collection of such data should be justifiable in that, although not federally endangered, several shrubland species are of conservation concern (U.S. Fish and Wildlife Service 2002). Further, managers should not assume that the needs of all shrubland songbirds will be met by our management for the black-capped vireo. Thus, there is a need to explicitly test the utility of using focal species like the black-capped vireo as umbrella species for shrubland conservation.

Finally, managers will need to broaden their temporal scope. Traditionally, songbird management has focused on the breeding season. However, there has been a recent shift towards assessing migratory stopover and winter habitats, which may be just as important for songbird conservation (Yong et al. 1998). Numerous species of conservation concern winter in central Texas shrublands (U.S. Fish and Wildlife Service 2002, Partners in Flight 2004). The needs of these species may be different from those that breed in central Texas shrublands. There are few data with which to assess the status of or the impacts of our management (positive or negative) on songbirds wintering in central Texas shrublands.

MANAGEMENT IMPLICATIONS

Detailed data on shrubland-dependent songbird response to widely applied management actions (e.g., mechanical disturbance and prescribed fire) in central Texas are generally lacking. Consequently, there are few data with which to compare shrubland songbird respon-ses to different types and intensities of disturbance. A better understanding of shrubland-dependent songbird responses to management would allow managers to make more robust management decisions. Managers should adopt an adaptive management approach for the implementation and assessment of management actions (Murphy and Noon 1991).

Further, managers must do a better job at communicating their results. To date, relatively little data related to the management of the black-capped vireo, much less other shrubland-dependent songbirds in central Texas, have been published in peer-reviewed journals. An ample gray literature on the black-capped vireo exists, but gray literature often lacks the fair and rigorous assessment of scientific merit received by papers published in peer-reviewed journals. Also, gray literature is not always readily available to the scientific and management community. Better assessment and communication of the results of management actions will allow managers to make more robust management decisions.

Finally, managers should adopt a broader management approach. Currently, most, if not all, research on and management of shrubland-dependent songbirds in central Texas is focused on the black-capped vireo. It is unknown whether the black-capped vireo can be used as a bellwether species for the status of other shrubland-dependent songbirds or shrubland habitat. There is a need to assess whether the black-capped vireo can be used as an umbrella species for the conservation of the entire shrubland-dependent songbird community.

Additionally, most research on and management of the black-capped vireo have been conducted at a few sites (i.e., Fort Hood [Kostecke et al. 2005, Cimprich and Kostecke 2006] and Kerr Wildlife Management Area [O'Neal et al. 1996, Dufault 2004]). All too commonly, but understandably, the boundaries of research and management are legal, administrative, or political (i.e., property lines) rather than ecological in nature. When possible, research and management needs to be expanded outside of these boundaries so that population- (e.g., source-sink dynamics) and landscape- (e.g., disturbance regimes) level processes beneficial to the black-capped vireo, other shrubland-dependent songbirds, and shrubland habitat can be promoted or maintained. Since the majority of research and management occurs on public lands, incentives in addition to current tax breaks (e.g., Texas Parks and Wildlife Department's Landowner Incentive Program) and safe harbor agreements (Environmental Defense 2003) may be needed.

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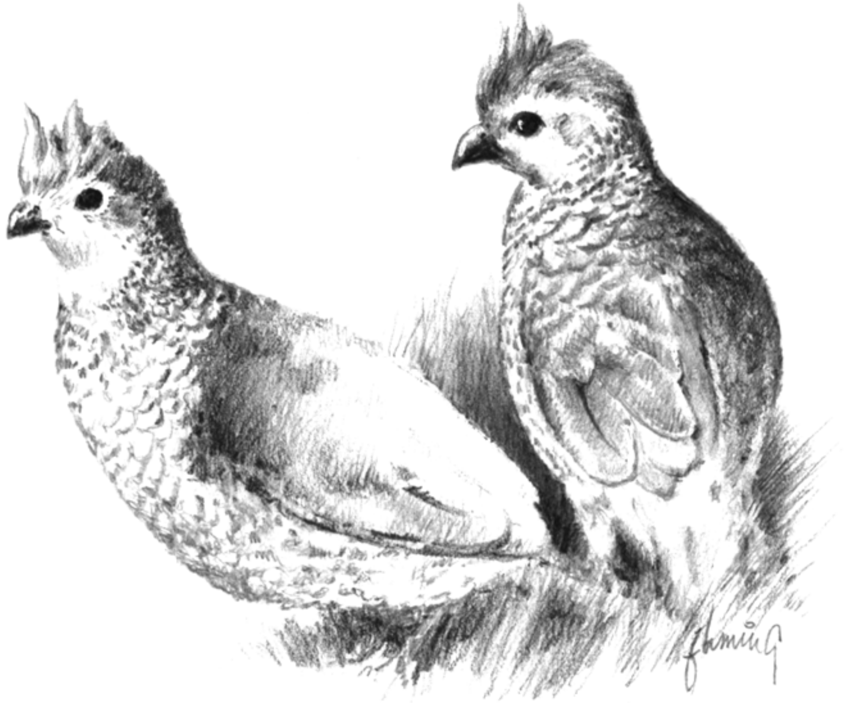

USING LAND COVER TO PREDICT WHITE-WINGED DOVE OCCURRENCE AND RELATIVE DENSITY

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Abstract: Throughout most of Texas, white-winged dove (*Zenaida asiatica*) breed mainly in urban areas. To design an efficient statewide white-winged dove survey, biologists must be able to identify a priori areas where white-winged dove will likely occur. We hypothesized that white-winged dove occurrence could be predicted on the basis of remotely sensed landcover data. To develop a spatial model of white-winged dove occurrence, we examined landcover data from Mason, Texas, and 6 surrounding counties, and identified residential areas according to the 1992 National Landcover Data Set. We estimated white-winged dove density in the residential and non-residential areas within 2 km of Mason, Texas using distance sampling on 275 points situated randomly along streets. Density declined with distance from residential areas, with estimates of 5.27 birds/ha (95% CI = 4.08–6.81) in the residential core, 1.16 birds/ha (95% CI = 0.80–1.68) at ≤500 m from the residential core, and 0.07 birds/ha (95% CI = 0.03–0.17) at 501–2,000 m from the residential core. Approximately 91–94% of white-winged dove occurred within 501 m of the residential core. When we applied our methodology to the 6 counties surrounding Mason County, we achieved similar results; 94% of white-winged dove occurred within 500m of residential areas. Our results suggest that landcover classification offers a powerful tool to predict white-winged dove occurrence and facilitate white-winged dove surveys.

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Key words: GIS, habitat, remote sensing, Texas, urban, white-winged dove, *Zenaida asiatica.*

In recent decades, white-winged dove have undergone a significant range expansion. Prior to the 1980s, white-winged dove in Texas were restricted to the Lower Rio Grande Valley of south Texas and the Big Bend region in west Texas. Since that time, the range of the white-winged dove has expanded to include every region in Texas except the Pineywoods of east Texas. However, white-winged dove populations outside of south Texas have been confined almost exclusively to urban areas (Schwertner et al. 2002).

The Texas Parks and Wildlife Department (TPWD) is required to monitor the population status of all game species, including white-winged dove. Prior to their range expansion, white-winged dove monitoring was conducted using a variety of techniques in the Lower Rio Grande Valley (Berger and George 2004). With the sudden appearance of white-winged dove in urban areas across Texas, a need has arisen to expand the survey effort statewide. Moreover, a recent review of TPWD surveys raised serious concerns regarding their design and the validity of their results (Wildlife Management Institute 2005). Thus, TPWD began an effort to redesign its white-winged dove surveys and implement them at the statewide level.

Because white-winged dove in much of Texas are restricted to urban areas, white-winged dove surveys must be effective in these environments. Recent research suggests that randomly placed point counts in urban areas using DISTANCE (Research Unit for Wildlife Population Assessment, St. Andrews, Scotland) methodology yielded reliable estimates of density (J. B. Breeden, Texas A&M University, Kingsville, unpublished data). However, any statewide survey effort must be designed to encompass the sampling universe of the population in ques-

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tion, to reduce unnecessary survey efforts in areas where the population density is known a priori to be zero.

Initial attempts to place random points in urban areas relied on the survey designer's personal judgment or political boundaries (i.e., city limits) to bound the urban areas of cities in question (M. C. Frisbie, TPWD, personal communication), but we consider neither of these methods satisfactory. Personal judgment is prone to bias and may differ between observers based on experience and opinion as to what constitutes urban areas. Likewise, city limits boundaries are set independent of the environmental characteristics of the surrounding landscape and differ widely on the types and proportions of landcover they encompass. Therefore, our objective was to develop an objective, cost- and time-efficient method for delineating white-winged dove distribution in urban areas to facilitate the placement of random survey points.

White-winged doves in urban areas tend to be associated with residential neighborhoods (West 1993, Mathewson 2002). We hypothesized that white-winged dove occurrence and relative abundance could be predicted based on the proximity to residential areas. Moreover, we hypothesized that this prediction could be made on the basis of remotely-sensed land cover data, without measuring actual environmental variables present at the site. Our approach was to conduct a pilot study of a single urban white-winged dove colony to collect exploratory data regarding white-winged dove population density relative to landcover. We then used these data to construct a simple conceptual model of white-winged dove occurrence relative to residential land cover. Finally, we expanded the project to urban areas in surrounding counties to determine whether the relationship between white-winged dove and landcover across the region was consistent with our initial results in Mason, Texas.

STUDY AREA

We conducted the initial phase of the project in and around Mason, Texas (Fig. 1) during 17 May – 24 June 2005. Mason had a population of 2,163 in 2002 (City-data.com 2005) and is located in Mason County, in the central Edwards Plateau of Texas. The city is relatively isolated, with the nearest other area of significant residential development >45 km away (i.e., Brady, Texas). White-winged doves colonized Mason in the early 1990s and the city currently has a robust white-winged dove population (Mathewson 2002).

During the expanded phase of the project (28 June – 22 July 2005), we collected landcover and white-winged dove abundance data from Gillespie, Kimble, Llano, McCulloch, Menard, and San Saba counties. These counties surround Mason County.

METHODS

Pilot Project

For our analysis, we used the 1992 National Land Cover Data Set. This dataset uses an unsupervised clustering algorithm to classify satellite imagery into 21 land cover classes (United States Geological Survey 1999). We used the ArcView (Environmental Systems Research Institute, Inc., Redlands, California, USA) Image Analysis extension to isolate High Intensity Residential and Low Intensity Residential land cover classes within Mason, combined the classes into a single Combined Residential class (hereafter the residential core), and converted it to an ArcView shapefile for further analysis. We then used the Random Point Generator extension in ArcView to generate 275 random points along streets and roads within the residential core of Mason and a zone extending 2 km from the residential core. We selected 2 km because white-winged dove in the Edwards Plateau nest almost exclusively in urban areas, and our personal experience suggested that few if any white-winged dove are observed >2 km from residential areas. We calculated the distance from each point to the nearest edge of the urban core using the Nearest Feature extension in ArcView.

We sampled each survey point using DISTANCE methodology (Buckland et al. 2001). Surveys were conducted 17 May–24 June 2005, between sunrise and 0930. A single observer recorded the number of white-winged dove seen or heard and the distance to each observation at each point during a 3-minute sampling period. Whenever possible, distances were measured using a laser rangefinder (Bushnell Yardage Pro 500, Bushnell Corporation, Over-

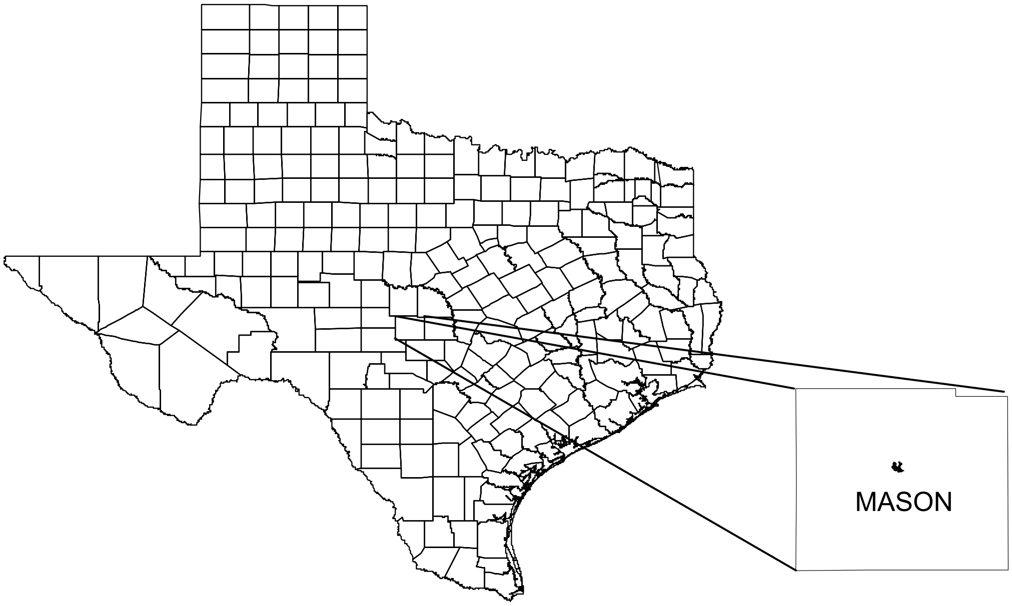

Fig. 1. Location of Mason, Mason County, Texas.

land Park, Kansas, USA); otherwise, distances were estimated by the observer.

We a posterior segregated all survey points into 5 categories based on distance from the residential core: <0 m (within the residential core), 0–500 m, 501–1,000 m, 1,001–1,500 m, and 1,501–2000 m from the residential core. We estimated dove densities in each zone using DISTANCE 4.0 and compared these densities by observing whether the resulting 95% CIs overlapped.

Expanded Phase

Our initial results suggested that the majority of white-winged dove were found within 500 m of the residential core. Thus, we simplified our approach during the expanded phase to determine whether this relationship was consistent across the 6-county area. We delineated all residential cores in the 6-county area as described above. We used a stratified, random sampling scheme to estimate white-winged dove densities in and around the combined residential cores. We randomly placed 100 survey points across the residential core areas of the 6 counties. We placed another 100 points in a zone extending from the residential core boundary to a distance of 500 m. Finally, we placed 100 points in a zone located 501–2,000 m outside the residential core boundary. We used DISTANCE sampling to estimate white-winged dove densities in each of these 3 areas as described above during 28 June–22 July.

RESULTS

Pilot Project

Density differed significantly among the 5 zones. Within the residential core, we estimated white-winged dove density to be 5.27 birds/ha (95% CI = 4.08–6.81). At 0–500 m from the residential core, we estimated white-winged dove density to be 1.16 birds/ha (95% CI = 0.80–1.68). Density declined sharply in zones >500 m from the residential core (Table 1). Because zone area increased with distance from the residential core, white-winged dove abundance in each of the zones declined even more moving outward (Table 1). A small number of observations in the outer 3 zones hampered our density estimates, so we pooled these zones and estimated density and abundance for the entire area located 501–2,000 m outside the residential core (Table 1). Our results indicated

Table 1. Estimated density and abundance of white-winged dove in the residential core and 4 surrounding concentric zones in Mason, Texas, 2005.

Zone	No. observations	Area (ha)	Estimated density (birds/ha)	95% CI	Estimated abundance	95% CI
Residential Core	187	311	5.27	4.08 – 6.81	1,638	1,268 – 2,116
0 – 500 m	99	1,131	1.16	0.80 – 1.68	1,311	904 – 1,901
501 – 1,000m	11	848	0.04	0.05 – 0.35	35	4 – 296
1,001 – 1,500m	18	972	0.13	0.04 – 0.40	124	40 – 385
1,501 – 2,000m	13	1,121	0.11	0.03 – 0.38	118	32 – 430
501 – 2,000m	42	2,941	0.07	0.03 – 0.17	192	76 – 488

that 91–94% of the white-winged dove population in Mason occurred within 500 m of the residential core.

Expanded Phase

The results of the expanded phase of the study were consistent with the pilot phase. White-winged dove density differed significantly among the 3 zones. We estimated white-winged dove density in the residential core zone to be 2.51 birds/ha (95% CI = 1.74–3.61). In the 0–500 m zone, white-winged dove density was estimated to be 0.79 birds/ha (95% CI = 0.42–1.49), while estimated density was 0.03 birds/ha (95% CI = 0.01–0.08) in the 501–2,000 m zone. Likewise, actual white-winged dove abundance was highest in the inner 2 zones, declining significantly past 500 m, with estimated populations of 11,316, 15,371, and 1,750 at <0 m, 0–500 m, and 501–2,000m zones, respectively. Thus, similar to the results of the pilot phase, we estimated that 94% of all white-winged dove occurred within 500 m of the residential core.

DISCUSSION

Our methodology appears to represent an efficient approach for predicting white-winged occurrence in Texas. White-winged dove occur in all ecological regions of Texas except the Pineywoods. The combined total of this area accounts for 617,651 km^2. However, given that white-winged dove nest mainly in urban areas, random sampling of the entire region would be inefficient and probably result in unreliable density estimates because of few observations. A more efficient approach would be restrict sampling to areas that, a priori, are determined to likely harbor birds.

Delineation of possible white-winged dove habitat has heretofore relied on subjective evaluation of habitat characteristics or irrelevant political boundaries. Our results suggest that we can predict the occurrence of 90—95% of white-winged dove in our study area, based on satellite imagery alone. The residential core areas and surrounding 500 m buffer account for <1.4% of the 1,896,907 ha encompassed by our study area; hence, the potential area to be surveyed can be reduced.

We acknowledge shortcomings in our study. Specifically, some white-winged dove populations probably occur outside of the 2-km limit we designated around the residential core areas. However, our experience suggests that, at least in our study area, these are rare. Moreover, our reliance on 1992 NLDS data probably resulted in inaccurate representation of residential areas, although the inclusion of a 500 m buffer probably encompassed change occurring since 1992 and the present.

We believe that this technique holds promise for facilitating TPWD white-winged dove surveys in Texas. In future studies, we intend to assess the usefulness of this technique in other ecological regions and use it as the basis for designating our sampling frame in those ecological regions where it is applicable.

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NESTING HABITAT OF RIO GRANDE WILD TURKEYS

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Abstract: We monitored nesting activity of Rio Grande wild turkeys (*Meleagris gallopavo intermedia*) in the southern Great Plains (i.e., 3 sites in Texas and 1 site in Kansas) from 2000 through 2004 to determine habitat selection, nesting rates and nesting success because some of these populations appeared to be declining. We measured nesting habitat at nests and paired random plots 50 m from each nest. In 2003 and 2004, we measured an additional 200 random plots to estimate available nesting habitat at each study site. We compare nest site selection at local and study area scales. Nest sites had greater height of visual obstruction than paired random plots ($P < 0.001$). This relationship held among all years and all sites except at the Matador Wildlife Management Area, Texas in 2001 ($P = 0.556$). Visual obstruction did not differ between successful nests and unsuccessful nests ($P > 0.05$). At the study area scale females used sites with visual obstruction > 0.4 m and avoided sites with visual obstruction < 0.2 m ($P < 0.001$). In 2004 nests sites with trees in the nesting area were used while sites without trees were avoided across all study sites ($P < 0.001$). Shrubs > 0.4m high provided >70% of the visual obstruction. Managers wishing to improve turkey nesting habitat could remove shrubs too tall to provide visual obstruction while enhancing vegetative structure providing visual obstruction in the 0.4 to 1.0 m heights. Deferring cattle grazing in riparian corridors during the nesting season may increase herbaceous vegetation that provides visual obstruction in the 0.2 to 0.4 m classes.

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Key words: Kansas, *Meleagris gallopavo intermedia,* nest site selection, nesting

Wild turkey population dynamics are poorly understood, despite considerable research (Vangilder 1992). Most studies of Rio Grande wild turkey (herein referred to as turkeys unless specifically stated) have been descriptive, site-specific, and of short duration (Weinstein et al. 1995). Understanding the factors that influence populations is essential to the development of effective management plans (Vander Haegen et al. 1988).

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Nesting success is important to annual population fluctuations of eastern wild turkeys (*M. g. silvestris*; Roberts et al. 1995). One way to prevent a population decline when survival is low is to increase reproductive output (Hubbard et al. 1999). Low nest survival is mainly attributed to nest depredation by mammalian predators (Martin and Roper 1988). Selecting nest sites with greater cover has reduced the risk of nest depredation with some birds (Martin and Roper 1988, Crabtree et al. 1989). Visual obstruction, measured as lateral cover at nests sites of wild turkeys, was greater than at random plots associated with the nest for turkeys in Colorado (Schmutz et al. 1989), South Dakota (Day et al. 1991) and Texas (Hohensee and Wallace 2001).

Studies conflict on whether differences in cover affect nest success. Successful nests had greater visual obstruction than unsuccessful nests for 38 nests of eastern wild turkeys in Mississippi (Seiss et al. 1990), and 40 nests in Arkansas (Badyaev 1995). From 67 nesting attempts by Merriam's wild turkeys (*M. g. merriami*) in Arizona, Wakeling et al. (1998) determined that successful nests had greater visual obstruction than unsuccessful nests. In contrast, no difference in visual obstruction was observed at 34 nest sites for turkeys in Colorado (Schumtz et al. 1989) and at 20 turkey nests in north- central Texas (Hohensee and Wallace 2001). No difference was found between successful and unsuccessful (*n* = 121) nest attempts for Merriam's wild turkeys in South Dakota (Rumble and Hodorff 1993). However, some of these studies had small sample sizes (*n* = 20) or covered $\leq$2 years possibly making it difficult to detect differences.

Our objectives were to determine the nesting rate and nesting success of turkeys at 3 study sites in the Texas panhandle and 1 in southwestern Kansas from 2000-2004. We made 3 predictions. Nesting rate and nesting success varied among study sites and years. Visual obstruction was greater at successful nests than unsuccessful nests. Turkeys used nest sites with greater visual obstruction at third and fourth order scales (Johnson 1980).

STUDY AREA

We studied turkeys nesting at 4 sites from 2000 through 2004. Three study sites were located in the Rolling Plains of the eastern Texas Panhandle, and the fourth site was in the High Plains of southwestern Kansas.

The northernmost Texas site was centered on the Gene Howe Wildlife Management Area (GHWMA) in Hemphill County east of Canadian, Texas. The Canadian River flowed along the southern edge of the GHWMA and contained water year-round. Dominant woody vegetation included sand sagebrush, western soapberry (*Sapindus saponaria*), and hackberry (*Celtis* spp.) in the upland areas, and tamarisk (*Tamarix chinensis*), Russian olive (*Elaeagnus angustifolia*), and western cottonwoods (*Populus deltoides*) in the riparian areas. Grass-es included bluestems (*Andropogon* spp.) and gramas *Bouteloua* spp.) (Hodge 2000). Major land use included livestock production with some dry land and irrigated cropland. The turkey population on the GHWMA was stable to increasing (W. Ballard et al. 2001, Changes in land use patterns and their effects on Rio Grande turkeys in the Rolling Plains of Texas, Annual Report. Texas Tech University, Lubbock, Texas).

The second study site was located south of the GHWMA along the Salt Fork of the Red River. The Salt Fork (SF) site was centered on privately owned land in Donley and Collingsworth counties near Clarendon, Texas. The Salt Fork of the Red River flowed intermittently through the site. Dominate riparian vegetation consisted of black locust (*Robina pseudoacacia)* and western cottonwood. The surrounding rangeland was dominated by grama and bluestem grasses, shinnery oak (*Quercus havardii*), honey mesquite (*Prosopis glandulosa*), and juniper (*Juniperus* spp.). Livestock production was the major land use in the area, but dry land and irrigated cropland were also present. The turkey population in the area was stable or increasing (W. Ballard et al. 2001)

The southern site was on the Matador Wildlife Management Area (MWMA) located north of Paducah, Texas in Cottle County. The Middle Pease River and the Tongue River flowed intermittently through the site. Woody vegetation was dominated by honey mesquite, juniper, hackberry, and western cottonwood. Major grasses included bluestems and gramas (Hodge 2000). Livestock production was the major land use, with other land use including dry land and irrigated cropland. The turkey population was stable or declining on the MWMA (W. Ballard et al. 2001).

The Kansas study site was centered on the Cimarron National Grasslands (CNG), Morton County, Kansas, and included private land in Stevens County, Kansas and Baca County, Colorado. The Cimarron River passed through the study area, and contained surface water year round in some areas. Western cottonwood (and tamarisk dominated the riparian corridor. Sand sagebrush (*Artemisia filifolia*) prairie dominated the uplands adjacent to the river corridor (Cable et al. 1996). Dominate grasses in the area included sand bluestem (*Andropogon hallii*), blue grama (*Bouteloua gracilis*), sideoats grama (*Bouteloua curtipendula*), sand dropseed (*Sporobolus cryptandrus*), sand lovegrass, (*Eragrostis trichodes*), prairie sandreed (*Calamovilfa longifolia*), and buffalo grass (*Buchloe dactyloides*). Major land use on the CNG and adjacent private land was livestock production, along with gas and oil production. Dry land and irrigated agriculture were also present on portions of private land. Rio Grande wild turkeys were declining on the CNG (W. Ballard et al. 2001).

METHODS

We caught female Rio Grande wild turkeys while in winter flocks at each of the 4 study sites using rocket nets (Bailey et al. 1980), drop nets (Glazener et al. 1964), and walk-in traps (Davis 1994). We pre-baited birds at trap sites using milo, whole kernel, or cracked corn. Our target was to maintain 25 adult and 15 juvenile female transmittered turkeys each year at each study site. We captured and transmittered turkeys between January and March of each year, except at CNG in 2002 because no research was conducted at that site. Juveniles were reclassified as adults in January of their second winter.

We classified turkeys as adult or juvenile (Petrides 1942), and equipped them with a 110 g backpack-style transmitter with a 8-hour mortality switch (Advanced Telemetry Systems, Insanti, Minnesota, USA) attached using 3.2 mm shock cord (Miller et al. 1995). Each bird received an individually numbered aluminum leg band. During 2000 and 2001 we equipped birds at the CNG similarly with backpack transmitters with 4.5 hour mortality switches (AVM Instruments, Livermore, California, USA)

We relocated female turkeys ≥2 times/week, after dispersal from winter roosts, by triangulation or visual observation (Samuel and Fuller 1996), to monitor movements, mortalities and onset of nesting behavior. We determined that females were incubating at nests using 3 methods: 3 consecutive locations of females during nesting period (April-July) in the same place, females shifted to overnight ground roosts, and a female was directly observed on the nest.

Once a female was incubating we located the nest site by tracking the nesting female to within 20 m depending on vegetation thickness. We flagged 3 to 4 areas around the nest. Each flagged point was marked using global positioning system (Garmin eTrex Legend, Garmin, Olathe, KS, USA). We recorded a compass bearing from the flagged points to the strongest telemetry signal from the female to assist in relocating the nest after abandonment (Schmutz et al. 1989). After the female was incubating for 14 days based on telemetry relocations, the nest was approached and the female flushed. We recorded the numbers of eggs in the nest and ≥4 eggs were floated to more accurately predict a hatch date (Westerkov 1950). We returned the eggs to the nest after they were floated.

When individual nest completion dates approached we monitored nests daily to determine when hatching or nest abandonment occurred (Schmutz et al. 1989). If telemetry locations indicated females were off the nest before the expected completion date, we located nests immediately to determine nest fate. We used egg shell remains to determine hatching success (Roberts and Porter 1998) and any unhatched eggs were opened to determine fertility (Keegan and Crawford 1999).

We considered a nest successful if ≥1 egg in a clutch hatched. We determined nesting success by dividing all successful nests by the total number of nests attempted. Females that abandoned nests due to observer activities were excluded from nesting success analysis. We determined nesting rates annually for each study site by dividing known telemetered females alive before the earliest nest initiation date by the number of telemetered females that attempted a first nest. We only included females that were relocated on average at ≤5 day intervals in nesting rate analyses. We calculated renesting rate by dividing the number of second nest attempts by the number of females alive that failed on their first nest attempt.

We assessed nest selection at third and fourth order levels (Johnson 1980). We measured plots at turkey nests (nests), at 50 meters from the nest in a random cardinal direction (nest area), and at random (study area) plots (*n* = 200/site/year) using ArcView GIS (Environmenttal Systems Research Institute, Redlands, CA, USA) within the 100% minimum convex polygon of all female locations from previous years at each site. We compared vegetative structure measures between nest area and study area plots to determine whether nest areas differed from what was available within turkey home ranges (third order) and compared nests to nest area to determine whether nest site selection occurred at finer (fourth order) levels of selection. We compared use versus availability of horizontal visual obstruction, ground cover types, and overstory trees (Neu et al. 1974)

At each nest, nest area, or random study area plot, we established a 20-m transect oriented 10 m north and south of the nest or random point. We measured height of visual obstruction at the nest and in the surrounding area using a Robel pole (Robel et al. 1970). The Robel pole was placed along the transect and measured from 4 m away using a 1 m sighting pole. To measure visual obstruction and ground cover in the area around the nest or random plot we placed the Robel pole at the 0 m mark on the transect with the 1 m sighting pole placed perpendicular to the transect and lowest visible band was recorded by the observer standing 4 m from the transect. This procedure was repeated every 5 m, with the sighting pole alternating directions and starting to the west for 5 visual obstruction readings. At the nest bowl, we measured visual obstruction by placing the Robel pole in the nest bowl and recording height of lowest visible band from the sighting pole placed in each of the 4 cardinal directions. At each of the 5 transect Robel pole measurements, we also recorded 10 readings of ground cover at 40 cm intervals using an ocular tube between the sighting pole and Robel pole, for 50 readings/plot. We recorded ground cover in 8 cover types (i.e., crop, grass, shrub, bare, forb, litter, cactus [*Opuntia* spp.], and other) to calculate percent ground cover along the transect. We averaged Robel pole readings separately for the nest bowl (nest) and transect (nest area) for analyses. We used a wedge prism (10 factor) from the plot center to determine basal area of trees at the plot. Each tree that was tallied using the prism was then identified to species, and height (m), diameter at breast height (cm), and height to lowest branch (cm).

We used SAS v. 9.1 (SAS Institute Cary, NC, USA) for all statistical analyses. We used a likelihood ratio G-test (PROC CATMOD) contingency table to analyze nesting rate and nest success at 4 sites over 5 years and to test 3-way interactions, main effects and pair-wise comparisons among sites, years, and nesting rate or nesting success. We compared visual obstruction measured as horizontal screening cover at successful nests to visual obstruction at unsuccessful nests among years and study sites (PROC GLM). No data were collected at CNG in 2002. We tested differences in visual obstruction at nests versus nest-paired random plots for a 3-way interaction between site, year, and visual obstruction with a factorial ANOVA (PROC MIXED).

We compared use versus availability of visual obstruction, ground cover, and trees present using chi-square goodness-of-fit tests (Neu et al. 1974). If the χ^2 test was significant, simultaneous confidence intervals were calculated (α = 0.05) around used proportions using the Bailey method (Cherry 1996). Proportion of ground cover in 8 measured categories was reduced to 6 categories: grass, shrub, bare, forb, litter, and other. Crop and cactus were combined into other to avoid zero values often recorded in these categories. We grouped use versus availability of trees present in the nesting area as trees and no trees.

RESULTS

We captured and equipped 360 adult female and 282 juvenile female wild turkeys with transmitters from 2000 – 2004 at the 3 Texas sites and during 2000 – 2001, 2003-2004 at the CNG. We detected 396 nesting attempts, and 129 (33%) of these nests successfully hatched

≥1 live poult. Causes of nest failure included 195 (73%) depredated, 31 (11.6%) failed from unknown causes, 26 (9.7%) abandoned, 14 (5.2%) censored (disturbed by observers), and 1 (0.4%) infertile.

We did not include nesting rates from 2000 SF, 2003 GHWMA, or 2003 CNG in analyses because only 3 birds were monitored frequently enough (≤ 5 days between relocations) to accurately determine nesting rate. When nesting rate of first nest attempts was compared among years ($n = 5$) and sites ($n = 4$) there was no 3-way interaction ($P = 0.227$, $G_8 = 10.58$). Nesting rate did not differ among sites ($P = 0.229$, $G_{11} = 14.07$), but was different among years ($P < 0.007$, $G_{12} = 27.36$). Nesting rate was greatest in 2001; 65% of transmittered females attempted nests. Nesting rate was lowest in 2000 when only 39% of females attempted to nest.

There was also no 3-way interaction ($P = 0.604$, $G_8 = 6.39$) when we compared re-nesting rates among years and sites. Re-nesting rate was not different among sites ($P = 0.244$, $G_{11} = 13.81$) or years ($P = 0.224$, $G_{12} = 15.32$). The average re-nesting rate across all sites and years was 32% but was quite variable ranging from zero (2004 MWMA) to 67% (2003 SF).

We found no interaction among nesting success for all nesting attempts by site or year ($P = 0.291$, $G_{11} = 13.03$). Nesting success was similar across all sites ($P = 0.287$, $G_{14} = 16.44$), and years ($P = 0.245$, $G_{15} = 18.34$). Nesting success rates ranged from 15% (2002 MWMA) to 58% (2003 MWMA) and averaged 34% across all sites and years.

No 2 or 3 way interactions between site, year, and nest outcome (successful or unsuccessful) were significant ($P > 0.05$). However, the study site year interaction approached significance ($P = 0.067$, $F_{9,244} = 1.81$) suggesting that nesting cover may vary differently among sites over years. When we compared years by pooling study sites within years, and study sites by pooling years within study sites we found no difference in visual obstruction between successful nests and unsuccessful nests at any study site, or any year. Mean height of visual obstruction at ($n = 129$) successful nests was 0.450 m ± 0.015 (SE) and at ($n = 147$) unsuccessful nests it was 0.496 m ± 0.022.

We measured visual obstruction at nests and in nest areas in 2000 ($n = 67$), 2001 ($n = 110$), 2002 ($n = 75$), 2003 ($n = 62$), and 2004 ($n = 66$). We measured visual obstruction at random study area plots ($n = 651$) in 2003 and ($n = 688$) in 2004. Visual obstruction in the nest area differed from random study area availability in 2003 ($P < 0.001$, $\chi^2_7 = 140.58$) and 2004 ($P < 0.001$, $\chi^2_7 = 39.55$). In 2003 visual obstruction that was only 0.1 - 0.2 m tall was avoided in the nesting area and turkeys used nesting areas with ≥0.6 m tall visual obstruction. In 2004 turkeys avoided nesting where visual obstruction was < 0.1 m tall. Visual obstruction at nests also differed from the nest area in all years ($P < 0.001$, $\chi^2_7 < 140.58$). Nests had more visual cover than nest areas with visual obstruction < 0.2 m tall avoided in all years and sites.

We found that ground cover in the nest area were used in proportion to availability for all categories in 2003 ($P > 0.1$, $\chi^2_5 = 7.37$) and in 2004 ($P > 0.1$, $\chi^2_5 = 7.75$). Observed use of nest sites with and without trees was not different than expected in 2003 ($P > 0.05$, $\chi^2_1 = 3.71$). However, in 2004, sites with trees were used ($P < 0.001$, $\chi^2_1 = 103.98$) and areas without trees were not used for nesting.

We compared visual obstruction at fourth order (nest vs. nest area) scale for 2001, 2003, and 2004 only, because nest area plots were not measured in 2000 and no data were collected at the CNG in 2002. There was a significant 3-way interaction between study site, year, and visual obstruction ($P = 0.004$, $F_{6,211} = 3.35$.) When study site was held constant only MWMA had a significant 2-way interaction ($P = 0.002$, $F_{2,211} = 6.51$), meaning the difference in visual obstruction between nests and paired random plots at the MWMA depended on the year. Visual obstruction at the nest was greater than visual obstruction at nest paired random plots at all study sites and years ($P = 0.001$) except for the MWMA in 2001.

DISCUSSION

Nesting rates of turkeys (39-65%) during our study did not differ among study sites but they did vary among years. Rates were similar to those of Rio Grande turkeys in the Edwards Plateau of Texas (53%; Reagan and Morgan 1980), and north-central Texas (48%; Hohensee and Wallace 2001). However, they were lower than nesting rates reported in south-central Kansas (90 – 100%; Buford 1993, Hennen 1999), and nesting rates for introduced Rio Grande turkeys in Colorado (95 – 100%; Schmutz et al. 1989) and Oregon (94 - 99%; Keegan and Crawford 1999) where turkey populations were thought to be increasing or stable.

For all years and sites the re-nesting rate averaged 29% which was lower than the typical range reported for eastern wild turkeys in Minnesota (65%; Porter et al. 1983), Massachusetts (50%; Vander Haegen et al. 1988), New York (65%; Roberts et al. 1995), Mississippi (35%; Miller et al. 1998), and Wisconsin (55%; Paisley et al. 1998). It was also lower than reported for Rio Grande turkeys in Kansas (78%; Hennen 1999). Re-nesting rates in this study were highly variable (0 – 67 %) but did not show consistent patterns across years or sites.

Nesting success during our study did not vary by study site or by year, and averaged 33% across all sites. However, nesting success did range from a low of 15% at the MWMA in 2002 to a high of 58% in 2003. These differences may not be statistically significant, but they do have biological importance. Nesting success has been reported to be more important to population fluctuations than poult survival or annual adult survival (Roberts et al. 1995). A 43% increase in nesting success could result in a large increase in the number of poults recruited into the population.

The average nesting success across all years and sites during our study (33%) was greater than other rates reported in central Texas (19%; Reagan and Morgan 1980), and was similar to that reported in south-central Kansas (32%; Hennen 1999). Nesting success has been reported to be higher for populations in Colorado (58%; Schmutz et al.1989) and in Oregon (60%; Keegan and Crawford 1999) where populations were increasing. Vangilder et al. (1987) suggested that nesting success was lower in stable turkey populations than in expanding populations. Hennen and Lutz (2001) postulated that populations in Kansas persisted despite low nest success, because of periodic years with high recruitment. Populations in our study could also be maintained by periodic years of high nesting success.

Rio Grande wild turkey nest selection occurs hierarchically at both third and fourth order levels of selection. Turkeys avoided nesting areas that had little visual obstruction at <0.2 m height, selecting instead for areas with >0.2 m visual obstruction and preferred 0.6 m obstruction. Turkeys further selected nests sites with greater visual obstruction than at paired random plots in the nesting area at all study sites over all years except MWMA in 2001. The pattern at MWMA in 2001 may be due to precipitation patterns during that year. Pre-nesting precipitation was greater in 2001 than any other year at the MWMA while precipitation after the start of nesting was the lowest reported during our study (National Oceanic and Atmospheric Administration, unpublished data). The high level of precipitation may have increased the overall availability of visual obstruction, while the low precipitation during nesting may have limited the continued growth through the nesting season.

Many studies have shown that wild turkeys will select nest sites with greater cover than at random plots associated with the nests (Schmutz et al. 1989, Day et al. 1991, Rumble and Hodorff 1993, Badyaev 1995, Hohensee and Wallace 2001). Fewer studies have compared nest areas (use) to random plots across the study site (available) to determine selection of cover on a larger scale (Schmutz et al. 1989, Badyaev 1995, Lehman et al. 2002). Most of these studies did not determine selection of visual obstruction available, but selection of vegetation association (i.e., woodland, grassland; Schmutz et al. 1989, Lehman et al. 2002). Female turkeys avoided nests sites with low height of visual obstruction (0.1 m, 0.2 m) and used nest sites with greater height of visual obstruction at 0.4, 0.5, 0.7, and 0.8 m height categories.

In 2003 presence of trees at nest areas did not differ from the availability across the study sites. However, in 2004 nest sites with trees were used while nests sites without trees were avoided. This could be due to the timing and amount of precipitation, as pre-nesting precipitation was low at the GHWMA, CNG, and SF (National Oceanic and Atmospheric Administration, unpublished data) and could have resulted in less nesting habitat available across the study sites. The best cover available during this time was in areas with trees and female turkeys were using those sites. While we did not examine changes in nest site selection as the nesting season progressed, Merriam's wild turkeys use different cover types as they became more available during the nesting season (Schmutz et al. 1989, Day et al. 1991); woodland sites were used early in the season and grassland sites were used later as herbaceous cover grew and provided visual obstruction similar to woodland sites.

MANAGEMENT IMPLICATIONS

Nesting success has been reported as an important factor in recruitment. Nesting success varied widely across our sites and years such that differences between sites and years were not statistically different. However, small changes in nest success could result in large changes in recruitment. Nesting habitat is an important factor in the nesting success of turkey populations. Amount and timing of precipitation differed over years among our sites (Huffman 2005). Less nesting cover may be available in dry years (Hennen 1999). Shrubs in riparian corridors can provide nesting habitat when herbaceous vegetation is lacking due to weather or grazing patterns. Rio Grande turkeys on 4 sites from the Texas Panhandle to southwest Kansas used sites and nests with visual obstruction from ≥ 0.2 to 0.6 m tall, often provided by shrubs when obstruction from herbaceous vegetation was lacking. We recommend management practices that retain low growing shrubs that provide visual obstruction to 0.2 – 0.6 m tall and remove shrubs such as tamarisk that quickly grow too tall to provide visual obstruction at the important < 0.2 m heights. Deferring cattle grazing in riparian corridors during the nesting season could also increase the herbaceous vegetation available for nesting cover in the 0.2 to 0.4 m height classes (Hall 2005). Short (<0.2- 0.4 m) shrub cover and reduced grazing during the nesting season would also increase habitat for turkey poults (Spears 2002).

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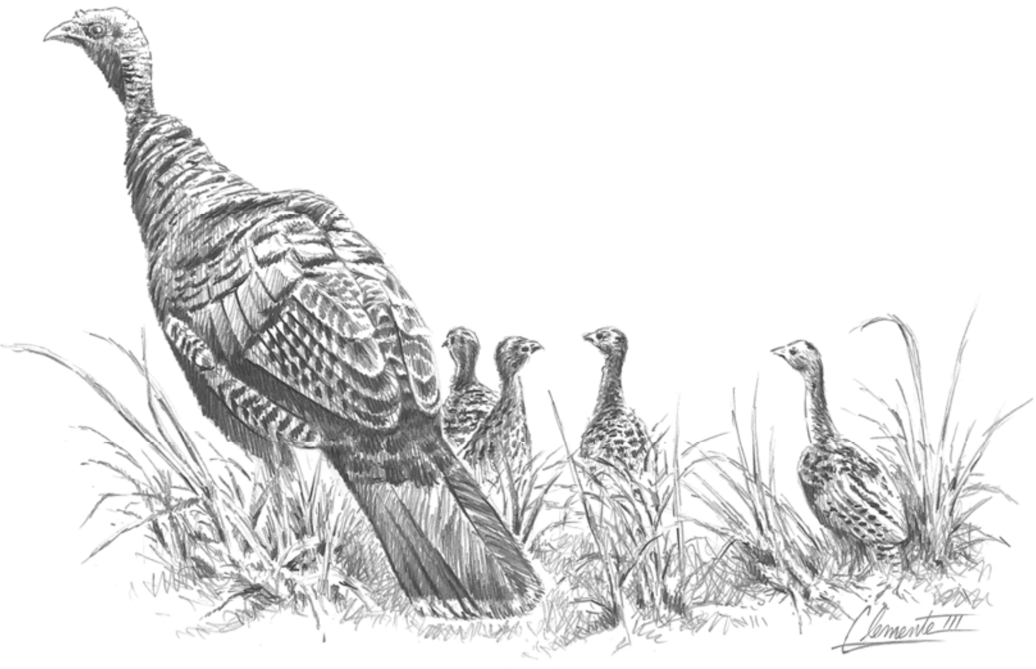

COMPARING TECHNIQUES FOR COUNTING RIO GRANDE WILD TURKEYS AT WINTER ROOSTS

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Abstract: Counting Rio Grande wild turkeys (*Meleagris gallopavo intermedia*) at winter roosts is a technique commonly used to index their abundance because they congregate in specific roost sites throughout the winter. We compared 5 techniques for counting wild turkeys on winter roosts. We used direct observation during evening and morning hours, and advanced technology such as a night vision device (NVD), a thermal infrared camera (thermal IR), and an automated video monitoring system (AVMS). Morning counts were 8.7 ± 5.9% (SE) larger than evening counts and 25.8 ± 5.4% larger than NVD counts. Morning counts were similar (28.6 ± 12.6%) to counts from the AVMS. Also, counts from the thermal IR were 46.6 ± 8.9% smaller than the evening counts and were similar (14.0 ± 31.5%) to the NVD counts. Overall, we found the advanced technology (e.g., NVD, thermal IR, or AVMS) was ineffective for counting wild turkeys at winter roosts and the more traditional morning counts provided the largest counts. Thus, we suggest using morning counts when counting Rio Grande wild turkeys at winter roosts.

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Key words: index, infrared night vision device, Kansas, *Meleagris gallopavo intermedia*, Rio Grande wild turkeys, roost counts, Texas, thermal infrared, winter roosts.

Rio Grande wild turkeys often congregate in specific roost sites throughout winter (Thomas et al. 1966, Watts and Stokes 1971). Many techniques have been used to count Rio Grande wild turkeys at winter roost sites. For example, counts have been obtained from surveying landowners (Thomas et al. 1966, Cook 1973), counting wild turkeys in the general area of known roosts (DeArment 1975, Healy and Powell 1999), and counting wild turkeys as they flew into or from roosts (Thomas et al. 1966, Cook 1973, Smith 1975). Also, increased interest in the use of advanced technology such as NVD, thermal IR, and AVMS for counting wild turkeys at winter roosts has emerged. However, little information comparing counting techniques is available.

Variation exists among the potential techniques for counting wild turkeys at winter roosts. However, only direct counts obtained by wildlife professionals and landowner surveys have been compared (Thomas et al. 1966, Cook 1973). Those comparisons suggested that landowner surveys were adequate to index direct winter roost counts in areas with stable winter roosting patterns. Thus, our objectives were to compare several techniques for counting wild turkeys at winter roosts and examine the relationships among counts generated with these techniques.

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Specifically, we were interested in counts obtained from direct observation of roosting wild turkeys during evening hours, from direct observation during morning hours, from a NVD, from a thermal IR, and from an AVMS. Our results will help researchers and managers eliminate ineffective techniques and focus future evaluation and validation efforts.

STUDY AREA

We conducted counts of wild turkeys at winter roosts at 3 study sites in the Texas Panhandle and 1 site in southwestern Kansas (Fig. 1). The Texas study sites were centered on (1) the Matador Wildlife Management Area (WMA), located northwest of Paducah in Cottle County along the confluence of the Middle and South Pease rivers; (2) the Gene Howe WMA, located east of Canadian in Hemphill County along the Canadian River; and (3) private ranches surrounding the Salt Fork of the Red River, located north of Hedley in western Collinsworth and eastern Donley counties. The Kansas study site was centered on the Cimarron National Grasslands north of Elkhart in Stevens and Morton counties, Kansas, and Baca County, Colorado along the Cimarron River. The riparian areas of the 4 study sites were dominated by eastern cottonwood (*Populus deltoides*), western soapberry (*Sapindus drummondi*), hackberry (*Celtis occidentalis*), netleaf hackberry (*C. reticulate*), sugarberry (*C. laevigata*), honey locust (*Gleditsia triacanthos*), black locust (*Robinia pseudoacacia*), tamarisk (*Tamarix chinensis*), and sand plum (*Prunus angustifolia*). More detailed descriptions of the study sites were provided by Holdstock (2003), Hall (2005), Huffman (2005), and Butler et al. (2005).

METHODS

Rio Grande wild turkey winter roosts were identified from historical data (Texas Parks and Wildlife Department, unpublished data), recent radiotelemetry efforts (Holdstock 2003, Phillips 2004), and communication with land managers. Wild turkeys were counted periodically at these winter roosts from mid-November through mid-March during 2003–2005. We conducted counts after leaf fall in autumn and before leaf emergence in spring.

We conducted morning and evening counts to obtain direct observations of wild turkeys congregating on the ground, flying to and from the roost, and settling in the roost trees. During morning and evening counts, observers used 10 power binoculars as needed; roosts were typically observed from <100 m. We used infrared technology to observe roosts after dark (i.e., 1 hr after sunset to 1 hr before sunrise). We used a Generation-III NVD (U.S. Night Vision® Goggle PVS-7B Ultra with an attachable 3X lens, U.S. Night Vision Corporation, Costa Mesa, California, USA) and a handheld thermal IR (Raytheon Thermal-Eye 250D thermal-infrared camera, L-3 Communications, New York, New York, USA) to observe roosting wild turkeys; roost were typically observed from <50 m. The NVD cost approximately $3,500 (U.S.) and relies on reflected light in the visible and near-infrared wavelengths. The thermal IR cost approximately $13,000 (U.S.) and relies on infrared light emitted from thermal sources.

We also used the AVMS to record roosting activities on 1.25 cm vertical helical scan (VHS) tape (Sony® T-160 VHS, Sony Corporation, Tokyo, Japan). The AVMS was a hybridization of the designs of King et al. (2001), Kristan et al. (1996), and Lewis et al. (2004) as reported in McGee et al. (2005). The AVMS had zoom capabilities and recording times were programmable (McGee et al. 2005). The AVMS cost approximately $1,600 (U.S.). After monitoring roosts several times to determine typical roost trees, we positioned the AVMS to allow the best, but likely incomplete, coverage of roost trees. The AVMS was programmed to record approximately 1 hr before and after sunrise and sunset allowing recording of wild turkeys as they flew into and from the winter roost. Counts of wild turkeys in the roosts were obtained from the VHS recordings.

Wild turkeys fly into a roost around sunset and usually remain there until sunrise. We considered the period of time from the evening flight to the morning flight as a roosting event. For each roosting event, ≥ 2 of the 5 techniques were used to count wild turkeys on the winter roost, which allowed pairing of the techniques for comparison with paired *t*-tests (Zar 1999). To avoid potential bias from prior knowledge of the number of turkeys in a roost, different observers were used for each technique during a specific roosting

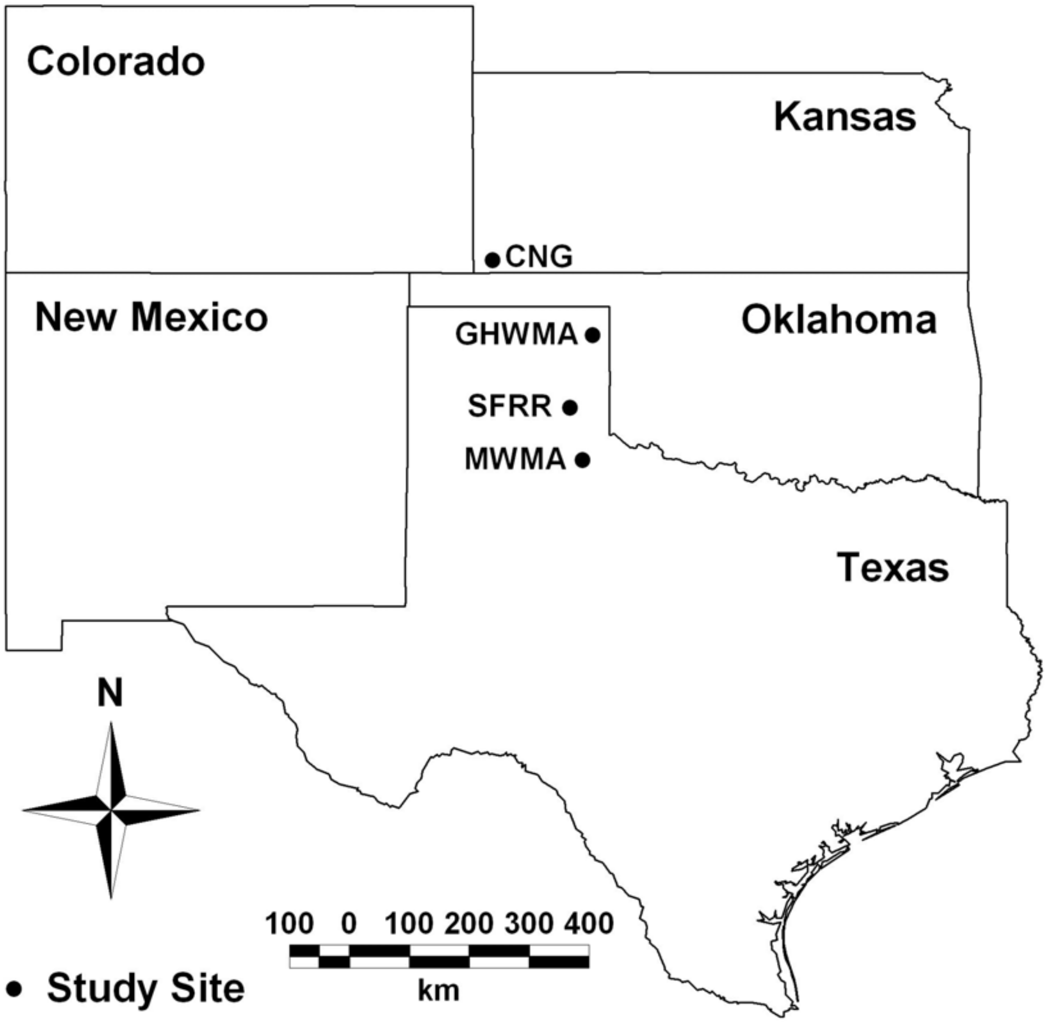

Fig. 1. Locations of study sites from north to south: Cimarron National Grasslands (CNG), Gene Howe Wildlife Management Area (GHWMA), private ranches surrounding the Salt Fork of the Red River (SFRR), and Matador Wildlife Management Area (MWMA) used to compare techniques for counting Rio Grande wild turkeys at winter roosts during mid-November through mid-March, 2003–2005.

event. Data for comparing thermal IR counts with the morning and AVMS counts were not available because the thermal IR was only available to us for a short period of time.

RESULTS

Comparisons were conducted on 105 roosting events. The number of Rio Grande wild turkeys observed during those roosting events ranged from 2 to 319. Evening counts resulted in 20.9 ± 3.5% larger counts than the NVD counts (Table 1). Morning counts were 8.7 ± 5.9% and 25.8 ± 5.4% larger than evening and NVD counts, respectively (Table 1). Counts from the thermal IR were 46.6 ± 8.9% smaller than evening counts but were similar (14.0 ± 31.5%) to the NVD counts (Table 1).

Though we attempted to position the AVMS to allow the best coverage of a roost, on 9 of 17 events we did not record wild turkeys in the roost because turkeys did not use their usual trees. Also, on 2 occasions, no wild turkeys

Table 1. Paired comparisons of techniques used for counting Rio Grande wild turkeys at winter roosts in the Texas Panhandle and southwestern Kansas during mid-November through mid-March, 2003–2005

		Percent difference		Paired *t*-test	
Paired comparison	*n*	$\bar{x}$	SE	*t*	*P*
Morning – evening	32	8.7	5.91	2.121	0.042
Morning – NVD	38	25.8	5.39	3.174	0.003
Morning – AVMS	8	28.6	12.57	1.841	0.108
Evening – NVD	77	20.9	3.52	5.340	<0.001
Evening – thermal IR	10	46.6	8.94	3.344	0.009
Evening – AVMS	8	15.2	13.51	1.492	0.179
AVMS – NVD	8	3.3	15.11	0.617	0.557
NVD – thermal IR	7	14.0	31.46	0.523	0.620

were recorded during the morning because of dew on the lens or foggy conditions. Otherwise, 8 attempts resulted in a count. Morning (28.6 ± 12.6%), evening (15.2 ± 13.5%), and NVD (3.3 ± 15.1%) counts were similar to those 8 counts obtained from the AVMS (Table 1).

DISCUSSION

Though the AVMS counts were similar to morning counts, it was not an effective technique. To properly setup the AVMS, roosts were observed for several days to determine the typical roost trees used by wild turkeys in a roost area. However, because of unstable roosting patterns, only 8 of 17 attempts at using the AVMS were successful. Other techniques allowed the observer to move in response to wild turkey movements (e.g., wild turkeys sometimes choose to roost in different trees on different nights). Thus, without stable roosting patterns, we do not recommend using an AVMS to count Rio Grande wild turkeys at winter roosts.

Direct observation of roosts during the morning resulted in the largest counts. Because wild turkeys may not roost in the same trees from night to night, direct observation was more difficult during the evening. In large stands of trees, it was more difficult to predict where wild turkeys would roost during a particular roosting event. But, during the morning it was easier to locate roosts (via visual location and sound) before turkeys begin to depart, resulting in larger counts. Also, wild turkeys often fly into open areas during morning, allowing for better counts. However, few morning roost counts of vultures (*Cathartes aura* and *Coragyps atratus*) were successful because a large number of vultures would depart the roost at once (Sweeney and Fraser 1986). We also found it was difficult to count wild turkeys when many departed the roost at once. However, this behavior occurred during morning and evening hours; it was unpredictable and varied among roosts and days.

Many factors such as tree, limb, and twig densities and illumination strongly affected NVD ability. With the NVD, wild turkeys appeared as silhouettes against the night sky. However, in low light conditions, wild turkey silhouettes were often indistinguishable from other shadows. In more illuminated conditions (e.g., full moon), wild turkey silhouettes were usually obscured by limbs and twigs that were much more visible due to reflected moon light.

Thermal IR counts were smaller than evening counts and similar to NVD counts. We observed that wild turkeys tuck their heads under their wings at night leaving little exposed skin. Because of the insulating capacity of their feathers, little heat escapes for detection. Wakeling et al. (2003) had similar difficulties with aerial thermal IR counts for Merriam's wild turkeys (*M. g. merriami*). However, we learned counts could be improved by emulating a yelp with a box or diaphragm call because wild turkeys would usually expose their heads revealing a thermal signature.

Advanced technology such as NVD, thermal IR, or AVMS was not as effective at determining the number of wild turkeys in a winter roost. But, the more traditional morning counts typically provided the largest counts. Thus, we suggest researchers and managers strive to obtain the best possible counts using morning counts; however, several survey attempts may be necessary to obtain the best possible count.

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BIBLIOGRAPHY OF LITERATURE PERTAINING TO THE SCALED QUAIL: 1847 – 2006

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Abstract: Scaled quail (*Callipepla squamata*) have received limited interest from researchers relative to other quail species. However, since 1980 population declines have enhanced interest in the species and its management. To assist future research efforts I conducted a literature search to compile a comprehensive bibliography of published scaled quail literature. The search was conducted via the internet, university library catalogs and databases, and by searching citations in the quail literature. All sources of scaled quail information discovered are cited.

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Key words: bibliography, *Callipepla squamata*, literature, scaled quail.

This bibliography contains literature citations for every publication discovered involving scaled quail, or containing noteworthy information on the species (*n* = 265). I examined as many of these sources as possible but did not investigate each listed source. Some are out of print and difficult to find, others are recent and easily obtained. I did not find a citation for the original description of scaled quail (Vigors 1830). The earliest citation found was Abert (1847), the latest in press (Cantu et al).

This review was able to clear up some anomalies. For example, Wallmo (1956) and Wallmo (1957) are identical publications, based on his work on scaled quail in the Trans-Pecos of Texas during the mid-1950s. Wallmo (1956) was a research report turned into the Texas Game and Fish Commission in December while the 1957 citation was a dissertation turned into Texas A&M University in January.

Peer-reviewed articles accounted for 38.5% (*n* = 102) of citations, reports and bulletins account for 24.5% (*n* = 65), dissertations and theses 15.8% (*n* = 42), proceedings and transactions 12.8% (*n* = 34), books 6.4% (*n* = 17 citations), and magazine articles 1.9% (*n* = 5 citations). Of the peer-reviewed articles, and monographs, 11.6% (*n* = 31) were found in publications of The Wildlife Society (Journal of Wildlife Management – 26, Wildlife Society Bulletin – 3, Wildlife Monographs – 2). I found 13.2% (*n* = 35) of the articles in avian specific journals (Condor – 15, Auk – 13, Wilson Bulletin – 5, Poultry Science – 2). Wildlife disease and veterinary journals published 3 articles, range science journals 6 articles, and regional journals (e.g., The Southwestern Naturalist), 14 articles. The remaining peer-reviewed articles were found in sources such as American Midland Naturalist, Animal Behavior, Journal of Field Ornithology, and Science.

Forty-two citations, 15.8%, are represented by theses (*n* = 33) and dissertations (*n* = 9). New Mexico State University produced 35.7% (*n* = 15; 13 theses, 2 dissertations), Texas Tech University 16.6% (*n* = 7; 5 theses, 2 dissertations), Texas A&M University 12% (*n* = 5; 4 theses, 1 dissertation), Angelo State and Sul Ross State universities in Texas 3 theses each, Oklahoma State University 2 (1 dissertation and 1 thesis), 1 dissertation each from the University of Nebraska, the University of New Mexico, and Oregon State University, and 1 thesis each from the University of Arizona, Auburn University, Colorado State University, and Texas A&M University – Kingsville.

Scaled quail articles focused primarily on populations and habitats in Texas 26.8% (*n* = 71), New Mexico 24.2% (*n* = 64), Arizona 11.7% (*n* = 31), Oklahoma 8.0% (*n* = 21), and a few from Colorado (*n* = 13), Mexico (*n* = 6), and Kansas (*n* = 3). Twenty-four articles addressed general biology, 21 with range-wide issues, and a few miscellaneous articles from extralimital or

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experimental populations in Nebraska and Nevada. There are 159 different primary authors represented in these citations. Campbell (*n* = 14), Schemnitz (*n* = 9), and Wallmo (*n* = 8) authored the most articles.

Twenty-three articles were found dated 1847 – 1940, with an additional 16 from 1941 – 1950. Since 1951, articles were somewhat equally balanced during each of the following decades with 16.9% (*n* = 45) articles from 1951-1960, 15.4% (*n* = 41) from 1961 – 1970, 14.3% (*n* = 38) from 1971 – 1980, 14.0% (*n* = 37) from 1981 – 1990, 16.6% (*n* = 44) from 1991 – 2000, and 20 from 2001 – 2005. Topics covered fell into 10 categories: natural history (*n* = 104), management (*n* = 45), habitat (*n* = 37), diet (*n* = 23), distribution (*n* = 17), reproduction (*n* = 17), disease (*n* = 10), water use (*n* = 6), population trends (*n* = 5), and movements (*n* = 1).

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FROM THE FIELD: PRESCRIBED BURNING RULES AND REGULATIONS IN TEXAS

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Abstract: Biologists and politicians in Texas have come together to create prescribed burning legislation to manage wildlife habitats. To initiate the effort, the Texas Prescribed Burning Coalition was organized in April 1998 to address concerns about prescribed burning. The Texas Prescribed Burning Law (HB 2599) became official in September 1999. The law, administered through the Texas Department of Agriculture, guarantees the right of every landowner in the state to burn on their own property, set up a prescribed burn manager certification system, and set up a Prescribed Burning Board (PBB) and an Advisory Committee to the PBB. The legislation placed liability directly upon a certified prescribed burn manager (CPBM), removing the landowner from $1,000,000 of liability. In a second bill, (HB 1080) passed in 2001, damage claims were capped at $2,000,000/insured/year. A third bill, (HB 3315) passed in 2001, set up a mechanism for counties to grant prescribed burning permits to CPBMs during county burn bans. Since creation of the PBB, prescribed burning rules have been written, certification training and re-certification have been established, and a number of individuals have been trained; however, no one has been certified because PBB has not located a company willing to insure CPBMs. These three bills are important to wildlife management in Texas because they set up a mechanism through which landowners and managers can use this valuable tool in wildlife habitat management.

MANAGING WILDLIFE IN THE SOUTHWEST 2006:130–134

Key words: fire, habitat, wildlife management, politics, Texas.

Prescribed fire is an economical habitat management tool. Researchers have found numerous positive effects of fire (Wright and Bailey 1982, Drawe 2003). Fire removes decadent herbage and rejuvenates stagnant grassland, thus improving the value of the herbage as food for wildlife (Scifres 1993). Fire causes resprouting of brush, thus improving palatability and nutritional value of woody plants for browsing animals (Box et al. 1967, Box and White 1969). Fire opens up the habitat for ground nesting and ground feeding birds such as northern bobwhite quail (*Colinus virginianus*), mourning doves (*Zenaidura macroura*), and some sparrows (Wilson and Crawford 1979, Reynolds and Krausman 1998, Marx 2003). Fire has been used to improve habitat for several species of wildlife, including mammalian and avian species (Wright and Bailey 1982). Burning increased quail numbers in south Texas (Wilson and Crawford 1979). Proper timing of burning can create forbs for use by white-tailed deer (*Odocoileus virginianus*) (Hansmire et al. 1988). Rodents may be reduced temporarily through increased avian predation because of reduced ground cover (Tewes 1982). Tick populations are reduced following fire, benefiting wildlife and cattle (Oldham 1983). In areas where strips or blocks have been cleared, burning herbaceous vegetation while leaving unburned woody areas enhances wildlife habitat by causing an increase in forb growth.

The negative aspects of fire are outweighed by the positive aspects. The few negative impacts are primarily the effect of fire on the environment. Smoke from prescribed fire can have a negative impact on air quality and can cause traffic hazards on roadways and waterways. If a fire escapes from the boundaries of a prescribed burn it can have a negative impact, including loss of property and even loss of human lives. However, if the prescription for fire is followed and if all safety rules for the use of prescribed fire are followed, losses of these types are minimized.

Our efforts in Texas have demonstrated

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how biologists and politicians can work together to create laws that work to the benefit of a natural resource important to all the people of a state. Prescribed burning in Texas is conducted under an exemption to the Outdoor Burning rule of the Texas Commission on Environmental Quality (TCEQ; Texas Commission on Environmental Quality 2005). Texas Commission on Environmental Quality rules must be followed in conducting prescribed burns on rangelands and wildlife habitats. These rules are specific in their requirements for certain factors influencing a prescribed burn such as weather conditions and type of vegetation. Texas Prescribed Burning rules seek to fill in where TCEQ rules end and make the use of prescribed fire a safe and effective tool for use in managing vegetation in the state.

Texas Prescribed Burning Coalition

The Texas Prescribed Burning Coalition (TPBC) was organized to have a positive impact on prescribed burning legislation in the state, to foster and support training in the art and science of the use of prescribed fire in Texas, and to disperse accurate information to the public about prescribed burning. The TPBC was organized because it became obvious that urbanization, a general fear and misunderstanding of fire, and lack of information on the proper use and positive benefits of fire might soon preclude the use of this valuable wildlife habitat management tool. Concerned professionals in the wildlife and rangeland management community in Texas organized the TPBC, an ad hoc independent group, in April 1998 at a meeting in Kerrville. The initial meeting identified approximately 60 organizations and individuals interested in prescribed burning in Texas. Major players in the initial meeting were members of the Texas Parks and Wildlife Department who had already instituted prescribed fire training for its employees to assist landowners with wildlife habitat management. Three committees were named: legislative, education and training, and public information. The TPBC met once in 1998 and once 1999 following the initial meeting.

Members of the TPBC were aware that recent court rulings in the South had increased the potential liability associated with prescribed burning (Alabama Forestry Commission, Long 2002, Oklahoma Department of Agriculture Forestry Services 2002.). These rulings defined prescribed burning as inherently dangerous. To solve this problem, the TPBC presented a prescribed burning bill to the 1999 Texas legislature. Representative Jim McReynolds introduced the Texas Prescribed Burning Bill, (HB 2599), which passed both chambers of the legislature, was signed by Governor George W. Bush, and became law in September 1999. This legislation guarantees the right of every landowner in the state to burn on their property. It also set up a prescribed burn manager certification system, a Prescribed Burning Board (PBB), and an Advisory Committee.

Texas PBB membership consists of representatives of 7 state agencies, (i.e., Texas Department of Agriculture [TDA], Texas Commission on Environmental Quality, Texas State Soil and Water Conservation Board, Texas Parks and Wildlife Department, Texas Cooperative Extension, Texas Agricultural Experiment Station, and Texas Forest Service), Texas Tech University, and 5 private landowners. The PBB, housed in the TDA, was formed to write the rules for prescribed burn managers in Texas and to oversee the certification and re-certification process.

House Bill 2599 addresses the landowners' right to burn and places liability directly on the prescribed burn manager, thus removing the landowner from $1,000,000 of liability. This amount of liability passes to the Certified Prescribed Burn Manager (CPBM). The legislation was designed in part to provide landowners with a positive incentive to use prescribed fire on their properties. Two provisions of HB 2599 address this goal: Texas landowners now have the right to burn on their own property and the bill created the CPBM along with a system of training and certification. Initially, the PBB interpreted the liability insurance requirement to mean that an individual must purchase the $1,000,000 policy; however, the PBB recently changed the insurance rule to allow a company or burn association to purchase the liability coverage and specifically name the individual covered in the policy. No other state has placed this kind of protection between a landowner and the possibility of damage claims or lawsuits that might occur as a result of the use of prescribed fire. With this kind of protection, landowners in Texas can now utilize prescribed fire as a wildlife management tool.

House Bill 2599 failed to place a cap on the amount of claims an insurance company might be obligated to pay annually as a result of damages created by prescribed fires conducted by a CPBM. Since providing fire liability insurance with an upper limit on claims per insured burner would be more appealing to insurers, HB 1080 was introduced into the 2001 Texas legislature. The legislation passed both houses, and the governor signed the bill into law. House Bill 1080 limits claims to $2,000,000/insured CPBM/year.

Additional legislation was still needed to make prescribed burning a viable option in Texas. In many cases the proper time to conduct prescribed burns is during dry times when counties are under burn bans. A remaining stumbling block limiting the application of prescribed burns in Texas was the inability of prescribed burn managers to obtain permits from county commissioners courts to conduct prescribed burns during county burn bans. At best, the system was haphazard with some counties having an acceptable mechanism for allowing prescribed burns and others having no mechanism for the approval process. In some counties it has been impossible to obtain permits during burn bans. A third bill, House Bill 3315, set up a mechanism for counties to grant permits to CPBMs during county burn bans.

House Bill 2599 made the Texas PBB responsible for setting up an Advisory Committee to PBB, defining prescribed burning standards, detailing certification and re-certification, setting training standards for prescribed burn managers, defining educational and professional requirements for burning instructors, and setting minimum insurance requirements for CPBMs. The PBB met monthly over 2 years from December 1999 through December 2001 to write the rules outlined by HB 2599. The PBB completed all of its tasks, appointed the Advisory Committee, set burning standards, certification standards, training standards, and set requirements for burning instructors.

In August 2001 the PBB approved the rules resulting from HB 2599 and in November 2001 the PBB added needed rule changes resulting from enactment of HB 1080 and HB 3315. The certification and training process is now in place and functional. A number of officially sanctioned prescribed burning schools have been taught since PBB's organization. Upcoming schools plus the rules and enabling legislation are available on the TDA web site (http://www.agr.state.tx.us/pesticide/).

The Advisory Committee consists of 12-15 prescribed fire professionals and others involved in prescribed burning on Texas rangelands. The charge to the committee has been to provide the PBB with technical information and advice on questions the PBB does not have the time or resources to address. The Advisory Committee has provided the PBB with valuable information throughout the process of rule writing, including rules from other states, details of particular rules for Texas, details of prescribed burn plans from various state, federal and private agencies, insurance details, and other needs of the PBB.

Certification and Re-Certification

Certified prescribed burn manager standards address the need for a written plan, personnel requirements, notification requirements, and insurance requirements. Certification and training have been set up by the PBB on a regional basis. The PBB has divided the state into 5 training regions (Table 1) based on similar vegetation requiring unique burning techniques. Within each region a contact agency has been selected to coordinate training and certification for the region. Each region has a certified burn manager training coordinator who has been approved by the PBB. These contact agencies and coordinators are responsible to TDA for coordination of training, issuance of certificates, and record keeping. The TDA keeps certification records and coordinates statewide training and re-certification activities. A prescribed burn manager is initially certified to practice only in the region in which they have received training. Later they may decide to become certified in other regions. If so, they must attend a 1-day specialized regional course for each new region.

Future

A major impediment to completing the task of the PBB has been in finding insurance companies willing to offer liability policies for this specialized work. To date, the PBB has not located a company willing to insure prescribed burners. The insurance problem is not unique to Texas. Other states (e.g., Florida and the southeastern states) that have recently approved burn certification have encountered similar problems obtaining insurance. Recently AGREN, Inc., Carroll, Iowa and the Iowa Department of Natural Resources have teamed up to develop a prescribed fire insurance liability product. The study is funded by the U. S. Department of Agriculture to conduct a survey of prescribed burners in 5 states (e.g., Texas, Iowa, Nebraska, Oklahoma, and Kansas). The study was initiated in February 2004 and completed during 2005. Questionnaires were mailed to practicing burners in early 2005. The study proposes to build a foundation for constructing an insurance tool to protect private contractors from personal liability when conducting prescribed burns.

Prescribed burning is on its way to becoming a standardized, accepted practice on Texas rangelands. Currently, there are approximately 100 individuals throughout the state who have completed PBB-sanctioned training courses and who may potentially become CPBMs when insurance becomes available. This has been a grass-roots effort to enhance the wise and safe use of fire as a wildlife habitat management tool in Texas. In the process, wildlife and range professionals from many state agencies, federal agencies, and NGOs became advocates to influence the creation of a much-needed state agency to promote the use of prescribed fire for wildlife habitat management. This combination is an example of how politicians and biologists can work together to make a positive impact on a valuable natural resource management tool.

ACKNOWLEDGEMENTS

This is Welder Wildlife Foundation Contribution No. 655.

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Table 1. Prescribed fire training regions in Texas, 2005.

Region no.	Organization	Contact
1	Texas Tech University, Lubbock, Texas	Carlton Britton 806-742-2842 Carlton.britton@ttu.edu
2	Texas Agricultural Experiment Station, San Angelo, Texas	Darrell N. Ueckert 325-653-4576 D-ueckert@tamu.edu
3	Texas Parks and Wildlife Department, Tyler, Texas	Jeff Sparks 903-566-5698 Jeff.sparks@tpwd.state.tx.us
4	Texas Forest Service, Gilmer, Texas	Ernie Smith 903-734-7007 Esmith@tfs.tamu.edu
5	Texas Cooperative Extension Corpus Christi, Texas	C. Wayne Hanselka 361-265-9203 C-hanselka@tamu.edu
	Texas State Soil and Water Conservation Board	Andres Garza 956-421-5841 Agarza27@hotmail.com

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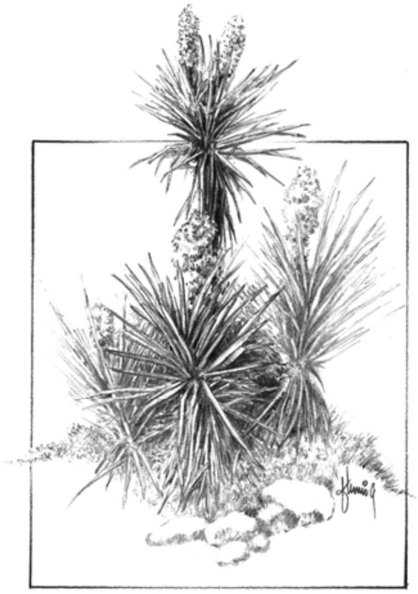

FROM THE FIELD: WILDLIFE CONSERVATION CAMP – THE TEXAS CHAPTER OF THE WILDLIFE SOCIETY INVESTS IN THE FUTURE

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Abstract: The motto of the Texas Chapter of The Wildlife Society (TCTWS), like The Wildlife Society (TWS), is "excellence in wildlife stewardship through science and education". Specifically, TWS policy encourages members of the wildlife profession to interpret and make readily available results of wildlife research that citizens require for decision-making and to actively participate in the implementation of sound, publicly oriented programs in conservation education. To accomplish these objectives the TCTWS developed a youth education program in 1993 entitled Wildlife Conservation Camp (WCC). The camp, organized and run entirely by volunteer efforts of TCTWS members, is designed for high school students in grades 10-12. Since 1993, TCTWS held 12 annual WCC, and has reached 230 first-year campers, with many of those young men and women returning in subsequent years as camper mentors, counselors, and professional staff.

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Key words: camp, conservation, education, management, stewardship, Texas Chapter of The Wildlife Society, volunteer, youth.

The motto of the Texas Chapter of The Wildlife Society, like The Wildlife Society, is "excellence in wildlife stewardship through science and education." Specifically, TWS policy encourages members of the wildlife profession to interpret and make readily available results of wildlife research that citizens require for decision-making and to actively participate in the implementation of sound, publicly oriented programs in conservation education. These goals and objectives have been met using camp programs in environmental education (Dresner and Gill 1994, Kruse and Card 2004).

In 1993, to help young people strive for excellence in wildlife stewardship, TCTWS developed a 1-week, all-volunteer WCC, for high school sophomores, juniors, and seniors. The chapter's goals for the camp were to create conservation-wise voters, and to create an avenue for youth interested in careers in natural sciences to interact with professionals and learn more about the natural resource profession. The objectives of the WCC are to promote conservation-oriented education of high school students, to promote a positive public image of TCTWS, and to organize this public education project in a cost-effective manner.

The first camp held in 1993 was called "The Wildlife Ecology Camp," and was initiated by R. M. Whiting, then president of TCTWS. It was organized and chaired by R. M. Whiting and J. S. Greene (then chair of the student activities committee). Week-long volunteers were limited for this first camp, so the camp was staffed by R. M. Whiting, J. S. Greene, and the Welder Wildlife Foundation staff, who were site hosts. Six female and 10 male campers received a broad overview of wildlife topics including game and non-game management, management techniques, and habitat management. Activities included field work, classroom sessions, and individual mini-research projects, which were presented by the campers at the end of their week at camp.

The first 4 camps were at the Welder Wildlife Foundation, Sinton, Texas. Since 1998,

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in alternating years, the camp has moved around the state. Texas Tech Center, in Junction, Texas, was the site for camp in 1998. Mason Mountain Wildlife Management Area (WMA), in Mason, Texas, hosted camp in 2000. In 2002, WCC was held at Elephant Mountain WMA in Alpine, Texas. In 2004 WCC was held at H. S. Estelle 4-H & Youth Camp in Huntsville, Texas.

Changing the site of WCC not only brought camp closer to a larger number of prospective campers and staff, and it allowed for the possibility of investigating new natural regions and wildlife habitats, and the application of different types of conservation and management. For instance, the Welder Wildlife Foundation is located in the Gulf Coast Prairies and Marshes region. It is a working ranch and wildlife refuge, and campers have studied plant and animal habitats and management techniques of wetlands, chaparral, live oak chaparral, and riparian woodland sites. Elephant Mountain WMA is in the Trans-Pecos. While it is also a wildlife refuge and ranch, campers saw much different flora and fauna, and the prospective conservation and management techniques in desert scrub, desert grassland, pinyon (*Pinon* spp.)-juniper (*Juniperus* spp.) woodlands, and mountain prairies and grassland. In addition, seeing and learning about management of desert bighorn sheep (*Ovis canadensis nelsoni*) was a special experience. Three natural regions (Blackland Prairie, Oak Woods and Prairies, and Pineywoods) meet near the H. S. Estelle 4-H & Youth Camp. Instead of ranching, urban development in a state park and a national forest added new conservation and management twists. Here, red-cockaded woodpecker (*Picoides borealis*) conservation and management was emphasized.

The location of WCC, with the knowledge and experience of the all-volunteer staff, determines camp activities. Camp is conducted from 0600 to 2300, and activities are many and varied. During the daily 3.5 hours of field work, teams of campers rotate through the week to different sites, where they learn how to identify flora and fauna, practice survey techniques, and discuss management practices. Some of the field activities from the 2005 camp included: mammal transects and herpatile arrays, scent stations, remote-sensing cameras, radiotelemetry, bird identification and point counts, wetlands plant and animal identification (in fresh and saltwater environments), vegetation transects, and succession and brush management activities. Afternoons were filled with cooler activities, including: introduction to global positioning system (GPS), a hunter safety trail, wildlife track casting, presentation etiquette and PowerPoint rules, water quality analysis based upon invertebrate assemblages, white-tailed deer (*Odocoileus virginianus*) aging and scoring, ocelot (*Felis pardalis*) ecology, nuisance wildlife, conservation hot topics discussions, and a writing exercise using *A Sand County Almanac* for inspiration. Before the 2300 lights out, the evenings were full, too. Activities included archery, shot gun shooting, bat mist-netting, fishing, frog call monitoring, alligator spotlighting, a talent show that featured skits on the history of wildlife conservation in the United States, wildlife calls that called in 3 owls, a *Tacky Deer* survey, and a "career campfire." More educational fun and games were sprinkled throughout each day, such as: getting-to-know-you games, Project Wild's *Habitat Lap Sit* and *Oh Deer*, Bobwhite Brigade's *Run For Your Life*, a succession activity called *Cow Patty Ecology*, airboat safaris, and learning to kayak, then paddling around a rookery in Rockport Bay. Survivor: WCC, The Amazing Race, and Welder Olympics were adaptations of familiar themes, with a wildlife twist.

The individual mini-research projects from the early years of camp were very difficult to fit into the activity schedule. Now, the camper's projects include a plant collection, journal, and notebook. A team presentation, about their week at camp, is presented to the TCTWS executive board, and then given again to the family and friends who come to the end-of-camp awards ceremony.

With 12 camps, since 1993, TCTWS has taught 230 first-time campers. Many of the participants returned as campers, counselors, and even staff. We believe WCC is a successful way to strive toward excellence in wildlife stewardship, and we would like to see this program grow. In 2004, we conducted a workshop at The Wildlife Society's national meeting, in Calgary. Using the Texas Chapter's Operations Manual as our guide, we developed a camp organizer's manual. It includes everything from the camp's history and philosophy to organizing, planning, and camp implementation to tying up loose ends after camp is over (e.g., sending articles to campers' home-town newspapers, writing thank-you notes to sponsors and donors, and paying the bills). This manual is on CD, and is available from TCTWS. Also, if there is enough interest, the Welder Wildlife Foundation has offered to host a camp organizers workshop. Perhaps soon, we can start discussing a returning camper exchange program between various chapters' camps. It may not happen soon enough to include the 2005 WCC camper alumnus; but we can hope!

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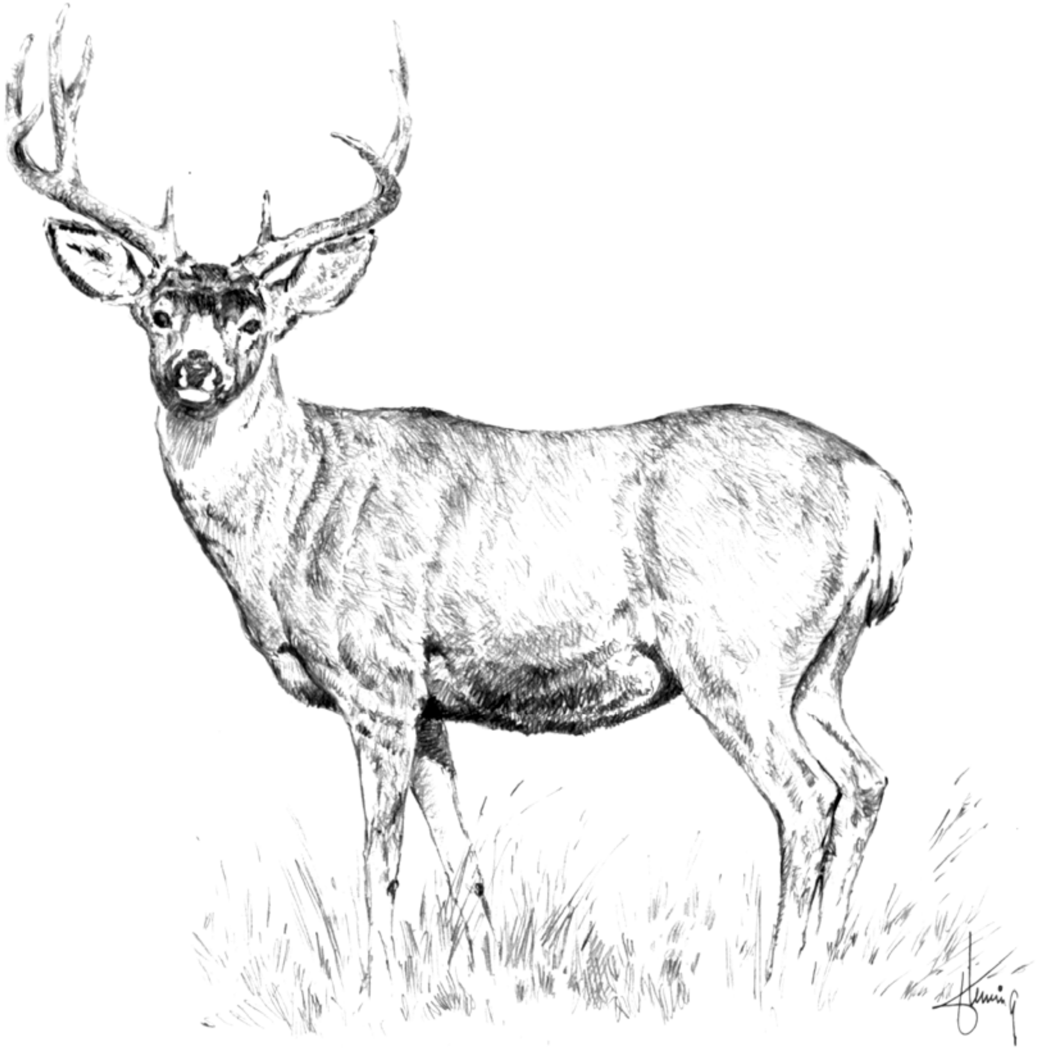

IN MY OPINION: SHINNERY OAK IS NOT A REQUIREMENT FOR LESSER PRAIRIE CHICKEN HABITAT

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Abstract: The historic range of lesser prairie chicken (*Tympanuchus pallidicinctus*) has decreased significantly in the 1900s due to numerous human-induced changes to the landscape. Today, lesser prairie chicken are in Colorado, Kansas, Okalahoma, New Mexico, and Texas in areas of sand sagebrush (*Artemisia filifolia*) and shinnery oak (*Quercus harvardii*) rangelands, with recent expansion into Conservation Reserve Program plantings, especially in western Kansas. Many researchers working with lesser prairie chickens have suggested shinnery oak as preferred habitat for the birds. I have reviewed the literature and found that shinnery oak is not and probably never was the preferred habitat for lesser prairie chicken. Areas now dominated by shinnery oak (formally dominated by tall grasses) are probably the result of human-induced changes (e.g., overgrazing and fire suppression) in the landscape and lesser prairie chickens have survived within these areas since the late 1800s or early 1900s because their preferred habitat (i.e., native prairie) have been lost due to human-induced changes in the landscape. The very fact lesser prairie chickens can exist in areas now dominated by shinnery oak is testimony to their ability to adapt to these human-induced changes; without this ability, they would have perished long ago. I have stated that shinnery oak is not essential for lesser prairie chicken habitat. Some biologists have disagreed with this assertion. Therefore, my objective is to use the literature to support my position that sandy soil areas dominated today by shinnery oak are not and were not the best habitat for lesser prairie chickens, and the majority of these birds during their peak abundance were distributed over areas now in agricultural crops.

MANAGING WILDLIFE IN THE SOUTHWEST 2006:138–142

Key words: habitat, lesser prairie chicken, *Quercus harvardii*, shinnery oak, *Tympanuchus pallidicinctus.*

BACKGROUND

Rangewide

Since the late 1800s, the range and numbers of lesser prairie chicken has been reduced from historically occupied regions of eastern New Mexico, southeastern Colorado, southwestern Kansas, western Oklahoma, and the Texas Panhandle (Crawford 1980, Taylor and Guthery 1980, Giesen 1998). Range-wide declines (>97%) in populations have resulted primarily from habitat loss (Crawford 1980, Taylor and Guthery 1980, Giesen 1998, Mote et al. 1998, Hagen et al. 2004). Robb and Schroeder (2005) estimated that lesser prairie chicken had declined by 92% since settlement by people of European descent, and an estimated 78% since the early 1960s. They also noted that concurrent with this decrease in occupied range, numbers of lesser prairie chicken have declined ≥90% since European settlement, resulting in smaller, more isolated populations.

Though considerable research has been conducted on lesser prairie chicken, declines in all of their ranges and populations have continued since 1950. In 1995, the United States Fish and Wildlife Service (USFWS) was petitioned to list the lesser prairie chicken as threatened under the Endangered Species Act, and in 1998 a "warranted but precluded" listing was given (Federal Register 1998, 50 CFR 17).

Texas Populations

Litton (1978) estimated ≤2,000,000 lesser prairie chickens were in Texas prior to 1900. By 1974, the estimated number of lesser prairie chickens in Texas was estimated at 17,000 (Litton 1978). Concerns of extinction in Texas initially arose in the 1930s, when population levels reached record lows, thus a ban on hunt-

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ing was enforced from 1937 until 1967 (Litton 1978). In 1940, lesser prairie chickens inhabited portions of 20 counties (1,366,578 ha), in the Texas Panhandle, but by 1989 occupied range had decreased by 58% (573,230 ha) and lesser prairie chickens were restricted to portions of 12 counties (Sullivan et al 2000). Though numbers of lesser prairie chickens in Texas increased to huntable levels in the 1960s, populations declined in the 1990s due to drought and continued habitat loss (Sullivan et al 2000).

SUPPORTING EVIDENCE

Rangewide

The first indication that lesser prairie chicken do not need shinnery oak can be inferred from an overlay of the former distribution (Peterson and Boyd 1998) of shinnery oak and the former distribution (Copelin 1963) of lesser prairie chicken (Fig. 1). Shinnery oak did not cover all of the former distribution of lesser prairie chicken. Peterson and Boyd (1998) estimated the historic distribution of shinnery oak to be about 2.4 million ha, whereas Taylor and Guthery (1980) estimated the historic distribution of lesser prairie chicken to be nearly 35.8 million ha suggesting that former ranges of lesser prairie chickens consisted of <7% shinnery oak habitat. In addition, Peterson and Boyd (1998) estimated that <607,000 ha of shinnery oak habitat has been lost from historic times, whereas Crawford (1980) estimated that 97% of the lesser prairie chickens have been lost since historic times.

Further supporting my contention that shinnery oak is not required by lesser prairie chickens is supported by Donaldson (1969) who found lesser prairie chickens favored sites in Oklahoma where shinnery oak was treated with 2,4,5-T and sand sagebrush with 2,4-D. Sites sprayed twice with a satisfactory kill had more display grounds and more birds when compared with untreated areas. However, such observations do not support my conjecture that lesser prairie chickens were once abundant on areas that are presently cropland. To support this conjecture, I have relied on the literature about lesser prairie chickens.

Copelin (1963) noted the lesser prairie chicken were widely distributed in western Oklahoma before the prairie sod was tilled. He stated the occupied range was greatly reduced by cultivation of most of the land, which began with settlement in 1890. In contrast, Jones (1963) noted that in Oklahoma that lesser prairie chicken habitat consisted of small units of short-grass prairie intermixed with larger units of shrub or half-shrub vegetation. Jones (1963) also noted that most display grounds of lesser prairie chickens were on the short-grass association and the only nest he was able to locate was in a short-grass community consisting of purple three-awn (*Aristida purpurea*) and sand sagebrush. Baker (1953) noted the areas in Kansas occupied by lesser prairie chicken prior to the drought of 1930–1940 decade supported tall grasses. He noted these tall grasses were eliminated over wide areas and were replaced by sagebrush (*Artemisia* spp.) and noted that in non-grazed areas, tall grasses were crowding out sagebrush, but in thousands of ha of rangeland, sagebrush and short grasses still predominated. He noted that until the native tall grasses recovered, lesser prairie chickens would not approach their former abundance. Colorado also supports lesser prairie chickens today without areas of shinnery oak (Peterson and Boyd 1998). Hoffman (1963) noted that through overgrazing, many of the mixed-grass plant communities were converted to short-grass prairie and farmland, which provided less favorable cover for lesser prairie chickens.

Lesser prairie chickens were historically found in Missouri and Nebraska (Crawford 1980), but these states did not have shinnery oak. Kansas also has no areas of shinnery oak and today supports more lesser prairie chickens than any other state. Recently their occupied range has increased as croplands have been placed into native grasses (Jensen et al. 2000) through the Conservation Reserve Program (CRP). Robb and Schroeder (2005) also noted that conversion of native grassland for production of row crops was largely responsible for the range-wide decrease in occupied habitat.

From Texas

The historic distribution of shinnery oak in Texas consisted of about 1.4 million ha and currently consists of 1.0 million ha (Peterson and Boyd 1998). Silvy et al. (2004) noted lesser prairie chickens were once found in 100 of the 254 counties of Texas, whereas by 1989 they were observed in only 11 counties. Historic

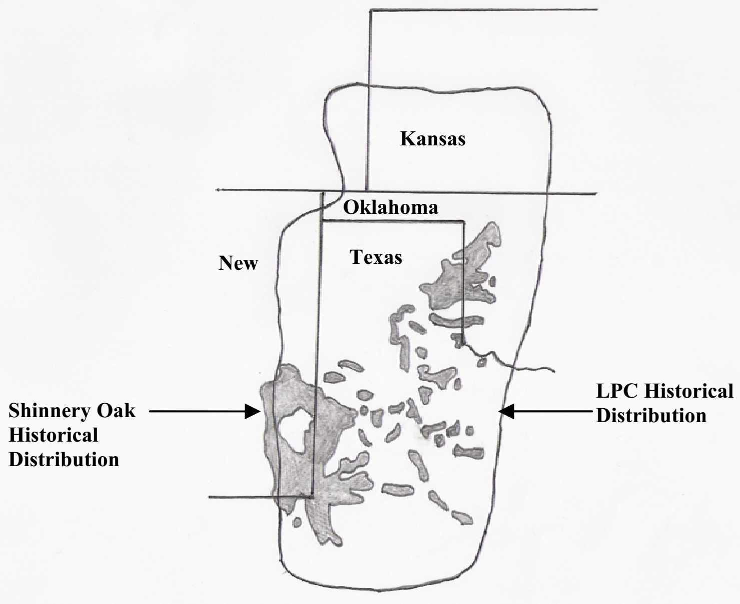

Fig. 1. Historical distribution of shinnery oak (from Peterson and Boyd 1998) and lesser prairie chicken (LPC; from Coplin 1963)

population estimates for lesser prairie chicken in Texas was 2,000,000 birds (Litton 1978), whereas today < 3,500 birds remain (Texas Parks and Wildlife Department, unpublished data). Since most of the historical shinnery oak habitat still remains in Texas, it is inconceivable the nearly 2,000,000 lesser prairie chickens historically found in Texas were tied to the 600,000 ha of shinnery oak that has been lost. In fact, much of the Texas Panhandle once occupied by lesser prairie chickens is currently in cropland (supporting my contention that most of the lesser prairie chickens in Texas were associated with areas now in cropland). The Texas Panhandle consists of the High Plains and Rolling Plains ecological regions of Texas (Gould 1962), with the vegetation on the High Plains being classified as short-grass prairie, mid-grass prairie, and in some areas as tall-grass prairie (Gould 1962). The region was essentially free from brush, but mesquite (*Prosopis* spp.) and yucca (*Yucca* spp.) had invaded some areas (Gould 1962). The Rolling Plains consists of gently rolling tall and mid grasses and brush covered plains (Gould 1962) with about 66% of the area in rangeland with the rest in crops.

In Texas, previous research on lesser prairie chickens occurred primarily in the shinnery oak rangelands, of the southwestern Texas Panhandle (Crawford and Bolen 1976, Sell 1979, Haukos and Smith 1989, Olawsky and Smith 1991), therefore, much of the habitat needs of lesser prairie chicken are biased by this research. Until recently (Toole 2005), no habitat research has been conducted in the Rolling Plains region of the Texas Panhandle where sand sage rangelands dominate. Although, Jackson and DeArment (1963) evaluated ranges, movements, and breeding success of lesser prairie chicken in Hemphill (sand sagebrush) and Wheeler (shinnery oak) counties through general observation and annual spring lek surveys from the 1940s until the 1960s. Toole (2005) concluded the more dense populations of lesser prairie chicken in Texas occur on the sand sagebrush rangelands and not the shinnery oak areas.

Toole (2005) hypothesized changes in land use, particularly the consolidation of croplands and loss of native rangeland patches, contributed to the decline of lesser prairie chickens in Wheeler County from 1940 to 1996. In contrast, lesser prairie chickens on sand sagebrush rangeland in Hemphill County remained relatively stable (525 males in 1942 to 475 males in 2001; Texas Parks and Wildlife Department, unpublished data) because large contiguous blocks of sand sagebrush rangeland remained intact. Observations of land use and interviews with local residents supported this hypothesis (Toole 2005). In Wheeler County, landowners and local residents (Toole 2005) related the historical importance agriculture had on the lesser prairie chicken population that inhabited the shinnery oak rangelands. Until the early 1970s, lesser prairie chickens in Wheeler County were routinely observed feeding in grain fields and nesting in alfalfa fields. Over time, agricultural practices were replaced with monocultures of exotic grasses (CRP fields) and lesser prairie chickens became increasingly confined to shinnery oak rangeland habitat and populations decreased from 300 males in 1942 to 11 males in 2001 on the 8,129-ha study area (Texas Parks and Wildlife Department, unpublished data). These data indicate that shinnery oak rangeland without grain fields and alfalfa fields could not support viable populations of lesser prairie chicken.

In summary, it is my contention the majority of former range of the lesser prairie chicken, as is much of the current range (i.e., Kansas, Colorado, and portions of Oklahoma and Texas), do not contain shinnery oak, and therefore, shinnery oak can not be a habitat requirement for lesser prairie chickens. Also, because <7% of the historical distribution (38.5 million ha) of lesser prairie chickens was shinnery oak (2.4 million ha) and 97% of the estimated historical numbers of lesser prairie chickens have been lost, shinnery oak could not have been the preferred habitat of lesser prairie chickens. Further, because most of the land area where lesser prairie chickens were formerly found is currently under cultivation, I reaffirm my contention that agricultural practices have been the primary demise of lesser prairie chickens over much of its former range. Lesser prairie chickens can exist in areas now dominated by shinnery oak, which is testimony to their ability to adapt to these human-induced changes; without this ability, they would have perished long ago.

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