POPULATION ENUMERATION AND THE EFFECTS OF OIL AND GAS DEVELOPMENT ON DUNE-DWELLING LIZARDS

A Thesis

by

NICOLE LIMUNGA SMOLENSKY

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

May 2008

Major Subject: Wildlife and Fisheries Sciences

POPULATION ENUMERATION AND THE EFFECTS OF OIL AND GAS DEVELOPMENT ON DUNE-DWELLING LIZARDS

A Thesis

by

NICOLE LIMUNGA SMOLENSKY

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Approved by:

Chair of Committee,	Lee Fitzgerald
Committee Members,	Kirk Winemiller
	Urs Kreuter
	Gillian Bowser
Head of Department,	Thomas Lacher Jr.

May 2008

Major Subject: Wildlife and Fisheries Sciences

ABSTRACT

Population Enumeration and the Effects of Oil and Gas Development on Dune-dwelling Lizards. (May 2008)

Nicole Limunga Smolensky, B.S., University of California, Santa Cruz

Chair of Advisory Committee: Dr. Lee Fitzgerald

Habitat loss is one of the leading causes of species decline across all taxa and conservation practices require information on population trends. The Mescalero Sands ecosystem, New Mexico, USA, is experiencing landscape changes associated with oil and gas development. The dune-dwelling lizard community contains a habitat specialist, *Sceloporus arenicolus*, that is of particular interest because it has a very limited geographic distribution that is entirely subject to oil and gas development. Distance sampling is widely used to estimate population densities of many vertebrate taxa however assumptions can be difficult to satisfy with certain species or in certain habitats. Researchers must investigate the likelihood that assumptions can be satisfied before implementing any population sampling method. I had two objectives. First to investigate the precision of population densities of dune-dwelling lizards estimated via distance sampling that was coupled with double-observer surveys. Second to compare abundances of dune-dwelling lizards among sites that varied in oil and gas development. I conducted distance line transects and compared those density estimates to densities obtained from total removal plots. I quantified the amount of oil and gas development, habitat quantity and quality and correlated those to lizard abundances to investigate the effects of oil and gas development on lizard populations.

I found large differences in density estimates from distance sampling and total removal plots that resulted from violation of distance sampling assumptions. Although distance sampling is a low cost method, it does not produce reliable density estimates for dune-dwelling lizards and is not an appropriate sampling method in this system. I did not find oil and gas development effects on the habitat quantity, quality or on the abundances of lizards. Lizard abundances were most strongly correlated to habitat quantity. Lizard abundances may be influenced by complex interactions between oil and gas development and habitat quantity and quality but controlling for those interactions was beyond the scope of my study. Before and after experiments and long-term studies at multiple sites would be required to more fully address the effects of oil and gas development on lizard populations in the Mescalero Sands.

ACKNOWLEDGEMENTS

This work would not have been possible without the aid of many people. First and foremost, I'd like to thank my committee chair Dr. Lee Fitzgerald. Much of my recent development as a scientist has been fostered by Lee. He was readily available to provide suggestions and insight throughout my master's research. I'm truly thankful for his endless concern and interest in my development as a scientist and his continuous support in my research career.

Some aspects of the data collection were laborious and would not have been possible without the tremendous help from the following individuals; I thank C. M. Bell, H. Boostrom, B. Bowers, D. Laurencio, and R. Loughridge for their assistance in data collection. Other persons including A. Padilla, D. Henderson, M. Hill, C. W. Painter, C. Shephard, and Dr. R. Jennings and his field class also helped in various aspects of data collection. Personnel from the Bureau of Land Management, New Mexico Office, especially T. Allen, S. Bird, and Q. Northrip provided information on and access to study site locations. Laura Laurencio was of great assistance in Geographic Information Systems (GIS) analyses. Dr. B. Collier and J. B. Johnson provided statistical guidance. I thank A. Subalusky, Drs. G. Bowser, L. Fitzgerald, T. Hibbitts, U. Kreuter, W. Rogers, and K. Winemiller for comments on the manuscript. Financial support for the research was provided by Texas A&M University, Bureau of Land Management, New Mexico, and the New Mexico Game and Fish Department.

Lastly I’d like to thank my family, friends and the Fitzgerald Herpetology Lab for their un-faltering support in my master’s career. These individuals provided emotional support and motivation when I needed it most. I am deeply grateful for a myriad of folks including persons not mentioned above who have helped me grow as an individual and as a scientist on the arduous path to becoming an academic.

TABLE OF CONTENTS

LIST OF FIGURES

FIGURE Page

LIST OF TABLES

CHAPTER I

INTRODUCTION

The purpose of this chapter is to present an outline of the thesis and my research objectives. Habitat loss is one of the leading causes of species decline across all taxa. Herpetofauna are especially sensitive to habitat alteration because many species occupy restricted habitats, and their relative low mobility and physiological constraints exert strong influences on their dispersal among suitable patches of habitat (Welsh et al. 2005). Management and conservation strategies of threatened and endangered species often require monitoring of multiple populations (Grumbine 1994, Meffe and Carroll 1997). As such, it is clear that rigorous methods of population enumeration need to be developed and tested.

Quantifying population densities for reptile species is notoriously difficult due to their small body size, secretive behavior, habitat preferences, physiological constraints on activity periods, and relative immobility (Turner 1977). The mark-recapture method is commonly used in herpetofaunal studies (Alberts 1993, Ballinger and Congdon 1981, Bull 1987, Hager 2001 and Hayer et. al 1994 p.183–205); however, this technique is time and labor intensive, restricted in its applicability to small spatial scales, and is better suited for monitoring single populations.

To investigate anthropogenic effects at the species level, studies must be conducted at a spatial scale large enough to incorporate multiple populations. Transect methods, such as distance sampling, are ideal for estimating populations at the landscape

---

Format and style follows Journal of Wildlife Management.

scale because they can be easily implemented, do not require the capturing or handling of animals, nor require many personnel. The accuracy and reliability of estimates from distance sampling is contingent on how well individuals can be detected during surveys. To have confidence in resulting population estimates, detection biases should be quantified to determine the accuracy of the method in the setting in which it is applied.

I am interested in understanding how well distance sampling methods may work for lizards inhabiting arid environments. This is a relevant topic with direct application for conservation and management of sensitive species of herpetofauna. A better understanding of population estimation methods for herpetofauna is important, because information on population densities across space and time is increasingly sought by natural resource agencies charged with developing conservation strategies to mediate the effects of landscape changes that may be impacting herpetofauna.

The Mescalero Sands ecosystem, located in New Mexico, USA, is experiencing landscape changes associated with livestock grazing and oil and gas development. This ecosystem is home to seven lizard species, including the endemic habitat specialist, *Sceloporus arenicolus*. This habitat specialist and other habitat generalist species may be affected differently by oil and gas development. For example, *Uta stansburiana* is a habitat generalist that is less likely to be affected by oil and gas development than a habitat specialist with very narrow habitat preferences. *Sceloporus arenicolus* is a habitat specialist that may be impacted by oil and gas development because of its limited geographic distribution and narrow habitat preferences. *Sceloporus arenicolus* is listed as endangered by New Mexico Department of Game (2006) and Fish and categorized by the U.S. Fish and Wildlife Service as a candidate for federal listing with a priority number of 2 (U.S. Fish and Wildlife Service 2004). Oil and gas development causes significant land-scarring and fragmentation of habitat due to construction of oil wells, caliché (decomposed limestone) well pads and networks of caliché roads.

In addition to outright conversion of land area to caliche, the network of roads and well pads may also influence the quality of remaining habitat for specialist species. Road building that accompanies oil and gas development directly reduces the surface area of habitat otherwise available to dune-dwelling lizards. Because of underlying geomorphological processes in dune landscapes, there may be a relationship between the amount of habitat disturbed by road-building and the quantity and quality of remaining habitat that bears long-term consequences for the makeup of the lizard community. It is unknown whether land scarring and fragmentation affects the formation and maintenance of landforms in the shinnery oak sand dune habitat of the Mescalero Sands.

No data are available on population densities of dune-dwelling lizards, and stakeholders including oil and gas companies, the Bureau of Land Management, New Mexico Department of Game and Fish, and members of the public are interested in understanding how *S. arenicolu*s densities vary across space and time in the context of increasing land development. My primary objectives were twofold:

1) Investigate the efficacy of distance sampling for population enumeration of dune-dwelling lizards.
2) Compare lizard populations among varying levels of oil and gas development, and varying levels of habitat quantity and quality.

By obtaining baseline data on population densities or relative abundance of multiple dune-dwelling lizard species, we can begin to understand how land practices and management affect the lizard assemblage over space and time. This study will produce a more complete picture of how populations of lizards vary with habitat condition and anthropogenic pressures.

CHAPTER II

IS DISTANCE SAMPLING APPROPRIATE FOR ESTIMATING POPULATION DENSITIES OF DUNE-DWELLING LIZARDS?

Introduction

Population quantification and monitoring is a fundamental aspect of management and conservation practices. Wildlife management policies, for example, are often designed according to population estimates from field-based surveys. Cost-effective methods for population estimation are essential; however, there is a trade-off between reliability of estimates and simplicity of the method (Anderson 2001, Rabe et al. 2002).

Transect-based methods that rely on visual encounters of individuals are easily implemented, do not require capture and processing of animals, can be conducted by few observers and allow for great coverage of the study area. However, accuracy and reliability of estimates from transect sampling is sensitive to variance in detection probability of individuals, which can vary as a result of observer, species, and habitat heterogeneity (Burnham and Anderson 1984, Diefenbach et al. 2003). Although mark-recapture methods require more time and resources, studies have shown the resulting population density estimates are more accurate and reliable than those based on transect methodologies (Funk et al. 2003 and Grant and Doherty 2007). The accuracy and precision of transect-based methods ultimately depend on satisfying assumptions related to detectability of animals.

Reptile populations are notoriously difficult to measure due to their small body size, secretive behavior, habitat preferences, physiological constraints on activity periods, and relative immobility (Turner 1977). Despite these well known constraints, transect sampling is commonly used for estimating population densities of herpetofauna. Several studies have reported population estimates of lizards and tortoises based on distance transect sampling (e.g. Iverson 1978, Cassey and Usher 1999, Germano et al. 2003, Reisinger et al. 2006).

Distance sampling (Buckland et al. 1993) is a transect method that can account for detection biases associated with distance between the object and observer on the transect line. Therefore, it is considered to be a better estimator of population density than traditional transect sampling, which is based solely on the encounter rate of animals (Otto and Pollock 1990 and Buckland et al. 2001). A key assumption of distance sampling is detection of all objects on the transect line. Density estimates will be biased low, for example, if detection probability is less than 1.0 on the transect line. Biases may be introduced when subjects are behaviorally or morphologically cryptic, or if observers differ in their ability to detect subjects (Graham and Bell 1989, Marsh and Sinclair 1989 and Laake and Borchers 2004). Distance sampling has been coupled with other methods (e.g. radio-tracking, double-observer counts) to compute a correction factor that accounts for objects not detected on the transect line (Laake et al. 1997, Borchers et al. 1998, Nichols et al. 2000, Anderson et al. 2001, Hounsome et al. 2005, and Kissling and Garton 2006). The double-observer method adjusts the assumed detection probability of 1.0 to a more realistic detection probability for the focal species in its habitat. Although double-observer methods may serve to adjust the detection probability, it is not a validation of the population density estimate. Therefore, independent measures of population density are crucial to assess efficacy of transect-based and other methods that rely on visual encounters.

Rodda and colleagues (2001) suggested use of total removal plots for validating the accuracy of several population estimation methods. Rodda and Campbell (2002) tested accuracy of distance sampling on several gecko species (*Hemidactylus frenatus*, *Lepidodactylus lugubris*, and *Gehrya mutilata*) and the Brown Treesnake (*Boiga irregularis*) in Guam. Total removal plots produced the best density estimates, which were significantly larger than estimates from mark-recapture and distance sampling conducted during the same study. They attributed the disparity in population estimates to missing individuals on the transect line either because they were blocked by vegetation or were inactive and unavailable for detection.

The principal objectives of my study were to quantify population densities for a community of dune-dwelling lizards and assess the accuracy and validity of distance sampling methodology in my study system. I compared population density estimates from distance sampling, coupled with double-observer survey adjustments, to densities based on total removal plots for seven species of dune-dwelling lizards. The endemic habitat specialist, *Sceloporus arenicolus*, was of particular interest because it is listed as endangered by New Mexico Department of Game and Fish (2006) and is a Candidate species with a listing priority 2 according to the U.S. Fish and Wildlife Service (U.S. Fish and Wildlife Service 2004). *Sceloporus arenicolus* (Phrynosomatidae) occurs in the Mescalero Sands and Monahans Sandhills of New Mexico and adjacent west Texas, USA, and is only found in open sandy depressions called blowouts in the shinnery oak sand-dune landscape (L. A. Fitzgerald, Texas A&M University, unpublished report). The majority of *S. arenicolus* habitat is within a landscape that is subject to extensive oil and gas development. No data are available on population densities of this species, and stakeholders including oil and gas companies, the Bureau of Land Management, New Mexico Department of Game and Fish, and members of the public are interested in understanding how *S. arenicolu*s densities vary across space and time in the context of increasing land development.

Herein, I present findings from a two year study of population density estimation for *S. arenicolus* alone, and for all seven species pooled, based on distance sampling and total removal plots. I predicted total removal plots would yield higher density estimates than distance sampling because inactive individuals are captured that would be missed during transect sampling. I was also interested in quantifying the consistency of distance estimates to total removal plot densities, to determine if distance sampling estimates in this study system could be corrected via a linear regression equation using densities from total removal plots.

Study Area and Methods

Study Area

The Mescalero Sands and Monahans Sandhills are part of the Chihuahuan Desert Ecoregion characterized by sandhill, sagebrush and shrubland habitat. Shinnery oak (*Qurecus havardii*) is the dominant vegetation and is interspersed with blowouts. Other dominant vegetation include sand sage (*Artemisia filifolia*), bunchgrasses (*Aristida* sp., *Schizachyrium* sp., *Andropogon* sp.) and mesquite (*Prosopis gladulosa*). Mean monthly temperatures for this area during my sampling months (May–July) were 19.3 C, 23.8 C and 25.4 C respectively. Total monthly precipitation for those months were 3.6 cm, 4.8 cm, and 5.8 cm respectively. Neither temperature nor precipitation was markedly different from the mean over a 68 yr period. Climate data was obtained from the National Climatic Data Center <http://www.ncdc.noaa.gov/oa/ncdc.html>.

Methods

Distance Sampling. I conducted distance sampling over two field seasons (May – July 2005–06) at 14 localities throughout the range of *S. arenicolus* (L. A. Fitzgerald, Texas A&M University, unpublished report) in Roosevelt, Chaves, Eddy, and Lea counties, New Mexico (Figure 1). *Sceloporus arenicolus* exhibit an extremely strong fidelity to dune blowouts (L. A. Fitzgerald, Texas A&M University, unpublished report); thus, transects were oriented to remain within dune complexes with blowouts. Transect starting points and headings were randomized within the shinnery oak sand dune matrix. To reduce spatial autocorrelation in the data, transect starting points were 100 m from previous transects. Starting and ending Global Positioning System (GPS) coordinates were taken for each transect. I noted cloud cover and measured substrate and air temperatures (2 cm above ground) using a quick-reading cloacal thermometer. I used Multiple Analysis of Variance (MANOVA) to test the hypothesis of no significant effect of these variables on the number of lizards seen. Transects were standardized by time rather than length because of the strong correlation between time and the number of lizards seen as well as between temperature and activity of lizards (Grant 1990, Radder et al. 2005). Transects lasted 25 minutes and were conducted during peak daily (0800 –

1300 h) and seasonal (May–August) activity periods. Transects were not conducted during rain or when substrate temperature was below 20 C or above 50 C (H. L. Snell, personal communication, University of New Mexico). When a lizard was seen, the perpendicular distance from the lizard to the transect centerline was recorded. Distances were measured by pacing; observers' paces were calibrated to the nearest 1.0 meter.

Double-observer Protocol. I conducted double-observer surveys (Nichols et al. 2000) to estimate the error associated with missing individuals on the transect line during typical distance sampling. Two observers walked the same transect in single file, with the primary observer recording every lizard detected and its associated distance from the transect line. The secondary observer recorded all detections made by the primary observer, and additional lizards missed by the primary observer. The primary and secondary observers switched roles between each transect. This double-observer approach functions similar to a mark-resight method, and allows estimation of detection probability. I analyzed the double-observer data in Program MARK 4.3 (White 2000) using a Huggins closed capture model (Huggins 1989, Huggins 1991) with constant probability of recapture among observers. The estimated detection probability was subsequently used as the multiplier to correct for lizards that may have been missed on the transect line.

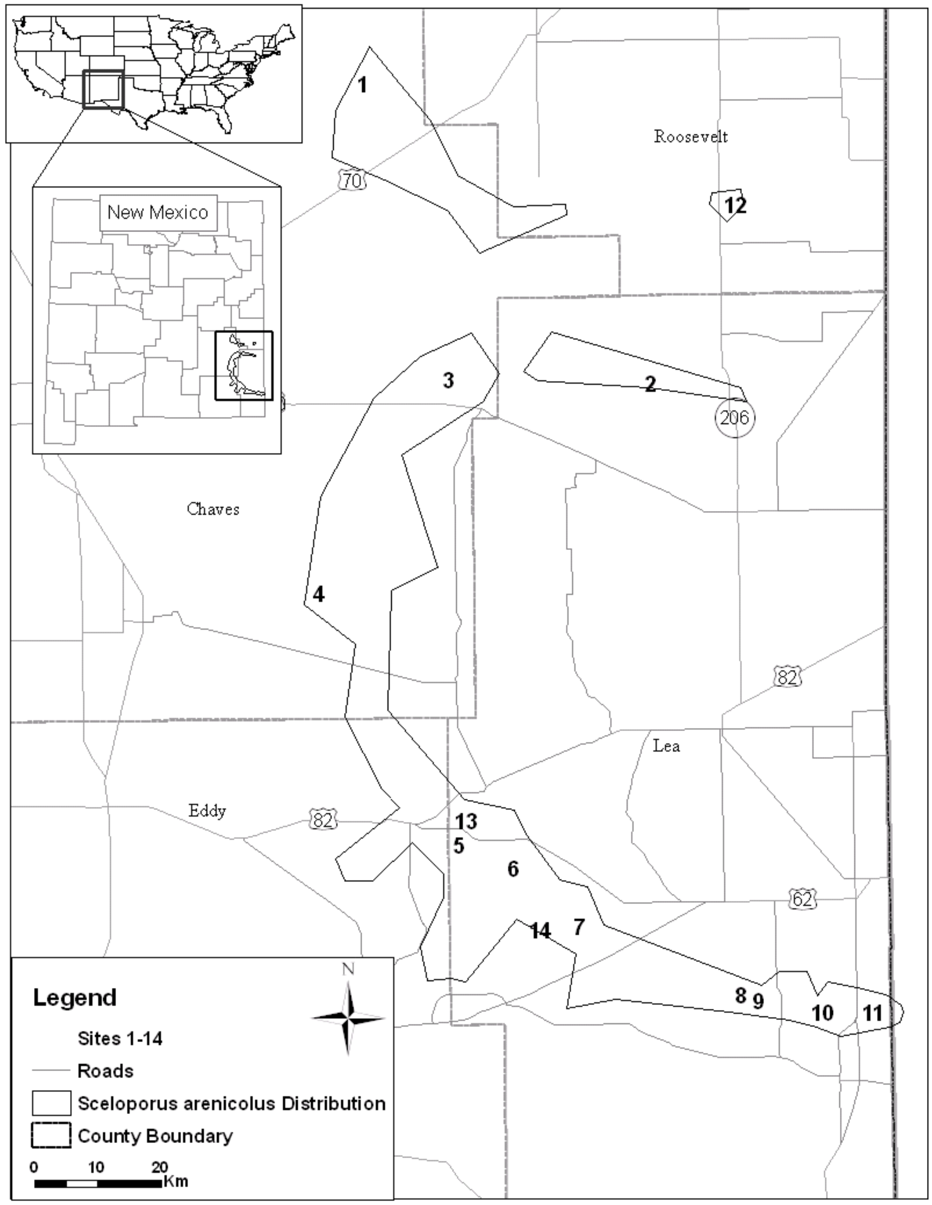

Figure 1. Map of 14 sites where distance sampling was conducted during May – July 2005–06, in New Mexico, USA. Total removal plots were constructed at sites 1–6, 8, 10, 12, 13.

DISTANCE Model Fitting. I estimated densities of *S. arenicolus* and all six lizard species pooled (collective lizards) using the DISTANCE Program Version 5.0 Release 1 (Thomas et al. 2006). Three models: uniform, half-normal and hazard rate key functions are available in DISTANCE. I followed the recommendation of Buckland and others (2001) to truncate the largest 10% of detections and include series expansions on the models to improve model fitting. I estimated variance in the detection function by a bootstrap analysis (n = 999 re-samples). I used Akaike's Information Criteria (AIC) weights and Chi-square goodness of fit tests for model selection. The hazard rate and half-normal models were used to estimate the collective lizard density and *S. arenicolus* densities among all 14 sites respectively (Table 1). I used a half-normal key function with a polynomial adjustment to model *S. arenicolus* density estimates across all 14 sites. To obtain both *S. arenicolus* and collective lizard density estimates at each site, I stratified the data. Detection functions for each site for both lizard collectively and *S. arenicolus* are summarized in Table 1.

Table 1. Model selected to estimate collective lizard densities (lizards/ha) and *Sceloporus arenicolus* densities using DISTANCE v. 5.0. Line transects occurred at 14 sites in southeast New Mexico, USA in 2005-06. Site names follow USGS DOQQ map names and numbers correspond to site locations on S. arenicolus distribution map (Figure 1) where transects were conducted. The 'N' refers to the number of individuals detected on transects.

Site name/location	Pooled species		*Sceloporus arenicolus*	
	Model selected	N	Model selected	N
Sites Pooled	Hazard-rate simple polynomial	1324	Half-normal	221
Connor Well / 4	Half-normal cosine	194	No model selected	0
Hobbs SE / 11	Hazard-rate	148	Hazard-rate	56
Hobbs SW / 10	Hazard-rate simple polynomial	91	Half-normal	13
Ironhouse Well / 6	Hazard-rate simple polynomial	83	Half-normal	6
Johnson Ranch / 2	Hazard-rate	131	Hazard-rate	34
Laguna Gatuna NNW / 7	Hazard-rate	47	Hazard-rate	5
Laguna Gatuna NW / 14	Half-normal	9	No model selected	0
Maljamar / 5	Half-normal cosine	93	Half-normal	10
Maljamar NW / 13	Half-normal	5	No model selected	0
Mescalero Point / 3	Half-normal cosine	232	Hazard-rate	73
Milnesand / 12	Half-normal	12	No model selected	1
Monument SE / 9	Half-normal	98	Hazard-rate	9
Monument SW / 8	Half-normal cosine	125	Half-normal	23
San Juan Mesa / 1	Half-normal key simple polynomial	56	Half-normal	12

Total Removal Plots. Twenty total removal plots were constructed among 10 sites (Figure 1). The number of sites that contained plots, and the number of plots constructed at each site was constrained by personnel available to construct plots. Six of the ten sites had one total removal plot. The remaining four sites had two or more total removal plots. Plots were constructed after 2200 hours to minimize disturbance of lizards. Plots were 10 meter x 10 meter (100 m^2), and consisted of a barrier made from plastic sheeting tied to wooden stakes (Figure 2). The plastic was buried 25 cm to prevent escape by lizards. Plots were left standing for two days with four 20 L pitfall traps positioned in the corners, and one in the center. All vegetation was removed on the first day and sand was raked thoroughly on the second day to ensure all lizards were found. Lizard densities from total removal plots were compared to estimates obtained from distance sampling to determine if distance sampling accurately estimated population densities. To evaluate my ability to find lizards in the plots and test whether lizards could escape, I stocked two plots with lizards. The first was stocked with 12 marked *S. arenicolus* (8 adults and 4 juveniles; both male and female) and the second with 3 *S. arenicolus* and 2 *Uta stansburiana*. I recovered all but one adult *S. arenicolus* from the two plots combined. I do not know if it escaped or simply went undetected in the plot, but the 94.1% recapture success gave us confidence in the method. Lizards were found in pitfalls before and after vegetation was removed, as well as during the raking process.

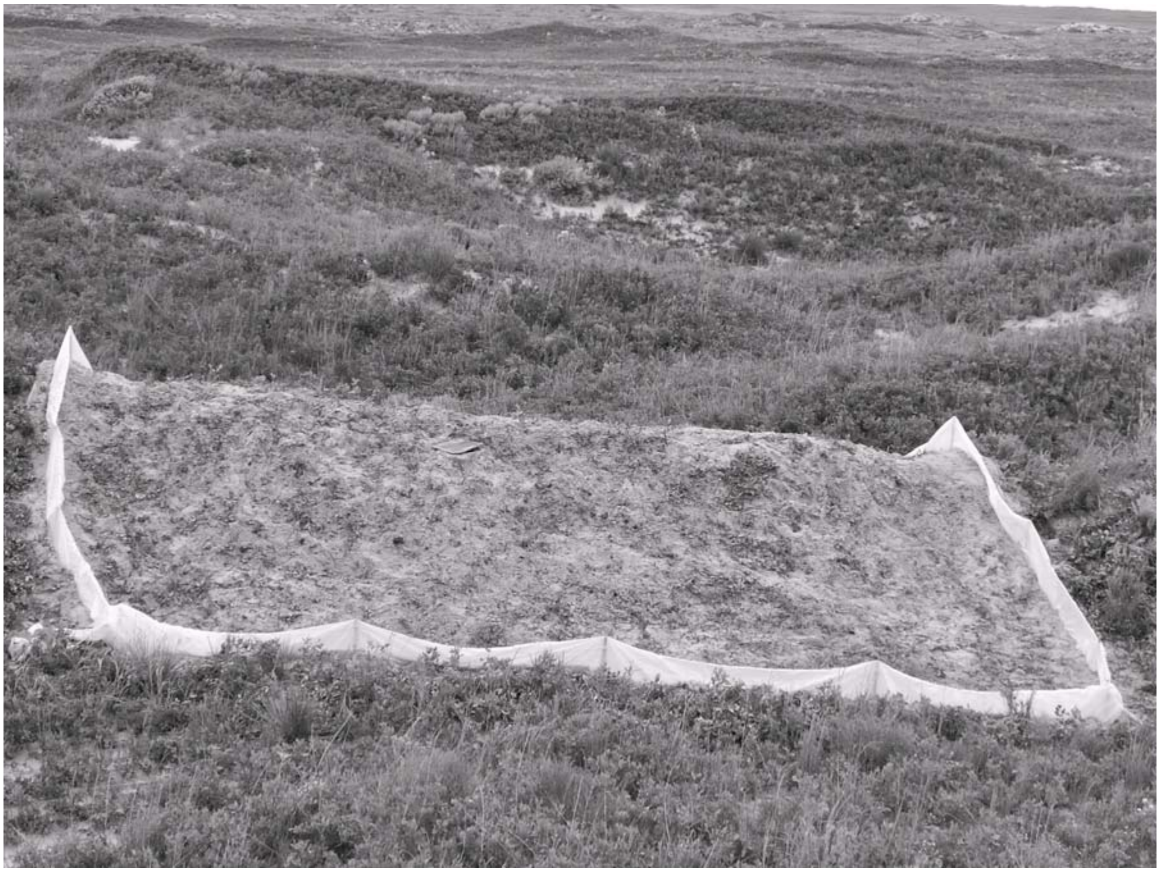

Figure 2. Example of a total removal plot.

Results

Descriptive Results

Seven lizards species were observed on 237 transect surveys during May–June 2005, and May–July 2006. The Side-blotched Lizard, *Uta stansburiana* was most frequently seen (n = 498) followed by the Sand Dune Lizard (*S. arenicolus* n = 283), Marbled Whiptail *(Aspidoscelis marmoratus* n = 228) Lesser Earless Lizard (*Holbrookia maculata* n = 101), Prairie Lizard (*S. undulatus* n = 38), Six-lined Racerunner (A. *sexlineatus* n = 37), and Texas Horned Lizard (*Phrynosoma cornutum* n = 2*)*. The

MANOVA testing for effects of substrate temperature, air temperature, cloud cover and observer revealed no significant effect of these variables on the number of lizards detected per meter ($F_{3, 213}$ = 1.49, P = 0.15). The mean number of lizards detected (collective lizard species) on a given transect was 5.70 lizards (n = 1,350; SD = 3.35). The mean number of *S. arenicolus* detected per transect was 1.07 (n = 254; SD = 1.45).

Population Density

I observed 70 lizards of six species on six double-server transects conducted during the 2006 field season. Small sample sizes precluded my ability to determine if detectability differed among species. The estimated detection probability obtained from Program MARK for all species pooled was 0.489 (± 0.065). The population density estimates for collective lizards using the estimated detection probability was 26.14 lizards/ha (n = 1,319; SD =139.46). Total variance in density estimates was attributed to three components: detection on the transect line (81.2%), probability of detecting individuals in the area surveyed (7.1%), and encounter rate during the survey (11.7%). Estimated density for *S. arenicolus* at all sites was 4.6 lizards/ha (n = 221; SD = 12.34). The component percentages of the variation in density were: detection on the transect line (53.3 %), probability of detecting individuals in the area surveyed (9.3 %) and encounter rate (37.4 %). Lizards population densities at each site ranged from 8.4 – 106.7 lizards / ha; and for *S. arenicolus* ranged from 0.0 – 8.1 lizards / ha (Table 1, Figures 3, 4).

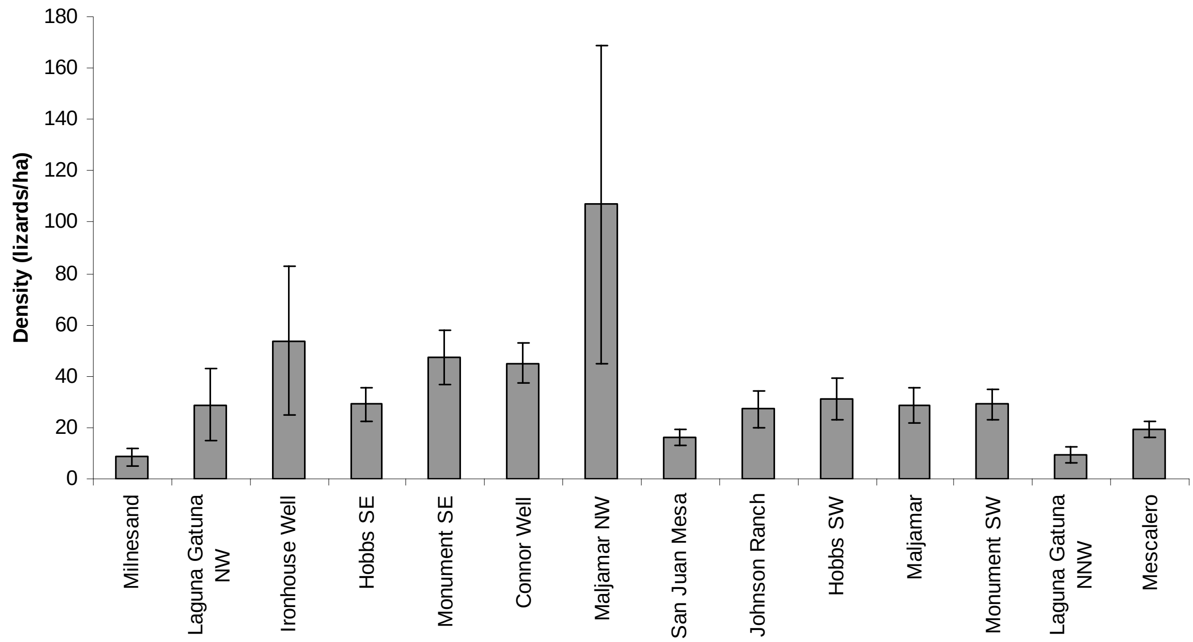

Figure 3. Distance density estimates of all lizard species at each site. Sites were located in southeastern New Mexico, U.S.A., and sampling was conducted May–July 2005–06.

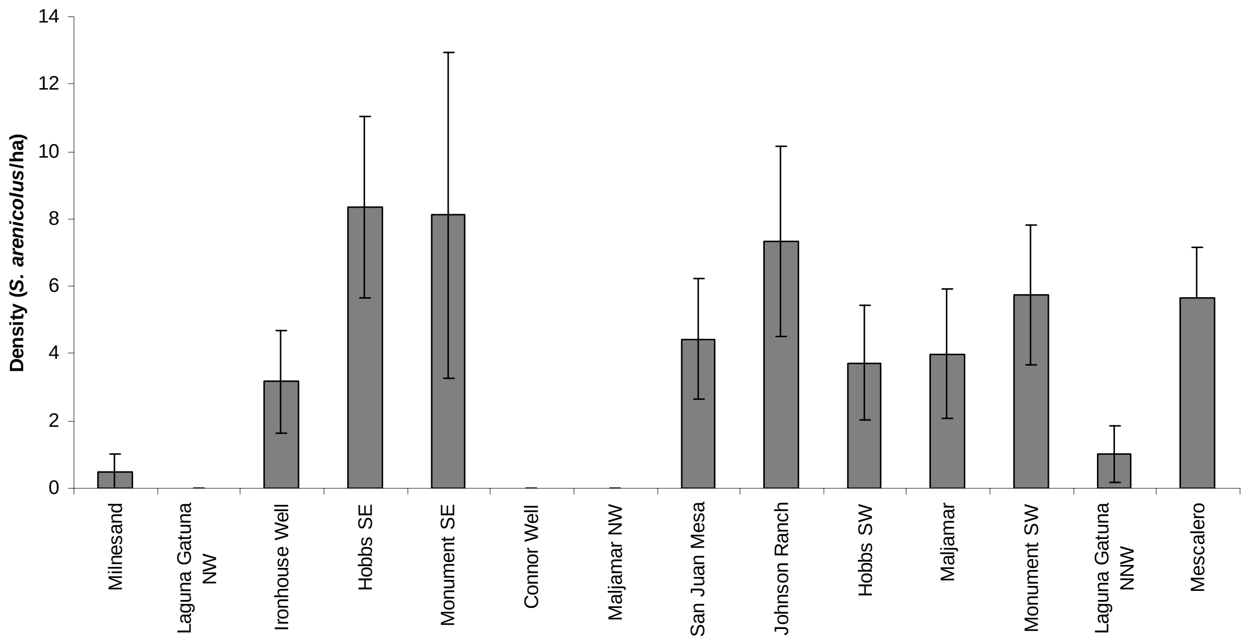

Figure 4. Distance density estimates of *Sceloporus arenicolus* at each site. Sites were located in southeastern New Mexico, U.S.A., and sampling was conducted May–July 2005–06.

Total Removal Plots

Twelve of the 20 total removal plots contained lizards ($\bar{x}$ = 0.85 lizards / 0.01 ha, SD = 0.88, range = 0–3, n = 17 lizards) (Table 2). *Sceloporus arenicolus* was the species most commonly caught in the plots with a mean 0.30 per plot (SD = 0.58, range 0 – 2, n = 6 lizards). Five other lizard species were captured in the plots (most to least captured): *U. stansburiana* (n = 3), *A. sexlineatus* (n = 3), *A. marmoratus* (n = 3), *S. undulatus* (n = 1) and *H. maculata* (n = 1). The mean density of all species pooled based on total removal plots was 85 lizards/ha (SD = 87.51). The density of *S. arenicolus* based on total removal plots was 30.0 lizards/ha (SD = 57.11). Standard error bars are plotted at sites where more than one plot was constructed (Figure 5).

Table 2. Total removal plot data illustrating the number of lizards and *Sceloporus arenicolus* caught in plots.

Location	Collective Lizards	*Sceloporus arenicolus*	Species
1	0	0	
1	1	0	As
2	2	0	As, Hm
2	1	1	Sa
3	1	0	Us
3	1	1	Sa
4	1	0	Am
5	1	1	Sa
6	0	0	
8	3	0	2 Us, Am
10	0	0	
10	0	0	
10	2	0	As, Su
10	0	0	
10	0	0	
10	0	0	
10	1	1	Sa
10	2	2	2 Sa
12	0	0	
13	1	0	Am

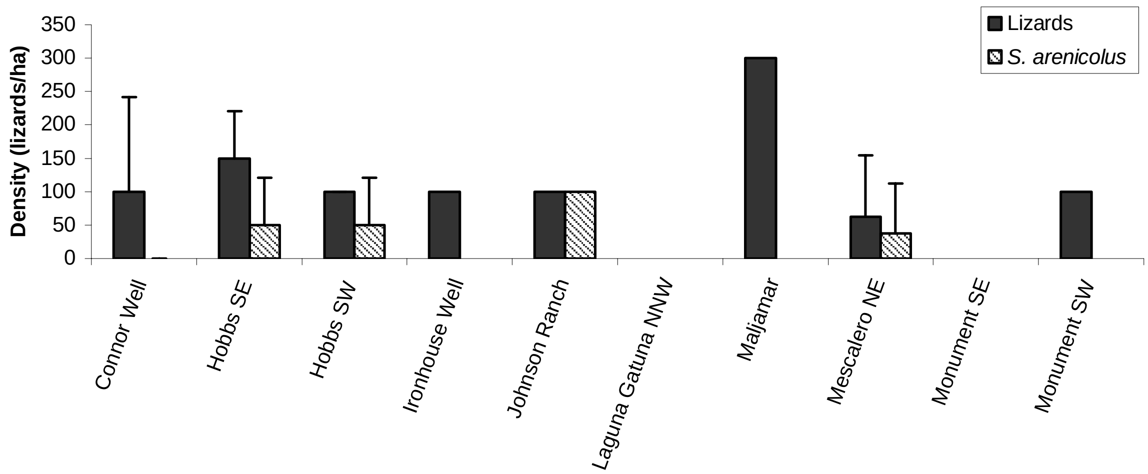

Figure 5. Total removal plot densities at each site where plots were constructed for lizards collectively, and *Sceloporus arenicolus*. Error bars are added to sites that had more than one total removal plot. Sites were located in southeastern New Mexico, U.S.A., and sampling was conducted May–July 2005–06.

Comparison of Distance Sampling to Total Removal Plots

Lizard densities estimated via distance sampling were significantly lower then densities estimated via total removal plots ($t_{18,\ 10,\ 0.05}$ = - 2.57, P = 0.02). The mean density from total removal plots for *S. arenicolus* was 6.5 times greater than the mean density from distance sampling. However, the variance in the total removal plot densities was large such that the difference between the two sampling methods was not significant for *S. arenicolus* densities (T = 98, P = 0.70) (Figures 6, 7). The large variation in total removal plot densities prevented deriving a regression equation to correct the distance density estimates with total removal plot density estimates.

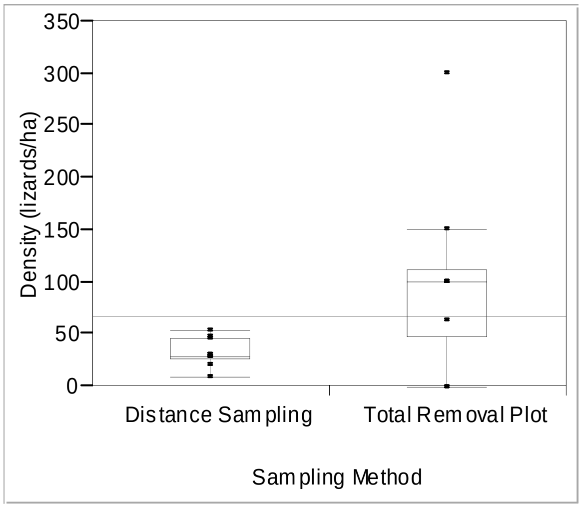

Figure 6. Comparison between two sampling methods for lizard densities from 14 sites. The horizontal line represents the mean density (66.7 lizards/ha) for both sampling methods. Sites were located in southeast New Mexico, U.S.A., and sampling was conducted May–July 2005–06.

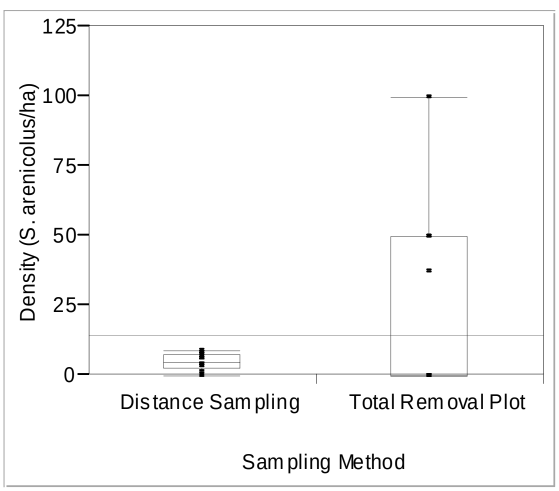

Figure 7. Comparison between two sampling methods for *Sceloporus arenicolus* densities from 14 sites. The horizontal line represents the mean density (14.2 lizards/ha) for both sampling methods. Sites were located in southeast New Mexico, U.S.A., and sampling was conducted May–July 2005–06.

Discussion

Distance sampling has become ubiquitous in population monitoring of fauna (Research Unit for Wildlife Population Assessment <http://www.ruwpa.st-and.ac.uk/distance/distanceusers.html>) and specifically herpetofauna (Dodd 1990, Akin 1998, Jenkins et al. 1999 and Dickinson and Fa 2000). Few studies verify distance sampling assumptions are not violated (Bächler and Liechti 2007). Physiological constraints on herpetofauna make it unlikely that all individuals on a transect line are detected during distance sampling of herpetofauna. In my assessment of distance sampling for lizards in a dune-dwelling landscape, I found estimates from distance sampling coupled with a corrected detection probability, still greatly underestimated densities as compared to my total removal plots. My DISTANCE density estimates for pooled species were 69.4% biased low as compared to densities from total removal. For *S. arenicolus,* the difference in DISTANCE density estimates was even larger with an 84.7% negative bias. The findings of this study and Rodda and Campbell (2002) suggest the discrepancy between the two methods was due to two factors: 1) missing lizards on the transect line, and 2) low sample size of total removal plots.

Missing Lizards on the Transect line

Detectability on the transect line is a key issue in distance sampling and the assumption that all individuals on the transect line are detected is rarely satisfied. Even conspicuous, slow moving reptiles can be missed on the transect line (Freilich et al. 2000). Anderson and colleagues (2001) found a 12% and 19.5% negative bias of adults and sub-adult desert tortoises (*Gopherus agassizzi*) respectively compared to known population densities. This bias resulted from observers missing tortoises that were above the ground and available for detection during distance sampling surveys.

Double-server correction factors have been successful in accounting for this type of bias (Graham and Bell 1989, Nichols et al. 2000 and Kissling and Garton 2006); however, this method did not perform well with dune-dwelling lizards. The correction factor derived from double-server surveys only accounted for visibility bias (Pollock and Kendall 1987). Only active lizards that were missed by the first observer and detected by the second observer were quantified. Double-observer surveys did not account for lizards that were inactive and beneath the surface during surveys. The total removal plots contained both active and inactive lizards that were detected during the removal of vegetation and raking of sand. The disparity between the density estimates from distance sampling and total removal plots suggests the proportion of inactive lizards missed and beneath the surface, albeit unknown, was much larger than the proportion of missed lizards above the ground and accounted for via double-server surveys. Consequently, the use of the double-server correction factor was not suitable method to overcome the limitations of distance sampling methods for lizards.

Quantifying a realistic estimate of detection probability on the transect line for reptiles and amphibians is challenging. The double-server correction factor in this study resulted in an increase in the coefficient of variation in my DISTANCE density estimates that translates to a loss of precision. Funk et al. (2003) used a correction factor derived from the ratio of mark-recapture densities to distance sampling densities for frogs in the genus *Eleutherodactylus*. Similar to my study, the corrected DISTANCE density estimates had a substantial decrease in precision and resulting density estimates that were ineffective for monitoring population trends of *Eleutherodactylus*.

The unpredictable activity levels or reptiles and amphibians can bias both the detection correction factor and DISTANCE density estimates (Nussear and Tracy 2007). Laake et al. (1997) and Anderson et al. (2001) recommend the use of radio telemetry to estimate availability bias and correct DISTANCE estimates. However, availability of individuals is contingent upon activity patterns that vary both temporally and amongst individuals (Whitford and Creusere 1977, Dunham 1981 and Dorcas and Peterson 1998). Consequently, a single correction factor derived from radio telemetry may not be applicable over time. To date, there is no suitable correction factor for detection biases on the transect line during distance sampling of herpetofauna.

Sample Size of Total Removal Plots

Total removal plots were well suited for detecting both active and inactive lizards as illustrated by the high success rate of detecting all lizards present in my plots. Although the density of lizards within the 100 m^2 plots were accurate, extrapolation of these densities to larger areas may have been positively biased given the limited number and size of my total removal plots. This is turn may have contributed to large differences in total removal plot densities and DISTANCE density estimates. The variance in population densities from the plots was large and I was unable to devise a corrected DISTANCE density estimate from the plots. The population estimates reported in this study serve as indices rather than precise estimates and my research suggests distance sampling is not appropriate for population density estimation of dune dwelling lizards.

Management Implications

Oil and gas development will continue throughout *S. arenicolus* habitat, increasing the threat of extirpation of fragmented *S. arenicolus* populations. Monitoring *S. arenicolus* populations at the landscape scale will become increasingly important for conservation of this highly endemic species. The secretive nature and varied activity patterns of herpetofauna impede the use of population monitoring techniques commonly used for other fauna. Though total removal plots eliminate detection biases, they are labor intensive and are better suited for assessing precision of other population estimation methods than as a population estimation technique. Some authors (Goldberg and Schwalbe 2004 and Barrows 2006, Mazorolle et al. 2007) have suggested the use of alternative measures, such as demographic parameters, modeling population dynamics or site occupancy modeling, may be more appropriate to monitor populations at landscape scale and determine species persistence. Future studies will compare mark-recapture sampling with total removal plots in the Mescalero Sands ecosystem. Although mark-recapture methods cannot be conducted at many locations simultaneously to obtain the population density of the entire species, it may be the best method to quantify and monitor populations in areas subject to high development.

CHAPTER III

EFFECTS OF OIL AND GAS DEVELOPMENT ON DUNE-DWELLING LIZARDS

Introduction

Anthropogenic habitat degradation and fragmentation is one of the leading threats to biodiversity (Fahrig 1997). Changes in landscapes can hinder dispersal of organisms (Andrén 1994), increase both intra- and interspecific competition for resources (Ballinger and Watts 1995 and Fahrig 2003) and alter predator prey dynamics (Andrèn 1994, Crooks and Soulé 1999, Ryall and Fahrig 2005). Whereas habitat degradation can alter entire communities of flora and fauna (Tilman et al. 1994, Laurance 2000), the effects of habitat loss are manifested differently in different species (Fischer et al. 2004, Rizkalla and Swihart 2006). These differences can complicate conclusions about impacts of habitat loss on entire assemblages (MacNally 2002). For example specialist species may be more negatively impacted by habitat change than generalist species in the same community (Hecnar and M'Closkey 1998, Vega et al. 2000), and the number of species within the community may either remain unchanged or even increase however the specific species within the community may change (Polus et al. 2007).

Habitat loss is one of the leading causes of herpetofaunal population decline (Shine 1991, Gibbons et al. 2000, Collins and Storfer 2003, Gardner et al. 2007). Despite the obvious importance of landscape effects of many decades of oil and gas development, there is no mention of the effects of oil and gas development on reptiles in the aforementioned literature reviews. Oil and gas development causes significant land-scarring and fragmentation of habitat due to construction of oil wells, caliche (decomposed limestone) well pads and networks of caliche roads. Many studies have documented specific impacts of oil and gas development on other classes of wildlife (Bradshaw et al 1997, Fiori and Zalba 2003, Lyon and Anderson 2003, Trail 2006), and research is needed to show how reptile populations respond to this development.

The Mescalero Sands ecosystem of southeast New Mexico and adjacent Texas, USA supports a lizard assemblage consisting of generalist and specialist species that occupy a sand dune system semi-stabilized by shinnery oak (*Quercus havardii*). Included among the seven lizard species that occupy this shinnery oak, sand dune habitat is the endemic habitat specialist *Sceloporus arenicolus* and the widespread habitat generalist *Uta stansburiana*. *Sceloporus arenicolus* has the second smallest geographic range of lizards in the United States and is listed as candidate species with a listing priority 2 (U.S. Fish and Wildlife Service 2004). This species is exclusively found in and around open bowl-shaped depressions called blowouts and does not use other habitat types in the surrounding landscape. *Sceloporus arenicolus* prefers relatively large blowouts based on their availability in the landscape (L. A. Fitzgerald, unpublished report, Texas A&M University). Other lizard species use both sand dune blowouts and the surrounding shinnery oak matrix. The status of *S. arenicolus* has drawn the attention of natural resource agencies, the oil and gas industry, conservation organizations, and other stakeholders because *S. arenicolus* populations occur on land leased by the State Leasing Office of New Mexico and Bureau of Land Management of New Mexico (L. A.

Fitzgerald, unpublished report, Texas A&M University). All stakeholders are interested in knowing more about natural variation in populations of *S. arenicolus,* both temporally and spatially, especially in the context of oil and gas development.

I investigated the effects of oil and gas development on the dune-dwelling lizard assemblage in the Mescalero Sands Ecosystem. Specifically, I compared lizard abundances in areas with varying amounts of oil and gas development, as indicated by the surface area of the landscape covered by caliche well pads and roads. In addition to outright conversion of land area to caliche, the network of roads and well pads may also influence the quality of remaining habitat for specialist species. Blowouts are an important landscape feature for *S. arenicolus*, and were used as my index of habitat. I compared lizard abundances among areas varying in size and total area of blowouts. I predicted that lizard abundances would decrease in association with increasing oil and gas development, and in association with reduced habitat quality and quantity. There is evidence that suggests individual oil well pads have a negative effect on *S. arenicolus* numbers, a pattern not found for other lizard species (H. L. Snell, unpublished report, University of New Mexico). Therefore, I predicted that the specialist, *S. arenicolus*, would have low abundances in areas containing relatively high amounts of oil and gas development and relatively low habitat quantity and quality. Because *U. stansburiana* is a habitat generalist that can use various types of habitats, I predicted *U. stansburiana* abundances would not change with the amount of oil and gas development or quantity or quality of habitat.

This study provides insight into the effects of an understudied form of habitat degradation on herpetofauna. Specifically, oil and gas development may affect *S. arenicolus* abundances as well as other lizard species that occupy shinnery oak, sand dune habitat and the findings from this research are applicable to land management and conservation of the Mescalero Sands ecosystem (MacNally et al. 2002). Comparing and contrasting effects of oil and gas development on a species of habitat generalist and a species of habitat specialist will elucidate how vertebrate community dynamics may change with habitat degradation.

Study Area and Methods

Study Area

Study sites in the Mescalero Sands Ecosystem were located in Chaves, Eddy and Lea counties, New Mexico, USA. This ecosystem is characterized by stabilized and semi-stabilized dunes interspersed with shinnery oak (*Quercus havardii*), sagebrush (*Artemisia filifolia*), bunchgrasses (*Aristida* sp., *Schizachyrium* sp., *Andropogon* sp.) and mesquite hummocks (*Prosopis gladulosa*). I quantified lizard abundance at 11 sites based on presence of *S. arenicolus* at those sites, and amount of oil and gas development (Figure 8).

Methods

Lizard Abundance. Lizard abundances were estimated from line transects surveyed in May – July 2005–06. Observers were trained to accurately identify lizards before surveying transects that were constrained to 25 minutes. Transects were standardized by time because of the correlation between the number of individual seen and time spent searching (Radder et al. 2005). Transects were randomly located within shinnery oak, sand dune habitat after presence of *Sceloporus arenicolus* was verified. I did not consider other habitat types, because *S. arenicolus* does not use them. As such, my results only apply to the effects of oil and gas development on lizards in the shinnery oak, sand dune habitat. All transects were surveyed between 0800 – 1300 h. Transects were not surveyed during rain or when substrate temperature was (< 20° C or > 50° C).

Measures of Habitat: Alteration, Quantity, and Quality. My indicator of oil and gas development on the landscape was total surface area of caliche (TSAC), which was the total area of oil well pads and roads connecting them. I used a Geographic Information System (ArcMap9.0; ESRI 2005) to quantify TSAC in a in a 259 ha (1 mi^2) area of shinnery oak, sand dune habitat surrounding the locations of transects at each study site. This spatial scale was large enough to include caliche well pads and roads in the immediate area surrounding transects. The New Mexico State Land Office (NMSLO 2006) provided GIS data in the form of shapefiles that represent the locations of oil wells and roads used in this study. The size of well pads and width of roads were standardized at 6400 m^2 and 4 m, respectively. An assumed road width of 4 m was conservative based on guidelines for caliche road development suggested by both the New Mexico Commission of Public Records (http://www.nmcpr.state.nm.us/NMAC/parts/ title19/19.002.0020.htm accessed December 2007) and Chaves County Commissioners (http://co.chaves.nm.us/agendas/2006/101906/101906-A3.pdf m accessed December

2007). Total road area and total well pad area were summed at each site.

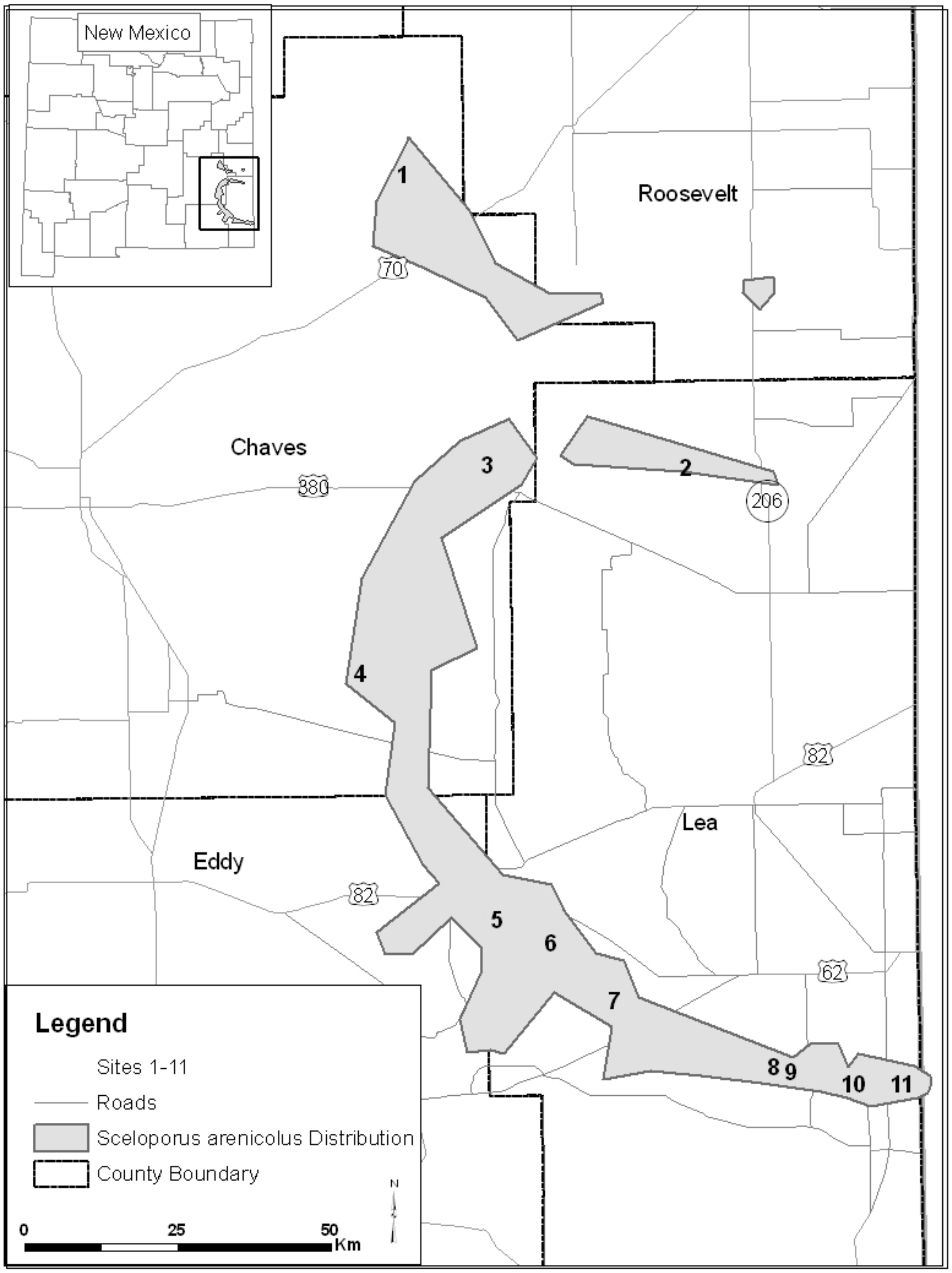

Figure 8. Location of study area and eleven sites where transects were conducted during May–July 2005–06.

Total area of blowouts (TAB) was used as my index of habitat quantity, because blowouts are integral to the shinnery oak sand dune habitat, and the only landscape feature that *S. arenicolus* uses. *Sceloporus arenicolus* are more likely to occur in larger blowouts than smaller blowouts (L. A. Fitzgerald, unpublished report, Texas A&M University) thus larger blowouts are considered better habitat quality than smaller blowouts. Many small blowouts can result in the same total area as a few large blowouts; for this reason, habitat quantity does not necessarily equal habitat quality. Thus, I used mean blowout size (MBS) as my index of habitat quality. I measured the area of all blowouts within the 259 ha sites in ArcMap 9.0 (ESRI 2005). A polygon shapefile of all shinnery oak blowouts was created from 2004 aerial photos obtained from NMSLO.

Analyses

Data from all species encountered were pooled for analysis of collective lizard abundance. Abundances of *S. arenicolus* and *U. stansburiana* were analyzed separately to compare effects of oil and gas development on a habitat specialist and a habitat generalist. To compare abundances among sites, I standardized the count data per unit effort. The total number of transects surveyed at a site was multiplied by 25 minutes (each transect was surveyed for 25 minutes) resulting in total search effort. Sites that had less than five transects were excluded from analyses. The total counts for a site was divided by total search effort. This resulted in an encounter per unit effort (EPUE) which was used as my abundance estimate.

I used Analysis of Variance (ANOVA) to compare EPUE distributions for *S. arenicolus, U. stansburiana* and lizards collectively across all sites. I rank transformed

EPUE of *S. arenicolus* and *U. stansburiana* prior to ANOVA to satisfy ANOVA assumptions. I used Pearson product-moment correlation and linear regression to investigate the relationships between oil and gas development and EPUE of collective lizards, *S. arenicolus* and *U. stansburiana*. I also used correlation and linear regression to assess the relationship between my habitat quantity and quality parameters and EPUE of collective lizards, *S. arenicolus* and *U. stansburiana*. Collective lizard abundance and *S. arenicolus* abundance were normally distributed. I used a square root transformation on *U. stansburiana* abundance to normalize the data prior to correlation and regression.

Results

Two hundred twenty-seven transects were surveyed at the eleven sites that varied in amounts of TSAC, TAB and MBS (Table 3). A total of 1,321 lizards (0.232 lizards/minute) comprised of seven lizard species were encountered on transects. *Uta stansburiana* (0.081 lizards/minute) was most frequently detected followed by *S. arenicolus* (0.046 lizards/minute), *Aspidoscelis marmoratus* (0.036 lizards/minute), *Holbrookia maculata* (0.016 lizards/minute), *A. sexlineatus* (0.009 lizards/minute), and *S. undulatus* (0.006 lizards/minute) (Figure 9). Overall lizard abundance and abundance of *U. stansburiana* varied significantly among sites ($F_{10,216} = 12.09$ $P < 0.01$ and $F_{10,216} = 5.69$ $P < 0.01$, respectively). Multiple comparisons showed five groups of sites had similar abundance indices of lizards overall, and three groups of sites had similar mean ranks for *U. stansburiana* (Table 4). I did not find a significant difference in ranks of *S. arenicolus* abundance among sites, but there clearly was a great deal of variation in transect counts for this species ($F_{10,216} = 1.83$ $P < 0.057$; Figure 9). Abundances of other species of lizards were not tested because sample sizes were relatively low at many sites.

I did not find a statistical correlation between TSAC and TAB, nor between TSAC and MBS ($r = -0.32$, $P < 0.34$ and $r = -0.08$, $P < 0.82$, respectively, Figure 10). Thus there was no apparent signal of oil and gas development associated with habitat quality or quantity at these 11 sites. Total area of blowout was significantly positively related to MBS ($R^2 = 0.41$ $P < 0.03$, Figure 11), indicating that habitat quality increased with habitat quantity.

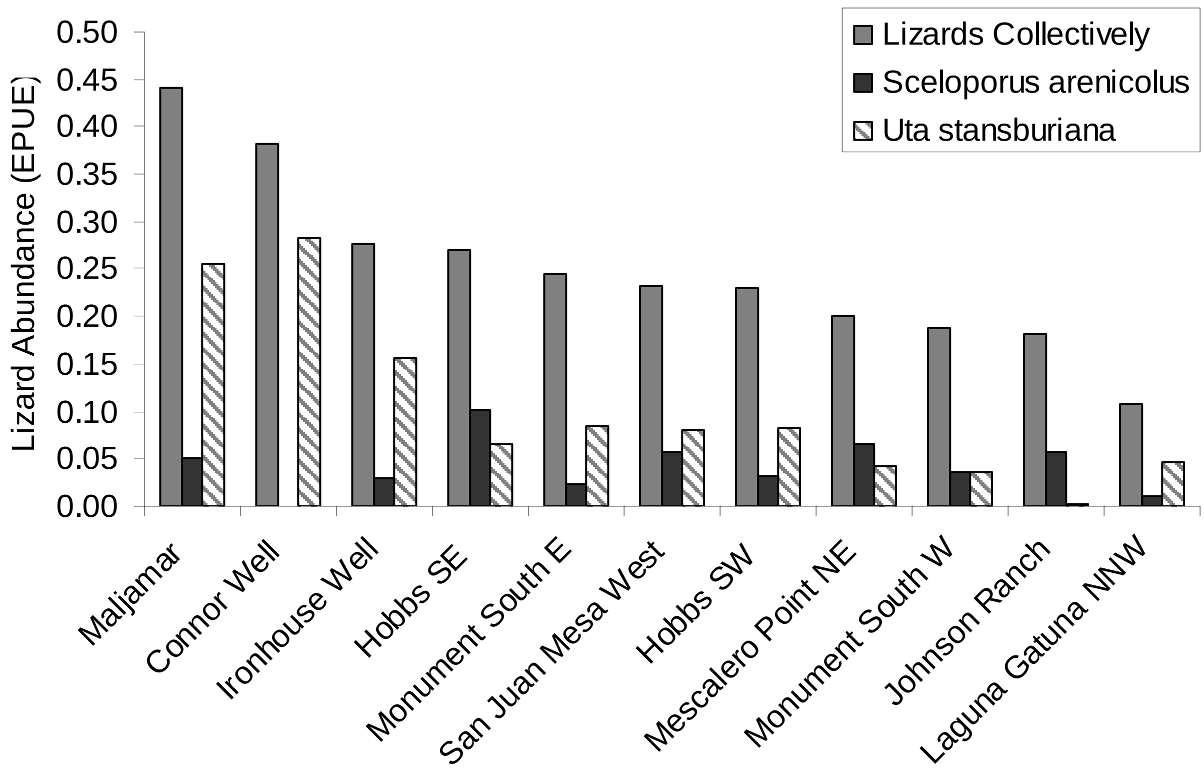

Figure 9. Distribution of lizard abundances of all seven species of dune-dwelling lizards collectively, *Sceloporus arenicolus*, and *Uta stansburiana* among 11 sites in New Mexico, USA. Abundance data are depicted as Encounter Per Unit Effort (EPUE).

Table. 3 Total surface area of caliche (TSAC), total area of blowout (TAB), mean blowout size (MBS) and encounter per unit effort (EPUE) of lizards collectivley, *Sceloporus arenicolus*, and *Uta stansburiana* at eleven sites in the Mescalero Sands ecosystem, New Mexico. The proportion of land area comprised of caliche well pads and roads or blowouts within each 259 ha sites is in parantheses.

Site Number corresponding to Map	Site Location	TSAC ha	TAB ha	MBS m^2	Lizard EPUE	*Sceloporus arenicolus* EPUE	*Uta stansburiana* EPUE
9	Monument South E	23.88 (9%)	26.4 (10%)	22.78	0.245	0.023	0.085
8	Monument South W	23.28 (9%)	25.23 (10%)	29.04	0.188	0.036	0.036
6	Laguna Gatuna NNW	14.45 (6%)	14.47 (6%)	19.8	0.104	0.011	0.047
5	Maljamar	12.24 (5%)	43.54 (17%)	25.78	0.387	0.036	0.231
11	Hobbs SE	7.52 (3%)	32.35 (12%)	22.33	0.275	0.101	0.066
2	Johnson Ranch	6.96 (3%)	41.06 (16%)	28.15	0.182	0.057	0.003
7	Ironhouse Well	4.20 (2%)	31.71 (12%)	15.25	0.277	0.030	0.157
10	Hobbs SW	3.73 (1%)	20.78 (8%)	16.96	0.230	0.033	0.083
4	Connor Well	2.47 (1%)	50.82 (20%)	37.85	0.377	0.000	0.282
3	Mescalero Point NE	1.52 (1%)	31.37 (12%)	27.42	0.205	0.066	0.043
1	San Juan Mesa West	1.32 (1%)	26.76 (10%)	28.15	0.232	0.056	0.080

Table. 4 Tukey's honest significant difference test of lizard abundance and *Uta stansburiana* abundance among 11 sites in the Mescalero Sands ecosystem, New Mexico. *Uta stansburiana* abundances were ranked transformed prior to analyses. Sites with similar mean abundances are grouped by letter a - d.

Site	Number of Transects	Lizards					*Uta stansburiana*		
		a	b	c	d	e	a	b	c
Hobbs SE	23	0.115					71.8		
Johnson Ranch	29	0.141					80.0	80.0	
Mesaclero Point NE	47	0.196	0.196				110.0	110.0	110.0
Hobbs SW	16	0.203	0.203	0.203			111.0	111.0	111.0
San Juan Mesa West	10	0.204	0.204	0.204			116.7	116.7	116.7
Ironhouse Well	12	0.230	0.230	0.230	0.230		119.6	119.6	119.6
Connor Well	21		0.270	0.270	0.270		127.5	127.5	127.5
Laguna Gatuna NNW	18		0.271	0.271	0.271		132.5	132.5	132.5
Monument South W	27			0.321	0.321	0.321		145.3	145.3
Monument South E	16				0.328	0.328			152.2

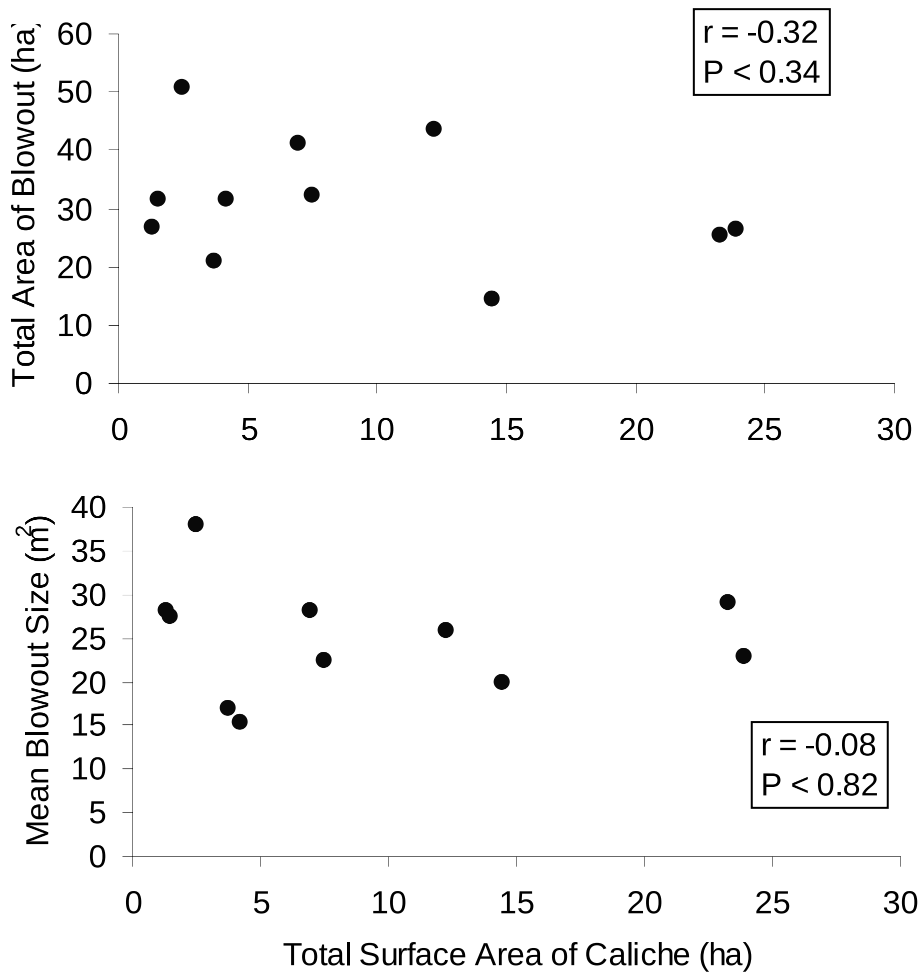

Figure 10. Relationship between total surface area of caliche and both total area of blowout and mean blowout size among 11 sites located in the Mescalero Sands ecosystem, New Mexico USA.

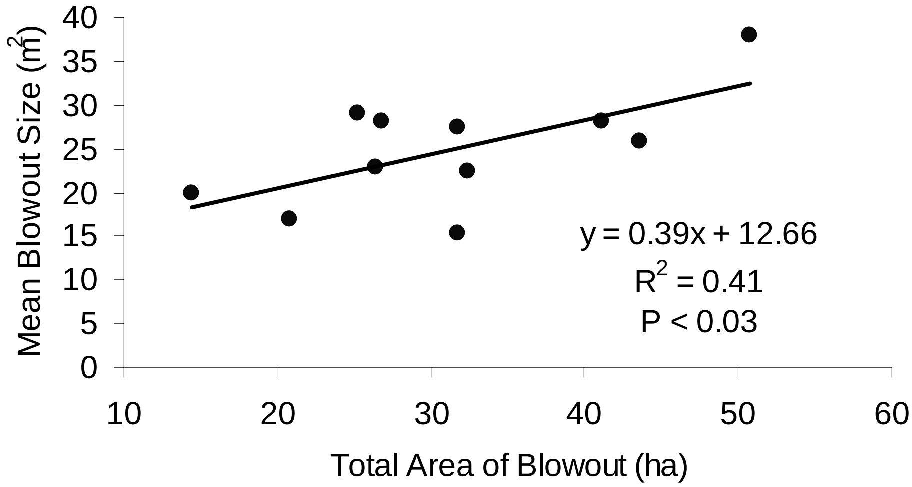

Figure 11. Relationship between total area of blowout and mean blowout size among 11 sites in the Mescalero Sands ecosystem, New Mexico USA.

Total Surface Area of Caliche and Abundance

I did not find a significant correlation between oil and gas development and EPUE of lizards collectively, *S. arenicolus* or *U. stansburiana* (r = -0.23, P < 0.50, r = -0.25, P < 0.45, r = -0.19, P < 0.58, respectively). Consequently these relationships could not be predicted by simple linear regression (R^2 = 0.05, P < 0.50, R^2 = 0.07 P < 0.45, R^2 = 0.04, P < 0.58, respectively; Figure 12).

Total Area of Blowout and Abundance

The total area of blowout (TAB) had a significantly positive relationship with collective lizard EPEU (r = 0.77, P < 0.01, Figure 13). Fifty nine percent of the variance of collective lizard EPUE could be explained by the linear regression equation: collective lizard EPUE = 0.006 x TAB + 0.57. My hypothesis that lizard abundances would increase with an increase in the amount of habitat was supported. I found a nearly significant relationship between TAB and log-transformed EPUE of *S. arenicolus* ($r = 0.57$, $P < 0.08$) with 33% of the variance in log *S. arenicolus* EPUE predicted by the linear regression equation: Log *S. arenicolus* EPUE = 0.02 x TAB – 1.94. There was more variation in *S. arenicolus* EPUE at higher levels of TAB reducing the strength of a linear relationship between the two variables. There was no relationship between TAB and square root transformed EPUE of *U. stansburiana* ($r = 0.46$, $P < 0.16$) and consequently the linear regression equation, (square root transformed *U. stansburiana* EPUE = 0.01 x TAB + .0.10) could not predict EPUE of *U. stansburiana* ($R^2 = 0.21$, $P < 0.16$).

Mean Blowout Size and Lizard Abundance

I did not find significant correlations between mean blowout size and EPUE of collective lizards ($r = 0.32$, $P < 0.33$), *S. arenicolus* ($r = -0.10$, $P < 0.76$) and *U. stansburiana* ($r = 0.16$, $P < 0.65$). Similarly, the regressions did not explain the relationship between mean blowout size and EPUE of collective lizards ($R^2 = 0.10$, $P < 0.33$), *S. arenicolus* ($R^2 = 0.01$, $P < 0.76$) and *U. stansburiana* ($R^2 = 0.02$, $P < 0.65$, Figure 14).

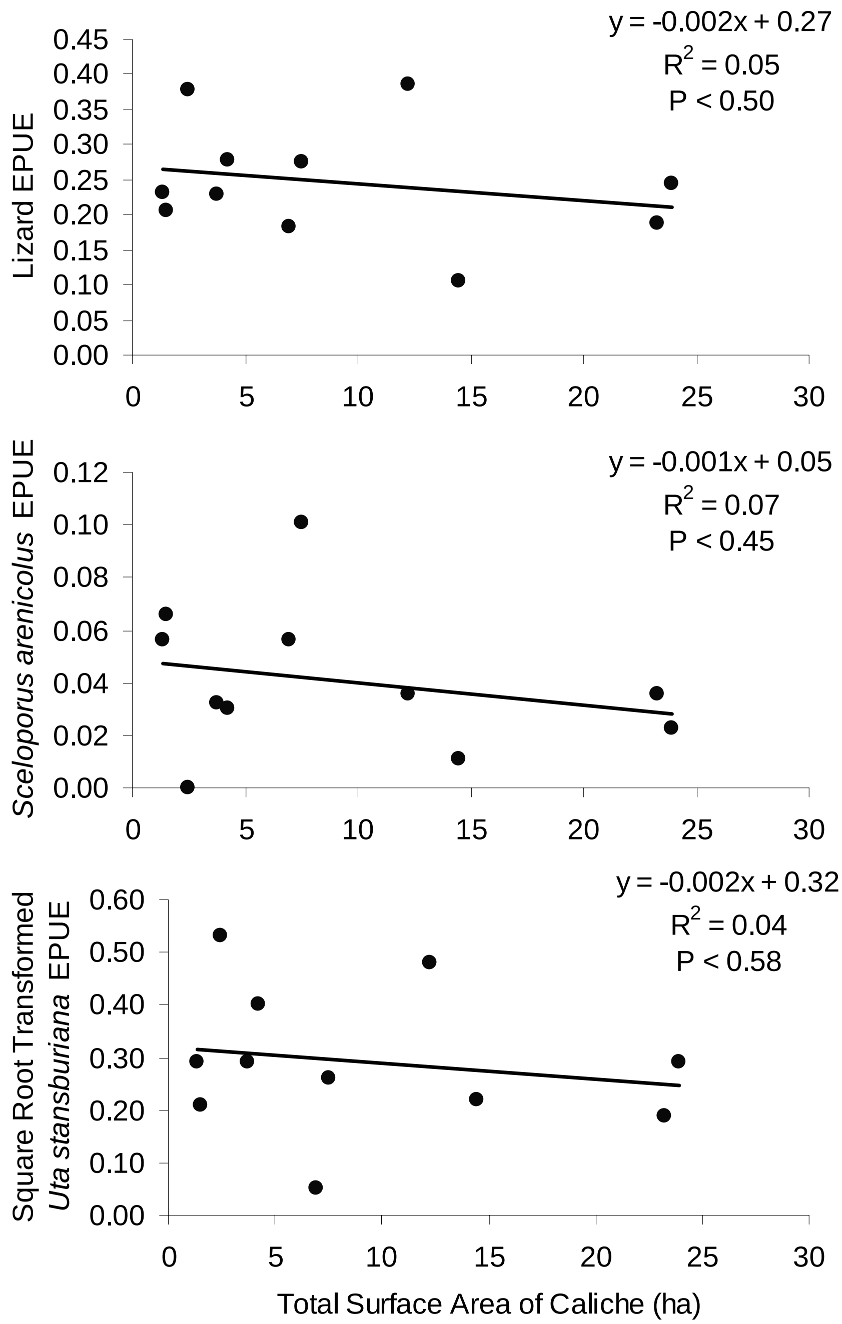

Figure 12. Relationship between total surface area of caliche and encounter per unit effort (EPUE) of lizards collectively, *Sceloporus arenicolus* and *Uta stansburiana* among 11 sites in the Mescalero Sands ecosystem, New Mexico USA.

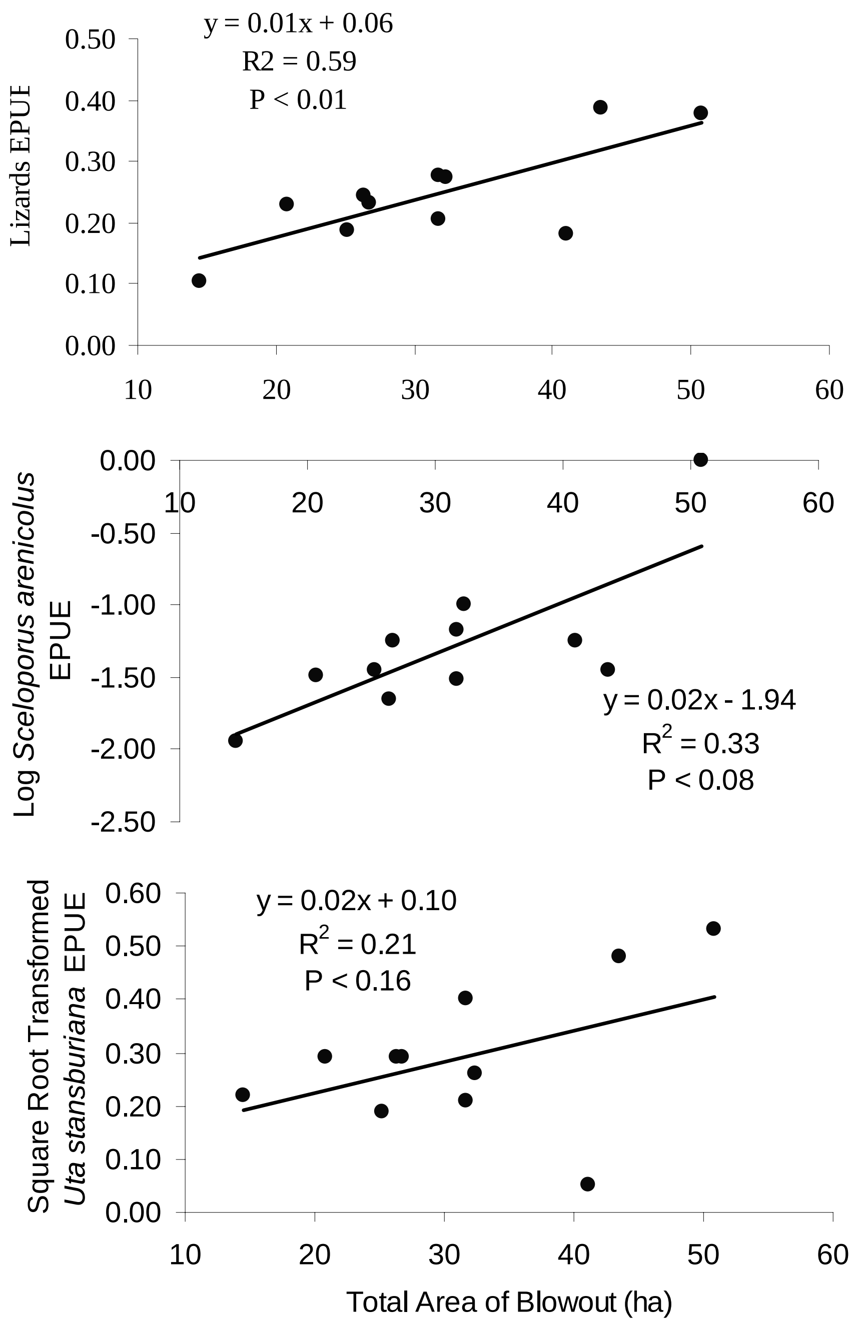

Figure 13. Relationship between total area of blowout and encounter per unit effort (EPUE) of lizards collectively, *Sceloporus arenicolus* and *Uta stansburiana* among 11 sites in the Mescalero Sands ecosystem, New Mexico USA.

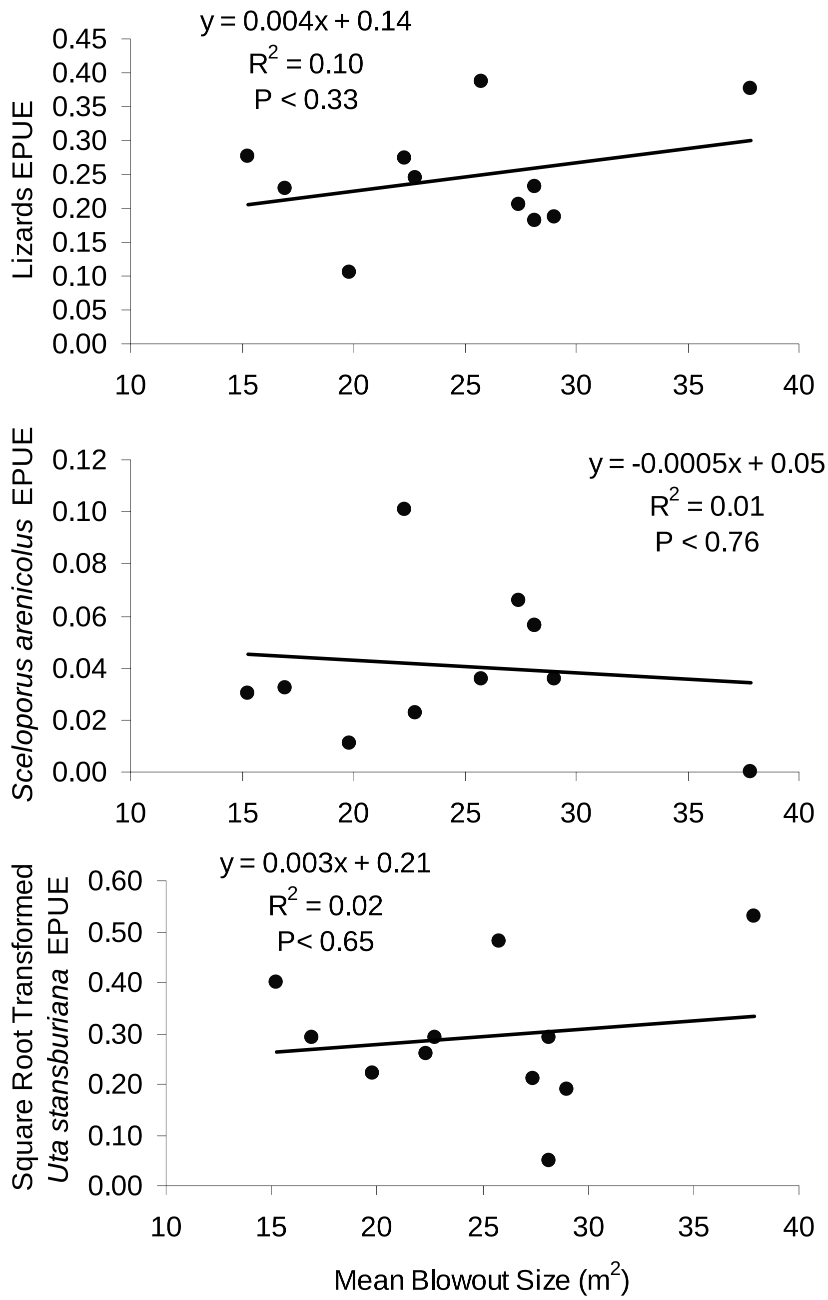

Figure 14. Relationship between mean blowout size and encounter per unit effort (EPUE) of lizards collectively, *Sceloporus arenicolus* and *Uta stansburiana* among 11 sites in the Mescalero Sands ecosystem, New Mexico USA.

Discussion

Although a large portion of the Mescalero Sands landscape has been modified by oil and gas development, I did not find evidence to support my predictions of a negative effect of oil and gas development on habitat quantity, quality or abundances of collective lizard abundances and *S. arenicolus.* All of the TSAC correlations had negative slopes, but none were significant. My hypothesis that *U. stansburiana* abundances would not be associated with oil and gas development was supported.

Lizard abundances varied significantly across the Mescalero Sands and were strongly correlated to the amount of habitat – total area of blowout (TAB). Specific lizard species differed in their respective abundance correlations to TAB. Abundances of *U. stansburiana* were not correlated to TAB, supporting my hypothesis that the habitat generalist would not have a relationship with habitat quantity. Abundances of *S. arenicolus* were not correlated to TAB and these results did not support my hypothesis that abundances of the habitat specialist would increase with increasing amounts of habitat quantity. I did find a non-linear trend between S. *arenicolus* abundances and TAB though this trend was not significant.

Lizard abundances and *S. arenicolus* abundance were not correlated to habitat quality – mean blowout size (MBS). These results, in particular for collective lizards and the habitat specialist *S. arenicolus* were surprising, given the significant positive correlation between TAB and MBS, the positive relationship between TAB and collective lizard EPUE and the nearly significant relationship between TAB and *S. arenicolus* EPUE.

Differences in habitat quality among sites made it impossible to isolate effects of TSAC from habitat quantity with these data. For example, several sites with medium amounts of TAB and MBS had high counts of *S. arenicolus*. Hobbs SE had relatively medium amounts of TSAC, TAB, and MBS yet had high abundance of *S. arenicolus*. These sites could be described as having relatively good habitat for *S. arenicolus*, despite moderate oil and gas development. These sites may have reduced my ability to detect effects of oil and gas development on *S. arenicolus* populations. Two of the three sites had relatively high amounts of TAB yet had low abundance of *S. arenicolus* (e.g. Maljamar). It is possible that *S. arenicolus* was experiencing negative effects from oil and gas development (H. L. Snell, unpublished report, University of New Mexico) that led to a decrease in abundance at Maljamar, but this remains inconclusive because of the interaction between available habitat at sites that also contain oil and gas development.

Without long-term data on these populations and habitats, I cannot infer that populations are persisting or declining due to the effects of TSAC. It is likely that the level of oil and gas development *S. arenicolus* can tolerate is associated with the quantity of shinnery oak, sand dune habitat and the connectivity sand dune complexes to other dune- shinnery oak matrices (Stacey and Taper 1992). If this is true, then S. *arenicolus* may be more susceptible to fragmentation effects than habitat loss alone (Bender et al. 1998).

Interspecific interactions between *U. stansburiana* and *S. arenicolus* may further confound effects of oil and gas development on these species. *Uta stansburiana* abundances were lowest where *S. arenicolus* abundance was highest. The factors associated with interspecific interactions among the lizard species in the community could not be separated from the factors associated with total area of blowout, mean blowout size and total surface area of caliche. Oil and gas development may not affect *U. stansburiana* directly, but indirectly via changes in lizard community structure associated with *S. arenicolus* populations (Andrén 1992).

This is one of the first papers to investigate effects of oil and gas development on a dune-dwelling lizard assemblage. Though I did not find significant effects of oil and gas development on lizard abundance, I cannot conclude that oil and gas development does not have an impact on dune-dwelling lizards. My results may have been confounded by my study design. For example, my indicators of oil and gas development, habitat quantity and quality were measured at a scale that may not have been the appropriate spatial scale to detect an ecological effect on lizard abundances (Fischer et al. 2004). Additionally, my analyses did not control for lizard numbers at individual sites before caliche roads and well pads were constructed, which may have influenced lizard numbers during this study. I also could not control for the influence of oil and gas development on the habitat and how that in turn affects lizard abundances. The interaction between habitat quality and quantity and influence of TSAC on each should remain a topic of investigation.

In some cases, it is not the amount of habitat that affects lizard populations, but rather changes in vegetation types that degrade habitat quality (Jellinek et al. 2004). *Sceloporus arenicolus* is known to be sensitive to habitat alteration. *Sceloporus arenicolus* numbers decreased by 78% at sites where shinnery oak was removed by herbicide spraying compared to control sites (H. L. Snell, unpublished report, University of New Mexico). Changes in shinnery oak and other vegetation types could serve as an additional habitat quality parameter for future studies on the impacts of oil and gas development on the Mescalero Sands ecosystem.

CHAPTER IV

CONCLUSIONS AND IMPLICATIONS

Population quantification and monitoring is a fundamental aspect in applied ecological research and conservation and sampling methods must be cost effective, allow for adequate spatial coverage, and produce reliable estimates. There is a tradeoff between cost of population sampling methods and reliability of derived estimates (Pollock et al 2002). For example, methods that are easily implemented at large spatial scales such as indices produce less reliable population estimates as compared to rigorous methods conducted at small spatial scales (Anderson 2001). More so, there is no single population method that is appropriate for all species in all habitat types (Pollock et al. 2002, Doan 2003). Conservation studies investigating the effects of habitat loss on flora and fauna require data collected at broad spatial scales and often the easily implemented designs are chosen over costly designs (Engeman 2003).

Distance sampling has been described as method that can be easily implemented across a large landscape and produces reliable results (Buckland et al. 1993). Distance sampling was neither accurate nor effective in estimating population densities of lizards in the Mescalero Sands due to detection biases. Even when coupled with double-observer methods distance density estimates were largely negatively biased compared to density estimates from total removal plots. The total removal plots contained both active and inactive lizards that were detected during the removal of vegetation and raking of sand. The disparity between the density estimates from distance sampling and total removal plots suggests the proportion of inactive lizards missed and beneath the surface, albeit unknown, was much larger than the proportion of missed lizards above the ground and accounted for with double-server surveys.

Detectability on the transect line is a key issue in distance sampling, and if it is not adequately estimated, resulting density estimates will be biased. Availability of individuals to be detected is contingent upon activity patterns that vary unpredictably both temporally and amongst individuals (Whitford and Creusere 1977, Dunham 1981 and Dorcas and Peterson 1998). This unpredictable, yet large variation in activity makes it extremely challenging to develop a correction factor for missed individuals. The population estimates reported in this study can thus serve only as indices of abundance rather than precise population density estimates. My research suggests distance sampling is not appropriate for population density estimation of dune dwelling lizards.

Total removal plots produce reliable population densities but are confounded by the cost of construction and spatial coverage. Additionally, the removal of vegetation may not be permissible or feasible in other places. Grant and Doherty (2007) suggest a modification to traditional mark-recapture sampling that addresses the issue of assuming closed population. They placed several pitfall traps within and enclosure and in essence conducted a hybrid of total removal plots and mark-recapture sampling. This technique may be the most appropriate method to quantify population densities of dune-dwelling lizards in the Mescalero Sands.

Based on the results of Chapter II in which I showed distance population density estimates were biased, I used an abundance index in the form of encounter per unit effort (EPUE) to evaluate the effects of oil and gas development, habitat quality, and habitat quality. Despite the significant amount of oil and gas development throughout the Mescalero Sands landscape, I did not find an effect of oil and gas development on abundances of dune-dwelling lizards, habitat quantity or habitat quality. While there may, in fact, be no effect of oil and gas development on lizard abundances in this landscape, it is also possible that the design of this study was simply unable to detect the impacts of oil and gas development. It can be challenging to detect anthropogenic effects with short-term studies. The alteration of the landscape via oil and gas development may have more complex effects on the flora and fauna of the Mescalero Sands ecosystem and simple correlations and linear regressions may not adequately detect these effects.

Many studies investigating anthropogenic effects on herpetofauna have failed to document deleterious effects due to the dearth of long-term population data (Gibbons et al. 2000, Storfer and Collins 2003, Gardner et al. 2007). There may be effects of oil and gas development on the habitat and species occupying the Mescalero Sands landscape, but these effects may not have been detected due to lack of temporal data on population densities of lizards and on development (e.g. when oil well pads and roads were constructed). It may take several years for the effects of oil and gas development to manifest itself on lizard populations, (e.g. ‘extinction debt’ Tilman et al. 1994). Below I discuss some of the potential impacts of oil and gas development on the habitat and lizard species occupying the Mescalero Sands ecosystem.

Potential Impacts of Oil and Gas Development

Changes in Abiotic Features and Associated Implications

Oil and gas development can cause changes in the geomorphology of the dunes directly via sand erosion (Matherne 2006) or indirectly via removal of shinnery oak. Removal of shinnery oak leads to an increase in grasses that in turn increase wind erosion of dunes (Peterson and Boyd 2000). Given that dune-dwelling lizard densities were strongly correlated to blowouts, temporal data on total area of blowouts and changes in vegetation types are important parameters to include in future investigations of the impacts of oil and gas development on dune-dwelling lizards.

Fragmentation and Spatial Arrangement of Patches

The Mescalero Sands ecosystem is fragmented naturally and by human practices. Though I did not study habitat fragmentation per se, fragmentation may have a greater impact on habitat specialists than habitat loss (Bender et al. 1998). Increased fragmentation is likely to have greater impacts on habitat specialist that have limited dispersal capabilities and cannot utilize habitat types in between fragments (Hokit et al. 1999). Spatial arrangement of remaining habitat patches can confound the effects of habitat loss on species (With and King 2001). The connectivity of shinnery oak sand dune habitat across the landscape likely influences the persistence of dune-dwelling lizard populations (Pulliam 1988). For example, Connor Well is a site with historical *S. arenicolus* populations, has low amounts of oil and gas and high amounts of habitat quantity. This site is relatively isolated from other existing *S. arenicolus* populations, and it is possible the extirpated population resulted from low recruitment of immigrants from distant source populations. Conversely, *S. arenicolus* subpopulations in the southern region of the species distribution, where there is more oil and gas development, are in closer proximity to each other and may be sustained via immigration from source subpopulations.

Edge Effects

The Mescalero Sands ecosystem contains several species of lizards that utilize different microhabitat types that occur in landscape. Some species, like *U. stansburiana*, occur throughout the various habitat types and are considered habitat generalists. Other species, such as *S. arenicolus*, *S. undulatus* and *Holbrookia maculata,* specialize in blowouts, shinnery oak flats or inter-dune flats with little vegetation respectively. The amount of habitat loss and resulting size of remaining patches is likely to affect theses species differently because of their respective levels of habitat specialization (Bender et al. 1998, Fagan et al. 1999).

Community Dynamics

Different lizard species are likely to respond to habitat changes differently (Rizkalla and Swihart 2006), and this can lead to changes in the proportion of species within the community (Busack and Bury 1974, Attum et al. 2006). Removal of shinnery oak leads to reductions in areas suitable for foraging, predator avoidance and thermoregulatory processes (Sanchez and Parmenter 2002, Attum and Eason 2006). This may result in increased intra- and interspecific competition and greater susceptibility to predation. Changes in the proportion of species within the community also may affect other faunal components that are trophically linked to the lizards as prey or predators (Hawlena and Bouskila 2006).

Limitations of the Study and Suggestions for Future Studies

The number of sites used in this study may have been insufficient to detect meaningful patterns of lizard abundances in association with oil and gas development across the landscape. More than twenty sites known to have historical *S. arenicolus* populations were visited. I did not detect *S. arenicolus* populations at one third of the sites visited. The absences of *S. arenicolus* from historical localities suggest investigations of changes in *S. arencoulus* distribution are needed. Sites with baseline population estimates can be used in future population monitoring programs that may in turn be able to address the impacts of oil and gas development on lizard populations.

My indicators of habitat quantity and quality may not have been the factor that determines the presence and abundance of *S. arenicolus*. Other abiotic factors, such as sand grain size (L. A. Fitzgerald, unpublished report, Texas A&M University) or blowout depth, may have stronger associations with *S. arenicolus* abundances than mean blowout size or total area of blowouts. Alternative landscape parameters, such as vegetation changes and habitat connectivity, should be included in future models investigating the presence and abundance of *S. arenicolus*.

Complex Interactions

The interactions between oil and gas development, habitat quantity, and habitat quality were complex and could not be controlled in my study. It is possible that the life history attributes of the lizards (e. g. dispersal abilities and fecundity) allow populations to persist in areas of moderate oil and gas development (With and King 1999). Additionally, interspecific interactions between *U. stansburiana* and *S. arenicolus* may further confound effects of oil and gas development on these species. Multiple modeling approaches (With and King 1999) may be able to address the complex interactions between species, their respective life-history attributes and changes in landscapes.

Conclusion

Despite several decades of research, conservation biologists are still challenged with the issues of anthropogenic impacts and the threat of species extinction. Some species naturally occur in low numbers, are cryptic, and fossorial thus complicating population enumeration and conservation. The desert tortoise (*Gopherus agassizii*) was federally listed as ‘threatened’ in 1990 (U. S. Fish and Wildlife Service1990). This species spends much of its time in burrows and juveniles are especially difficult to detect. Several studies have investigated ways to enumerate and monitor *G. agassizii* populations (Anderson et al. 2001, Freilich et al. 2005, Nussear and Tracy 2007) and yet there is still uncertainty about the population status of this species.

Funding limitations can impose additional challenges when designing studies faced with low population numbers and restricted habitats. Impacts of habitat loss and fragmentation must be investigated at multiple scales, including the scale of habitat destruction and the scale at which the species of interest interact with the changed landscape. A combination or hybrid of existing methods (With and King 1999, Ryan et al. 2001, Pollock et al. 2002, Grant and Doherty 2007) may be the best approach to addressing factors the commonly confounding studies. In the Mescalero Sands ecosystem, data are needed in areas before and after oil and gas development to better determine if oil and gas development has a negative impact on *S. arenicolus* populations.

LITERATURE CITED

Akin, J. 1998. Fourier series estimation of ground skink populations. Copeia 1998: 519–522.

Alberts, A. C. 1993. Relationship of space use to population density in an herbivorous lizard. Herpetologica 49: 469–479.

Anderson, D. R. 2001. The need to get the basics right in wildlife field studies. Wildlife Society Bulletin 29: 1294–1297.

Anderson, D. R., K. P. Burnham, B. C. Lubow, L. Thomas, P. S. Corn, P. A. Medica, and R. W. Marlow. 2001. Field trials of line transect methods applied to estimation of desert tortoise abundance. Journal of Wildlife Management 68: 583–597.

Andrèn, H. 1992. Corvid density and nest predation in relation to forest fragmentation - a landscape perspective. Ecology 73: 794–804.

Andrèn, H. 1994. Can one use nested subset pattern to reject the random sample hypothesis - examples from boreal bird communities. Oikos 70: 489–491.

Attum, O. A. and P. K. Eason. 2006. Effects of vegetation loss on a sand dune lizard. Journal of Wildlife Management 70: 27–30.

Attum, O. A., P. K. Eason, G. Cobbs and S. M. B. E. Din. 2006. Response of a desert lizard community to habitat degradation: do ideas about habitat generalist/specialists hold? Biological Conservation 133: 52–62.

Bächler, E. and F. Leichti. 2007. On the importance of *g*(0) for estimating bird population densities with standard distance-sampling: implications from a telemetry study and a literature review. Ibis 149: 693 –700.

Ballinger, R. E. and J. D. Congdon. 1981. Population ecology and life history strategy of a montane lizard (*Sceloporus scalaris*) in southeastern Arizona. Journal of Natural History 15: 213–222.

Ballinger, R. E., and K. S. Watts. 1995. Path to extinction -impact of vegetational change on lizard populations on Arapaho Prairie in the Nebraska Sandhills. American Midland Naturalist 134: 413–417.

Barrows, C. M. 2006. Population dynamics of a threatened sand dune lizard. Southwestern Naturalist 51: 514–532.

Bender, D. J., T. A. Contreras, and L. Fahrig. 1998. Habitat loss and population decline: A meta-analysis of the patch size effect. Ecology 79: 517–533.

Borchers, D. L., S. T. Buckland, P. W. Goedhart, E. D. Clarke, S. L. Hedley. 1998. Horvitz-Thompson estimators for double-platform line transect surveys. Biometrics 54: 1221–1237.

Bradshaw, C. J. A., S. Boutin, and D. M. Herbert, 1997. Effects of petroleum exploration on woodland caribou in northeastern Alberta. Journal of Wildlife Management 61: 127–1133.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling. Oxford University Press, Oxford, United Kingdom.

Buckland, S. T., D. R. Anderson, K. P. Burnham, and J. L. Laake. 2004. *Advance Distance Sampling*. Oxford University Press, New York, New York, USA.

Buckland, S. T., K. P. Burnham, D. R. Anderson, and L. Thomas. 1993. Distance sampling: estimation of abundance of biological populations. Chapman and Hall, London, United Kingdom.

Bull, C. M. 1987. A population study of the viviparous Australian lizard *Trachydosaurus rugosus* (Scincidae). Copeia 1987: 749–757.

Burnham, D. R., and D. R. Anderson. 1984. The need for distance data in transect counts. Journal of Wildlife Management 48: 1248–1254.

Busack, S. D. and R. B. Bury. 1974. Some effects of off-road vehicles and sheep grazing on lizard populations in the Mojave Desert. Biological Conservation 6: 179–183.

Cassey, P. and G. T. Usser. 1999. Estimating abundance of tuatara. Biological Conservation 88: 361–399.

Collins, J. P., and A. Storfer. 2003. Global amphibian declines: sorting the hypotheses. Diversity and Distributions 9: 89–98.

Corn, J. L. and M. J. Conroy. 1998. Estimation of density of mongooses with capture-recapture and distance sampling. Journal of Mammology 79: 1009–1015.

Crooks, K. R. and M. E. Soulè. 1999. Mesopredator release and avifaunal extinction in a fragmented ecosystem. Nature 400: 563–566.

Dickinson, H. C., and J. E. Fa. 2000. Abundance, demographics and body condition of a translocated population of St. Lucia whiptail lizards (*Cnemidophorus vanzoi*). Journal of Zoology 251: 187–197.

Diefenbach, D. R., D. W., Brauning, and J. A. Mattices. 2003. Variability in grassland bird counts related to observer differences and species detection rates. The Auk 120: 1168–1179.

Doan, T. 2003. Which methods are most effective for surveying rain forest herpetofauna. Journal of Herpetology 37: 72–81.

Dodd, C. K. 1990. Line transect estimation of red hills salamander burrow density using a Fourier series. Copeia 1990: 555–557.

Donnely, M. A., Guyer, C. and Hayek, L .C. 1994. Estimating populations. Pages 183–205 *in* R. W. Heyer, M. A. Donnelly, R. W. McDiarmid, L. C. Hayek and M. S. Foster editors. Measuring and Monitoring Biological Diversity: Standard Methods for Amphibians. Smithsonian Institution Press, Washington, USA.

Dorcas, M. E. and C. R. Peterson. 1998. Daily body temperature variation in free-ranging rubber boas. Herpetologica 54: 88–103.

Dunham, A. E. 1981. Populations in a fluctuating environment: the comparative population ecology of the iguanid lizards *Sceloporus merriami* and *Urosaurus ornatus*. Miscellaneous Publications, Museum of Zoology, University of Michigan No.158.

Engeman, R. M. 2003. More on the need to get the basics right: population indices. Wildlife Society Bulletin 31: 286–287.

Fagan, W. F., R. S. Cantrell, C. Cosner. 1999. How habitat edges change species interactions. The American Naturalist 153: 165–182.

Fahrig, L. 1997. Relative effects of habitat loss and fragmentation on population extinction. Journal of Wildlife Management 61: 603–610.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology Evolution and Systematics 34: 487–515.

Fiori, S. M. and S. M. Zalba. 2003. Potential impacts of Potential impacts of petroleum exploration and exploitation on biodiversity in a Patagonian Nature Reserve, Argentina. Biodiversity and Conservation 12: 1261–1270.

Fischer, J., D. B. Lindenmayer, and A. Cowling. 2004. The challenge of managing multiple species at multiple scales: reptiles in an Australian grazing landscape. Journal of Applied Ecology 41: 32–44.

Freilich, J. E., R. J. Camp, J. J. Duda, and A. E. Karl. 2005. Problems with sampling desert tortoise: a simulation analysis based on field data. Journal of Wildlife Management 69: 45–56.

Funk, C., D. Almedia-Reinoso, F. O. Nogales-Sornosa, and M. R. Bustamante. 2003. Monitoring population trends of *Eleutherodactylus* frogs. Journal of Herpetology 37: 245–256.

Gardner, T. A., J. Barlow, and C. A. Peres. 2007. Paradox, presumption and pitfalls conservation biology: the importance of habitat change for amphibians and reptiles. Biological Conservation 138: 166–179.

Germano, J. M., J. M. Sander, R. W. Henderson, and R. Powell. 2003. Herpetofaunal communities in Grenada: a comparison of altered sites, with an annotated checklist of Grenadian amphibians and reptiles. Caribbean Journal of Science 39: 68–76.

Gibbons, J. W., D. E. Scott, T. J. Ryan, K. A. Buhlmann, T. D. Tuberville, B. S. Metts, J. L. Greene, T. Mills, Y. Leiden, S. Poppy, and C. T. Winne. 2000. The global decline of reptiles, Deja Vu amphibians. Bioscience 50: 653–666.

Goldberg, C. S., and C. R. Schwalbe. 2004. Consideration for monitoring a rare anuran (*Eleuthrodactylus agusti*). Southwestern Naturalist 49: 442–448.

Graham, A., and R. Bell. 1989. Investigating observer bias in aerial survey by simultaneous double-counts. Journal of Wildlife Management 53: 1009–1016.

Grant, B. W. 1990. Tradeoffs in activity time and physiological performance for thermoregulating desert lizards, *Sceloporus merriami*. Ecology 71: 2323–2333.

Grant, T. J., and P. F. Doherty, Jr. 2007. Monitoring of the flat-tailed horned lizard with methods incorporating detection probability. Journal of Wildlife Management 71: 1050–1056.

Grumbine, R. E. 1994. What is ecosystem management? Conservation Biology 8: 27–38.

Hager, S. B. 2001 Microhabitat use and activity patterns of *Holbrookia maculata* and *Sceloporus undulatus* at White Sands National Monument, New Mexico. Journal of Herpetology 35: 326-330.

Hawlena, D., and A. Bouskila. 2006. Land management practices for combating desertification cause species replacement of desert lizards. Journal of Applied Ecology 43: 701–709.

Hayer, R. W., M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster. 1994. Measuring and monitoring biologicaldiversity: standard methods for amphibians. Smithsonian Institution Press, Washington USA.

Hecnar, S. J., and R. T. M'Closkey. 1998. Species richness patterns of amphibians in southwestern Ontario ponds. Journal of Biogeography 25: 763–772.

Hokit, D. G., B. M. Smith, and L. C. Branch. 1999. Effects of landscape structure in Florida scrub: A population perspective. Ecological Applications 9: 124–134.

Hounsome, T. D., R. P. Young, J. Davison, R. W. Yarnell, I. D. Trewby, B. T. Garnett, R. J. Delahay, and G. J. Wilson. 2005. An evaluation of distance sampling to estimate badger (*Meles meles*) abundance. Journal of Zoology London 266: 81–87.

Huggins, R. M. 1989. On the statistical analysis of capture experiments. Biometrika 76: 133–140.

Huggins, R. M. 1991. Some practical aspects of a conditional likelihood approach to capture experiments. Biometrics 47: 725–732.

Iverson, J. B. 1978. The impact of feral cats and dogs on populations of the West Indian rock iguana (*Cyclura carinata*). Biological Conservation 14: 63–73.

Jellinek, S., D. A. Driscoll, and J. B. Kirkpatrick. 2004. Environmental and vegetation variables have a greater influence than habitat fragmentation in structuring lizard communities in remnant urban bushland. Austral Ecology 29: 294–304.

Jenkins, R. K. B., Brady, L. D., Huston, K., Kauffmann, J. L. D., Rabearivony, J. Raveloson, G., and J. M Rowcliffe. 1999. The population status of chameleons within Ranomafana National Park, Madagascar, and recommendations for future monitoring. Oryx 33: 38–46.

Kissling, M. L., and E. O. Garton. 2006. Estimating detection probability and density from point-count surveys: a combination of distance and double-server sampling. The Auk 123: 735–752.

Laake, J. L., J. Calambokidis, S. D. Osmek, and D. J. Rugh. 1997. Probability of detecting harbor porpoise from aerial surveys: estimating g(0). Journal of Wildlife Management 6: 63–75.

Laake, J. L., and D. L. Borchers. 2004. Methods for incomplete detection at distance zero Pages 108–189 *in* S. T. Buckland, D. R. Anderson, K. P. Burnham, and J. L. Laake, editors. Advanced Distance Sampling. Oxford University Press, New York, New York USA.

Lacki, M. J., J. W. Hummer, and J. L. Fitzgerald. 1994. Application of line transects for estimating population density of the endangered copperbelly water snake in Southern Indiana. Journal of Herpetology 28: 241–245.

Laurance, W. F. 2000. Do edge effects occur over large spatial scales? Trends in Ecology & Evolution 15: 134–135.

Lyon, A. G. and S. H. Anderson. 2003. Potential gas development impacts on sage grouse nest initiation and movement. Wildlife Society Bulletin 31: 486–491.

MacNally, R., A. F. Bennett, G. W. Brown, L. F. Lumsden, A. Yen, S. Hinkley, P. Lillywhite, and D. A. Ward. 2002. How well do ecosystem-based planning units represent different components of biodiversity? Ecological Applications 12: 900–912.

Marsh, H. and D. F. Sinclair 1989. Correcting for visibility bias in strip transect aerial surveys of aquatic fauna. Journal of Wildlife Management 53: 1017–1024.

Matherne, A. M. 2006. Effects of roads and well pads on upland erosion in the Largo Canyon Watershed, New Mexico, 2001–02. U.S. Geological Survey. Scientific Investigations Report 2006–5039, U.S. Department of Interior, U.S. Geological Survey.

Mazerolle, M. J., L. L. Bailey, W. L. Kendall, J. A. Royle, S. J. Converse, and J. D. Nichols 2007. Making great leaps forward: accounting for detectability in herpetological field studies. Journal of Herpetology 41: 672–689.

McCoy, E. D., and H. R. Mushinsky. 1999. Habitat fragmentation and the abundances of vertebrates in the Florida scrub. Ecology 80: 2526–2538.

Meffe, G. K., C. R. Carroll, and Contributors editors. 1997. The species in conservation. Pages 57–86 *in*. Principles of conservation biology. Second edition. Sinauer Associates, Inc., Sunderland, Massachusetts, USA.

National Climatic Data Center. Monthly temperature and precipitation data for New Mexico <http://www.ncdc.noaa.gov/oa/climate/stationlocator.html>. Accessed 21 March 2007.

New Mexico Game and Fish Department. 2006. Threatened and endangered species of New Mexico: draft biennial review and recommendations. Authority: Wildlife Conservation Act (NMSA 17-2-37+B1 through 17-2-46, 1978).

New Mexico State Land Office (NMSLO). 2006. GIS point layer of well pad points for all wells in NMSLO database. New Mexico State Land Office, Santa Fe, NM.

Nichols, J. D., J. E. Hines, J. R. Sauer, F. W. Fallon, J. E. Fallon, and P. J. Heglund. 2000. A double-observer approach for estimating detection probability and abundance from point counts. The Auk 117: 393–408.

Nussear, K. E. and C. R. Tracy. 2007. Can modeling improve estimation of desert tortoise population densities? Ecological Applications 17: 579–586.

Otto, M.C., and K. H. Pollock. 1990. Size bias in line transect sampling: a field test. Biometrics. 46: 239–245.

Pechmann, J. H. K., D. E. Scott, R. D. Semlitsch, J. P. Caldwell, L. J. Vitt, and J. W. Gibbons. 1991. Declining amphibian populations - the problem of separating human impacts from natural fluctuations. Science 253: 892–895.

Pechmann, J. H. K., and H. M. Wilbur. 1994. Putting declining amphibian populations in perspective - natural fluctuations and human impacts. Herpetologica 50: 65–84.

Peterson, R. S., and C. S. Boyd. 2000. Ecology and management of sand shinnery communities: a literature review. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado USA.

Pollock, K. H., and W. L. Kendall, 1987. Visibility bias in aerial surveys: A review of estimation procedures. Journal of Wildlife Management 51: 502–510.

Pollock, K. H., J. D. Nichols, T. R. Simons, G. L. Fansworth, L. L. Bailey and J. R. Sauer. 2002. Large scale wildlife monitoring studies: statistical methods for design and analysis. Environmetrics 13:105–119.

Polus, E., S. Vandewoestijne, J. Choutt, and M. Baguette. 2007. Tracking the effects of one century of habitat loss and fragmentation on calcareous grassland butterfly communities. Biodiversity Conservation 16: 3423–3436.

Pulliam, H. R. 1988. Sources, sinks, and population regulation. The American Naturalist 132: 652–661.

Rabe, M. J., S. S. deVos, and J. C. Rosenstock, Jr. 2002. Review of big-game survey methods used by wildlife agencies of the western United States. Wildlife Society Bulletin 30: 46–53.

Radder, R. S., S. K. Saidapur, and B. A. Shanbhag. 2005. Population density, microhabitat use and activity pattern of the Indian rock lizard, *Psammophilus dorsalis* (Agamidae). Current Science 89: 560–566.

Reisinger, W. J., D. M. Stuart-fox, and B. F. N. Erasmus. 2006. Habitat associations and conservation status of an endemic forest dwarf chameleon (*Bradypodion* sp.) from South Africa. Oryx 40: 183–188.

Research Unit for Wildlife Population Assessment. 2007 Jul 31. Who uses distance page. <http://www.ruwpa.st-and.ac.uk/distance/distanceusers.html>. Accessed Feb 2008.

Rizkalla, C. E., and R. K. Swihart. 2006. Community structure and differential responses of aquatic turtles to agriculturally induced habitat fragmentation. Landscape Ecology 21: 1361–1375.

Rodda, G. H., and E. W. Campbell. 2002. Distance sampling of forest snakes and lizards. Herpetological Review 33: 271–274.

Rodda, G. H., E. W. Campbell, and T. H. Fritts. 2001. A high validity census technique for herpetofaunal assemblages. Herpetological Review 32: 24–30.

Ryall, K. L., and L. Fahrig. 2005. Habitat loss decreases predator–prey ratios in a pine-bark beetle system. Oikos 110: 265–270.

Ryan, T. J., T. Philipi, Y. A. Leiden, M. E. Dorcas, T. B. Wigley, and J. W. Gibbons. 2001. Monitoring herpetofauna in a managed forest landscape: effects of habitat types and census technique. Forest Ecology and Management 167: 83–90.

Sanchez, B.C., and R. R. Parmenter. 2002. Patterns of shrub dwelling arthropods diversity across a desert shrubland-grassland ecotone: a test of island biogeographic theory. Journal of Arid Environments 50: 247–265.

Shine R. 1991. Australian Snakes: A Natural History. Cornell University Press, Ithaca, New York, USA.

Skuag, H. J., and T. Schweder. 1999. Hazard models for line transect surveys with independent observers. Biometrics 55: 29–36.

Stacey, P. B., and M. Taper. 1992. Environmental variation and the persistence of populations. Ecological Applications 2: 18–29.

SPSS. 2001. SPSS Advanced Models. Version 11. SPSS, Inc., Chicago, IL, USA.

Swann, D. E., R. C. Averill-Murray, and C. R. Schwable. 2002. Distance sampling for the Sonoran desert tortoise. Journal of WildlifeManagement 66: 969–975.

Thomas, L., J. L. Laake, S. Strindberg, F. F. C. Marques, S. T. Buckland, D. L. Borchers, D. R. Anderson, K. P. Burnham, S. L. Hedley, J. H. Pollard, J. R. B. Bishop, and T. A. Marques. 2006. Distance 5.0. Release 1. Research Unit for Wildlife Population Assessment, University of St. Andrews, United Kingdom. Available at: http://www.ruwpa.st-and.ac.uk/distance/.

Tilman, D., R. M. May, C. L. Lehman, and M. A. Nowak. 1994. Habitat destruction and the extinction debt. Nature 371: 65–66.

Toft, C., and P. J. Shea. 1983. Detecting community-wide patterns: estimating power strengthens statistical inference. The American Naturalist 122: 618–625.

Trail, P. W. 2006. Avian mortality at oil pits in the United States: a review of the problem and efforts for its solution. Environmental Management 38: 532–544.

Turner, F. B. 1977. The dynamics of populations of squamates, crocodilians, and rhynchocephalians. Pages 157–262 *in* C. Grans and D. W. Tinkle editors Biology of the Reptilia. Academic Press, New York, USA.

U. S. Fish and Wildlife Service. 1990. Endangered and Threatened Wildlife and Plants; Determination of Threatened Status for the Mojave Population of the Desert Tortoise. Federal Register 55: 12178–12191.

U. S. Fish and Wildlife Service. 2004. Endangered and Threatened Wildlife and Plants; Review of Species That are Candidates or Proposed for Listing as Endangered or Threatened; Annual Notice of Findings on Resubmitted Petitions; Annual Description of Progress on Listing Actions; Notice of Review; Proposed Rule. Federal Register 69: 24876– 24904.

Vega, L. E., P. J. Bellagamba, and L. A. Fitzgerald. 2000. Long-term effects of anthropogenic habitat disturbance on a lizard assemblage inhabiting coastal dunes in Argentina. Canadian Journal of Zoology-Revue Canadienne De Zoologie 78: 1653–1660.

Vogt, R. C., and R. L. Hine. 1982. Evaluation of techniques for asessment of amphibian and reptile populations in Wisconsin. *in* Scott, N.J., Jr., editor. Herpetological communities: Wildlife Research Report 13: 201–217. U. S. Fish and Wildlife Service, Washington, DC USA.

Welsh, H. H. Jr., J. R. Hodgson, and A. J. Lind. 2005. Ecogeography of the herpetofauna of a northern California watershed: linking species patterns to landscape processes. Ecography 28: 52–526.

White, G. C. 1990–2000. MARK 4.3. Colorado State University,Ft. Collins, CO USA.

Whitford, W. G., and F. M. Creusere. 1977. Seasonal and yearly fluctuations in Chihuahuan desert lizard communities. Herpetological 33: 54–65.

With, K. A., and A. W. King. 1999. Extinction thresholds for species in fractal landscapes. Conservation Biology 13: 314–326.

With, K. A., and A. W. King. 2001. Analysis of landscape sources and sinks: the effect of spatial pattern on avian demography. Biological Conservation 100: 75–88.

VITA

NICOLE LIMUNGA SMOLENSKY

ADDRESS

Department of Wildlife and Fisheries Sciences, Texas A&M University
210 Nagle Hall College Station, TX 77843-2258
nsmo@tamu.edu

EDUCATION

Master of Science, Wildlife and Fisheries Sciences, Texas A&M University, 2008

Bachelor of Science, Biology, University of California at Santa Cruz, 2002

WORK EXPERIENCE

Graduate Teaching Assistant, Natural History of the Vertebrates, and Herpetology, Wildlife and Fisheries Sciences Department, Texas A&M University, 2006 - present

Graduate Research Assistant, Texas A&M University, 2004 - present

Lab Technician, Calstate Los Angeles, Reproductive ecology of the gastropod species *Alderia modesta*, 2003 – 2004

Field Technician, University of Cantebury, New Zealand, Conservation biology of New Zealand Fur Seal (*Arctocephalus fosteri)*, 2002 – 2003

PUBLICATIONS AND REPORTS

Laurencio, D., Laurencio, L, **Smolensky, N.** 2007. Geographic distribution. *Chelydra serpentina*. Herpetological Review 38:327.

Smolensky, N. and Fitzgerald, L. 2006. Population density estimates of the sand dune lizard (*Sceloporus arenicolus*) in the Mescalero Sands ecosystem. Interim technical report submitted to New Mexico Department of Game and Fish (December 2006).

Smolensky, N., Hill, M. and Fitzgerald, L. 2005. Population density estimates of the sand dune lizard (*Sceloporus arenicolus*) and movements of gravid females in the Mescalero Sands ecosystem. Interim technical report submitted to New Mexico Department of Game and Fish (December 2005).