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COLLEGE OF AGRICULTURE
AND LIFE SCIENCES

DEPARTMENT OF WILDLIFE AND FISHERIES SCIENCES
210 NAGLE HALL

Texas Cooperative Wildlife Collection

October 5, 2011

Dear Jeanette Martinez,

This letter and attached pdf file summarize our ongoing research on the dunes sagebrush lizard carried out with BLM funding during this reporting period. This report corresponds to Year 2 of our current 4-year research project, "*Effects of Management Practices for Oil and Gas Development on the Mescalero Dune Landscape and Populations of the endemic Dunes Sagebrush-lizard, Sceloporus arenicolus*." Our research team consists of Dr. Lee Fitzgerald, Dan Leavitt, Graduate Student, Dr. Wade Ryberg, Postdoctoral Research Associate, Laura Laurencio, Research Associate, Nicole Smolensky, Graduate Student, and Dr. Lauren Chan, Collaborator at Duke University.

We made significant progress on all aspects of research during the 2011 field season. In addition to executing the best management practices research project, we were able to engage in numerous ancillary research activities that add tremendous value to the overall research effort.

The report is organized by Sections, each pertaining to these topics:

- Core Project: *Effects of Management Practices* (Leavitt et al.)*;*
- Population dynamics of *S. arenicolus*: a multi-site population viability approach (Ryberg and Fitzgerald);
- Study of effects of trenching on lizards and other wildlife (Leavitt et al.);
- Update on geographic distribution of *S. arenicolus* (Laurencio and Fitzgerald);
- Advances on conservation and landscape genetics of *S. arenicolus* (Chan et al.);
- Using this research to inform and design research on *S. arenicolus* in Texas (Ryberg and Fitzgerald).

Enclosed you will find a status report on the core project, research on best management practices (Leavitt et al. 2011a). We have also included a draft of the manuscript describing our analysis of population density of *S. arenicolus* at 6 independent mark-recapture sites at Caprock Wildlife Area (Ryberg and Fitzgerald 2011a). Research on the core project provided an opportunity to quantify the effects of oil and gas trenching on mortality of vertebrates living within the Mescalero Sands ecosystem. A draft of the manuscript describing these observations, which is currently in revision for *Western North American Naturalist* (Leavitt et al. 2011b), has been included here. Field work was conducted in 2011 in order to update the atlas of distribution and habitat published in 2010. Several surveys for *S. arenicolus* were conducted in June 2011 by Lee Fitzgerald and Dan Leavitt. These surveys resulted in 5 new important localities, which we document here in an update of the Atlas of Distribution and Habitat of *Sceloporus arenicolus* in New Mexico (Laurencio and Fitzgerald 2011). *Sceloporus arenicolus* tissues collected from the core project, as well as additional surveys in 2011, continue to inform on-going research on the genetic structure of *S. arenicolus* populations at multiple spatial and temporal scales in both fragmented and unfragmented habitats (Chan and Fitzgerald 2011). Finally, all of the research described in this report has provided the foundation for a complementary research program on *S. arenicolus* behavior, population dynamics, genetic structure, and distribution in Texas. We have included a summary of this nascent research program here (Ryberg and Fitzgerald 2011b).

During 2011, research on the core project was presented and discussed in both outreach and professional settings (see table below). In May, Dan Leavitt gave an "on-site" lecture to high-school students from Bosque High School in Albuquerque about the conservation and management objectives of the project. In August, Dan

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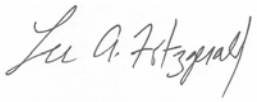
also gave an oral presentation titled "Landscape fragmentation disrupts lizard metacommunity structure in a sand-dune ecosystem" at the annual national meeting of the Ecological Society of America in Austin, TX. Lee Fitzgerald presented 2 invited research seminars to universities in Argentina. We also wish to call attention to a recent publication that resulted from Nicole Smolensky's masters thesis (citation below; pdf attached). Although not funded by BLM, the paper presents important findings for conservation and management of dunes sagebrush lizards.

Smolensky, N. and L.A. Fitzgerald. 2011. Population variation in dune-dwelling lizards in response to patch size, patch quality, and oil and gas development. Southwestern Naturalist 56(3):315-324.

Annotated list of meetings attended and events occurring in 2011

Date	Meeting or event
January 2011	Participate in USFWS update on proposed listing Roswell
April 2011	Begin field work
May 2011	Participate in USFWS update on 2011 Dunes Sagebrush Lizard Surveys and Protocol
May 2011	High-school students from Bosque High School in Albuquerque visit core project with on-site lecture by Leavitt
June 2011	Participate in conference meeting to discuss lizard issues in Texas and habitat boundaries
June 2011	Meeting in Loco Hills to discuss research on restoration ecology with scientists from Texas Tech and CEHMM
August 2011	"Landscape fragmentation disrupts lizard metacommunity structure in a sand-dune ecosystem" oral presentation by Leavitt Ecological Society of America national meeting
August 2011	Research update to BLM, FWS, NMGF, CEHMM, and others
September 2011	End field work
October 2011	Annual Report

Thank you for your attention and please contact any of our team with questions,
Sincerely,



Lee A. Fitzgerald
Professor and Curator of Amphibians and Reptiles
Co-Director Applied Biodiversity Science NSF-IGERT Doctoral Program

RESULTS FROM THE THIRD YEAR OF RESEARCH: EFFECTS OF MANAGEMENT PRACTICES FOR OIL AND GAS DEVELOPMENT ON THE MESCALERO DUNE LANDSCAPE AND POPULATIONS OF THE ENDEMIC DUNES SAGEBRUSH LIZARD, *SCELOPORUS ARENICOLUS*

A REPORT SUBMITTED TO THE BUREAU OF LAND MANAGEMENT, CARLSBAD OFFICE

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October, 2011

PROJECT GOAL

The aim of this project is to gain insight into the effects of management practices on patterns of landscape fragmentation and populations of the endemic lizard, *Sceloporus arenicolus*. Management practices currently used by the Bureau of Land Management and other agencies rely on controlling or minimizing the placement of caliche well pads and roads within shinnery oak sand dune complexes that are presumed to be occupied by *S. arenicolus*. The effectiveness of this strategy and the specific stipulations placed on oil and gas development have not been evaluated with an experimental ecology approach. It is critical to understand at what spatial scale the management practices may be most effective, and at which scale they may be ineffective. Our research is assessing this issue with both mensurative and manipulative experimental approaches. Initially, we will draw comparisons of landscape characteristics and population size of *S. arenicolus* between areas that have already been developed by oil and gas to those that have not. In the future, we will use a before-after-control-intervention (BACI) design to quantify landscape change as a result of oil and gas development and to document changes in population size of *S. arenicolus*. We have created nine study areas with three trapping grids within each. This design will allow us to describe effects at small scales, ~ 1 hectare, which corresponds to a single well pad sites, and at larger landscape scales, 100 hectares which matches the extent of development in resource rich areas.

INTRODUCTION

Sceloporus arenicolus (Dunes Sagebrush Lizard) is a small lizard that occurs within the Mescalero-Monahans shinnery-sands ecosystem of west Texas and southeastern New Mexico (Degenhardt and Jones 1972, Degenhardt et al. 1996, Fitzgerald and Painter 2009). Within the shinnery-sands ecosystem, *S. arenicolus* demonstrates habitat specialization, selecting sites that contain large depressions (blowouts) with little vegetation over the other available habitats (Sena 1985, Fitzgerald et al. 1997). As a consequence of a small geographic distribution, habitat specialization and concern regarding

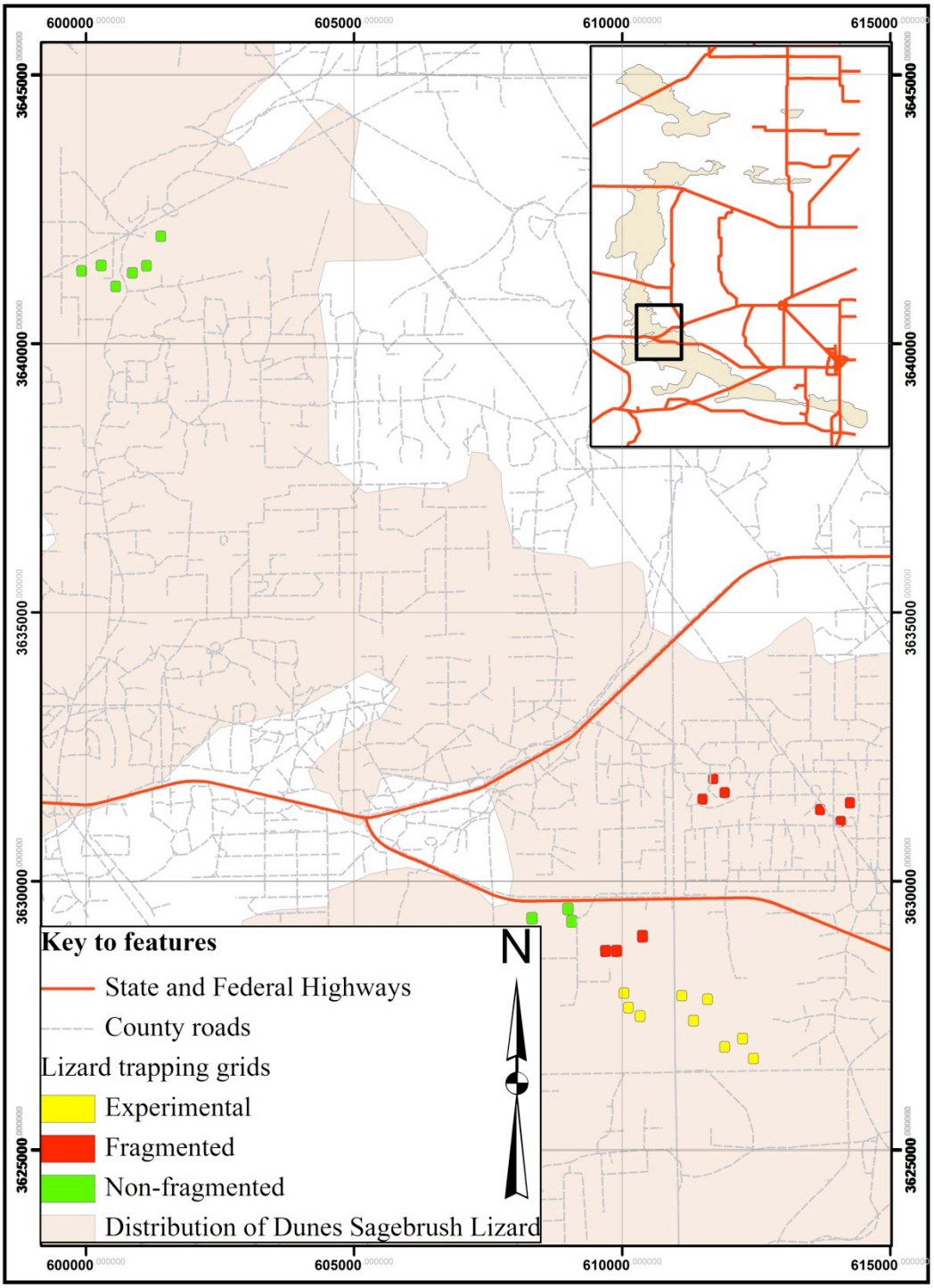

Figure 1. Locality of Study areas (100 ha.) with inset map of Dunes Sagebrush Lizard distribution in southeastern New Mexico (datum: NAD 83, zone 13).

human impacts in this region, *S. arenicolus* is listed as endangered by the New Mexico Department of Game and Fish (2006) and is proposed for federal listing as endangered by the US Fish and Wildlife Service (Federal Register December 14, 2010). State and federal agencies charged with natural resource conservation in southeastern New Mexico and west Texas need basic ecological information on *S. arenicolus* in order to formulate scientifically defensible conservation plans.

Landscape management practices will likely decide the fate of *S. arenicolus*. Researchers have explored the effects of land-use practices in this region on *S. arenicolus*. Snell et al. (1997), in a mensurative experiment, demonstrated the negative effect of shinnery oak (*Quercus havardii*) removal by the herbicide tebuthiuron on lizard captures, and specifically on *S. arenicolus*. Significantly fewer lizards were found on treated sites than on untreated sites resulting in the management suggestion to cease use of such herbicides within a 500 m buffer surrounding occupied habitat (Painter et al. 1999). Research conducted by Sias and Snell (1996, 1998) demonstrated the potential for effects of oil and gas development on *S. arenicolus*. They found a significant negative correlation between proximity to oil well pads and presence of *S. arenicolus*. This research initiated current management practices to locate caliche oil well pads and roads outside of occupied habitats in the shinnery-oak flats (Painter et al. 1999) and the suggestion to restrict surface occupancy 200 m away from occupied sites (U. S. Fish and Wildlife Service et al. 2008).

Because previous research has identified this pattern of fewer observations of *S. arenicolus* near well pads, we intend to see if these patterns exist across larger scales with populations. Additionally, the landscape alteration associated with oil and gas development fragments the landscape. Thus, it is unknown what may occur to *S. arenicolus* populations once the surrounding landscape becomes fragmented. This project aims to assess the current land management practices utilized in the Mescalero-Monahans shinnery sands ecosystem in southeastern New Mexico through manipulative and mensurative experiments on landscape fragmentation and population monitoring of *S. arenicolus*. With the before-after experimental design, we intend to tease apart the effects of land management practices through manipulation and avoid some of the potential effects associated with differences observed in mensurative experiments. We intend to analyze these effects alongside other relevant ecological patterns and processes at multiple scales. Patterns such as resource availability and landscape structure, and processes such as interaction rates and community turn-over are also of interest to this study. The following report is written following the conclusion of the third field season of research. Herein we ask if there are distinct differences between fragmented and non-fragmented sites regarding: 1) lizard communities, 2) populations of *S. arenicolus* and 3) landscape environmental features.

METHODS

Study area site selection.— Our study areas are 100 hectare regions, each containing 3 trapping grids, of which each is 1.2 ha. Nine study areas were selected to accommodate three treatment groups. These groups are: fragmented, non-fragmented, and experimental (non-fragmented areas that were intended to become fragmented following year two). The criteria used to select treatments were as follows: study areas 1) must lie in shinnery-oak dune landscape and 2) be less than 0.5 kilometer distant from a known *S. arenicolus* specimen locality 3) and if density of well pads exceeded 12 active wells per 100 hectares treatments were considered fragmented if not they were selected to be either non-fragmented or experimental. All study areas are located in Eddy and Lea counties, NM (Fig. 1) in the vicinity of the

villages of Loco Hills and Maljamar. Within each study area we randomly chose three areas large enough to place a pitfall trapping grid (1.2 ha.). Grids were randomly positioned using a random number generator and an x/y grid in Microsoft Excel. Grid size was determined based upon the known home range size variability for *S. arenicolus* (Hill and Fitzgerald 2007). Additionally, all trapping grids are approximately 100 m distant from each other, to minimize the potential of regular *S. arenicolus* movements between grids (Fitzgerald et al. 2005).

Pitfall trapping.— To capture lizards for population and community analysis we used pitfall trapping grids. Trapping grids consist of 30 pitfall buckets spaced 20 meters apart in a 6 x 5 (rows to columns) format. Following installation, each trapping grid was operated for five-day operation cycles (S) equaling 30 trap-days per grid per day (2009 S = 3; 2010 S = 6; 2011 S = 6). The operation cycles were staggered between 11 April and 30 September to account for any seasonal or climatic related lizard activity. Daily operation of a trapping grid consisted of visiting each trap, processing each lizard captured, and removing all living organisms from the trap. Processing lizards included identifying individuals to species, determining sex, measuring the snout-to-vent length (SVL: a straight line distance from rostrum to cloaca), measuring tail, measuring any regenerated tail, recording mass (g) of each individual with a Pesola® spring scale, individually marking by toe-clip (Waichman 1992), recording general information regarding body condition or reproductive state and releasing them back to the study grid near the area of capture. Additionally, all other vertebrates captured during trapping were individually marked, measured and weighed. The nomenclature used for reptiles and amphibians and mammals follows recent trends by Crother et al. (2008) and Schmidly (2004), respectively.

On site habitat comparison.— To account for structural and environmental variability, both coarse and fine scale measures of habitat structure were measured on each grid. In 2009, we measured leaf litter and relative cover. At each trap in 2010, we calculated 16 more microhabitat variables (described in detail in Leavitt and Fitzgerald 2010). In 2011, an additional measure of environmental variability was measured in the blowouts of each grid. We measured blowout depth by taking a distance measurement with a laser range finder (Nikon Riflehunter 1000) from the highest and lowest points in a depression. We took a gps (Garmin GPSMap 60CSx; ±3m) point at each of these locations so to calculate distance on the ground. From this we can estimate the depth of the blowouts using the Pythagorean Theorem.

Statistical analysis.— Because landscape manipulation has not yet occurred, non-fragmented and experimental treatments were grouped for statistical comparison to fragmented treatments. All means are reported, and we determined a relationship to be significant if two –tailed P-values were < 0.05. We compared lizard communities with t-Tests using the following measures: total lizard captures, total lizard recaptures, unique lizard captures and lizard diversity. Using this same test we individually analyzed each common species' total captures, recaptures, and unique captures per treatment. We calculated a species diversity index ($\Delta 1$; Hurlbert 1971) from the species compositions at each site per year. Hurlbert's diversity measure was chosen due to its robustness at low sample sizes (Olszewski 2004). To assess *S. arenicolus* populations in this study we provide descriptive statistics for captures, recaptures, male:female ratios and demographics. In addition, we provide a measure of dispersion calculated from recaptured individuals and the total distance moved between traps over time. Further, we describe the phenology of *S. arenicolus* over the three trapping years to help plan the trapping schedule for future work. To assess landscape environmental differences between treatments for each grid, correspondence analysis was conducted with the environmental variables calculated in 2010. This analysis was conducted on a list of

242 environmental variables measured both remotely [with program FRAGSTATS on the vegetation class layers of each of the trapping grid (McGarigal et al. 2002)] and at the site of each trap (Leavitt and Fitzgerald 2010).

RESULTS

There were a total of 11,559 vertebrates captured in pitfall traps between 2009 and 2011 (of this 11,263 were lizards). No new species of vertebrates were captured in 2011 that had not yet been captured previously. A total of 48,600 trap days were recorded between May 2009 and August 2011, and the lizard capture rate was 23.2 %.

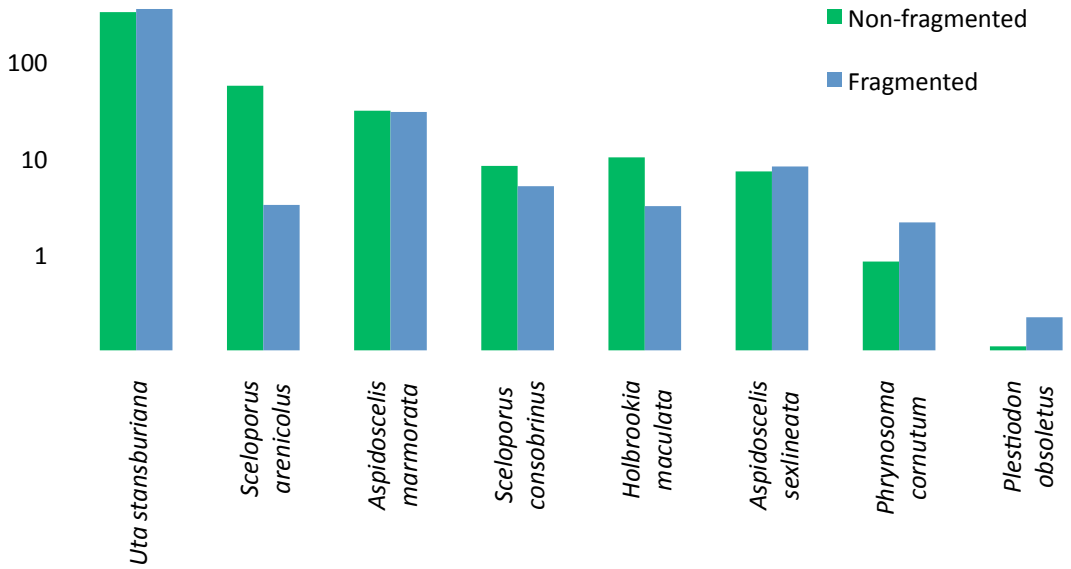

Figure 2. Rank abundance of lizard captures (y-axis, log-scale) between two treatment types.

Lizard communities.—The total number of lizards captured per grid between treatments was not significantly different for the pooled data between 2009 and 2011 ($t = 1.06$, $P = 0.15$), with the non-fragmented grids averaging more captures (441.28) than the fragmented (400.78). Total recaptures were not significantly different with slightly fewer recaptured individuals trapped on the fragmented grids ($t = 0.28$, $P = 0.78$; non-fragmented: 99.2, fragmented: 95.4). When corrected for recaptures (total – recaptured), the total number of unique individuals captured was not significantly different ($t = 1.41$, $P = 0.17$) with fragmented grids having fewer unique lizard captures than the non-fragmented grids (305.3 and 342.1, respectively). Diversity was significantly different between treatments with non-fragmented grids being more diverse than fragmented grids ($t = 5.99$, $P < 0.01$; non-fragmented: 0.44, fragmented: 0.27).

Table 1. Summary of total, male (M), female (F), recaptured (R), gravid female, and hatchling *Sceloporus arenicolus* along with male female ratio, percentage recaptured, dispersion rate (m/day) and treatment (Non-fragmented [NF]; fragmented [F]; experimental [EX]) for trapping grids where *S. arenicolus* was present between 2009 and 2011.

Grid	Total	M	F	R	Gravid F	Hatchlings	M:F	% R	Dispersion	Treatment
LQ3†	1	0	1	1	0	0	0.00	1.00	0.00	NF
TR2	1	0	1	0	1	0	0.00	0.00	0.00	F
LAB2	3	1	2	1	0	1	0.50	0.33	0.00	F
LAB1	4	2	2	0	0	1	1.00	0.00	1.54	F
CS3	8	6	2	1	1	2	3.00	0.13	6.67	NF
LAB3	9	5	4	2	1	2	1.25	0.22	0.00	F
PR3	12	5	7	0	1	5	0.71	0.00	0.00	F
CS1	16	8	8	7	4	0	1.00	0.44	9.61	NF
CS2	17	11	6	7	5	0	1.83	0.41	10.03	NF
VI2	22	14	8	5	0	0	1.75	0.23	0.51	NF
SI1	24	14	8	1	3	5	1.75	0.04	0.00	EX
CAJ1	30	22	8	11	1	2	2.75	0.37	10.38	EX
CAJ2	40	19	20	4	3	1	0.95	0.10	13.33	EX
SI3	40	20	20	12	8	2	1.00	0.30	0.00	EX
LQ2	57	26	30	23	7	5	0.87	0.40	2.43	NF
NS2	57	31	25	28	5	1	1.24	0.49	1.94	EX
CAJ3	59	36	22	16	3	6	1.64	0.27	1.54	EX
LQ1	63	39	24	31	2	2	1.63	0.49	8.02	NF
NS1	65	29	36	21	7	3	0.81	0.32	3.58	EX
SI2	85	43	42	22	10	11	1.02	0.26	0.00	EX
VI3	110	61	49	31	6	13	1.24	0.28	2.11	NF
NS3	119	59	60	37	9	11	0.98	0.31	4.94	EX
VI1	172	88	83	62	10	22	1.06	0.36	5.44	NF
Total	1014	539	468	323	87	95	1.15	0.32	4.88	

†This record is for individual A2B1 described below in the movements section.

Capture rates per lizard species were compared by treatment (Fig. 2). The species having significantly different capture rates between treatments were *S. arenicolus* and *H. maculata* (Fig. 2). Both of these species were in significantly higher abundances on the non-fragmented trapping grids (*S. arenicolus*: t = 4.91, $P < 0.01$; *H. maculata*: t = 2.26, P = 0.02). Conversely, no differences were detected between treatments for *U. stansburiana*, *S. consobrinus*, *A. marmorata*, *A. sexlineata*, *P. cornutum* or *P. obsoleta*.

Dunes Sagebrush Lizard captures.—This species occurs in much lower frequency on fragmented grids versus non-fragmented grids (Table 1). Four fragmented grids have not yet produced a capture of *S. arenicolus* (TR1, TR3, PR1, PR2). All of these grids exist between US 82 and SH 529 and between Maljamar and Loco Hills. These grids are excluded from the summary statistics in Table 1.

Dunes Sagebrush Lizard phenology.— Seasonal activity patterns for *S. arenicolus* can be inferred from our capture rates (Fig. 3). Captures were highest in May (corresponding with the peak of adult *S. arenicolus* captures), with a second, smaller peak occurring in August (coinciding with the peak of hatchling captures). April had the second highest monthly capture rate of all *S. arenicolus*. Both adult male *S. arenicolus* and adult female *S. arenicolus* captures peaked in May. However, there were more adult male *S. arenicolus* captured every month with the exception of August. Gravid females were captured between May and July, with the highest capture rates in June. The date for first hatchling observed was 22-25 July, yet the peak of the hatchling captures occurred in August.

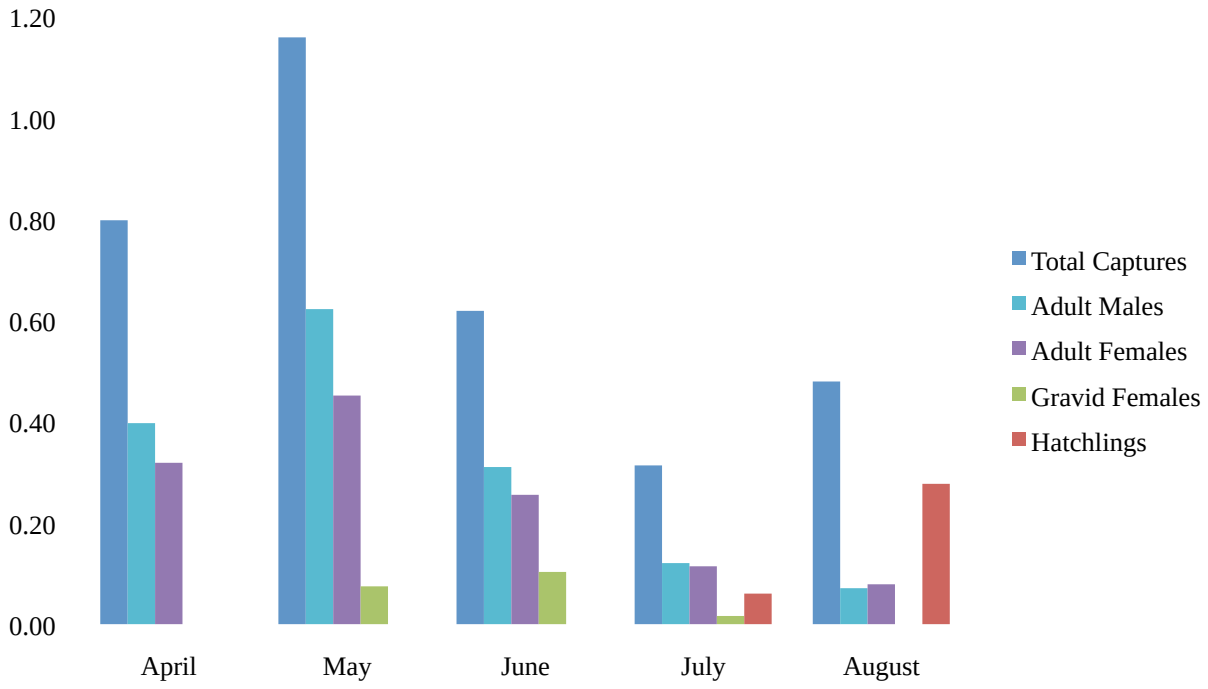

Figure 3. *Sceloporus arenicolus* captures per monthly trapping effort between 2009 – 2011.

Dunes Sagebrush Lizard movements.— The following two records are interesting, because they describe the long-distance movements of *S. arenicolus* between sites.

On 27 June 2011, a female *S. arenicolus* marked A2B1 was captured at the LQ3 site in trap 9. This lizard was given this mark on 22 May 2011 at the VI3 site in trap 13 (we can be certain of this due to similarities in tail regeneration patterns). The straight line distance between these traps is 843m ± 5m (measured in Garmin MapSource using GPS localities taken at each trap). Further, these two trap locations are separated by a road (Square Lake Rd.) and a patchwork of different types of vegetative classes including shinnery-dunes, shinnery-flats, and shinnery/mesquite-flats.

On 29 June 2011, a male *S. arenicolus* marked A2B5 was captured at the VI3 site in trap 16. It is possible (but not certain) that this individual was marked at VI1 site in trap 10 on 21 May 2011. However, differences in the descriptions of the individual call this observation into question. If this is the same individual it represents a movement of 117m, between trapping sites and across a caliche road.

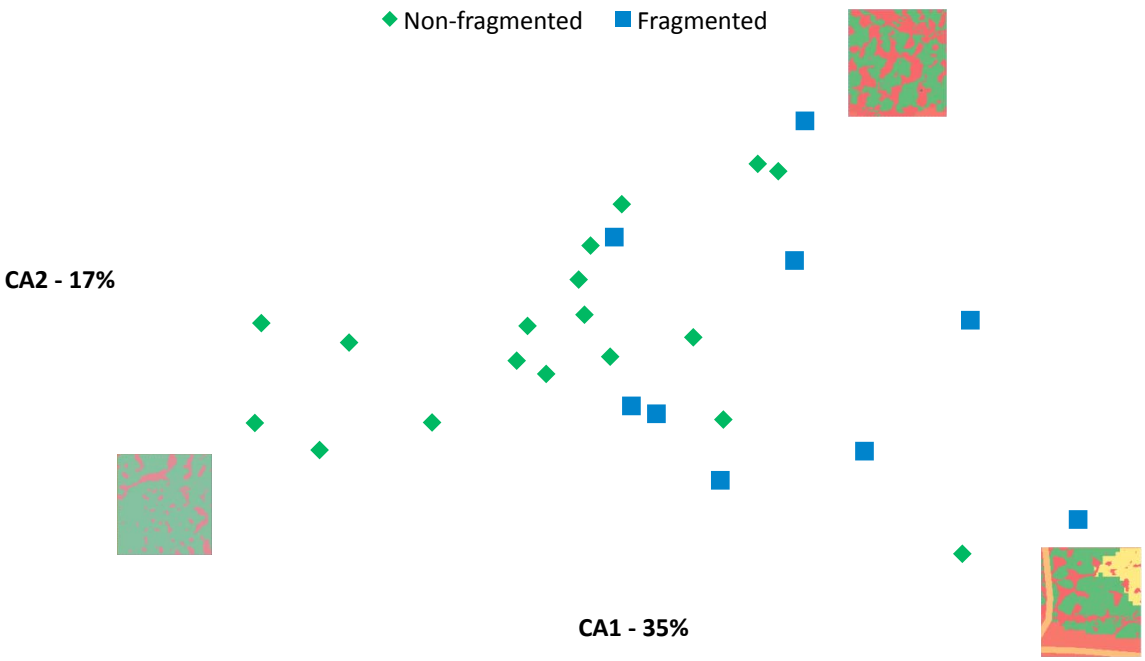

Figure 4. Correspondence analysis of environmental variation on 27 sites in the Mescalero-Monahans shinnery sands ecosystem of New Mexico. Two axes represent 52% of total variation in the dataset. Axis 1 (CA1) represents variation in dominant cover types (shinnery oak duneland – open blowout). Axis 2 represents a gradient of patch clumpiness (i.e. how far apart are patches from each other). Inset maps represent trapping grids found at each end of the spectrum, color corresponds with cover type (red = blowout, open ground; green = shinoak duneland; yellow = grassland; orange = road).

Environmental variation.— Of the 242 environmental variables calculated both on site and remotely, the most explanatory variables relate to the configuration of the landscape (Fig. 4). This analysis suggests that fragmented sites have as much open space as the non-fragmented and the configuration appears to be similar. The differences in environmental structure may not be represented in landscape metrics

calculated from satellite imagery, but rather from differences of on-the-ground comparisons of blowout depths.

DISCUSSION AND WORK IN PROGRESS

These data demonstrate significant differences exist in lizard communities among the treatment types. Non-fragmented sites (experimental and non-fragmented) have significantly greater lizard diversity and a higher average number of captures for *S. arenicolus* and *H. maculata*. Our data suggest differences exist in lizard diversity where land management practices differ. Study areas in fragmented treatments contained fewer species, including fewer *S. arenicolus* and *H. maculata*. As a result of this, fragmented study areas have lower lizard diversity than those that are non-fragmented. The goal of our research is to relate the effects of management practices on patterns of landscape fragmentation and populations of the endemic lizard, *S. arenicolus.* At this stage, it is too early to estimate population sizes of *S. arenicolus* in all of our study areas (Table 1). By the end of the 4-year study, we will be able to calculate good population estimates that will provide an important addition to the relationships we have already identified. Currently, we are drawing comparisons between fragmented areas and areas that have not yet been fragmented. Until the proposed experimental treatments become fragmented we cannot identify the proximate mechanism that is responsible for the differences observed between fragmented and non-fragmented habitats.

Capture rates for the different demographic groupings of *S. arenicolus* in Table 1 demonstrate the extent of the differences between fragmented and non-fragmented populations of *S. arenicolus*. Indeed, only three recaptured individuals have been captured on fragmented grids and these rates are well below that of the non-fragmented control sites. In addition to building our datasets on lizard demography, home range and movements, we gained additional insights on lizard movements and activity: One *S. arenicolus* moved >800 m, and it is possible another moved >100m; *Sceloporus arenicolus* are sometimes active during winter months (Fig. 3; Leavitt et al. 2011); Environmental variation between fragmented and non-fragmented landscapes may be best examined with metrics that account for the depth of depressions on the landscape rather than the size of the patch of sand (Fig. 4; Fitzgerald et al. 1997, Smolensky and Fitzgerald 2011).

Our ongoing research is focused on both landscape patterns and resource availability. Between now and the next field season we will accomplish a variety of tasks. More thorough analysis of landscape characteristics associated with community compositions will be conducted. Analyses of functional characteristics of species in relation to their capture rates and environmental variation at capture sites will be conducted. Finally, we will evaluate whether *S. arenicolus* serves as an umbrella species in the Mescalero sands ecosystem based on patterns of its presence and that of other biodiversity. Taken together, this information will provide results and insights necessary for promoting conservation strategies on BLM land.

ACKNOWLEDGEMENTS

The field work conducted was accomplished with the superb assistance provided by Matthew R. Acre, Levi T. Cole, Johanna A. Huntley, Drew E. Dittmer, Allison F. Leavitt, Anthony J. Romano, and Christopher M. Schalk (Texas A&M). Ty Allen (US DOI: BLM, now USFWS) provided technical assistance and had a crucial role in establishing and implementing this research. Ty, John Chopp, and

Steve Bird (US DOI: BLM) helped schedule assistance from technicians from both the US DOI: BLM and the Center for Excellence in Hazardous Materials Management. Debra Hill and Tim Breen (US DOI: USFWS) kindly provided and transported our accommodations for the duration of the research in 2009. Bill and Linda Gideon provided the finest field accommodations in Maljamar for 2010 and 2011. Additional assistance with the sorting arthropod samples was provided by Johanna A. Huntley, Paul Lenhart, Marion Le Gall, Leslie E. Ruyle and Wade A. Ryberg. Additionally we thank the following individuals for on-going collaborations, assisting in planning, technical, and/or physical support: L. M. Chan, T. B. Cotten, J. R. Dixon, G. W. Ferguson, A. L. Fitzgerald, C. D. Hathcock, T. J. Hibbitts, M. T. Hill, L. Laurencio, B. D. Marks, C. W. Painter, H. L. Prestridge, D. R. Roelke, W. E. Rogers, A. P. Sena, N. L. Smolensky, H. L. Snell, M. L. Treglia, L. D. Walker, K. O. Winemiller, and T. J. Zaffarano. This research was supported by the (US DOI: BLM), received research approval from the NMDGF (permit: 3409 SCI) and follows the institutional animal use protocols under the following Texas A&M University approval: AUP# 2008-95.

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SOURCE-SINK DYNAMICS IN A POPULATION OF THE DUNES SAGEBRUSH LIZARD (*SCELOPORUS ARENICOLUS*) AT CAPROCK WILDLIFE AREA

A REPORT SUBMITTED TO THE BUREAU OF LAND MANAGEMENT, CARLSBAD OFFICE

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October, 2011

OVERVIEW

The goal of this project is to describe spatial variation in population density and movement rates of *Sceloporus arenicolus* (Dunes Sagebrush Lizard) and other co-occurring lizard species found in Shinnery Oak sand-dune habitats within the Mescalero Sands. In this report, we use 5 years of mark-recapture data collected from 6 sites at Caprock Wildlife Area, NM (Hill and Fitzgerald 2007) to estimate survival, recruitment, population density, and diffusion rates of a *S. arenicolus* population occupying a large patch of Shinnery Oak sand-dune habitat. These data identify source-sink dynamics within this population of *S. arenicolus* that are associated with the quality and configuration of blowouts within Shinnery Oak sand-dune habitat. We present these data as a manuscript co-authored with Michael T. Hill (TAMU) and Charles W. Painter (NMDGF). General management recommendations discussed in the manuscript are listed below.

- Source and sink Shinnery Oak sand-dune habitats should be managed as a collective landscape.
- Management approaches labeling sink habitats as expendable should be discouraged.
- Source habitats are likely most important in determining population size, but sink habitats increase population connectivity, which supports population persistence through time.

Our current and future work on this project includes analyses of spatial variation in population density and/or occupancy and movement rates of the remaining lizard species co-occurring with *S. arenicolus*. We will describe the relative importance of abiotic (e.g., landscape features) and biotic mechanisms (e.g., competition) in determining the patterns of occupancy observed in this lizard assemblage. In doing so, we hope to better understand the role of co-occurring lizard species in determining variation in *S. arenicolus* presence/absence from small scales (e.g., 1 hectare, about one well pad site) up to larger landscape scales (e.g., 100 hectares, many well pads in a resource rich site).

NEIGHBORHOODS, SOURCES, AND SINKS: LANDSCAPE PATTERN GENERATES METAPOPULATION DYNAMICS IN AN ENDEMIC HABITAT SPECIALIST

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Abstract

Populations can exhibit spatial subdivision at multiple scales, and this subdivision can have considerable influence on population dynamics. Landscape features that restrict or direct organism movements define the distribution of individuals and therefore have strong effects on the spatial subdivision of populations and their dynamics. At large spatial scales, discrete habitat patches across landscapes define patterns of population subdivision, and population dynamics are well described by metapopulation theories incorporating the characteristics of individual habitat patches (e.g., size or isolation). At smaller spatial scales, however, landscape features within habitat patches may also generate patterns of subdivision within populations, and the localized dynamics within these subdivided populations may best be described by landscape pattern within habitat patches rather than individual habitat patch characteristics. Here, we describe a multisite mark-recapture study designed to link processes affecting localized population dynamics (e.g., mortality, recruitment) of a lizard, *Sceloporus arenicolus*, with landscape pattern within a patch of habitat. We make this link using the neighborhood concept, which reduces population dynamics to the scale of individual lizard movements. We found that landscape pattern within habitat patches generated variation in the density, movement, and thus spatial organization of individual lizards into neighborhoods of different sizes. Areas containing smaller neighborhoods exhibited recruitment rates that were lower than estimated mortality and diffusion rates that were much smaller than the spatial extent of our sampling areas, and areas containing larger neighborhoods exhibited the opposite pattern. This spatial pattern of neighborhood dynamics is closely aligned with the metapopulation concept of source-sink populations, defined as net exporters or importers of individuals. We conclude that patterns of population subdivision traditionally described by metapopulation dynamics among populations occupying multiple habitat patches may also describe patterns of population subdivision within populations occupying individual habitat patches that vary in landscape pattern. As such, population dynamics viewed in a landscape context must consider the explicit distribution and movement of individuals in space within habitat patches as well as among them. Our findings bear important implications for the conservation of populations in general, and specifically for the conservation of *S. arenicolus* populations throughout its range.

Key words: diffusion; landscape; movement; neighborhood size; patch; population subdivision; recruitment

Introduction

Populations of most species are subdivided at multiple scales (Kareiva 1986, Gilpin 1987, Amarasekare 1994, Tilman and Kareiva 1997). The scale of population subdivision often emerges as a consequence of movements of individuals through heterogeneous landscapes (Johnson et al. 1992). Thus, even for species with narrow habitat requirements, there are potentially many ways of conceptualizing population subdivision at a variety of spatial scales that reflect underlying habitat patchiness and landscape patterns (Wiens 1989). Characterizing population subdivision as a consequence of landscape pattern is important not only for understanding persistence of populations, as it can influence the degree to which a population is buffered from natural and anthropogenic disturbances (Hinrichsen and Holmes 2010), but also for understanding genetic structure and evolution within populations (Hanski and Gilpin 1997). As such, quantifying landscape-scale population subdivision is especially critical for species that are endangered, harvested, invasive, or disease-causing. However, defining population subdivision at the most appropriate spatial scale is not always obvious (Amarasekare 1994, Thomas and Kunin 1999).

At larger temporal and spatial scales where discrete habitat patches support entire populations, population subdivision can be described using metapopulation approaches (Hanski and Gilpin 1997, Murphy 2001). Classic approaches designate population dynamics at “local” and “regional” spatial scales (Hanski 1999). Local population dynamics are determined by the survival and recruitment of individuals within discrete habitat patches. Regional metapopulation dynamics are determined by the extinction and colonization of local populations. Importantly, metapopulation dynamics arise from the influence of patch heterogeneity (e.g., patch area, isolation, shape, context) on colonization and extinction rates (Royle and Dorazio 2008). As such, many patterns of population subdivision reflecting different metapopulation dynamics and persistence are possible (e.g., source-sink, core-satellite) depending on how local population extinction and colonization rates covary among habitat patches across landscapes (Harrison and Taylor 1997).

Subdivision may also exist within populations at scales below the habitat patch-level (Amarasekare 1994). In particular, the spatial configuration of landscape features (i.e., landscape pattern) and quality of resources within habitat patches may facilitate or constrain the movement of individuals and create spatial variation in population density (Ritchie 2009). This variation in population density can generate groups of strongly interacting individuals called “neighborhoods” that are organized regionally into continuous networks (Wright 1946). Individual neighborhoods can vary in rates of survivorship and recruitment, and the persistence of the network depends on dispersal or diffusion among neighborhoods just as in metapopulations (Pacala and Silander 1985, Flather and Bevers 2002, Fraterrigo et al. 2009, Roy et al. 2009). Patterns of population subdivision reflecting metapopulation dynamics (e.g., source-sink, core-satellite) can therefore also emerge among neighborhoods that are self-organized by landscape pattern within habitat patches (Pulliam 1988). Understanding how such habitat heterogeneity generates patterns of subdivision within populations and therefore influences their persistence is an important line of research that is part of a larger synthesis between landscape ecology and metapopulation theory (Harrison and Taylor 1997, Wiens 1997, Hanski 1999, With 2004).

Herein, we posit that many of the same patterns of population subdivision thought to manifest regionally in metapopulations also occur at smaller scales within populations due to the

response of individuals to landscape pattern within habitat patches. We illustrate this concept using multisite mark-recapture data from a population of the Dunes Sagebrush Lizard *Sceloporus arenicolus*, an extreme habitat specialist occuring only in Shinnery Oak (*Quercus havardii*) sand-dune habitats of southeastern New Mexico and west Texas (Fitzgerald and Painter 2009, Laurencio and Fitzgerald 2010). Briefly, within patches of habitat, *S. arenicolus* uses wind-hollowed depressions, called sand-dune blowouts. They prefer larger blowouts with specific topographic, thermal, and physical characteristics, but they also readily move among them incorporating several blowouts within a single home range (Fitzgerald and Painter 2009). Importantly, *S. arenicolus* are not uniformly distributed among and do not occupy all blowouts, even those with seemingly suitable characteristics. Thus, the blowout landform creates a landscape pattern that may facilitate or constrain the movement of individual lizards creating spatial variation in the density and therefore organization of lizards into neighborhoods with consequences for the spatial dynamics of lizard populations occupying patches of Shinnery Oak sand-dune habitat. As such, the goals of this study were to 1) determine how blowout heterogeneity organizes lizards into neighborhoods, 2) describe the "localized" dynamics of those neighborhoods, and 3) identify the pattern of population subdivision that emerges from spatial variation in the "localized" dynamics of neighborhoods across the landscape.

Methods

Study sites and data collection

Study sites ─ All study sites were situated within the Bureau of Land Management (BLM) Caprock Wildlife Area (33° 26'50.0" N, 103° 47'56.0" W), which occupies a large patch of Shinnery Oak sand-dune habitat located approximately 48 km east of Roswell, NM. Previous genetic research on *S. arenicolus* in this area failed to detect fine-scale genetic structure indicating that this patch of habitat hosts a single, extensive population (Chan et al. 2009). We selected six sites within this population of *S. arenicolus* and constructed a pitfall trapping grid of 36 buckets (20 liter) spaced 15 m apart in a 6x6 pattern at each site. We buried each bucket so that its top edge was flush with the sand and then covered each pitfall trap with a 41 cm x 41 cm plywood square elevated slightly off the surface to provide an entrance and shade for trapped lizards. With this design, each grid sampled an area of 5,625 m^2, which is large enough to include dozens of blowouts per site. Due to logistical constraints, the six grids were established over three years (2 per year). Grids at sites 1-2, 3-4, and 5-6 were constructed in 2005, 2006, and 2007 respectively. All trapping grids were operational from the year of construction to September 2009.

Data collection ─ All 6 sites hosted monthly trapping occasions from May-September in 2007-08 and from April-September in 2009 during the activity season of *S. arenicolus* (late April-September; Degenhardt et al. 1996). Due to the iterative construction of trapping grids, sites 1-4 hosted additional monthly trapping occasions from June-July 2006, and sites 1-2 hosted an additional trapping occasion in June 2005. For each trapping occasion, traps were opened on day 1, checked on days 2 through 6, and closed on day 6 resulting in 5 full days of continuous trapping. For all lizards captured, pitfall location was noted, snout-vent length and mass were measured, sex was determined, fertility was estimated via palpation (females only), and individuals were marked permanently by toe-clipping.

Analysis

Encounter histories and population parameter estimates — The parameters of interest in our population analyses were apparent survival probability (ϕ; probability that a lizard survives from time t to t+1, given it is alive at the beginning of time t), recapture probability (p; probability that a lizard alive at time t is recaptured), rate of population change (λ), recruitment rate (f; number of individuals recruited during the interval per member of the population alive at time t), and population size. We were also interested in estimating probabilities of movement among sites (ψ; probability of moving to a site in which the marked individual may potentially be encountered, given it is alive and at that site); however, no marked individuals were detected moving among sites. Thus, we restricted our analysis of lizard movements to estimates of site-specific diffusion rates (see *Population redistribution*).

To estimate population parameters, mark-recapture data from each monthly trapping occasion were summarized into encounter histories and then imported into Program MARK (White and Burnham 1999). Encounter histories included 16-19 trapping occasions over the five-year study. Our general approach for estimating parameters was to develop *a priori* models for analysis, evaluate goodness-of-fit and estimate an overdispersion parameter ($\hat{c}$) for each data set. Next we selected the most parsimonious model for inference based on AICc model selection (Burnham and Anderson 2002), which is a version of Akaike's information criterion (Akaike 1973, Sakamoto et al. 1986) corrected for small sample bias (Hurvich and Tsai 1989), and then estimated parameters from candidate models (adjusted for overdispersion if necessary; Lebreton et al. 1992, Anderson et al. 1994) using a model-average approach that incorporates model selection uncertainty.

Apparent survival and recapture rates — We used Cormack–Jolly–Seber open population models (Cormack 1964, Jolly 1965, Seber 1965, Burnham et al. 1987, Pollock et al. 1990, Franklin et al. 1996) to estimate apparent survival (ϕ) and recapture probability (p) of lizards for each monthly trapping occasion across all six sites independently. We examined sex-dependent characteristics and temporal variation of these parameters with models grouped by sex (g) and crossed with constant (.) or variable (t) time effects (16 models). The global model included estimates of sex and time effects plus the interaction between them for both apparent survival probability and recapture probability. For all six data sets (i.e., sites), we assessed goodness-of-fit with the global model using the median-$\hat{c}$ approach in Program MARK. Estimates of $\hat{c}$ ranged from 0.94 to 1.05 for all six data sets, which indicated no overdispersion and good fit of the data to the models. Consequently, all analyses were run with estimates of $\hat{c}$ = 1.00.

Recruitment and rate of population change — From the same encounter histories following a similar approach, we used models developed by Pradel (1996) to estimate recruitment (f) and the rate of population change (λ), which determined whether populations were increasing ($\lambda > 1.0$), decreasing ($\lambda < 1.0$), or static ($\lambda = 1.0$). We examined models where apparent survival probability, recapture probability, and recruitment or the rate of population change were either constant (.) or variable (t) (8 models per estimate of f and λ). The global model for each independent estimate of recruitment and the rate of population change was fully time-dependent. For all six data sets (i.e., sites), we assessed goodness-of-fit of these global models and estimated overdispersion ($\hat{c} = \chi^2$/df using the combined χ^2 values and degrees of freedom (df) from tests 2 and 3) in Program RELEASE (Burnham et al. 1987). Across all six data sets, estimates of $\hat{c}$ ranged from 0.7 to 1.06 for models of recruitment and the rate of population change. Analyses were adjusted for overdispersion where necessary.

We investigated synchrony within and among sites for population parameters identified as time-dependent. After determining that there was no serial correlation in these time-

dependent parameters, synchrony was determined by measuring the tendency of series to move in the same direction (Buonaccorsi et al. 2001). We also investigated synchrony between population parameters identified as time-dependent using the same methods.

Population and neighborhood size ─ We used the open-population POPAN parameterization (Schwarz and Arnason 1996, 2006) in Program MARK to estimate population abundance as modeled by N, super-population size. Super-population size can be thought of as the total number of animals available for capture during the study (Nichols 2005). Lizard density was calculated by dividing the N estimates by the area of each grid site. Some parameter estimates in the POPAN parameterization of fully time-dependent models are confounded preventing accurate parameter estimation. Because the same effort was used in all sampling occasions, we constrained variability in capture rates over time and also maintained constant survival to resolve this confounding and allow all parameters to be properly estimated. Because having an equal fraction of the super-population returning each sampling interval is highly unlikely biologically, we treated *PENT* (probability of entrance) as fully time-dependent within models across all sites. With these restrictions, a single model was fit to each data set to estimate super-population size across each site. For each data set, we assessed goodness-of-fit and estimated overdispersion using the fully time-dependent model as discussed above in Program RELEASE (Burnham et al. 1987). Across all six data sets, estimates of $\hat{c}$ ranged from 0.7 to 1.09. Analyses were adjusted for overdispersion where necessary.

Estimates of lizard density across sites were combined with site-specific measures of intra-grid movements calculated from the subsequent recaptures of individual lizards that generated each of the encounter histories described above. Following Wright (1946), these densities and intra-grid movements were used to calculate neighborhood size (NS) for each site using the equation below:

$$\boldsymbol{NS} = 4\pi\sigma^2 \boldsymbol{d}$$

where σ^2 is the variance of intra-grid movements for lizards along a single axis and d is the density of individuals.

Population redistribution ─ Although there are many different kinds of movement metrics, we focused on estimating the diffusion rate (D) at each of the six sites. The diffusion rate is a single metric of population spread through time that incorporates both the mean and variance in movement distances over time. We calculated the distance of intra-grid movements from the subsequent recaptures of individual lizards that generated each of the encounter histories described above. In order to use such diffusion model approaches to estimate population spread, several assumptions must be made. First, we assumed a quasi-homogeneous environment at each site, because the spatial scale of heterogeneity was less than the scale of movement (i.e., lizards could move readily among adjacent blowouts within Shinnery Oak sand-dune habitat). Second, we assumed there was no directional persistence or any other correlation between successive movements (i.e., displacements were best described by an uncorrelated random walk). Tests supporting independence among successive movements were conducted following the methods of Swihart and Slade (1985). Five of the six sites exhibited no autocorrelation between successive movements (i.e., at α=0.05, all observed t^2/r^2 > critical values for upper bound, resulting in failure to reject the null hypothesis of independence) and the remaining site (3) contained too few successive movements to carry out the test. With no reason to suspect dependence among successive movements at only site 3 versus the remaining five sites, we characterized the diffusion rate at each of the sites using uncorrelated random walk

procedures. The estimated diffusion rate for an uncorrelated random walk in two-dimensional space from *n* moves is:

$$\hat{D} = \frac{\sum_{i=1}^{n} l_i^2}{4\sum_{i=1}^{n} t_i}$$

where l_i is the length of the *i*-th move and t_i is its duration (Turchin 1998). We scaled the duration of movements by generation time of the lizard (i.e., m^2/generation). This re-scaling is common when estimating diffusion rates for territorial species that may move only once during their lifetime as juveniles.

Associations between landscape pattern, neighborhoods, and population dynamics ─ To identify relationships between landscape pattern and neighborhood size, we mapped the 6 grid sites on two different digitized landcover layers in a GIS. One layer was a classification of vegetation types for each site at a 1-m resolution derived from Landsat Enhanced Thematic Mapper (ETM) satellite imagery and aerial photos (Neville et al. 2005). The other layer was a classification of sand-dune blowouts derived directly from 1-m digitally rectified orthoquarterquads (DOQQ's) taken in 2004 using ArcInfo (Environmental Systems Research Institute 1999-2005b).

Landcover composition and structure were measured from 100 x 100 m plots, framing each sampling grid (75 m/side) with an additional 12.5 meters per side. These landcover plots were clipped from the two landcover layers, and landscape metrics were calculated using Program FRAGSTATS (McGarigal et al. 2002). Only Blowout and Shinnery Oak landcover types occurred in the study plots. Because *S. arenicolus* is only found in association with blowouts within Shinnery Oak sand-dunes (Fitzgerald and Painter 2009, Smolensky and Fitzgerald 2010, Laurencio and Fitzgerald 2010, Smolensky and Fitzgerald 2011), we focused landscape calculations on the Blowout landcover type. We used multiple metrics that quantified blowout area, edge, shape, and connectivity within the landscape plot surrounding each site (Appendix B1; formulas for all metrics described in FRAGSTATS manual McGarigal and Marks 1995).

In addition to these two-dimensional measures of blowout geometry, fine-scale topographical variation among blowouts was quantified at two different spatial scales to provide a third-dimension of Shinnery Oak sand-dune topography. We measured depth for all individual blowouts occurring within the landcover plots surrounding each grid site. We also measured elevation, slope, aspect, soil compaction, and percent vegetative cover at each pitfall trap using a handheld GPS unit (Garmin GPSMap 76CSx), level, and compass. Variation in these variables characterized the 'bumpiness' of the landcover plot surrounding each site. These three-dimensional measures of blowout heterogeneity and thus landscape pattern are thought to represent metrics of habitat quality for *S. arenicolus*, because they have been shown to be important for predicting habitat selection and presence-absence of the lizard at multiple scales throughout its range (Fitzgerald and Painter 2009, Smolensky and Fitzgerald 2011).

We used Pearson's correlations to assess relationships between neighborhood size and the different measures of landscape pattern described above. We also used Pearson's correlations to explore associations between neighborhood size, those population parameters estimated above, and diffusion rates across sites. Relationships with correlation coefficients > 0.70 were considered strong enough to warrant biological interpretation. This threshold represents the value of correlation coefficients for which the coefficient of determination

approaches 50% (i.e., variable x accounted for half of the variation in variable y under linear regression), implying a causal relationship.

Results

Encounter histories and population parameter estimates ─ From 2005 to 2009, 303 encounter histories were created from *S. arenicolus* marked, released, and recaptured 1,127 times across all six sites.

Apparent survival and recapture rates ─ Model structure among all competitive models ($\Delta AIC_c < 2.0$) generated for estimates of apparent survival probability (ϕ) using Cormack–Jolly–Seber open population models was similar across the six sites with sex-dependent (*g*) but constant trends (.) exhibiting the greatest support given the data (Table A1). Under these model structures, estimates of apparent survival probability were high for *S. arenicolus* in this region ranging from 0.88 to 0.95 with broadly overlapping 95% confidence intervals among sexes and across sites. Thus, while there was some evidence of sex-dependent survival in model selection procedures, estimates of male and female survival rates across sites were not distinguishable statistically (Table A2). The fact that survival rates were unified across sites reinforces inferences made from previous genetic research (Chan et al. 2009) that this patch of habitat hosts a single, extensive population.

Under these same model structures, estimates of recapture probability (*p*) varied among the six sites with sites 1, 3, and 5-6 exhibiting sex-dependent (*g*) but constant trends (.) and sites 2 and 4 exhibiting time-dependent (*t*) trends (Table A1). For those sites exhibiting sex-dependent but constant model structures, estimates of recapture probability ranged from 0.21 to 0.31 with broadly overlapping 95% confidence intervals among sexes across sites (Table A3). Thus, as seen above, there was no statistical evidence for sex-dependent recapture probabilities across sites despite support for such models in model selection procedures. Estimates of recapture probabilities for sites exhibiting time-dependent model structures were highly variable (Table A3); however, closer examination of these estimates in light of the broadly overlapping 95% confidence intervals suggested more subtle variation in recapture rates through time at those sites. Of the 99 trapping intervals (i.e., consecutive trapping occasions) across the six sites, only 3 intervals exhibited statistically distinguishable estimates of recapture.

Recruitment and rate of population change ─ Model structure among all competing models ($\Delta AIC_c < 2.0$) for estimates of recruitment (*f*) was consistently time-dependent across sites 1-2 and 4-6 with site 3 exhibiting constant trends in recruitment (Table A4). Estimates of recruitment ranged from 0.03 to 4.20 (Table A5). Temporally variable recruitment rates were weakly synchronized among certain sites within certain years (e.g., sites 1 and 6), but that synchrony varied from year to year (e.g., sites 2 and 4; Figure A1). Model structures generated for estimates of the rate of population change (λ) were consistently time-dependent across all sites with site 3 showing some support for constant trends (Table A6). Estimates of the rate of population change ranged from 0.20 to 1.70 (Table A7). Sites exhibiting temporal variation in the rate of population change were asynchronous, although most sites shared a characteristic dip and rebound in this estimate each year (Figure A1).

Models where apparent survival (ϕ) is time-invariant and the rate of population change (λ) varies over time are reasonable, as long as variation in recruitment (*f*) complements variation in the rate of population change. While this may be true in a general sense, we find no evidence that such synchrony between recruitment and the rate of population change is operating over the

trapping occasions in our mark-recapture study. Of 82 consecutive trapping occasions where both recruitment and the rate of population change varied, only 1 interval (site 4 in May-June 2007; Figure A1) exhibited synchrony between them. In the remaining 17 consecutive trapping occasions either recruitment or the rate of population change did not vary. This asynchrony indicates that recruitment varies both spatially and temporally (Franklin 1992), which can confound interpretations of apparent survival, recruitment, and the rate of population change. To resolve this confounding and because we are interested in spatial patterns of population dynamics, we use time-invariant estimates of recruitment and the rate of population change to determine associations between neighborhoods, population parameters, and diffusion rates across sites. Time-invariant estimates of recruitment across sites ranged from 0.09 to 0.16 (Figure 1A), while time-invariant estimates for the rate of population change across sites did not vary and were indistinguishable from 1.0.

Population and neighborhood size ─ Estimates of super-population size ranged from 30 to 144 across sites with densities ranging from 0.5 to 2.6 *S. arenicolus* per 100 m^2 (Table 1). Intra-grid movement distances ranged from 20.1 to 29.2 m across sites (Table 2). These density and movement components yielded neighborhood sizes ranging from 7.5 to 40 lizards across sites (Figure 1A).

Population redistribution ─ Diffusion rates were estimated across sites from 303 movement pathways comprised of the following components: mean step length, total path length, mean number of moves per path, and turning angle per move (Table 2). Diffusion rate estimates across the six sites ranged from 2,786 to 18,371 m^2/generation (Table 2).

Associations between landscape pattern, neighborhoods, and population dynamics ─ Neighborhood size was strongly correlated with blowout quality, geometry, orientation, and fine-scale topography within Shinnery Oak sand-dune habitats (Table 3). Specifically, neighborhood size was positively correlated with mean blowout contiguity, slope angle, east and west aspects, and the compactness of sand. Alternatively, neighborhood size was negatively associated with variation in blowout area, contiguity, and the compactness of sand (Table 3, bold values). This means that larger neighborhoods were associated with sites containing more compactly shaped blowouts of uniform size with steep east and west facing slopes and evenly compacted sand.

Recruitment was the only model parameter exhibiting spatial variation within this population. As such, associations between neighborhood size and estimated population parameters across sites were restricted to this relationship. There was a strong positive correlation between neighborhood size and recruitment across sites (Figure 1A). By plotting a conservative estimate of lizard mortality [1 - mean ϕ (lower 95% CI) across sites] against recruitment (horizontal dashed line), this graph shows the level of recruitment, 0.13, necessary to balance mortality across sites. Recruitment at Site 6 was estimated at 0.09, and that site is therefore labeled as a sink. Given a predictive relationship, the graph also shows the minimum neighborhood size, 22 (vertical dashed line), required to match that level of recruitment. Estimates of neighborhood size at Site 6 (7 lizards) were well below that minimum size.

There was also a strong positive correlation between recruitment and diffusion rates (Figure 1B). By plotting the area of each site, 5,625 m^2, against diffusion rates (horizontal dashed line), this graph shows how the level of diffusion determines whether localized populations are expanding or contracting. Site 6 exhibits diffusion rates smaller than the occupied area in the landscape indicating localized population contraction. Alternatively, four of the remaining five sites exhibit diffusion rates greater than the occupied area in the landscape

indicating localized population expansion. Because those sites also exhibit recruitment rates that are greater than mortality rates (now vertical dashed line), they are identified as sources.

Discussion

Our results showed how landscape pattern within a habitat patch created variation in the density, movement, and thus spatial organization of individual lizards into neighborhoods, whose dynamics influenced patterns of population subdivision across the landscape. Specifically, we found that the spatial configuration and quality of blowouts within Shinnery Oak sand-dune habitat determined the size of *S. arenicolus* neighborhoods (Table 3), which was positively related to recruitment (Figure 1A). Larger neighborhoods exhibited higher recruitment rates that resulted in population diffusion rates exceeding the spatial extent of our sampling areas. Conversely, recruitment in smaller neighborhoods did not balance mortality, and diffusion rates were much smaller than the spatial extent of our sampling areas (Figure 1B). This spatial pattern of neighborhood dynamics is closely aligned with the metapopulation concept of source-sink populations, defined as net exporters or importers of individuals, respectively (Pulliam 1988). Our results therefore converge on the conclusion that patterns of population subdivision traditionally described by metapopulation dynamics among populations may also describe patterns of population subdivision within populations.

This conclusion emphasizes the link between individual movements at small scales and patterns of population subdivision that can generate metapopulation phenomena at larger scales. Traditionally, the approach to making individual movements relevant to metapopulation theories has been to consider movements at the habitat patch-level, where population dynamics are more discrete (Hanski 1999). Conceptually, that approach directs our point of reference to broad-scale movements of individuals among patches that reflect migration among populations in a metapopulation. However, metapopulation dynamics depend on movements within patches as well as among them (Wiens 1997, With 2004). Because our study focused on subdivision within a population occupying a single habitat patch, we used the opposite approach and were able to reduce population phenomena to the scale of individual movements through the neighborhood concept.

The neighborhood concept, or more specifically the neighborhood size parameter, was devised by Wright (1946) to describe genetic structure within a population in terms of individual movements, although we used it here as an index of the number of individual lizards in the population that were coupled by strong interactions (e.g., potential mates, Ricklefs 1995). The ‘neighborhood’ therefore represented the basic demographic unit within this population, and neighborhood size and dynamics were defined by the movement of individual lizards. At larger spatial scales, individual neighborhoods were part of a continuous network whose connectivity was determined by regional diffusion among adjacent neighborhoods that were net exporters or importers of individual lizards. Importantly, the persistence of this network depended on diffusion-dispersal among neighborhoods just as in metapopulations. In this way, the neighborhood concept established a mechanistic link between individual lizard movements at small scales and patterns of population subdivision that generated metapopulation phenomena at larger scales.

By reducing the dynamics of populations to the scale of individual lizard movements, the use of the neighborhood concept in this study also provided an empirical link between landscape ecology and the conceptual domain of metapopulation theory. In lizard neighborhoods, both

localized dynamics and regional connectivity were determined by the movement of individuals within a habitat patch; therefore, anything affecting individual movements (e.g., landscape pattern, habitat heterogeneity) also altered the size, dynamics, and connectivity of neighborhoods. Indeed, the localized neighborhood dynamics and regional neighborhood connectivity that generated the metapopulation phenomena observed in this study are likely best characterized through an explicit understanding of how the spatial configuration and quality of blowouts within Shinnery Oak sand-dune habitats directed movements at the level of individual lizards.

Such an explicit consideration of how landscape pattern in this ecosystem leads to the emergence of spatial pattern in our population lies at the heart of the landscape ecology metapopulation synthesis, which has largely been theoretical (Wiens 1997, With 2004, Hanski 2010). Specifically, cell-based simulations of landscape mosaics have helped model how individual movements are shaped by fine-scale, within-patch heterogeneity (Pacala and Silander 1985, Flather and Bevers 2002, Ovaskainen et al. 2002, Fraterrigo et al. 2009, Roy et al. 2009). Our observations match the general conclusion from these models that fine-scale spatial pattern within populations can be generated completely from the localized dynamics of clusters of individuals interacting in neighborhoods. These models also conclude that such fine-scale spatial pattern within populations can have dramatic effects on the dynamics of populations and metapopulations across the landscape (Roy et al. 2009). More research at much larger scales than the 5 km^2 area we studied would be required to identify whether such fine-scale spatial pattern within our population had similar sweeping effects on the persistence of *S. arenicolus* across the landscape in our ecosystem.

As our empirical data illustrate how landscape pattern within habitats determines individual movements and the size of lizard neighborhoods, it follows that alterations to these habitats would disrupt neighborhood dynamics. Under such conditions, we would expect individual lizard movements to be redirected and neighborhoods to be reconfigured with consequences for the source-sink dynamics of the population at larger scales. Along these lines, it is noteworthy that the area around site 6, which we identified as a population sink, was sprayed with tebuthiuron herbicide to remove Shinnery Oak in the early 1990s. Although suitable abiotic conditions persist at this site post-spray (i.e., Shinnery Oak and blowouts are present), we suggest that this landscape change around site 6 probably impacted the movement of lizards and formation of lizard neighborhoods.

At larger scales, anthropogenic habitat loss and fragmentation would also alter the amount and spatial configuration of landscape heterogeneity both above and below the habitat patch-level. Several of the models listed above focus on this conservation issue and illustrate how changes in habitat configuration as it relates to individual movements can facilitate or have little effect on population persistence (Flather and Bevers 2002). Generally for source-sink populations, the amount, but not configuration, of source habitat is most important in determining population size and persistence (Ritchie 1997). Other studies on *S. arenicolus* in this system lend support to these ideas. For example, a recent study on population abundance of *S. arenicolus* throughout its range showed that amount and quality of Shinnery Oak sand-dune habitat were correlated and that lizard population size was correlated with habitat patch area rather than configuration (Smolensky and Fitzgerald 2011). As such, we suggest that the general recommendations for source-sink populations (Howe et al. 1991, Pulliam et al. 1992) apply for the conservation of *S. arenicolus*. These recommendations encourage the management of source and sink habitats as a collective landscape, because sink habitat can increase population size and

therefore persistence, and discourage management approaches that label sink habitats as expendable, because such habitats represent a form of landscape connectivity that supports colonization and thus population persistence (Pulliam and Danielson 1991, Howe et al. 1991).

Our study demonstrates for the first time that population processes in small-scale neighborhoods reflect those of metapopulations and thus provide a mechanism for the scaling of individual movements and recruitment up to dynamics traditionally reserved for metapopulations. This finding also calls attention to the importance of habitat conservation at multiple scales. To conserve populations, habitat conservation above and below the patch-level is needed to maintain processes that drive persistence of populations and metapopulations across landscapes. Conserving large areas of marginal habitat, for example, may not sustain the diffusion dynamics we documented that are likely required for long-term persistence of populations. With a better understanding of mechanisms that result in movement among neighborhoods and populations at different scales, future work can focus on disentangling the relative importance of diffusion and dispersal within and among habitat patches, and identify which is most important for population persistence under different anthropogenic threats.

Acknowledgments

We thank Don Miles for helpful comments on earlier versions of this manuscript. We also thank Drew Dittmer, Toby Hibbitts, Dan Leavitt, Chris Schalk, Nicole Smolensky, and Mike Treglia for helpful discussions on data analysis and interpretation. We are especially grateful to the following who provided valuable assistance in the field: (Texas A&M University) Taylor Cotten, David Laurencio, Laura Laurencio, Dan Leavitt, Nicole Smolensky, Toby Hibbitts, Brandon Bowers, Helen Boostrum, Waldo Ramirez, Don Hardeman, Jr.; (U. S. Fish and Wildlife Service) Debra Hill, Ty Allen, (New Mexico Department of Game and Fish) Lorraine McInnes, Andrea Currylow, Clay Crowder, James Borgmeyer, Austin Fitzgerald; (Bureau of Land Management) Steve Bird, Quincy Northrip, Johnny Chopp, Teal Richards Dimitrie, Jamie Hall, Danielle Grant, Christina Raving, Steven Torrez, Alma Padilla, Collin Shephard; (Western New Mexico University) Randy Jennings and his 2005 and 2009 Biology Classes; Erin Lynn, Jason Ashley, Matt Mathis, Casey Gilman, Maggie Fitzgerald, Evan Vincent, and Chuck Hathcock. Funding for this project was provided by the Bureau of Land Management and New Mexico Department of Game and Fish. This is publication number xxxx from the Texas Cooperative Wildlife Collection.

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Table 1. Site specific model estimates of super-population size (*N*) followed by estimated density per 100 m^2 for *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09. Also shown are the standard error (SE) and confidence interval (95% CI).

Site	*N*	SE	95% CI	Density
1	72	6	62-83	1.3
2	120	11	103-145	2.1
3	30	4	25-43	0.5
4	144	7	131-159	2.6
5	48	4	40-56	0.9
6	42	5	34-57	0.7

Table 2. Mean step length, total path length, and number of moves for *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09. Standard errors (SE) are also shown. Turning angles for *S. arenicolus* movements at each site are given. Population-level diffusion rates (m^2 per generation) at each site were estimated from movement data following Turchin 1998.

Site	Step Length (m)	Total Path Length (m)	Number of Moves	Turn Angle									Diffusion Rate
				0	45	90	135	180	225	270	315	Sum	m^2/gen
1	22.7 (0.6)	32.5 (5.2)	1.4 (0.2)	0	1	2	1	7	2	2	0	15	9,553
2	22.9 (1.1)	27.8 (3.1)	1.2 (0.1)	1	0	3	0	3	1	0	1	9	7,034
3	27.6 (4.2)	30.9 (6.4)	1.1 (0.1)	0	0	1	0	0	0	1	0	2	5,508
4	20.1 (0.7)	30.4 (2.8)	1.5 (0.1)	1	2	6	5	18	3	3	2	40	18,371
5	29.2 (2.6)	39.8 (7.3)	1.4 (0.2)	1	1	0	3	3	0	0	0	8	9,134
6	21.7 (1.5)	31.8 (5.1)	1.5 (0.2)	0	0	1	1	2	1	2	0	7	2,786

Table 3. Correlations (Pearson's) between *S. arenicolus* neighborhood size and mean and coefficient of variation in Blowout landcover metrics (Appendix B1) and blowout habitat variables (Appendix B2) at six sites in the Mescalero Sands from 2005-09. Correlation coefficients > 0.70 (coefficients of determination > 0.50) are in bold.

Landcover metric	**Correlation with neighborhood size**	
	Mean	Coefficient of Variation
Area	0.28	**-0.76**
Perimeter	0.25	-0.66
Gyrate	0.10	-0.28
Shape	0.23	-0.35
Fractal	0.01	0.02
Circle	0.60	-0.22
Contiguity	**0.79**	**-0.82**
Isolation	0.52	0.55
Habitat variable	Mean	Coefficient of Variation
Depth	0.26	-0.34
Elevation	-0.12	0.43
Soil compaction	**0.74**	**-0.86**
Slope	**0.83**	0.25
North-South facing	-0.61	0.41
East-West facing	0.49	**0.76**
Cover	0.06	0.47

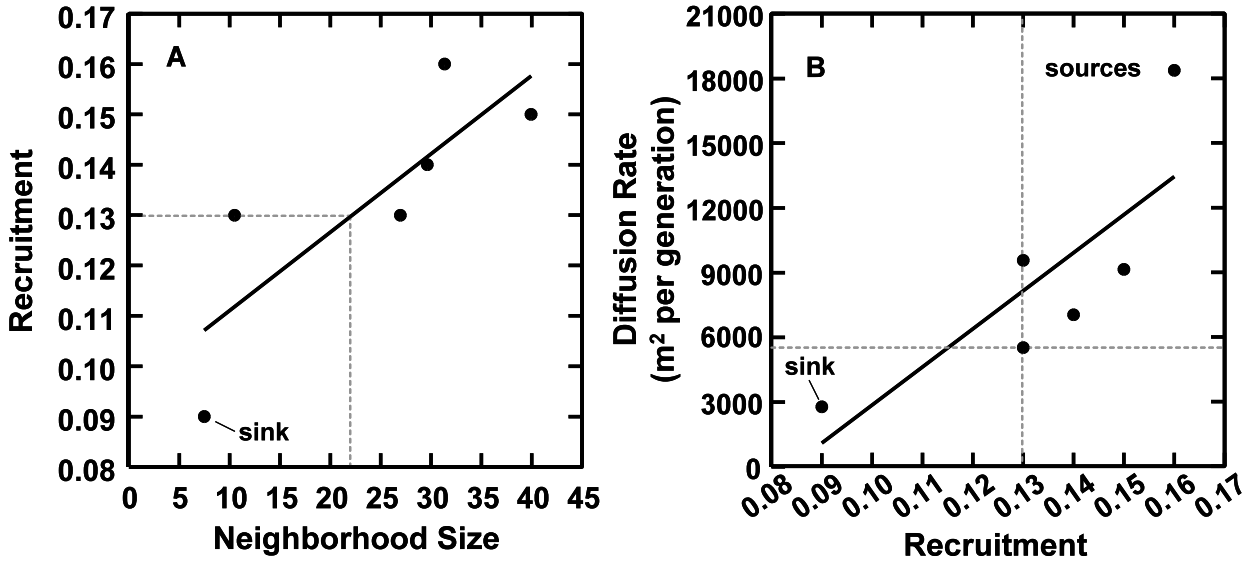

Figure 1. (A) Recruitment rate plotted as a function of neighborhood size (Pearson's correlation coefficient, R = 0.82; coefficient of determination, r^2 = 0.67). Given a predictive relationship, dashed lines indicate the level of recruitment, 0.13, and neighborhood size, 22, required to balance estimates of lizard mortality (see text). Site 6, with recruitment estimated at 0.09 and a neighborhood size of 7 lizards, is therefore labeled as a sink. (B) Diffusion rate (m^2 per generation) plotted as a function of recruitment rate (Pearson's correlation coefficient, R = 0.80; coefficient of determination, r^2 = 0.64). Dashed lines indicate the level of recruitment, 0.13, and diffusion rate, 5,625 m^2, required to balance estimated mortality of lizards and occupy the same area in the landscape. This panel identifies four sites as likely sources, because their recruitment rates are greater than mortality rates and their diffusion rates exceed the occupied area in the landscape indicating localized population expansion.

Appendix A

Table A1. Model ranking of Cormack–Jolly–Seber (CJS) mark–recapture models estimating apparent survival (ϕ) and recapture probability (*p*) for *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09. Shown are delta Akaike's information criteria corrected for small sample size (ΔAICc), the AICc weight (AICc wt), the number of parameters and the deviance for each model. A '(·)' denotes time-invariant parameters, '(*t*)' denotes time-variant parameters, and '(*g*)' denotes sex-dependent parameters.

Site	Model	ΔAICc	AICc wt	Parameters	Deviance
1	ϕ(.) *p*(.)	0.00	0.49	2	229.7
	ϕ(.) *p*(*g*)	1.73	0.20	3	229.3
	ϕ(*g*) *p*(.)	1.74	0.20	3	229.3
	ϕ(*g*) *p*(*g*)	3.08	0.10	4	228.5
	ϕ(.) *p*(*t*)	16.28	<0.01	19	202.5
2	ϕ(.) *p*(*t*)	0.00	0.78	19	255.2
	ϕ(*g*) *p*(*t*)	2.78	0.19	20	255.2
	ϕ(.) *p*(.)	8.49	0.01	2	304.4
	ϕ(.) *p*(*g*)	9.38	0.01	3	303.2
	ϕ(*g*) *p*(.)	10.58	<0.01	3	304.4
3	ϕ(.) *p*(.)	0.00	0.31	2	100.2
	ϕ(*g*) *p*(.)	0.03	0.31	3	97.8
	ϕ(*g*) *p*(*g*)	0.39	0.25	4	95.7
	ϕ(.) *p*(*g*)	1.94	0.12	3	99.7
	ϕ(*t*) *p*(.)	39.83	<0.01	18	74.1
4	ϕ(.) *p*(*t*)	0.00	0.75	18	428.1
	ϕ(*g*) *p*(*t*)	2.23	0.25	19	428.0
	ϕ(.) *p*(*g**t*)	12.31	<0.01	35	397.4
	ϕ(*g*) *p*(*g**t*)	14.30	<0.01	36	396.7
	ϕ(.) *p*(.)	21.13	<0.01	2	484.2
5	ϕ(.) *p*(.)	0.00	0.55	2	147.6
	ϕ(*g*) *p*(.)	2.08	0.19	3	147.5
	ϕ(.) *p*(*g*)	2.16	0.19	3	147.6
	ϕ(*g*) *p*(*g*)	4.24	0.07	4	147.4
	ϕ(*t*) *p*(.)	18.87	<0.01	15	131.8
6	ϕ(*g*) *p*(.)	0.00	0.31	3	145.7
	ϕ(.) *p*(.)	0.25	0.27	2	148.1
	ϕ(.) *p*(*g*)	0.26	0.27	3	145.9
	ϕ(*g*) *p*(*g*)	1.54	0.14	4	144.9
	ϕ(*t*) *p*(.)	13.87	<0.01	16	121.9

Table A2. Model-averaged estimates of apparent survival for female and male *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09 derived using Cormack-Jolly-Seber (CJS) mark-recapture models. Also shown are the standard error (SE), unconditional SE (USE), and 95% confidence intervals (CI) for each model-averaged estimate.

Site	Sex	ϕ	SE	USE	95% CI
1	Female	0.93	0.016	0.016	0.89-0.96
	Male	0.93	0.018	0.019	0.88-0.96
2	Female	0.95	0.010	0.010	0.93-0.97
	Male	0.95	0.010	0.010	0.93-0.97
3	Female	0.94	0.025	0.029	0.85-0.98
	Male	0.88	0.044	0.060	0.71-0.96
4	Female	0.94	0.010	0.010	0.92-0.96
	Male	0.94	0.010	0.010	0.92-0.96
5	Female	0.93	0.023	0.023	0.87-0.96
	Male	0.93	0.024	0.025	0.86-0.96
6	Female	0.93	0.027	0.032	0.83-0.97
	Male	0.95	0.020	0.023	0.88-0.98

Appendix A3. Sex-specific or time-variant (depending on model ranking) estimates of recapture probability for *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09 derived using Cormack-Jolly-Seber (CJS) mark-recapture models. Also shown are the standard error (SE), unconditional SE (USE) and 95% confidence intervals (CI) for each model-averaged estimate.

Site	Sex/Interval	*p*	SE	USE	95% CI
1	Female	0.27	0.042	0.043	0.19-0.36
	Male	0.29	0.049	0.054	0.19-0.40
2	June 2005-06	0.66	0.18	0.20	0.26-0.91
	June-July (2006)	0.20	0.08	0.08	0.09-0.41
	July-May (2007)	0.59	0.12	0.14	0.31-0.81
	May-June (2007)	0.13	0.07	0.07	0.04-0.33
	June-July (2007)	0.09	0.06	0.06	0.02-0.30
	July-Aug (2007)	0.05	0.05	0.05	0.01-0.30
	Aug-Sept (2007)	0.005	0.001	0.03	0.001-0.06
	Sept-May (2008)	0.37	0.13	0.13	0.16-0.64
	May-June (2008)	0.13	0.07	0.07	0.04-0.34
	June-July (2008)	0.18	0.08	0.08	0.07-0.40
	July-Aug (2008)	0.05	0.04	0.05	0.006-0.29
	Aug-Sept (2008)	0.13	0.07	0.07	0.04-0.35
	Sept-April (2009)	0.19	0.10	0.10	0.06-0.47
	April-May (2009)	0.27	0.10	0.10	0.12-0.51
	May-June (2009)	0.19	0.09	0.09	0.07-0.43
	June-July (2009)	0.24	0.10	0.10	0.10-0.49
	July-Aug (2009)	0.05	0.05	0.05	0.006-0.31
	Aug-Sept (2009)	0.25	0.09	0.10	0.11-0.47
3	Female	0.21	0.06	0.07	0.10-0.38
	Male	0.27	0.08	0.10	0.12-0.50
4	June-July (2006)	0.60	0.14	0.14	0.33-0.83
	July-May (2007)	0.69	0.23	0.23	0.21-0.95
	May-June (2007)	0.25	0.13	0.13	0.08-0.56
	June-July (2007)	0.38	0.11	0.11	0.19-0.60
	July-Aug (2007)	0.0002	0.00002	0.00003	0.00002-0.005
	Aug-Sept (2007)	0.32	0.10	0.10	0.15-0.54
	Sept-May (2008)	0.61	0.12	0.12	0.37-0.80
	May-June (2008)	0.46	0.08	0.08	0.31-0.62
	June-July (2008)	0.17	0.06	0.06	0.08-0.32
	July-Aug (2008)	0.18	0.06	0.06	0.09-0.33
	Aug-Sept (2008)	0.17	0.06	0.06	0.08-0.33
	Sept-April (2009)	0.23	0.07	0.07	0.12-0.41
	April-May (2009)	0.40	0.08	0.09	0.25-0.57
	May-June (2009)	0.38	0.08	0.08	0.24-0.54
	June-July (2009)	0.35	0.08	0.08	0.21-0.51
	July-Aug (2009)	0.15	0.06	0.06	0.07-0.30

	Aug-Sept (2009)	0.17	0.06	0.12	0.04-0.53
5	Female	0.30	0.05	0.05	0.21-0.42
	Male	0.31	0.06	0.06	0.21-0.43
6	Female	0.23	0.05	0.06	0.13-0.37
	Male	0.28	0.06	0.07	0.17-0.42

Table A4. Model ranking of Pradel mark–recapture models estimating apparent survival (ϕ), recapture probability (*p*), and recruitment (*f*) for *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09. Shown are delta Akaike's information criteria corrected for small sample size (ΔAICc), the AICc weight (AICc wt), the number of parameters and the deviance for each model. A '(·)' denotes time-invariant parameters, '(*t*)' denotes time-variant parameters.

Site	Model	ΔAICc	AICc wt	Parameters	Deviance
1	ϕ(.) *p*(*t*) *f*(*t*)	0.00	0.71	26	176.6
	ϕ(.) *p*(*t*) *f*(.)	3.03	0.16	21	196.6
	ϕ(.) *p*(.) *f*(.)	3.67	0.11	3	244.3
	ϕ(.) *p*(.) *f*(*t*)	7.11	0.02	13	223.9
	ϕ(*t*) *p*(.) *f*(.)	22.9	<0.01	20	219.7
2	ϕ(.) *p*(*t*) *f*(*t*)	0.00	0.95	24	222.0
	ϕ(.) *p*(.) *f*(*t*)	6.61	0.03	8	272.2
	ϕ(.) *p*(*t*) *f*(.)	7.76	0.01	20	236.3
	ϕ(.) *p*(.) *f*(.)	25.7	<0.01	3	298.8
	ϕ(*t*) *p*(.) *f*(*t*)	31.0	<0.01	15	313.5
3	ϕ(.) *p*(.) *f*(.)	0.00	0.95	3	87.7
	ϕ(*t*) *p*(.) *f*(.)	5.88	0.05	9	111.1
	ϕ(.) *p*(.) *f*(*t*)	21.0	<0.01	10	105.1
	ϕ(*t*) *p*(.) *f*(*t*)	34.7	0.00	18	76.5
	ϕ(.) *p*(*t*) *f*(.)	46.8	0.00	16	102.0
4	ϕ(.) *p*(*t*) *f*(*t*)	0.00	1.00	26	446.1
	ϕ(*t*) *p*(*t*) *f*(*t*)	26.7	0.00	42	429.8
	ϕ(.) *p*(*t*) *f*(.)	33.2	0.00	20	493.9
	ϕ(*t*) *p*(.) *f*(*t*)	53.3	0.00	28	494.4
	ϕ(*t*) *p*(*t*) *f*(.)	56.0	0.00	36	476.0
5	ϕ(.) *p*(.) *f*(*t*)	0.00	0.99	9	183.0
	ϕ(*t*) *p*(.) *f*(*t*)	20.3	<0.01	12	211.5
	ϕ(.) *p*(.) *f*(.)	23.8	<0.01	3	229.3
	ϕ(.) *p*(*t*) *f*(*t*)	25.2	0.00	20	182.2
	ϕ(.) *p*(*t*) *f*(.)	29.8	0.00	17	197.4
6	ϕ(.) *p*(.) *f*(*t*)	0.00	0.45	5	151.0
	ϕ(.) *p*(*t*) *f*(*t*)	0.18	0.41	16	118.2
	ϕ(*t*) *p*(*t*) *f*(*t*)	2.50	0.13	20	104.1
	ϕ(*t*) *p*(.) *f*(.)	7.50	0.01	9	148.1
	ϕ(*t*) *p*(*t*) *f*(.)	21.6	<0.01	20	123.2

Appendix A5. Time-variant (depending on model ranking) estimates of recruitment (*f*) for *Sceloporus arenicolus* across 6 sites in the Mescalero Sands from 2005-09 derived using Pradel mark-recapture models. Also shown are standard error (SE), unconditional SE (USE), and 95% confidence intervals (CI) for each estimate.

Site	Interval	*f*	SE	USE	95% CI
1	June 2005-06	0.00	0.00	0.00	(0.00-0.00)
	June-July (2006)	0.00	0.00	0.00	(0.00-0.00)
	July-May (2007)	0.09	0.03	0.03	(0.04-0.17)
	May-June (2007)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2007)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2007)	0.00	0.00	0.00	(0.00-0.00)
	Aug-Sept (2007)	0.39	0.14	0.14	(0.17-0.68)
	Sept-May (2008)	0.10	0.04	0.04	(0.05-0.21)
	May-June (2008)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2008)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2008)	0.00	0.00	0.00	(0.00-0.00)
	Aug-Sept (2008)	0.16	0.12	0.12	(0.03-0.54)
	Sept-April (2009)	0.02	0.01	0.01	(0.01-0.08)
	April-May (2009)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2009)	0.09	0.05	0.05	(0.01-0.12)
	June-July (2009)	0.07	0.05	0.05	(0.01-0.10)
	July-Aug (2009)	0.09	0.07	0.07	(0.01-0.39)
	Aug-Sept (2009)	0.00	0.00	0.00	(0.00-0.00)
2	June 2005-06	0.00	0.00	0.00	(0.00-0.00)
	June-July (2006)	0.00	0.00	0.00	(0.00-0.00)
	July-May (2007)	0.07	0.03	0.03	(0.03-0.16)
	May-June (2007)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2007)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2007)	0.29	0.14	0.14	(0.01-0.95)
	Aug-Sept (2007)	2.00	1.50	1.50	(0.01-4.94)
	Sept-May (2008)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2008)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2008)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2008)	0.00	0.00	0.00	(0.00-0.00)
	Aug-Sept (2008)	0.41	0.23	0.23	(0.10-0.82)
	Sept-April (2009)	0.00	0.00	0.00	(0.00-0.00)
	April-May (2009)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2009)	0.15	0.11	0.11	(0.01-0.66)
	June-July (2009)	0.31	0.17	0.17	(0.01-0.96)
	July-Aug (2009)	0.04	0.02	0.02	(0.01-0.99)
	Aug-Sept (2009)	0.11	0.04	0.04	(0.01-0.94)
3	Time-invariant	0.07	0.02	0.03	(0.03-0.14)

	June-July (2006)	0.00	0.00	0.00	(0.00-0.00)
	July-May (2007)	0.09	0.03	0.03	(0.04-0.18)
	May-June (2007)	1.02	0.58	0.58	(0.01-2.16)
	June-July (2007)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2007)	0.99	0.52	0.52	(0.01-1.00)
	Aug-Sept (2007)	0.00	0.00	0.00	(0.00-0.00)
	Sept-May (2008)	0.06	0.03	0.03	(0.02-0.17)
	May-June (2008)	0.00	0.00	0.00	(0.00-0.00)
4	June-July (2008)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2008)	0.17	0.14	0.14	(0.02-0.63)
	Aug-Sept (2008)	0.58	0.30	0.30	(0.11-0.94)
	Sept-April (2009)	0.00	0.00	0.00	(0.00-0.00)
	April-May (2009)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2009)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2009)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2009)	0.10	0.09	0.09	(0.02-0.44)
	Aug-Sept (2009)	0.00	0.00	0.00	(0.00-0.00)
	July-May (2007)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2007)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2007)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2007)	0.21	0.19	0.20	(0.02-0.76)
	Aug-Sept (2007)	0.12	0.05	0.05	(0.05-0.26)
	Sept-May (2008)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2008)	0.00	0.00	0.00	(0.00-0.00)
5	June-July (2008)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2008)	1.02	0.51	0.56	(0.02-2.12)
	Aug-Sept (2008)	0.06	0.04	0.04	(0.17-0.19)
	Sept-April (2009)	0.00	0.00	0.00	(0.00-0.00)
	April-May (2009)	0.07	0.06	0.06	(0.01-0.16)
	May-June (2009)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2009)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2009)	0.10	0.05	0.06	(0.01-0.78)
	Aug-Sept (2009)	0.00	0.00	0.00	(0.00-0.00)
6	July-May (2007)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2007)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2007)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2007)	4.21	2.87	2.87	(0.01-9.84)
	Aug-Sept (2007)	0.06	0.05	0.05	(0.01-0.27)
	Sept-May (2008)	0.00	0.00	0.00	(0.00-0.00)
	May-June (2008)	0.00	0.00	0.00	(0.00-0.00)
	June-July (2008)	0.00	0.00	0.00	(0.00-0.00)
	July-Aug (2008)	0.41	0.31	0.31	(0.05-0.89)
	Aug-Sept (2008)	0.58	0.30	0.30	(0.11-0.94)

Sept-April (2009)	0.00	0.00	0.00	(0.00-0.00)
April-May (2009)	0.00	0.00	0.00	(0.00-0.00)
May-June (2009)	0.00	0.00	0.00	(0.00-0.00)
June-July (2009)	0.00	0.00	0.00	(0.00-0.00)
July-Aug (2009)	0.00	0.00	0.00	(0.00-0.00)
Aug-Sept (2009)	0.00	0.00	0.00	(0.00-0.00)

Table A6. Model ranking of Pradel mark–recapture models estimating apparent survival (ϕ), recapture probability (*p*), and rate of population change (λ) for *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09. Shown are delta Akaike's information criteria corrected for small sample size (ΔAICc), the AICc weight (AICc wt), the number of parameters and the deviance for each model. A '(·)' denotes time-invariant parameters, '(*t*)' denotes time-variant parameters.

Site	Model	ΔAICc	AICc wt	Parameters	Deviance
1	ϕ(.) *p*(.) λ(*t*)	0.00	0.99	17	127.2
	ϕ(.) *p*(*t*) λ(*t*)	8.88	0.01	38	165.2
	ϕ(.) *p*(*t*) λ(.)	46.1	<0.01	21	122.0
	ϕ(.) *p*(.) λ(.)	52.5	<0.01	3	196.6
	ϕ(*t*) *p*(.) λ(*t*)	53.1	<0.01	26	244.3
2	ϕ(.) *p*(.) λ(*t*)	0.00	1.00	17	165.2
	ϕ(*t*) *p*(.) λ(*t*)	30.4	0.00	24	215.2
	ϕ(*t*) *p*(*t*) λ(.)	73.6	0.00	29	223.3
	ϕ(.) *p*(*t*) λ(.)	82.5	0.00	21	256.5
	ϕ(*t*) *p*(.) λ(.)	96.0	0.00	10	298.2
3	ϕ(.) *p*(.) λ(*t*)	0.00	0.66	14	70.9
	ϕ(.) *p*(.) λ(.)	1.38	0.33	3	111.1
	ϕ(*t*) *p*(.) λ(*t*)	10.6	<0.01	20	40.4
	ϕ(.) *p*(*t*) λ(.)	42.1	0.00	18	88.4
	ϕ(*t*) *p*(*t*) λ(.)	42.5	0.00	21	62.6
4	ϕ(.) *p*(.) λ(*t*)	0.00	1.00	18	436.7
	ϕ(*t*) *p*(*t*) λ(.)	38.0	0.00	28	450.4
	ϕ(.) *p*(*t*) λ(.)	61.9	0.00	20	493.9
	ϕ(*t*) *p*(.) λ(.)	97.6	0.00	12	547.9
	ϕ(.) *p*(.) λ(.)	113.2	0.00	3	582.8
5	ϕ(.) *p*(*t*) λ(*t*)	0.00	0.97	30	63.5
	ϕ(.) *p*(.) λ(*t*)	6.79	0.03	13	138.2
	ϕ(*t*) *p*(.) λ(.)	50.7	0.00	7	198.2
	ϕ(*t*) *p*(*t*) λ(.)	54.6	0.00	20	162.9
	ϕ(.) *p*(.) λ(.)	72.5	0.00	3	229.3
6	ϕ(.) *p*(.) λ(*t*)	0.00	1.00	11	74.8
	ϕ(*t*) *p*(*t*) λ(.)	69.1	0.00	18	119.3
	ϕ(.) *p*(*t*) λ(.)	73.8	0.00	15	135.5
	ϕ(*t*) *p*(.) λ(.)	75.6	0.00	10	153.4
	ϕ(.) *p*(.) λ(.)	81.8	0.00	3	177.5

Appendix A7. Time-variant (depending on model ranking) estimates of the rate of population change (λ) for *Sceloporus arenicolus* across 6 sites in the Mescalero Sands from 2005-09 derived using Pradel mark-recapture models. Also shown are standard error (SE), unconditional SE (USE), and 95% confidence intervals (CI) for estimates.

Site	Interval	λ	SE	USE	95% CI
1	June 2005-06	1.00	0.00	0.00	(1.00-1.00)
	June-July (2006)	1.00	0.00	0.00	(1.00-1.00)
	July-May (2007)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2007)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2007)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2007)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2007)	1.00	0.00	0.00	(1.00-1.00)
	Sept-May (2008)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2008)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2008)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2008)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2008)	1.00	0.00	0.00	(1.00-1.00)
	Sept-April (2009)	1.00	0.00	0.00	(1.00-1.00)
	April-May (2009)	0.66	0.16	0.17	(0.33-0.88)
	May-June (2009)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2009)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2009)	0.31	0.11	0.12	(0.14-0.55)
	Aug-Sept (2009)	1.00	0.00	0.00	(1.00-1.00)
2	June 2005-06	1.00	0.00	0.00	(1.00-1.00)
	June-July (2006)	1.00	0.00	0.00	(1.00-1.00)
	July-May (2007)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2007)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2007)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2007)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2007)	1.00	0.00	0.00	(1.00-1.00)
	Sept-May (2008)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2008)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2008)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2008)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2008)	1.00	0.00	0.00	(1.00-1.00)
	Sept-April (2009)	1.00	0.00	0.00	(1.00-1.00)
	April-May (2009)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2009)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2009)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2009)	1.00	0.01	0.01	(0.99-1.01)
	Aug-Sept (2009)	1.00	0.00	0.00	(1.00-1.00)
3	June-July (2006)	1.00	0.00	0.00	(1.00-1.00)
	July-May (2007)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2007)	1.00	0.00	0.00	(1.00-1.00)

	June-July (2007)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2007)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2007)	1.00	0.00	0.00	(1.00-1.00)
	Sept-May (2008)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2008)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2008)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2008)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2008)	1.00	0.00	0.00	(1.00-1.00)
	Sept-April (2009)	1.00	0.00	0.00	(1.00-1.00)
	April-May (2009)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2009)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2009)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2009)	0.98	0.01	0.01	(0.96-1.01)
	Aug-Sept (2009)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2006)	0.39	0.08	0.00	(0.26-0.58)
	July-May (2007)	1.08	0.04	0.05	(1.00-1.16)
	May-June (2007)	1.67	0.53	0.51	(0.92-2.00)
	June-July (2007)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2007)	1.54	0.36	0.37	(0.98-2.00)
	Aug-Sept (2007)	1.08	0.03	0.03	(1.02-1.14)
	Sept-May (2008)	0.65	0.10	0.11	(0.48-0.87)
	May-June (2008)	0.84	0.09	0.09	(0.68-1.02)
4	June-July (2008)	1.11	0.13	0.12	(0.89-1.39)
	July-Aug (2008)	1.32	0.20	0.21	(0.98-1.79)
	Aug-Sept (2008)	0.98	0.02	0.02	(0.93-1.03)
	Sept-April (2009)	1.15	0.21	0.21	(0.82-1.63)
	April-May (2009)	0.93	0.14	0.15	(0.70-1.25)
	May-June (2009)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2009)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2009)	0.93	0.09	0.09	(0.77-1.13)
	Aug-Sept (2009)	1.00	0.00	0.00	(1.00-1.00)
5	July-May (2007)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2007)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2007)	1.14	0.07	0.07	(1.02-1.29)
	July-Aug (2007)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2007)	1.00	0.00	0.00	(1.00-1.00)
	Sept-May (2008)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2008)	0.98	0.33	0.33	(0.52-1.86)
	June-July (2008)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2008)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2008)	1.00	0.00	0.00	(1.00-1.00)
	Sept-April (2009)	1.04	0.05	0.05	(0.95-1.14)
	April-May (2009)	0.74	0.14	0.14	(0.51-1.07)
	May-June (2009)	1.18	0.24	0.24	(0.79-1.75)

	June-July (2009)	1.09	0.22	0.23	(0.74-1.60)
	July-Aug (2009)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2009)	1.00	0.00	0.00	(1.00-1.00)
6	July-May (2007)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2007)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2007)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2007)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2007)	1.08	0.05	0.06	(0.98-1.20)
	Sept-May (2008)	1.17	0.32	0.32	(0.69-1.97)
	May-June (2008)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2008)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2008)	1.35	0.27	0.28	(0.91-2.00)
	Aug-Sept (2008)	1.02	0.03	0.03	(0.96-1.08)
	Sept-April (2009)	1.00	0.00	0.00	(1.00-1.00)
	April-May (2009)	1.00	0.00	0.00	(1.00-1.00)
	May-June (2009)	1.00	0.00	0.00	(1.00-1.00)
	June-July (2009)	1.00	0.00	0.00	(1.00-1.00)
	July-Aug (2009)	1.00	0.00	0.00	(1.00-1.00)
	Aug-Sept (2009)	1.00	0.00	0.00	(1.00-1.00)

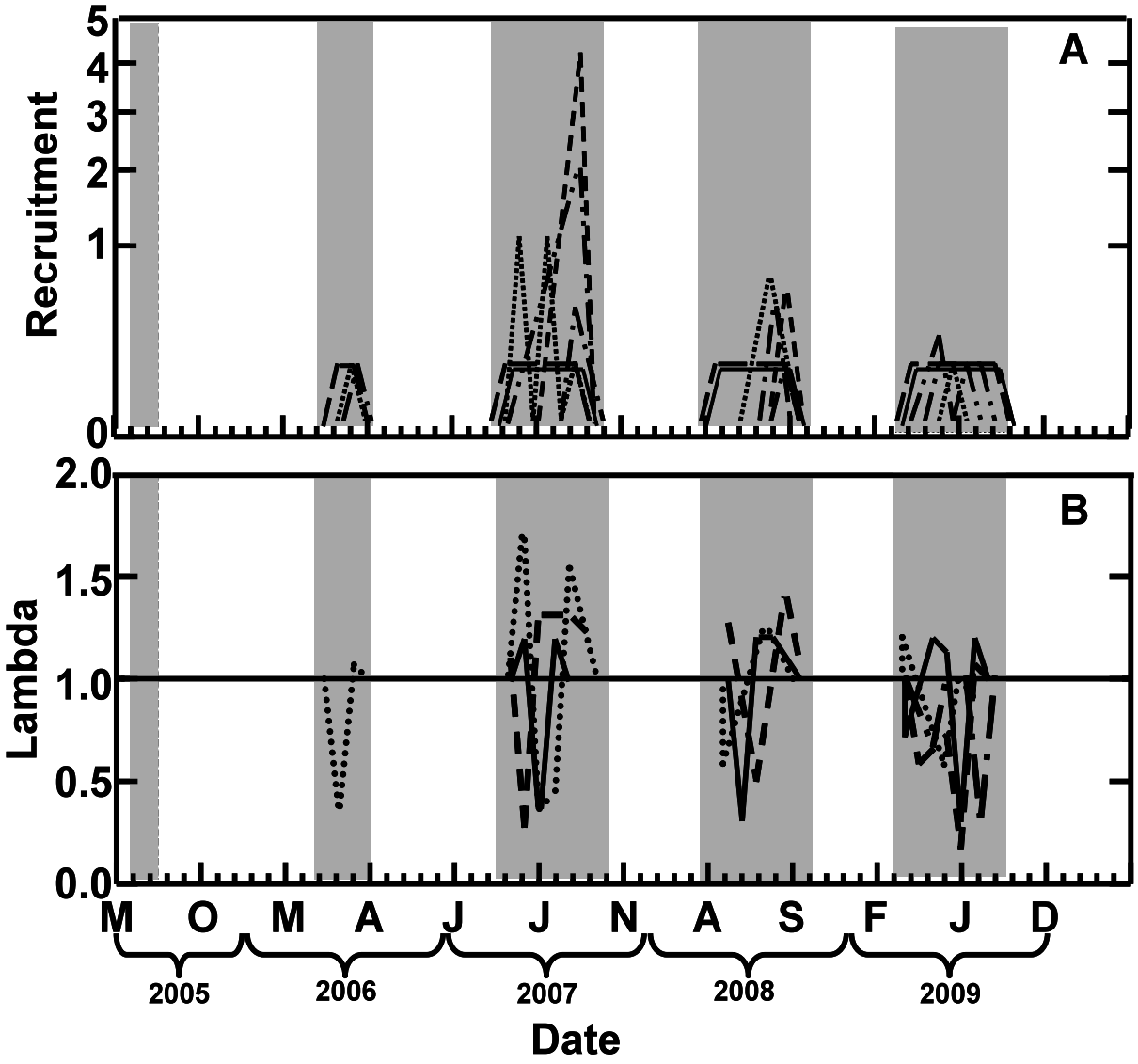

Figure A1. Estimates of recruitment and lambda (i.e., rate of population change) for *Sceloporus arenicolus* across six sites in the Mescalero Sands from 2005-09 (first month "M" = May). Shaded periods correspond to trapping occasions. Sites 1-6 are represented by dash-dot, dash-dot-dot, long-dash, dotted, solid, and short-dash lines, respectively. (A) Variable recruitment rates (note log scale) were observed across sites 1-2 and 4-6, but not site 3. Some sites exhibiting variable recruitment rates were synchronized within certain years (e.g., sites 1 and 6), but that synchrony varied from year to year at certain sites (e.g., sites 2 and 4). (B) All sites exhibited variation in lambda that was asynchronous within years across sites and within sites across years. No sites exhibited synchrony between recruitment rates and lambda that would suggest a positive correlation between recruitment and increases in population size.

Appendix B

Appendix B1. Mean (coefficient of variation) of Blowout landcover metrics (units = meters) for landcover plots surrounding each site calculated in FRAGSTATS. Blowout counts were 45, 45, 21, 29, 47, and 65 for sites 1-6, respectively.

Site	Area	Perimeter	Gyrate*	Shape†	Fractal‡	Circle§	Contiguity**	Isolation††
1	55.6 (4.68)	30.9 (3.10)	1.8 (1.72)	1.2 (0.39)	1.2 (0.12)	0.5 (0.27)	0.3 (0.88)	4.0 (0.46)
2	60.4 (4.25)	34.7 (3.17)	2.1 (2.14)	1.2 (0.47)	1.2 (0.12)	0.5 (0.27)	0.3 (0.91)	4.8 (0.61)
3	169.0 (3.37)	69.1 (2.79)	3.5 (2.25)	1.4 (0.59)	1.2 (0.13)	0.5 (0.35)	0.4 (0.83)	3.5 (0.55)
4	130.5 (4.66)	54.5 (3.61)	2.3 (2.07)	1.3 (0.54)	1.2 (0.10)	0.5 (0.23)	0.4 (0.79)	4.4 (0.47)
5	52.8 (2.90)	34.2 (1.69)	2.5 (1.38)	1.3 (0.33)	1.2 (0.14)	0.6 (0.25)	0.5 (0.61)	3.9 (0.52)
6	51.4 (5.15)	36.9 (3.90)	1.7 (2.12)	1.3 (0.59)	1.2 (0.13)	0.5 (0.29)	0.3 (1.01)	3.0 (0.45)

*mean radius of gyration; measures the average extent of blowouts from the blowout centroid
†mean shape index; measures mean blowout shape
‡mean fractal dimension index; measure of blowout shape complexity (1 for shapes with simple perimeters to 2 for shapes with convoluted perimeters)
§mean related circumscribing circle; measure of how compact blowouts are
**mean contiguity index; measures contiguity of cells within a blowout to provide an index of blowout boundary configuration and thus blowout shape
††mean Euclidian nearest-neighbor distance; measures blowout isolation by quantifying mean shortest distance from blowout edge to nearest blowout edge

Appendix B2. Means (coefficient of variation) of three-dimensional and habitat quality metrics for each grid site. Blowout counts for the depth metric were 10, 8, 11, 19, 18, and 8 for sites 1-6, respectively. All other metrics were measured at each of the 36 pitfall traps at each site.

Site	Depth*	Elevation (m)	Soil compaction†	Slope‡	North-South§	East-West**	Cover††
1	6.34 (0.361)	1292.7 (0.001)	2.59 (1.052)	11.7 (0.715)	0.08 (7.933)	-0.05 (-15.346)	51.7 (0.55)
2	9.19 (0.485)	1292.8 (0.002)	3.86 (0.828)	12.6 (0.869)	0.02 (30.82)	0.20 (3.677)	52.6 (0.47)
3	7.45 (0.264)	1298.5 (0.002)	5.64 (0.528)	11.8 (0.99)	-0.07 (-10.095)	-0.22 (-2.971)	54.4 (0.40)
4	6.60 (0.364)	1270.3 (0.002)	4.15 (0.696)	14.0 (0.776)	0.03 (25.663)	-0.14 (-5.106)	58.7 (0.61)
5	5.18 (0.415)	1279.9 (0.001)	4.06 (0.459)	13.2 (0.712)	0.04 (17.531)	0.28 (2.498)	34.6 (0.97)
6	4.89 (0.575)	1277.5 (0.001)	1.53 (0.963)	10.0 (0.723)	0.14 (5.433)	-0.10 (-6.18)	41.5 (0.57)

*vertical distance (in meters) measured perpendicularly from lowest to highest points in each blowout
†soil compaction measured at a point 2 m from each pitfall trap (Instrument, maker, and units)
‡degrees from level along dominant slope within $1m^2$ frame measured at a point 2 m from each pitfall trap
§northern to southern aspect (in radians) taken along dominant slope
**eastern to western aspect (in radians) taken along dominant slope
††percent sand cover vs. vegetative structure within $1m^2$ frame measured at a point 2 m from each pitfall trap

MORTALITY AND OTHER POTENTIAL IMPACTS OF PIPELINE TRENCHES ON SMALL VERTEBRATES IN SOUTHEASTERN NEW MEXICO

A REPORT SUBMITTED TO THE BUREAU OF LAND MANAGEMENT, CARLSBAD OFFICE

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October, 2011

ABSTRACT

Land use practices are well understood as a driver of biodiversity loss. Extraction of oil and gas resources occurs in a variety of ecosystems worldwide and is known to affect native flora and fauna during stages of exploration, development and transport. In southeastern New Mexico construction activities associated with oil and gas development pose risks to vertebrates. Construction of road networks, well pads and creation of open trenches during construction of pipelines are obvious activities that displace vertebrates and cause direct mortality. We surveyed 2 km of a 65 km trench created for a pipeline and found 24 individuals of 10 species of mammals and reptiles trapped in the trench. We extrapolated our capture and mortality rates to the total 65 km length of trench and estimated 130 captures/day and 21.6 deaths/day as result of this trench. Guidelines to reduce impacts on animals developed by the New Mexico Department of Game and Fish were not followed in the section of pipeline trench we surveyed. We recommend that trenching guidelines be refined and widely disseminated to all parties involved in construction of pipelines. Natural resource agencies may also consider adopting enforceable regulations related to construction of trenches.

Land use practices are an important driver of biodiversity loss (Tilman et al. 1994, Foley et al. 2005). Exploration, development and transport of oil and gas occurs in ecosystems worldwide (Doody et al. 2003) and land use practices in all stages of petrochemical operations influence organisms in a variety of ways (Moseley et al. 2010). Oil and gas development may increase exposure of organisms to pollutants (Ko and Day 2004) and cause immediate displacement or mortality during construction of roads, pipelines and facilities.

As a result of pipeline construction, that connects oil wells to collection facilities, a singular oil and gas operation can potentially impact multiple habitats or ecosystems. During pipeline construction trenches are dug that are often many kilometers in length and can remain open for extended periods. Animals that fall into these trenches are often trapped. Doody et al. (2003) studied the effects of the trench constructed for the 792 km

Eastern Gas Pipeline in southeast Australia, which traverses sclerophyll forests and heathlands, dry woodland, native grasslands, and improved pastures. Multiple visits to the trench throughout its length demonstrated the breadth of impacts of the trench. A total 7,438 animals of 103 species were captured from the trench (Doody et al. 2003). During 25 August – 19 September 2001, 298 reptiles of 16 species were trapped in a 302 km trench constructed near Albuquerque, New Mexico dug for the installation of a fiber optic cable (C.W. Painter *pers comm.*). Herein, we describe the effects of trenching for a natural gas pipeline on small vertebrates in the Mescalero-Monahans shinnery sands ecosystem of southeastern New Mexico.

Oil and gas extraction in the southwestern United States has resulted in fragmentation per se (*sensu*: Fahrig 2003) due to the creation of networks of roads, well pads and pipelines. In particular, the Mescalero-Monahans shinnery sands of southeast New Mexico and adjacent Texas is an ecosystem where oil and gas operations are prevalent and expanding (Laurencio and Fitzgerald 2010; Smolensky and Fitzgerald, in press). The landscape is characterized by shinnery oak sand dunes, which include expanses of vegetated dunes with open depressions, called blowouts, in a matrix of shinnery oak (*Quercus havardii*). The shinnery oak provides cover for fauna and is an integral part of the dune structure (Peterson and Boyd 1998, Hall and Goble 2006). Several endemic species occur in this ecosystem (Tinkham 1961, 1979, Hovore 1981, Degenhardt et al. 1996) including the dunes sagebrush lizard (*Sceloporus arenicolus*) an endemic habitat specialist that is only found in and around the blowouts (Fitzgerald and Painter 2009, Laurencio and Fitzgerald 2010).

In the summer of 2010, we surveyed a trench that was created for a pipeline in southeastern New Mexico. Our interests were: 1) to determine if *S. arenicolus* had been trapped by this trench; 2) to determine if other organisms were trapped; and 3) to remove all trapped individuals. The pipeline is approximately 65 km long, beginning from an area south of Maljamar, New Mexico to Artesia, New Mexico (east-end of trench line: 32° 48’ 59’’ N, 103° 48’ 21’’ W: datum WGS 84). The trench measured 1.5 m deep and 0.7 m wide. Between 24 and 31 July 2010, we surveyed the same 2 km section of the trench for approximately 45-90 minutes, at various times of the day, on each of the six visits.

We removed 24 individuals of 10 vertebrate species from the trench of which 4 were found dead (Table 1). All live animals were marked and released but none were re-captured. In addition, we encountered an ornate box turtle (*Terrapene ornata*) that had walked to the edge of the trench and turned around. We did not find any *S. arenicolus* in our surveys of this trench.

The following equations were used to estimate daily capture and mortality rates for the full length of the trench. We calculated daily capture rates (C_t) to be:

$$C_t = \left[\frac{c}{n}\Big/_{l}\right]$$

where c is the number of individuals captured, n is the number of days sampled, l is the distance (km) sampled. We calculated mortality rates (M_t) to be:

$$M_t = \left[\frac{m}{n}\Big/_{l}\right]$$

where m is the number of mortalities observed. We used these rates to estimate daily captures and mortalities for the total trench length, by multiplying by 65 km. The rates we calculated are conservative because we assumed a constant rate of capture and mortality for all species, and were unable to account for every organism that was captured by the

trench. Based on our survey we calculated $C_t = 2$ and $M_t = 0.33$. Presuming this rate of capture and mortality over the full length of the trench, we estimated the capture rate to be 130 animals captured/day and the mortality rate to be 21.6 deaths/day.

The 2 km of trench that we surveyed were entirely within the Mescalero-Monahans shinnery sands ecosystem; the remainder of the trench extended across a transition zone into Chihuahuan desert grasslands and Chihuahuan basins and playas. Although we assumed constant trap rates over the entire trench, it is possible that rates would vary in the two adjacent ecosystems. In addition, the organisms living in and near the trench exhibit a wide range of activity patterns and life histories, which should influence their susceptibility to falling into a trench. Our surveys were conducted at various times of day, and did not take into account daily activity patterns of animals and other factors, such as predators that also could remove animals from the trench. We did observe coyote (*Canis latrans*) tracks in and around the trench, and 2 active loggerhead shrike (*Lanius ludovicianus*) nests nearby, both of which are capable of preying on small vertebrates in the trench. The absence of *S. arenicolus* could be a result of them not being captured by the trench or their absence from the local lizard community. Yet, it should be noted that the location of this trench is near a known historic locality for *S. arenicolus* where they have not been captured in high abundances in recent surveys (Leavitt and Fitzgerald, data).

The effects of trenches on wildlife have been noticed by natural resource agencies. The New Mexico Department of Game and Fish, for example, developed recommendations intended to minimize the impact of trenching operations on native fauna from NMDGF (http://www.wildlife.state.nm.us/conservation/habitat_handbook/documents/TrenchingGuidelines.pdf). These guidelines recommend that operators:

> keep trenching and back-filling crews close together, to minimize the amount of open trenches at any given time. Trench during the cooler months (October – March). However, there may be exceptions (e.g., critical wintering areas) that need to be assessed on a site-specific basis. Avoid leaving trenches open overnight. Where trenches cannot be back-filled immediately, escape ramps should be constructed at least every 90 meters. Escape ramps can be short lateral trenches or wooden planks sloping to the surface. The slope should be less than 45 degrees (1:1). Trenches that have been left open overnight should be inspected and animals removed prior to backfilling, especially where endangered species occur (NMGFD 2003).

We observed trenching and back-filling crews working Monday through Friday only. The 2 km section of trench we surveyed was open for 51 days from 19 July to 7 September 2010. No escape ramps for wildlife were constructed in this section of trench. We did note there were 4 access ramps for people located at intersections with other pipelines, and some animals could have climbed out of the trench on these ramps.

Our analysis is the first we are aware of to estimate the impact of open trenches on fauna in southeastern New Mexico. As oil and gas operations are prevalent in ecosystems worldwide, there have been large scale efforts to reduce impacts of oil and gas development on biodiversity (Conservation International 2007). Towards this end, we

recommend NMDGF seek a mechanism to inform all parties of their trenching guidelines, receive acknowledgment that the guidelines are understood, and develop regulations that can be enforced.

ACKNOWLEDGEMENTS

The authors would like to thank J. R. Dixon, C. W. Painter and M. T. Hill for their comments on an earlier draft of this manuscript. The authors complied with all applicable institutional guidelines addressing animal care and use (approved protocol AUP 2008-95), and regulations set forth by the Bureau of Land Management and New Mexico Department of Game and Fish.

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Table 1 – Vertebrates captured while surveying an open trench in the Mescalero-Monahans Shinnery Sands of New Mexico.

Species	Captured	Found Dead
Reptiles		
Aspidoscelis marmorata	2	1
A. sexlineata	4	1
Phrynosoma cornutum	5	
Tantilla nigriceps	1	
Uta stansburiana	2	
Amphibians		
Anaxyrus cognatus	3	
Scaphiopus couchii	1	
Spea bombifrons	4	
Mammals		
Geomys knoxjonesii	1	1
Perognathus flavus	1	1

UPDATE ON THE ATLAS OF DISTRIBUTION AND HABITAT OF THE DUNES SAGEBRUSH LIZARD (*SCELOPORUS ARENICOLUS*) IN NEW MEXICO

A REPORT SUBMITTED TO THE BUREAU OF LAND MANAGEMENT, CARLSBAD OFFICE

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October 2011

The Atlas of Distribution and Habitat of the Dunes Sagebrush Lizard (*Sceloporus arenicolus*) in New Mexico has been published by the Texas Cooperative Wildlife Collection, Texas A&M University (Laurencio and Fitzgerald 2010). The atlas is a visual tool for understanding the geographic distribution of the Dunes Sagebrush Lizard (*Sceloporus arenicolus*) in the context of its unique and easily identifiable habitat, shinnery oak dune complexes. Because these features can be accurately perceived with remotely sensed imagery (Digital Orthophoto Quarter Quads), the atlas serves to identify potential and suitable habitat of *S. arenicolus*. The atlas illustrates the currently known extent, connectivity, and fragmentation of the species' habitat within New Mexico. It consists of text, 85 detailed maps covering the range of *Sceloporus arenicolus* in New Mexico, and an appendix of specimen information and coordinates for all localities.

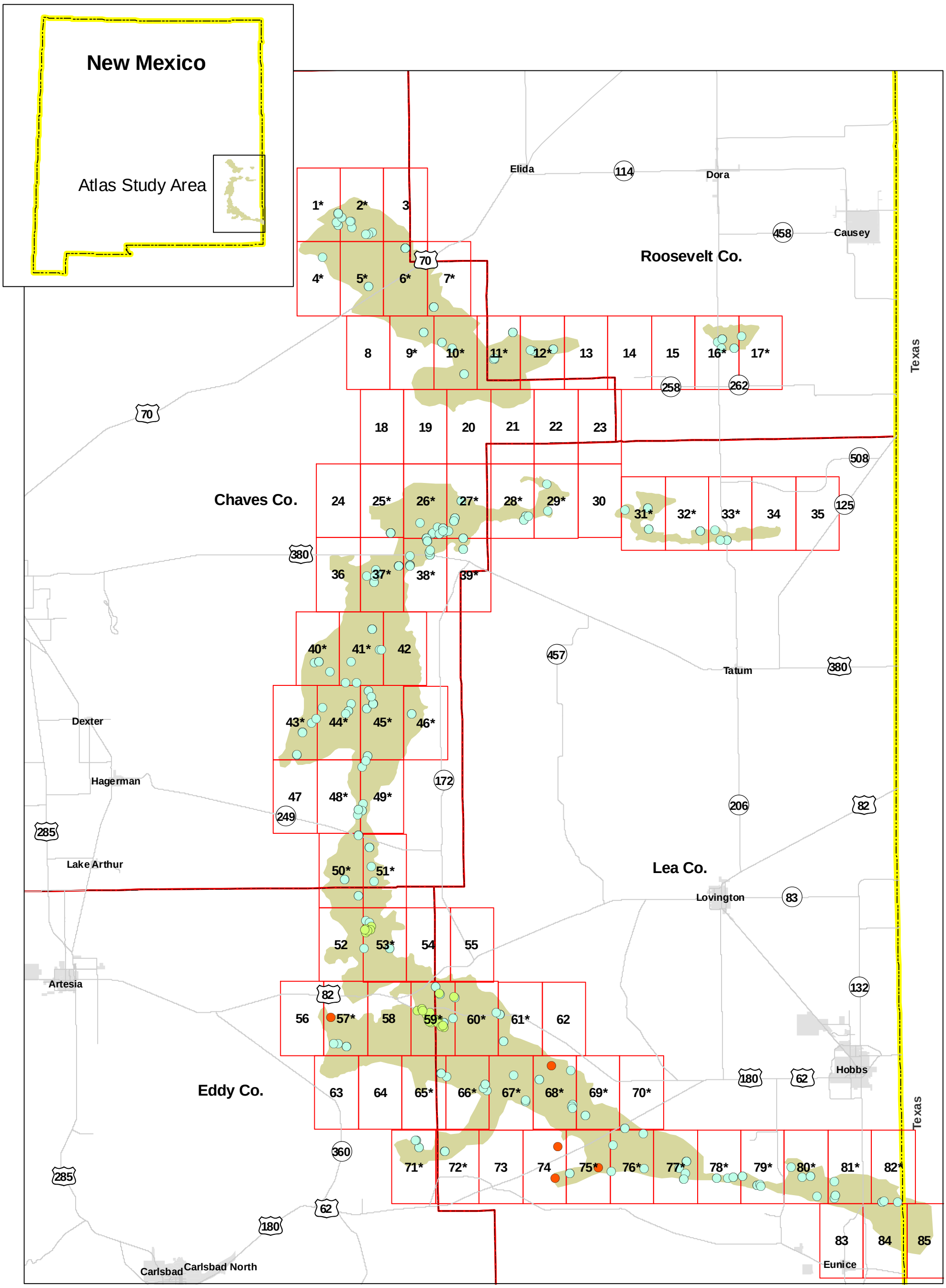
Printed and bound copies were distributed to several agencies, researchers, and both New Mexico University and Texas A&M University libraries in early February 2011. Additionally, both the pdf version of the atlas and the ArcGIS shapefile of the newly delineated range of *S. arenicolus* in New Mexico were made publicly available for download through Dr. Fitzgerald's website (http://herpetology.tamu.edu/Fitzgerald/arenicolus.html). As of October 1, 2011, the New Mexico atlas has been downloaded 102 times, and the distribution shapefile has been downloaded 46 times.

During Summer 2011, surveys conducted by Lee Fitzgerald and Dan Leavitt, utilizing the atlas to identify potential habitat, resulted in 5 new important localities. These localities fill in gaps between previously known *S. arenicolus* localities, with one of these localities falling outside of the known range. Additionally, Dan Leavitt provided coordinates corresponding to 23 trapping grids that have documented *S. arenicolus*

presence since 2009. This document includes an updated range map, illustrating the new localities, seven updated detailed atlas pages highlighting the newly obtained *S. arenicolus* localities, and a legend corresponding to the detailed atlas pages.

ATLAS CITATION:

Laurencio, Laura R. and Lee A. Fitzgerald. 2010. *Atlas of distribution and habitat of the dunes sagebrush lizard (Sceloporus arenicolus) in New Mexico*. Texas Cooperative Wildlife Collection, Department of Wildlife and Fisheries Sciences, Texas A&M University, College Station, TX 77843-2258. ISBN# 978-0-615-40937-5.


New Mexico
Atlas Study Area
Roosevelt Co.
Chaves Co.
Eddy Co.
Lea Co.
Elida
Dora
Causey
Tatum
Dexter
Hagerman
Lake Arthur
Artesia
Carlsbad
Carlsbad North
Lovington
Hobbs
Eunice
Texas
Texas
1* 2* 3 4* 5* 6* 7* 8 9* 10* 11* 12* 13 14 15 16* 17*
18 19 20 21 22 23 24 25* 26* 27* 28* 29* 30 31* 32* 33* 34 35
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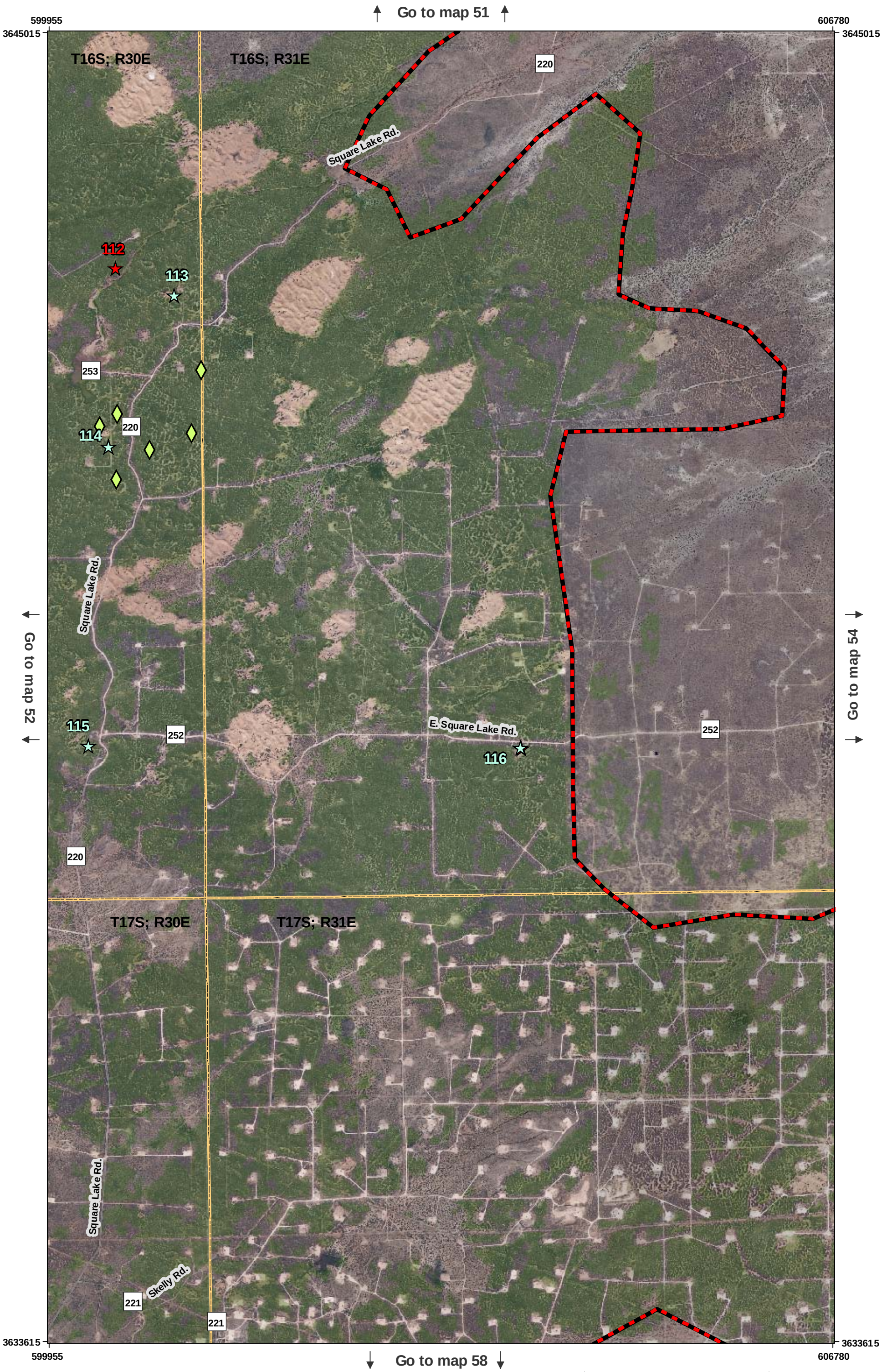
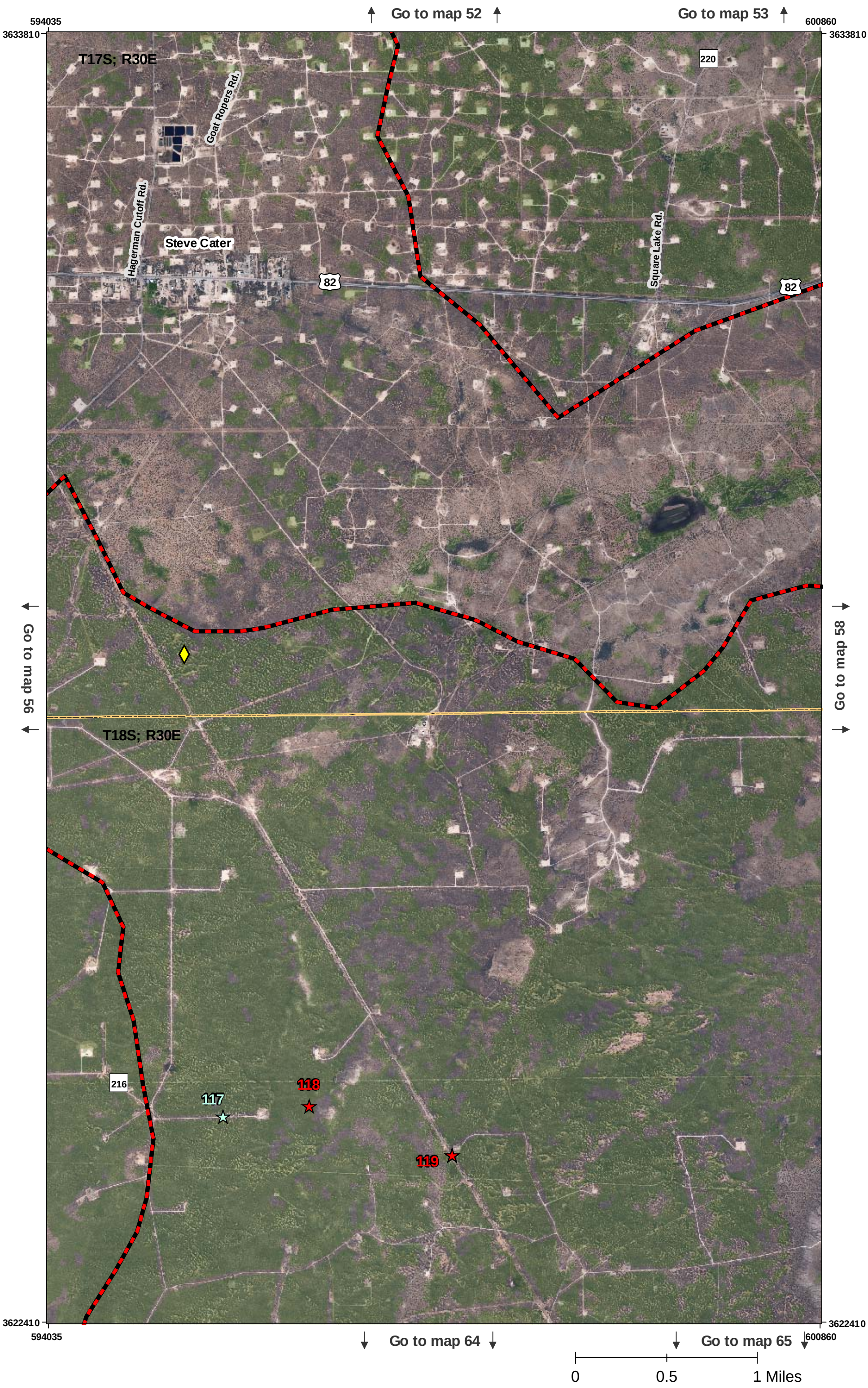
71* 72* 73 74 75* 76* 77* 78* 79* 80* 81* 82* 83 84 85
0 5 10 Miles
Legend
Dunes Sagebrush Lizard distribution
Dunes Sagebrush Lizard localities - Dan Leavitt's study sites
Dunes Sagebrush Lizard localities - June 2011
Dunes Sagebrush Lizard localities in atlas
State boundary
County boundary
State highway
U.S. highway

Key to Features

- *Sceloporus arenicolus* presence verified in June 2011
- *Sceloporus arenicolus* presence verified since 2011 (Dan Leavitt's study plots)
- *Sceloporus arenicolus* historical species occurrences
- *Sceloporus arenicolus* presence verified since 2005
- 55 U.S. route
- 55 State route
- 55 County road
- Road Road
- City Populated area
- Township boundary
- Distribution boundary
- Shinnery Oak dominated shrubland and duneland, classified by Natural Heritage New Mexico



Note: Vegetation classification unavailable for northeastern portion of map.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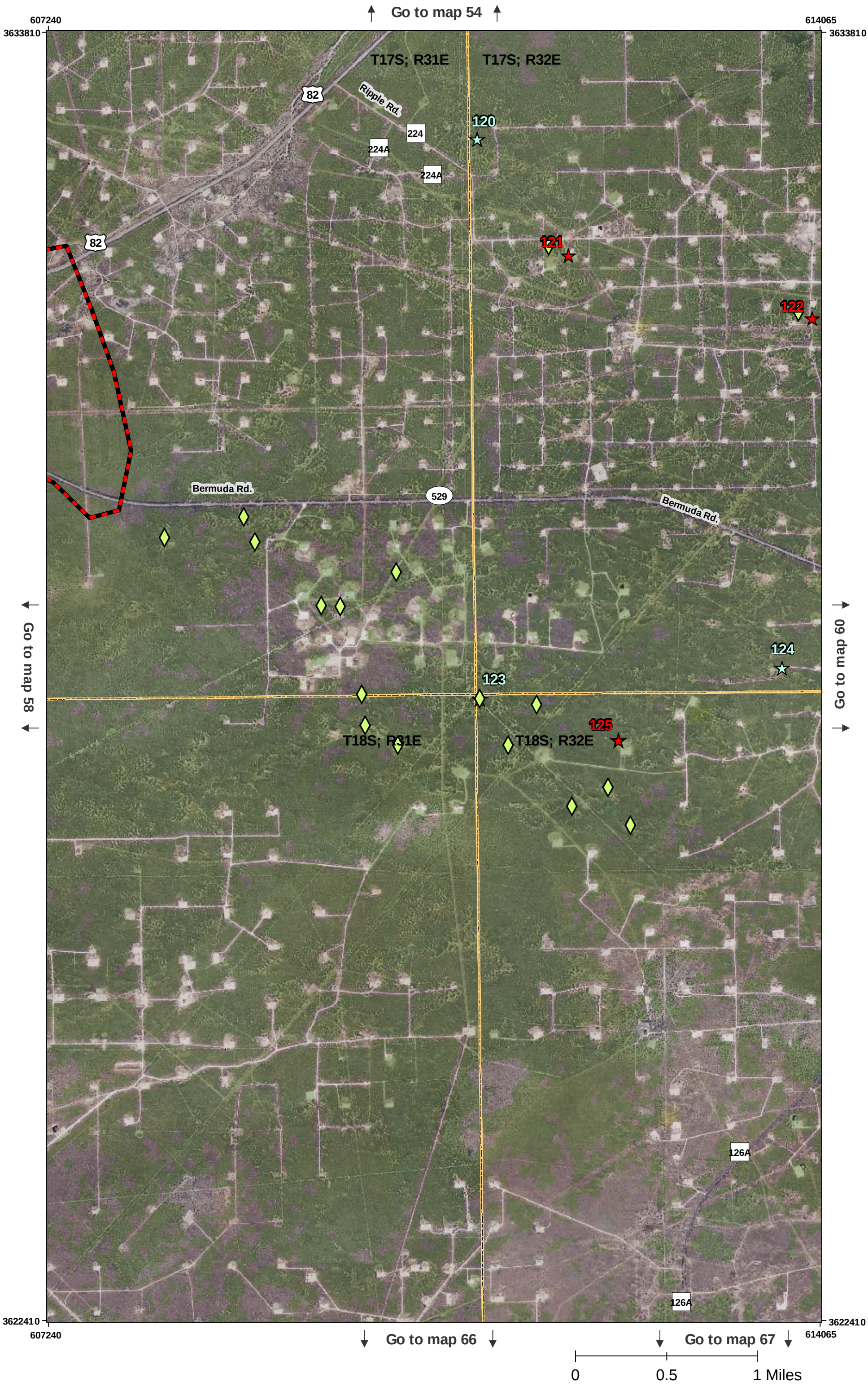
Go to map 53
594035
3633810
600860
3633810
T17S; R30E
220
Goat Ropers Rd.
Hagerman Cutoff Rd.
Steve Cater
82
Square Lake Rd.
82
Go to map 56
Go to map 58
T18S; R30E
216
117
118
119
3622410
594035
3622410
600860
Go to map 64
Go to map 65
0
0.5
1 Miles


Go to map 54
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3633810
614065
3633810
T17S; R31E
T17S; R32E
82
Ripple Rd.
224
224A
224A
120
82
121
122
Bermuda Rd.
529
Bermuda Rd.
Go to map 58
Go to map 60
124
123
125
T18S; R31E
T18S; R32E
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126A
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614065
Go to map 66
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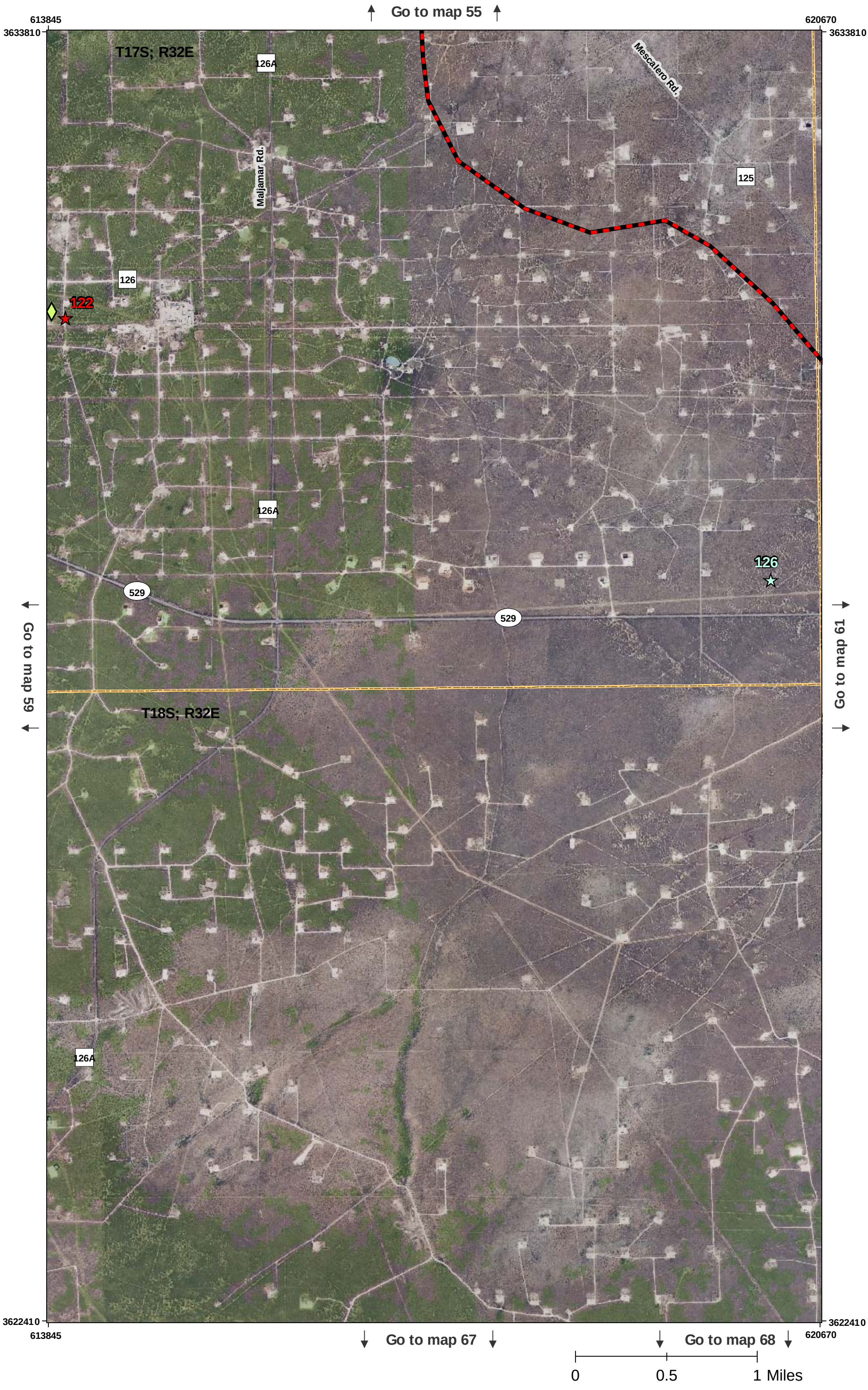
1 Miles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613845
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T17S; R32E
126A
Maljamar Rd.
Mescalero Rd.
125
126
122
126A
529
529
126
Go to map 59
Go to map 61
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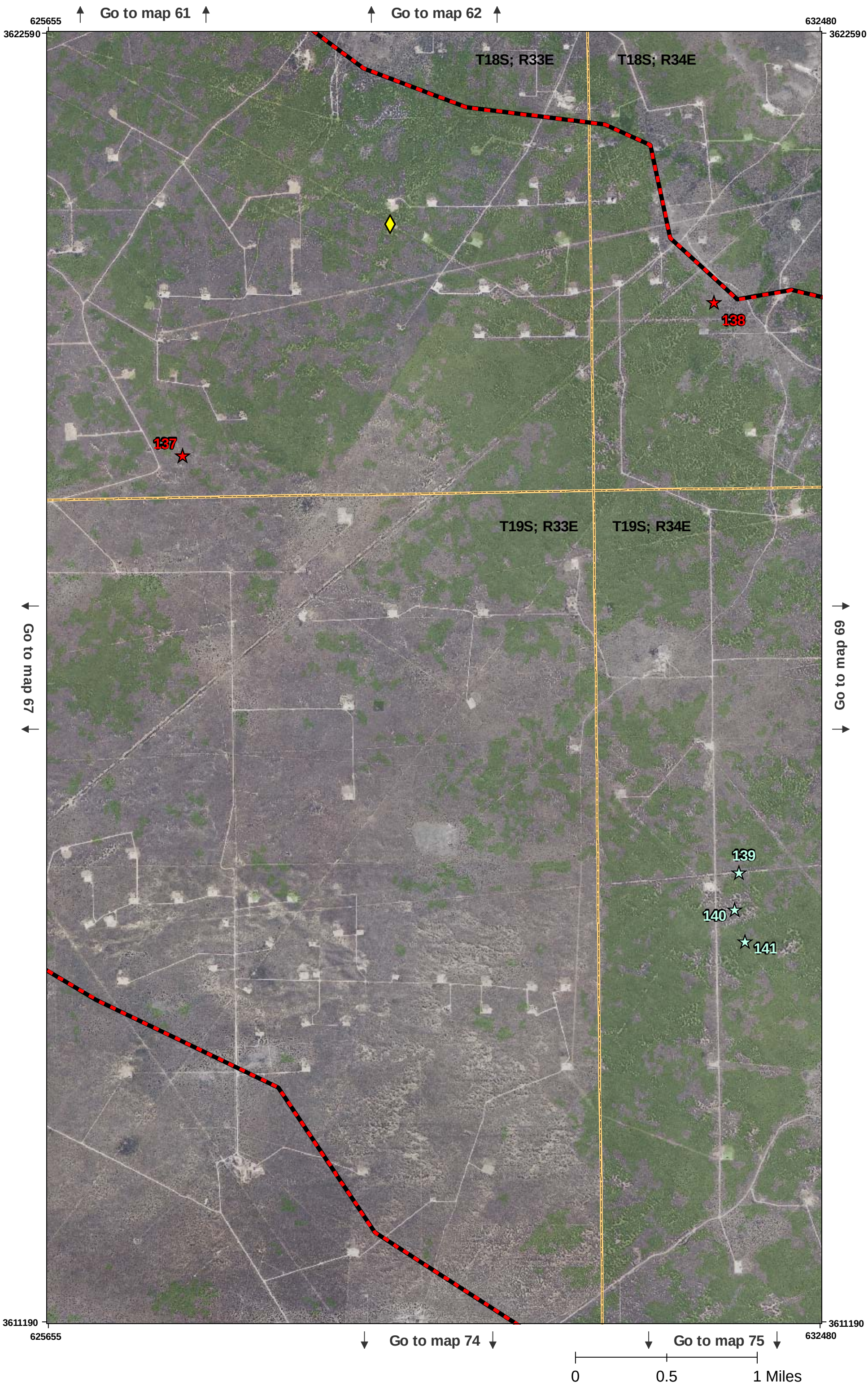
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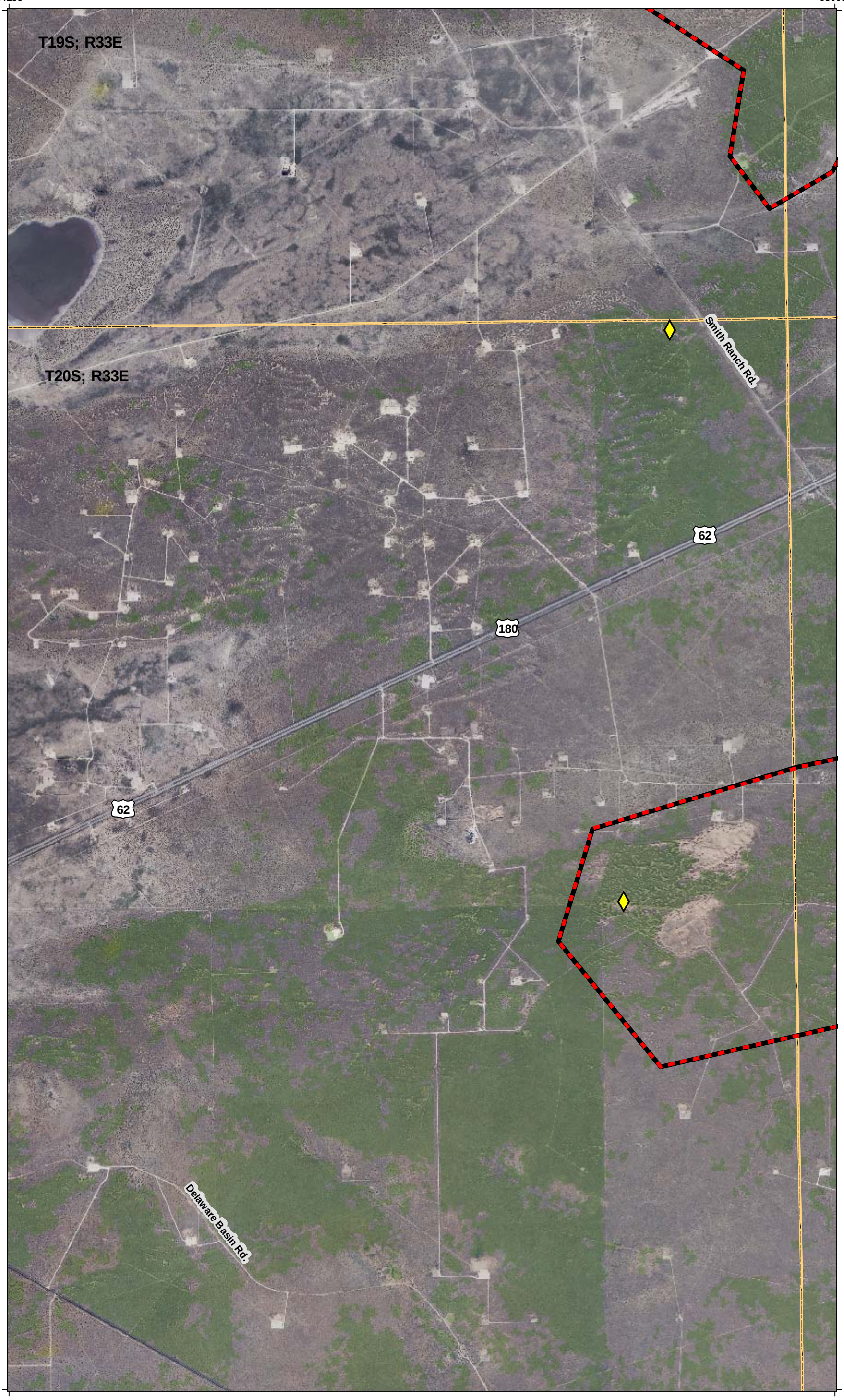

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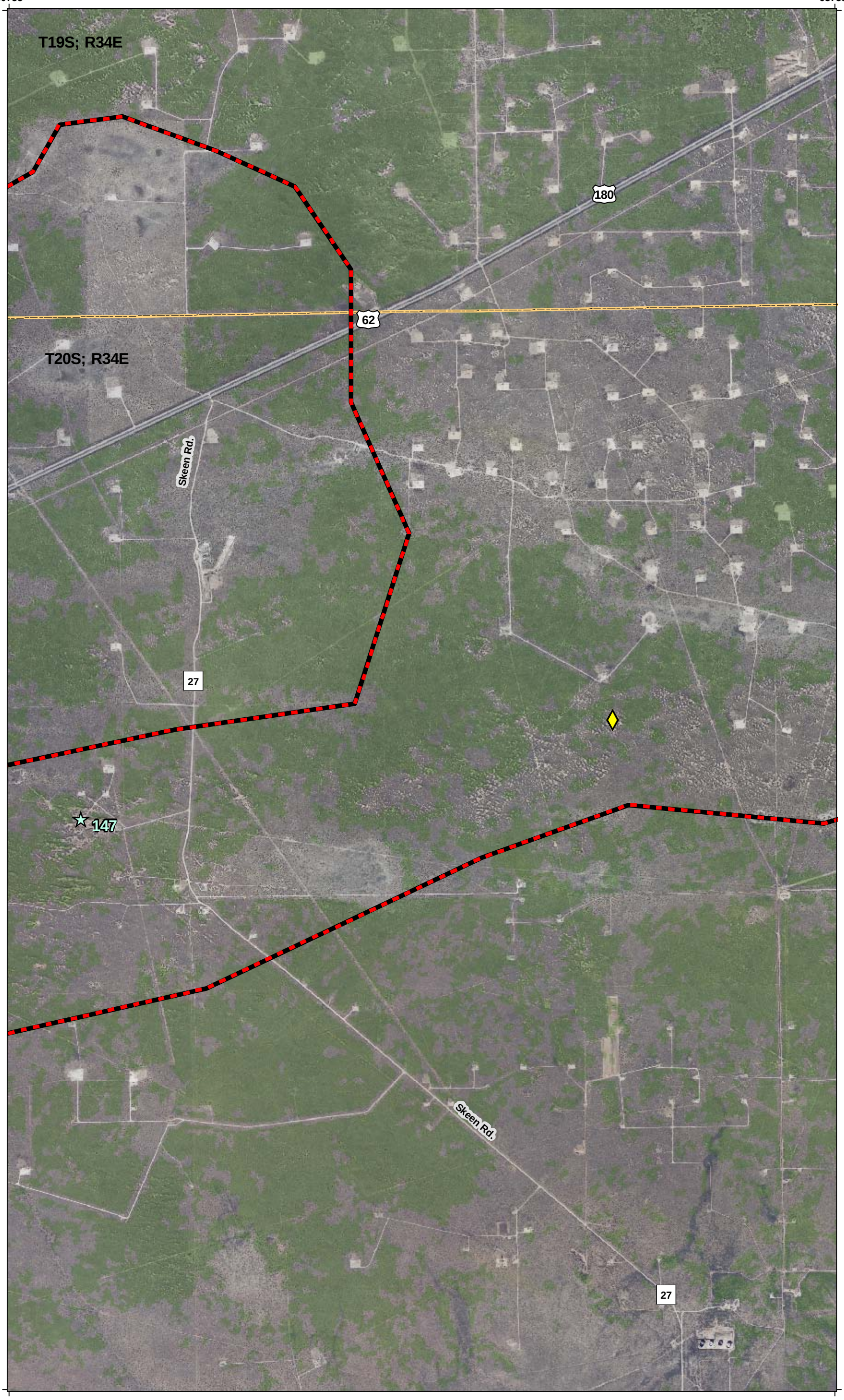

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EXPLORATION OF GENETIC DIFFERENTIATION IN THE DUNES SAGEBRUSH-LIZARD (*SCELOPORUS ARENICOLUS*) ACROSS MULTIPLE SPATIAL AND TEMPORAL SCALES

A REPORT SUBMITTED TO THE BUREAU OF LAND MANAGEMENT, CARLSBAD OFFICE

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October, 2011

OVERVIEW

We are examining historical and contemporary population genetic structure in the Dunes Sagebrush lizard in an effort to accurately characterize patterns of divergence and diversification in this species as they relate to habitat alteration and evolutionary history. This study is hierarchical in nature, looking at "natural" breaks in the population connectivity of *Sceloporus arenicolus* throughout their entire range as well as at the role of landscape features at fine scales (e.g. roads, caliche, sand dune blowout characteristics) on population connectivity and population persistence.

We have collected over 300 tissue samples for *S. arenicolus* from nearly all known localities in the Mescalero and Monahans shinnery sands ecosystem (Figure 1). For all tissue samples, we are obtaining DNA sequence data at two mitochondrial and two nuclear gene regions. These data allow us to estimate range-wide differentiation in *S. arenicolus* and determine the degree to which populations have been isolated from one another across geological and evolutionary time scales.

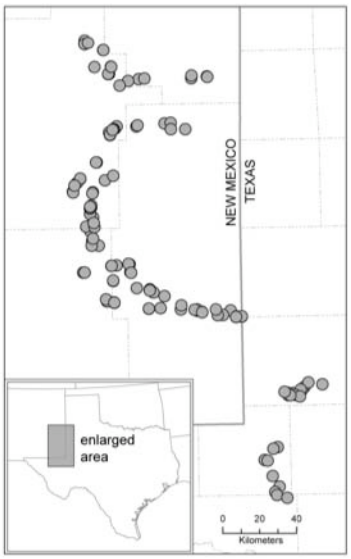

Figure 0: Map of sampled localities throughout the range of the Dunes Sagebrush Lizard.

Our preliminary analyses of DNA sequence data indicate that patterns of isolation and historical demographics vary spatially across the shinnery sand ecosystem. At the northern portion of the range we find strong population structure with little connectivity among four areas suggesting long-term isolation. In the southern extent of their distribution within New Mexico, there is low genetic diversity and also low genetic divergence among regions suggesting that historical populations may

have been small, but well-connected, or that these populations recently experienced a population bottleneck reducing overall diversity. Interestingly, these data also suggest that populations in Texas may be particularly diverse. Completion of this multi-locus genetic dataset will add resolution to these preliminary conclusions, adding to our understanding of divergence and diversity in *S. arenicolus*.

At finer spatial scales, we are investigating how human-altered aspects of the landscape influence population health and population connectivity. Tissue samples corresponding to over 1000 marked individuals of three species *(Uta stansburiana*, *Aspidoscelis marmorata*, and *Sceloporus arenicolus*) from twenty-seven pitfall arrays monitored over three years have been collected (Leavitt et al. 2011). These arrays are in three different treatment types: 1) non-fragmented Shinnery Oak sand-dune habitat, 2) habitat fragmented by oil and gas development, and 3) before experimental sites monitored before (pre-2010) and after (post-2011) oil and gas development (see Figure 1 in Leavitt et al. 2011). Microsatellite markers are currently being used to assess population genetic diversity in each of these three species within and among regions with respect to habitat features such as roads, caliche wellpads, and topography. Comparing across three species allows us to investigate the interaction between ecological specialization and landscape ecology and better understand how anthropogenic changes affect not only *S. arenicolus*, but other members of the community as well.

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EFFECTS OF MANAGEMENT PRACTICES FOR OIL AND GAS DEVELOPMENT ON POPULATIONS OF THE DUNES SAGEBRUSH LIZARD (*SCELOPORUS ARENICOLUS*) IN TEXAS

A REPORT SUBMITTED TO THE BUREAU OF LAND MANAGEMENT, CARLSBAD OFFICE

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Research proposed for Texas populations of *S. arenicolus* builds upon previous *S. arenicolus* research conducted in NM, much of which was funded by BLM. Proven methodologies from the research program in New Mexico have already provided insights into landscape-scale patterns of geographic distribution (Fitzgerald et al. 1997, Laurencio and Fitzgerald 2010), population estimation (Smolensky and Fitzgerald 2010, Ryberg and Fitzgerald 2011a), population genetics (Chan et al. 2009), habitat specificity and movements of gravid females (Fitzgerald et al. 1997, Hill and Fitzgerald 2007), and the effects of oil and gas development on lizard detectability (Sias and Snell 1998, Smolensky and Fitzgerald 2011).

Below we describe our priorities for future research that could take place in Texas and that would contribute to an overall research program for *S. arenicolus* throughout the species' range.

OVERVIEW

Sceloporus arenicolus is a habitat specialist restricted to Shinnery Oak sand-dune habitats that are comprised of sand-dunes stabilized by Shinnery Oak and wind-hollowed, open sandy depressions called blowouts. The configuration of sand-dunes and blowouts within this habitat arises from a dynamic interaction between wind, sand, and Shinnery Oak (*Quercus havardi*) creating the unique dune-blowout landform (Fitzgerald and Painter 2009). *Sceloporus arenicolus* occurs only in direct association with the blowout landform in this habitat, and recent research shows that subtle differences in amount, quality, and configuration of blowouts can have dramatic consequences for lizard population dynamics (Smolensky and Fitzgerald 2011, Ryberg and Fitzgerald 2011a). Any factors disturbing this landform within Shinnery Oak sand-dune habitats can therefore be considered threats to *S. arenicolus* population persistence.

Construction of networks of caliche roads and well pads for oil and gas development within the Shinnery Oak sand-dune landscape results in the loss and fragmentation of *S. arenicolus* habitat. Previous and ongoing research on *S. arenicolus* have identified potential

correlates between oil and gas development and lizard abundance. Using data from visual transect surveys and measurements of oil pad density, Sias and Snell (1996) found a significant, negative correlation between lizard abundance and oil pad density. Additionally, Smolensky and Fitzgerald (2011) identified a positive association between lizard abundances and the amount or extent of blowouts within the surrounding habitat, two landscape elements known to decrease with oil and gas development. Finally, in an ongoing mark-recapture study Leavitt et al. (2011a) found significantly fewer *S. arenicolus* in areas fragmented by networks of caliche roads and well pads versus areas that are not fragmented.

Despite these observed relationships between oil and gas development and *S. arenicolus* abundances, the processes by which such disturbed habitats lead to variation in lizard population dynamics and the scale at which those processes occur are not completely understood. As such, the primary objective of this research is to identify the processes by which road and well pad construction associated with oil and gas development may affect the population dynamics and persistence of *S. arenicolus* at multiple spatial and temporal scales in Texas.

To accomplish this objective, we propose a 3-part, multi-scale research program designed to simultaneously quantify the effects of oil and gas development on (1) *S. arenicolus* behavior and movement, (2) population dynamics, and (3) landscape-scale modeling of habitat suitability, connectivity, and fragmentation. Specifically, the TX study will nest fine-scale behavioral and dispersal studies within multi-year, mark-recapture, demographic studies that are replicated at sites with varying levels of oil and gas development. Information gathered from this behavioral and demographic research will be used to construct an individual-based population model that tests and evaluates how different land use scenarios associated with oil and gas development alter the suitability and connectivity of *S. arenicolus* habitats and therefore the abundance and distribution of lizards across the Shinnery Oak sand-dune landscape.

The research described here was devised in collaboration with Dr. Toby Hibbitts at Texas A&M University.

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THE SOUTHWESTERN NATURALIST 56(3):315–324 SEPTEMBER 2011

POPULATION VARIATION IN DUNE-DWELLING LIZARDS IN RESPONSE TO PATCH SIZE, PATCH QUALITY, AND OIL AND GAS DEVELOPMENT

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ABSTRACT—We studied relationships between quality and quantity of habitat and conversion of land to caliche roads and well pads associated with oil and gas development. We asked how these factors affected abundance of dune-dwelling lizards, with emphasis on a habitat specialist, the dunes sagebrush lizard *Sceloporus arenicolus*. Open depressions in dune complexes are a critical landscape feature for *S. arenicolus*, and extensively used by all species; thus, size and total area of open depressions in a study site were our measures of habitat quality and quantity. There were significant differences in habitat quality among sites, and habitat quality and quantity were correlated significantly. Abundances of all lizards, including *S. arenicolus*, varied significantly among sites and this variation could be explained by amount of habitat at a given site. Relationships between oil and gas development, quantity and quality of habitat, and abundances of lizards likely occur on different spatial scales constraining our ability to detect direct effects of oil and gas development alone. Our research is the first to investigate effects of oil and gas development on an assemblage of dune-dwelling lizards.

RESUMEN—Se estudiaron las relaciones entre la calidad y la cantidad de hábitat y la conversión de hábitat a caminos y plateas de caliche para pozos petroleros asociados al desarrollo petrolero. Preguntamos cómo dichos factores afectan la abundancia de lagartijas que habitan dunas, con énfasis en una especie especialista *Sceloporus arenicolus*. Las depresiones abiertas en los complejos de dunas de arena son una característica importante del paisaje para *S. arenicolus* y son usadas mucho por todas las especies, entonces el tamaño y área total de estas depresiones en un sitio fueron nuestras medidas de calidad y cantidad de hábitat. Se encontraron diferencias significativas en la calidad de hábitat entre los sitios, y la calidad y cantidad de hábitat estuvieron significativamente correlacionadas entre sí. Las abundancias de lagartijas de todas las especies, inclusive *S. arenicolus*, variaron significativamente entre sitios, y tal variación podría estar explicada por la cantidad de hábitat en un sitio determinado. Las relaciones entre el desarrollo petrolero, la calidad y cantidad de hábitat, y las abundancias de lagartijas probablemente ocurren a distintas escalas espaciales, limitando nuestra capacidad de detectar los efectos directos de desarrollo petrolero exclusivamente. Este trabajo es el primero en investigar algunos efectos del desarrollo petrolero en un ensamblaje de lagartijas de dunas.

The association between availability of habitat and persistence of species is clear, and recent work elucidates how cumulative loss of habitat at large spatial scales erodes capacity of the landscape to sustain biodiversity (Pope et al., 2000; Gaston et al., 2003). Farming croplands, extraction of timber, urban development, and construction of roads all contribute to loss of habitat and result in variable configuration of remaining patches of habitat and associated effects on biodiversity (Andren, 1994; Hokit et al., 1999; Fahrig, 2003). For example, with agricultural or urban development, large tracts of land may be converted leading to a few patches of habitat that are relatively isolated. Habitat also is lost to road construction or timber extraction, but the resulting fragmentation may lead to an increase in number of patches and variation in size and isolation of patches. Thus, it is important to understand how land-management practices influence availability and quality of habitats across landscapes.

Oil and gas exploration and development began in the 1800s and occurs on large spatial scales spanning hundreds of thousands of hectares (Bureau of Land Management, http://www.blm.gov/nm/st/en/prog/energy/oil_and_gas/lease_sale_notices.html). Yet there are few stud-

ies that discuss loss of habitat, scarring of land, and fragmentation caused by this type of land management (Fiori and Zalba, 2003; Schneider et al., 2003; Linke et al., 2005, 2008). During oil and gas development in the Southwest, natural habitat is lost and fragmented by networks of caliche (decomposed limestone) roads and well pads. The pattern of landscape fragmentation from oil and gas development is distinct from that caused by farming croplands, clear-cutting, and cattle grazing, and its effects on different forms of biodiversity are even less understood. Consequently, investigating responses of species to oil and gas development is warranted.

Among squamate reptiles, various forms of habitat loss and fragmentation are considered to be primary drivers of declines in populations (Shine, 1991; Gibbons et al., 2000; Collins and Storfer, 2003; Gardner, 2007). Although farming croplands, deforestation, and cattle grazing were discussed in these reviews, none addressed the potential effects of oil and gas development on reptiles. Possible direct and indirect effects of oil and gas development have been documented on other species of invertebrates and vertebrates. For example, changes in community structure of ground beetles (Carabidae) were attributed to fragmentation of forest by installation of an oil pipeline (Silverman et al., 2008). Birds have been trapped in oil pits, causing mortalities numbering in the thousands (Trail, 2006). Initiation of nesting by sage grouse (*Centrocercus urophasianus*) was reduced due to disturbance from oil wells and activity on roads (Lyon and Anderson, 2003), and mule deer (*Odocoileus hemionus*) shifted selection of habitat as a result of oil and gas development (Sawyer et al., 2006, 2009). Although it is intuitive that oil and gas development will affect reptiles, it also is apparent that species vary in response to change in landscape (Andren, 1992; Bender et al., 1998; Hokit et al., 1999; Vega et al., 2000). Herein, we investigate the response of an assemblage of lizards to various levels of oil and gas development. The goal of our research was to gain insight into thresholds of landscape degradation that elicit responses from habitat specialists versus from an entire assemblage of lizards.

The Mescalero Sands ecosystem of southeastern New Mexico and adjacent Texas supports an assemblage of lizards consisting of generalists and specialists that occupy a sand-dune system that is semi-stabilized by shinnery-oak (*Quercus havardii*). This ecosystem is best described as a restricted system of dune complexes that vary in size and connectivity. Dune complexes themselves are patchy, consisting of open sand-dune depressions, called blowouts, in a matrix of shinnery-oak. Size and density of blowouts vary as does topographic complexity of the dunes. Among the seven species of lizards that inhabit the shinnery-oak and sand-dune habitat is the endemic habitat specialist *Sceloporus arenicolus* (dunes sagebrush lizard). *Sceloporus arenicolus* has a restricted range (Degenhardt et al., 1996; Fitzgerald and Painter, 2009; Laurencio and Fitzgerald, 2010) and is a candidate for listing as endangered or threatened (United States Fish and Wildlife Service, 2009). This species occurs exclusively in and around sand-dune blowouts and it prefers relatively large blowouts based on availability in the landscape (Chan et al., 2009; Fitzgerald and Painter, 2009). Other species of lizards use both sand-dune blowouts and the surrounding matrix of shinnery-oak. Conservation status of *S. arenicolus* has drawn attention of natural-resource agencies, the oil and gas industry, conservation organizations, private landowners, and other stakeholders. 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.

The patchy nature of this shinnery-oak-sand-dune habitat, the species of lizards that use it, and the ongoing development of oil and gas resources within the Mescalero Sands ecosystem present an excellent system for studying relationships between quality and quantity of habitat patches, abundance of lizards, and effects of land conversion to caliche roads and well pads on both habitat and abundance of lizards. As such, we asked, what is the association between patch size (blowouts) and total patch area on abundance of lizards, especially for the specialist *S. arenicolus*, and what are the effects of networks of caliche roads on sand-dune blowouts? Blowouts represent important patches of habitat in the shinnery-oak, sand-dune matrix that are used by various dune-dwelling lizards, and size of individual blowouts (patch size) is a measure of habitat quality. Based on occupation by *S. arenicolus*, dune complexes with a high density of large, deep, sand-dune blowouts is presumed superior habitat to areas with small numbers of small blowouts. In addition to loss of habitat due

to outright conversion of the surface from natural habitat to packed caliche, the network of roads and well pads may also influence quality of remaining habitat by disrupting geomorphologic processes that create and maintain blowouts. For example, after fragmentation, remaining blowouts may be smaller in size and, thus, poorer quality for lizards. We predicted that abundance of lizards would increase with greater numbers of sand-dune blowouts and larger size of blowouts. We also predicted a decrease in number of lizards at sites with more oil and gas development. We tested these predictions by counting lizards and quantifying variation in size of individual blowouts and total area of blowouts at multiple study sites with varying amounts of caliche roads and well pads, then correlating number of lizards to quality and quantity of blowouts, and to amount of caliche. Because oil and gas development may affect *S. arenicolus* and other species of lizards that occupy the Mescalero Sands ecosystem, results of this research are directly applicable to land management and conservation (Mac Nally et al., 2002). This study is the first to provide insight into effects of alteration of habitat associated with oil and gas development on herpetofauna.

MATERIALS AND METHODS—Study sites in the Mescalero Sands ecosystem were in Chaves, Eddy, and Lea counties, New Mexico. This ecosystem is characterized by stabilized and semi-stabilized dunes interspersed with shinnery-oak, sand sagebrush (*Artemisia filifolia*), bunchgrasses (*Aristida, Schizachyrium, Andropogon*), and sandy hummocks with honey mesquites (*Prosopis glandulosa*). We quantified abundance of lizards at 11 sites (Fig. 1) based on presence of shinnery-oak-sand-dune habitat, presence of *S. arenicolus*, and amount of oil and gas development.

Our indicator of oil and gas development on the landscape was total surface area of caliche, which was the total area of oil-well pads and roads in a 259-ha area of shinnery-oak-sand-dune habitat surrounding transects at each study site. This spatial scale was large enough to include caliche-covered well pads and roads that immediately surrounded transects. We used ArcMap, version 9.0 (Environmental Systems Research Institute, Inc., Redlands, California) to quantify total surface area of caliche. The New Mexico State Land Office (in litt.) provided GIS data that included locations of oil wells and roads. Size of well pads was standardized at 6,400 m^2 and roads were standardized at 4 m wide. The 4-m width of roads was conservative based on guidelines for construction of caliche roads suggested by the New Mexico Commission of Public Records (http://www.nmcpr.state.nm.us/NMAC/parts/title19/19.002.0020.htm) and the Chaves County Commission (http://co.chaves.nm.us/agendas/2006/101906/101906-A3.pdf m).

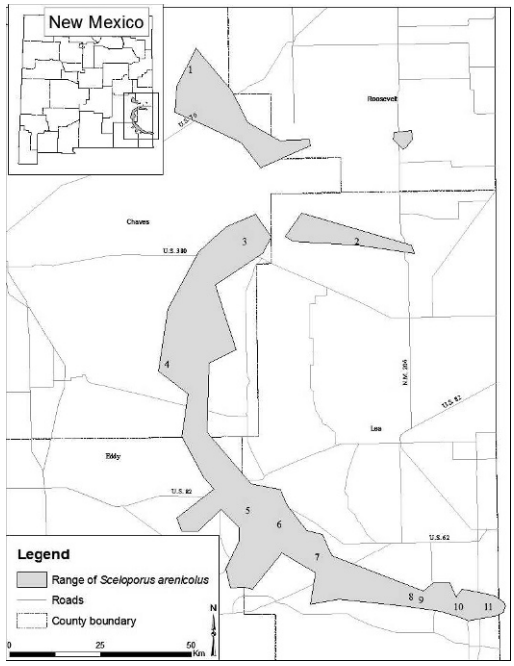

FIG. 1—Geographic range of the dunes sagebrush lizard *Sceloporus arenicolus* in New Mexico. Also shown are locations of 11 sites where survey transects were conducted and where habitat variables were quantified in the Mescalero Sands ecosystem of southeastern New Mexico during May–July 2005 and 2006.

Total area of blowouts at sites was our measure of quantity of habitat because blowouts are integral to the shinnery-oak-sand-dune habitat, they are used by multiple species of lizards, they are a critical feature of habitat for *S. arenicolus*, and they provide thermoregulatory sites for other species of lizards (Sartorius et al., 2002). *Sceloporus arenicolus* inhabits larger blowouts based on availability in occupied habitat; thus, larger blowouts are considered to be better quality of habitat than smaller blowouts. Many small blowouts can provide the same total area of blowouts as a few large blowouts; for this reason, quantity of habitat does not necessarily equal quality of habitat. We measured area of each blowout within the 11,259-ha sites in ArcMap 9.0 (Environmental Systems Research Institute, Inc., Redlands, California) to determine average size of blowout available to dune-dwelling lizards. A polygon shape file of all blowouts was created from aerial photographs taken in 2004 and obtained from the New Mexico State Land Office (in litt.). Mean size of blowout for the entire site was our index of site-wide quality of habitat.

We quantified encounters of lizards per unit effort by time from line transects conducted in May–July 2005 and 2006. Number of transects at a site was 8–48 due to availability of personnel. We standardized data on abundance of lizards by converting raw counts to encounters per unit effort for each transect and for all

transects at a site (site-wide encounters of lizards per unit effort). To compute site-wide encounters of lizards per unit effort we divided total minutes of searching effort by total number of lizards counted at a site. All transects were 25 min in duration. Transects were standardized by time because in studies of lizards, the correlation between number of individuals seen and time spent searching might be more meaningful than number of individuals seen over distance. All transects were surveyed during 0800–1300 h. Transects were not surveyed during rain or when substrate temperature was (<20 or >50°C). Transects were located randomly within shinnery-oak-sand-dune habitat after presence of *S. arenicolus* was verified. We did not consider other types of habitats, because *S. arenicolus* does not use them. As such, our results only apply to effects of oil and gas development on lizards in the shinnery-oak-sand-dune habitat.

We used a Kruskall-Wallis test to search for differences in quality of habitat (size of blowout) among sites, with size of individual blowouts as the sample unit. This was followed by a Nemenyi test (nonparametric Tukey test) for post-hoc multiple comparisons, with the standard error adjusted for unequal samples (Zar, 1999). Quality and quantity of habitat may be correlated and consequently confound our conclusions regarding the relationship between total area of blowouts, mean size of blowout, total surface area of caliche, and abundance of lizards. Thus, we used linear regression to test for a relationship between mean size of blowout (site-wide quality of habitat) and total area of blowouts (quantity of habitat), with total area of blowouts as the independent variable.

We analyzed pooled encounters of lizards per unit effort for all lizards combined, and separately for *S. arenicolus*. We used analysis of variance and a Kruskal-Wallis test to compare encounters of lizards per unit effort collectively and of *S. arenicolus* among sites. Encounters of lizards per unit effort for each transect was the sample unit in these tests. We used a Tukey test and an adjusted Nemenyi test for post-hoc multiple comparisons.

We computed linear regressions on site-wide encounters of lizards per unit effort against total surface area of caliche, total area of blowouts, and mean size of blowout for *S. arenicolus* and lizards collectively to test the null hypotheses of no relationship between abundance of lizards and oil and gas development, quantity of habitat, and quality of habitat. When necessary, data for encounters of lizards per unit effort were log-transformed prior to analysis to homogenize variances of groups and to meet assumptions of normality (Zar, 1999).

RESULTS—Range for total surface area of caliche among the 11 sites was 1.32–23.88 ha (mean = 9.23, *SD* = 8.24, $n = 11$; Table 1). A site that contained 23.88 ha of total surface area of caliche had ≤9% of the area comprised of oil and gas development. Range for total area of blowouts among the 11 sites was 14.47–50.82 ha (mean = 31.31, *SD* = 10.50, $n = 11$). A site that contained ≤50.82 ha of total area of blowouts

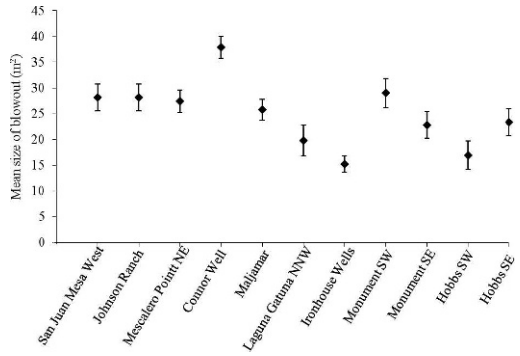

FIG. 2—Quality of habitat measured by mean size of blowout at 11 sites within shinnery-oak and sand-dune habitat in the Mescalero Sands ecosystem of southeastern New Mexico. Error bars indicate ±1 *SE*.

translated to 10% of the area comprised of blowouts. The rest of the area within our 259-ha sites was shinnery-oak vegetation. There were significant differences in quality of habitat (size of blowout) among sites ($\chi^2_{10} = 3{,}348.39$, $P < 0.01$; Fig. 2). Quality and quantity of habitat were significantly and positively correlated. Of the variation in quality of habitat among sites, 41% was explained by quantity of habitat (total area of blowout; $R^2 = 0.41$, $P < 0.03$; Fig. 3), suggesting that quality of habitat is linked to amount of habitat. The prediction that oil and gas development should be associated with fewer and smaller sand-dune blowouts was not supported. There was no significant correlation between total surface area of caliche and total area of blowouts nor between total surface area of caliche and mean size of blowout ($r = -0.32$, $P < 0.34$; $r = -0.08$, $P < 0.82$). All sites were chosen because they were in shinnery-oak-sand-dune habitat, and sites with the most area converted to packed caliche still had moderate (10%) total area of blowouts (Table 1).

We encountered 1,321 lizards (0.232 lizards/min) of seven species on 227 transects at the 11 sites. The Texas horned lizard *Phyrnosoma cornutum* was seen only on two transects; thus, it was excluded from analyses. These six species made up the collective dataset; the side-blotched lizard *Uta stansburiana* (0.081 lizards/min) was the most frequently detected followed by *S. arenicolus* (0.046 lizards/min), the marbled whiptail *Aspidoscelis marmoratus* (0.036 lizards/min), the lesser earless lizard *Holbrookia maculata* (0.016 lizards/min), the six-lined race runner *Aspidoscelis*

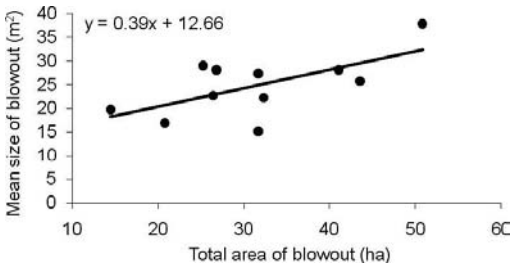

FIG. 3—Results of a linear regression showing the relationship between mean size of blowout and total area of blowout among 11 sites in the Mescalero Sands ecosystem of southeastern New Mexico.

sexlineatus (0.009 lizards/min), and the prairie lizard *Sceloporus consobrinus* (0.006 lizards/min). Mean encounters of lizards per unit effort varied significantly among sites for lizards collectively and nearly so for *S. arenicolus* ($F_{10,215} = 8.78$, $P < 0.01$; $\chi^2_{10} = 57.35$, $P < 0.06$, respectively; Fig. 4, Tables 2 and 3). Site-wide encounters of lizards per unit effort was correlated with total area of blowouts for lizards collectively ($R^2 = 0.59$, $\beta = 0.77$, $P < 0.01$; Fig. 5a) indicating a positive relationship between abundance of lizards and quantity of habitat. We did not detect a significant relationship between site-wide encounters of *S. arenicolus* per unit effort and total area of blowouts ($R^2 = 0.27$, $\beta = 0.57$, $P < 0.12$; Fig. 5b). Large variation in encounters of *S. arenicolus* per unit effort among sites with large total area of blowouts reduced the ability of linear regression to detect a relationship between the two variables. Interestingly, there was no strong relationship between quality of habitat (mean size of blowout) and abundances for lizards collectively or for *S. arenicolus* ($R^2 = 0.10$, $P < 0.33$; $R^2 = 0.01$, $P < 0.76$, respectively).

We did not detect significant relationships between encounters of lizards per unit effort and total surface area of caliche for lizards collectively or for *S. arenicolus* ($R^2 = 0.05$, $P < 0.50$; $R^2 = 0.07$, $P < 0.45$, respectively). Sites with 9% of the surface area converted to caliche had similar mean encounters of lizards per unit effort to sites that had 1% of the area converted (Table 1). The lack of significant predictive relationships between total surface area of caliche and encounters of lizards per unit effort of lizards collectively and *S. arenicolus* indicated that differences in abundance of lizards among sites could not be attributed to total surface area of caliche.

DISCUSSION—Our results clearly showed that abundance of lizards varied significantly and in complex ways across the landscape of the Mescalero Sands ecosystem. Total amount of blowout and mean size of blowout in the habitat also varied across the landscape. Quantity and quality of habitat were positively correlated. There was a significant predictive relationship between total area of blowouts and mean size of blowout, indicating that quality of habitat (i.e.,

TABLE 1—Total surface area of caliche, total area of blowout, mean size of blowout, and site-wide encounters per unit effort of lizards collectively and the dunes sagebrush lizard *Sceloporus arenicolus* at 11 sites (numbers correspond to Fig. 1) in the Mescalero Sands ecosystem of southeastern New Mexico. The proportion of each 259-ha site comprised of caliche well pads and roads is in parentheses.

Site	Total surface area of caliche (ha)	Total area of blowout (ha)	Mean size of blowout (m^2)	Site-wide encounters of lizards per unit effort	Site-wide encounters of *S. arenicolus* per unit effort
1. San Juan Mesa West	1.32 (1)	26.76 (10)	28.15	0.232	0.056
2. Johnson Ranch	6.96 (3)	41.06 (16)	28.15	0.183	0.059
3. Mescalero Point NE	1.52 (1)	31.37 (12)	27.42	0.197	0.064
4. Connor Well	2.47 (1)	50.82 (20)	37.85	0.383	0.000
5. Maljamar	12.24 (5)	43.54 (17)	25.78	0.330	0.035
6. Laguna Gatuna NNW	14.45 (6)	14.47 (6)	19.80	0.107	0.011
7. Ironhouse Well	4.20 (2)	31.71 (12)	15.25	0.277	0.030
8. Monument South W	23.28 (9)	25.23 (10)	29.04	0.188	0.036
9. Monument South E	23.88 (9)	26.40 (10)	22.78	0.245	0.023
10. Hobbs SW	3.73 (1)	20.78 (8)	16.96	0.230	0.033
11. Hobbs SE	7.52 (3)	32.35 (12)	22.33	0.282	0.105

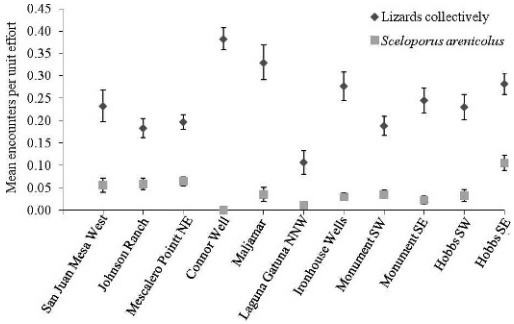

FIG. 4—Abundances of all seven species of lizards collectively and of the dunes sagebrush lizard *Sceloporus arenicolus* at 11 sites in the Mescalero Sands ecosystem of southeastern New Mexico. Abundances are mean encounters per unit effort (±1 *SE*) computed from 227 transects.

size of patch) for lizards in shinnery-oak sand-dunes depended on extent of habitat. Moreover, abundance of lizards also was associated positively with extent of habitat.

Abundances of *S. arenicolus* also varied across the landscape, and our results suggest that extent of sand-dune blowouts in the surrounding landscape was an important determinant of these abundances, although total area of blowouts could not explain entirely the observed pattern among sites. The relationship between total area of blowouts and encounters of *S. arenicolus* per unit effort was positive and explained 27% of variance in the dataset (Fig. 5b), but this relationship was not significant. Lack of a stronger statistical pattern for *S. arenicolus* was due to small mean encounters per unit effort and considerable variation among sites. For example, mean values were zero at one

TABLE 2—Results from Tukey's test with unequal samples on mean encounters per unit effort of all species of lizards collectively at 11 sites in the Mescalero Sands ecosystem of southeastern New Mexico. Sites that do not share the same letter are significantly different.

Site	*n*	Mean	Rank
Laguna Gatuna NNW	18	0.107	1 a
Johnson Ranch	28	0.183	2 ab
Monument SW	27	0.188	3 ab
Mescalero Point NE	48	0.197	4 abc
Hobbs SW	16	0.230	5 abc
San Juan Mesa West	10	0.232	6 abc
Monument SE	16	0.245	7 bc
Ironhouse Wells	12	0.277	8 bcd
Hobbs SE	22	0.282	9 bcd
Maljamar	8	0.330	10 cd
Connor Well	21	0.383	11 d

TABLE 3—Results from Nenyemi's test (nonparametric Tukey's test) with unequal samples for mean encounters of *Sceloporus arenicolus* per unit effort at 11 sites in the Mescalero Sands ecosystem of southeastern New Mexico. Sites that do not share the same letter are significantly different.

Site	*n*	Mean	Rank
Connor Well	21	55.50	1 a
Laguna Gatuna NNW	18	72.69	2 a
Monument SE	16	91.16	3 ab
Hobbs SW	16	98.09	4 ab
Monument SW	27	105.46	5 ab
Ironhouse Wells	12	107.13	6 ab
Maljamar	8	107.44	7 ab
Johnson Ranch	28	128.64	8 ab
Mescalero Point NE	48	135.58	9 ab
San Juan Mesa West	10	136.30	10 ab
Hobbs SE	22	167.43	11 b

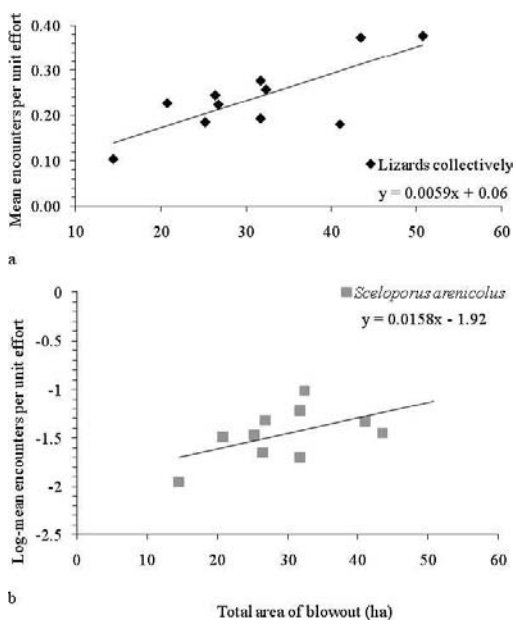

FIG. 5—Relationship between total area of blowout and site-wide encounters per unit effort for a) lizards collectively and for b) the dunes sagebrush lizard *Sceloporus arenicolus* among 11 sites in shinnery-oak and sand-dune habitat in the Mescalero Sands ecosystem of southeastern New Mexico. Mean encounters per unit effort for *S. arenicolus* were log-transformed.

site and close to zero at several sites (Fig. 5b). Other factors associated with fragmentation (e.g., isolation of patch) or meta-population dynamics (Hanski and Gilpin, 1991; Hokit et al., 1999) may also have influenced the patterns of abundance that we observed. Populations of lizards studied in other patchy environments showed responses similar to what we detected. Area of habitat was correlated with abundances of nine species of squamates in the patchy Florida scrub (McCoy and Mushinsky, 1999). In a coastal-dune system in Argentina, a habitat specialist, the sand dune lizard *Liolaemus multimaculatus*, nearly disappeared within 7 years after construction of a road reduced its habitat by 90% (Vega et al., 2000). Thus, it is likely that populations of *S. arenicolus* exhibit similar responses and were correlated with area of blowouts in the habitat.

Despite specialization of *S. arenicolus* on sand-dune blowouts and its presence in relatively large patches (Chan et al., 2009; Fitzgerald and Painter, 2009), we did not find a significant relationship between average size of patch (mean size of blowout) and encounters of *S. arenicolus* per unit effort, or for all lizards combined. Although there is evidence that abundance of *S. arenicolus* and other species of lizards increase with size of patch (Fitzgerald and Painter, 2009), the form of the relationship is not clear, e.g., the relationship may be curvilinear or a step-function. Understanding size of populations and demographic variation among dune-dwelling lizards in relation to size of patch is a topic currently being studied.

We demonstrated that area of habitat and size of patch co-varied, but it was difficult to disentangle effects of size of patch from quantity of habitat. Size of patch is correlated with abundance, survivorship, and recruitment in other lizards studied at single sites (Hokit and Branch, 2003*a*, 2003*b*). Thus, we predict both features probably are important to lizards in the Mescalero Sands ecosystem. Importantly, effects of area and size of patch on abundance of lizards probably also are scale-dependent. Sandel and Smith (2009) pointed out three basic observations from which scale-dependence arises: abundance increases with area, environmental conditions vary across space, and the effect of an organism on its environment is spatially limited. Our analyses demonstrated the first two conditions clearly exist for lizards in shinnery-oak-sand-dune habitats, and we may presume that environmental effects on local populations of lizards, especially the specialist *S. arenicolus*, were independent among our study sites. Interestingly, because of scale-dependence between area of habitat (total area of blowouts) and size of patch (mean size of blowout) among geographically dispersed sites in the Mescalero Sands ecosystem, effects of patch size at individual sites probably were not detected in our study. At the regional scale, habitat area (total area of blowouts) apparently drives abundance, whereas patch size may determine local abundance of lizards.

Furthermore, variation in abundances of lizards across the Mescalero Sands ecosystem probably did not correspond to only two habitat variables and one measure of landscape alteration. Changes in vegetation, for example, affect abundances of lizards (Jellinek et al., 2004). Abundances of *S. undulatus* and *H. maculata* remained relatively stable during initial years of brush encroachment on a dune landscape, but then declined after the open vegetation on dunes was significantly reduced (Ballinger and Watts, 1995). We did not quantify vegetative cover in patches of sand-dune blowouts, but we could see differences in diversity of grasses and plants in blowouts at the study sites. Thus, it is reasonable to speculate that vegetative differences also were associated with abundances of *S. arenicolus* and other species of lizards. Indeed, there is evidence that *S. arenicolus* is sensitive to changes in cover and type of vegetation. Numbers of *S. arenicolus* decreased by 78% within 5 years at sites where shinnery-oak was removed by spraying with herbicide compared to paired control sites (H. L. Snell et al., in litt.). Since the time when shinnery-oak was killed at those study sites >15 years ago, dune blowouts became vegetated with grasses and forbs, blowouts became smaller and flatter, and *S. arenicolus* was no longer present.

Despite the observation that areas of the landscape in the Mescalero Sands ecosystem have been fragmented extensively by oil and gas development, we did not find clear statistical evidence to support our hypotheses that oil and gas development at our study sites had a direct negative effect on quantity of habitat, quality of habitat, and populations of lizards. Thus, we did not detect a threshold for which oil and gas development correlated to reduced abundances of dune-dwelling lizards, or of *S. arenicolus*

specifically. Among the 11 sites scattered throughout the Mescalero Sands ecosystem, total area of blowouts and mean size of blowout were variable, and we could not control for natural variation in habitats in our analyses. For example, the two sites with greatest total area of blowouts, Maljamar and Connor Well, had low abundance of *S. arenicolus* (Table 1). These sites also had relatively low total surface area of caliche (5 and 1%, respectively). In contrast, Hobbs SE had relatively low total surface area of caliche, total area of blowouts, and mean size of blowout, yet had the highest encounters of *S. arenicolus* per unit effort.

In addition to natural variation in area of habitat and patchiness among sites, stochastic variation in populations of lizards among sites further confounded our ability to detect effects of conversion of land to caliche, if such effects exist. All studies of lizard populations have documented spatial and temporal variation in vital rates that determine population growth and size. A classic 11-year study of the sagebrush lizard *Sceloporus graciosus* (closest relative of *S. arenicolus*) documented two-fold differences in population size between two sites and among years at each site (Tinkle et al., 1993). Distinguishing variation in natural populations from variation caused by anthropogenic mechanisms can be challenging, especially when long-term data on populations are not available (Tinkle, 1979; Pechmann et al., 1991; Tilman et al., 1994). In the face of demographic stochasticity such as this, it becomes extremely difficult to ascertain the cause of differences in populations among multiple sites, even with long-term data (Fitzgerald, 1994).

Our work represents the first attempts to enumerate lizard populations in the Mescalero Sands ecosystem (Smolensky, 2008; Smolensky and Fitzgerald, 2010; this study). This is the first study to investigate effects of oil and gas development on a dune-dwelling assemblage of lizards, and our results provide insights for future research and establishment of monitoring protocols. What is the next step? Research on landscape fragmentation and connectivity, and on associations among landscape metrics and occurrence of *S. arenicolus* at multiple scales is needed to gain better insight into what factors, and at which spatial scale, best predict persistence of this species and other lizards. A good approach to directly elucidate effects of oil and gas development, or other disturbances on populations of lizards at individual sites, would be long-term studies coupled with before-after-control-intervention experiments (Murtaugh, 2002; Stewart-Oaten, 2003).

We thank C. M. Bell, H. Boostrom, B. Bowers, D. Henderson, M. Hill, R. Jennings and his field class, D. Laurencio, R. Loughridge, and C. W. Painter for assistance in the field. L. Laurencio helped with GIS analyses and B. Collier provided statistical advice. One anonymous reviewer and R. Jennings gave constructive comments on the manuscript. M. Bonino helped translate the abstract. Funding was provided by Texas A&M University, with in-kind support from the Bureau of Land Management, New Mexico Department of Game and Fish, and Texas Parks and Wildlife Department. All procedures were in compliance with Texas A&M University Lab Animal Care Committee protocol 2003-82. This is publication 1368 of the Texas Cooperative Wildlife Collection.

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Submitted 2 February 2010. Accepted 6 April 2011.
Associate Editor was Michael L. Kennedy.