DESERT BIGHORN SHEEP RESTORATION IN TEXAS:

SURVIVAL, POPULATION DYNAMICS, AND HABITAT

A Dissertation

by

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ABSTRACT

Bighorn sheep (*Ovis canadensis*) once occupied mountain ranges from western Canada to northern Mexico in North America. The distribution and abundance of mountain sheep in North America have declined from >500,000 historically, to 185,000 in the 1990s. In Texas, there were 1,000-1,500 desert bighorn (*O. c. mexicana*) living in 16 mountains ranges within the Trans-Pecos region during the late 1800s. Declines resulted from a combination of factors including competition for forage with domestic livestock, introduced diseases from domestic animals, unrestricted hunting, and restriction of movements by net-wire fencing. By the mid-1940s, bighorn sheep populations were estimated at 35 individuals, and by early 1960s the last Texas native desert bighorn was extirpated. One successful approach to the conservation of large mammals has been their translocation into former habitats. While translocation strategies have been successful for many species, translocations of large ungulates can be expensive and time consuming, as well as logistically and politically challenging. Beginning in 1957, the Texas Game and Fish Commission brought desert bighorn from Arizona to a breeding facility to initiate a restoration process. Over the next 4 decades, a total of 146 desert bighorn were transplanted to Texas facilities from other states. This study was initiated to fill gaps in the autecological knowledge of desert bighorn in order to inform management decisions and maximize the potential for long–term success of translocated desert bighorn populations. The objectives of this study included: (1) analysis of survival and cause-specific mortality, (2) assess various strategies to conduct translocations of desert bighorn in Texas using a system modeling approach, and (3) evaluation of potential desert bighorn distributions utilizing a

probability occurrence distribution model at a landscape scale within the Trans-Pecos region of Texas. Results for the first objective, from the 172 collared individuals a total of 57 mortalities was recorded (25 M, 32 F). Causes of mortality were: 27 undeterminable, 20 by mountain lion predation (*Puma concolor*), 5 were attributed to contagious ecthyma (parapox orf virus), 1 poached in Mexico, 1 birth complication, 1 infection due to a broken jaw, 1 ingestion of toxic vegetation (cloakfern, *Astrolepis sinuate*), and 1 fell from a cliff. For the second objective, results indicated that the number of years required for the population to reach carrying capacity (1) was reduced when proportionally more females than males were reintroduced, (2) was reduced slightly more by shorter than by longer time lags between the initial and the second reintroduction, although differences were negligible, and (3) was reduced when a larger number of animals (representing a larger proportion of carrying capacity) was reintroduced. Results for objective 3 showed slope (49.74%) to have the greatest variability explanation followed by elevation (21.26%). The model was able to explain 95.73% of variability by using 4 variables. Distribution values for slope demonstrated selection values ranging from 0.09 to 314, having a median of 56.6 with a lower quartile of 38.2 and upper quartile of 76.3. Elevation values showed greater selection for elevations between 1,200 m and 1,600 m having the median of 1,459 m. Elevation values ranged from 721 m to 2,024 m. In conclusion, reintroductions are increasingly used to re-establish populations of threatened species. However, many reintroduction attempts have been unsuccessful and the main reasons of failure are seldom understood. Monitoring should continue to provide the primary tool by which we learn about the success or failure of conservation investments.

DEDICATION

To my family. For my sister's unconditional support and model of willpower over the years, my mother's much-needed nights of praying for me, and to my father's teaching that our duty in life is not to be better than others but to become the best we can day-to-day.

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Contributors

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

Page

CHAPTER I

INTRODUCTION

Bighorn sheep (*Ovis canadensis*) of North America once occupied mountain ranges from western Canada to northwestern Mexico and Baja California (Valdez and Krausman 1999). Currently, desert bighorn are distributed throughout the western and southern portions of Arizona, southeastern California, the western half of Colorado, central and southern Nevada, western New Mexico, western Texas, and southeastern Utah (Monson 1980). In Mexico, desert bighorn inhabit the states of Sonora, Baja California, and Baja California Sur, Chihuahua, Coahuila, and Nuevo Leon (Tarango and Krausman 1997, Espinosa et al. 2006).

The distribution and abundance of mountain sheep in North America have declined from >500,000 (Seton 1929, Valdez 1988) historically, to 185,000 in the 1990s (Valdez and Krausman 1999). Currently, desert bighorn populations are estimated to be <20,000 individuals in the contiguous United States, and several populations are state- and federally-listed as endangered (e.g., peninsular bighorn sheep [*O. c. cremnobates*]). In Texas, Carson (1941) believed there to be 1,000-1,500 desert bighorn (*O. c. mexicana*) living in 16 mountains ranges (Bailey 1905, Davis and Taylor 1939) in the Trans-Pecos region in the late 1800s. Declines resulted from a combination of factors including competition for forage with domestic livestock, introduced diseases from domestic animals, unrestricted hunting, and restriction of movements by net-wire fencing (Davis and Taylor 1939, Buechner 1960). Protective measures for desert bighorn were initiated as early as 1903 with the enactment of a hunting prohibition. However, by the mid-1940s, the Texas desert bighorn population was estimated at 35 individuals (Carson 1945). Further protective measures occurred in 1945 with the establishment of the Sierra Diablo WMA to serve as a sanctuary for the last remaining Texas desert bighorn (Brewer 2001), but it is believed that the last native Texas desert bighorns were gone by the early 1960s (Schmidly 1977).

One successful approach to the conservation of large mammals has been their translocation into former habitats (Krausman 2002). Beginning in 1957, the Texas Game and Fish Commission (now the Texas Parks and Wildlife Department [TPWD]) brought desert bighorn from Arizona to a breeding facility to initiate a restoration program (Kilpatric 1990). Over the next 4 decades, a total of 146 desert bighorn was transplanted to Texas facilities from Nevada (n = 107), Arizona (n = 31), Mexico (n = 6), and Utah (n = 2) (Brewer and Hobson 2000). Initial efforts focused on propagation of desert bighorn in captivity to provide a source of stock for transplanting into suitable habitat. The first propagation facility was constructed on Black Gap Wildlife Management Area (BGWMA) and was operational by 1959. Additional facilities were constructed on Sierra Diablo Wildlife Management Area (SDWMA) in 1970 and 1983, and Chilicote Ranch in 1977. As free-ranging desert bighorn numbers increased, the need for brood facilities and out-of-state augmentations diminished (Cook 1994). By 1997, 237 desert bighorn had been captured from re-established Texas populations and released into other Texas mountain ranges (Cook 1994, Brewer and Hobson 2000).

Currently, the Trans-Pecos region of Texas supports 7 free-ranging populations of desert bighorn (Brewer and Hobson 2000). These occur within the mountain ranges of Baylor, Beach, Sierra Diablo, Sierra Vieja, Van Horn, Bofecillos, BGWMA and Elephant Mountain Wildlife Management Area (EMWMA). In 2000, less than 50 years after desert bighorn restoration efforts began, Pittman et al. (2001) reported observing 381 free-ranging individuals during annual helicopter surveys in 7 of Texas' mountain ranges.

While translocation strategies have proven successful thus far in Texas, translocations of large ungulates can be expensive, time consuming, and logistically and politically challenging (Beck et al. 1994, Biggins and Thorne 1994, Wolf et al. 1996, Fritts et al. 1997), as such is the case for desert bighorn (Gilad et al. 2013). Success can be measured by survival and population growth (Singer et al. 2000). Because of this, studies have utilized very high frequency radio collars and computer mapping software to help assess the movements and survival of resident (Longshore and Douglas 1995) and translocated (Ravey and Schmidt 1981, Roy and Irby 1994, Singer et al. 2000) bighorn populations. If success is measured by survival and population growth (Singer et al. 2000), monitoring a translocated bighorn population post-release could allow managers to determine if the new location is in fact a successful site reflected by survival and population growth.

While population models have the potential to guide reintroduction programs, to date decisions have been made largely based on intuition (Armstrong and Reynolds 2012). Because of this, my study objectives were as follows: (1) determine the survival and cause-specific mortality of desert bighorn by monitoring radio-collared individuals post-translocation, (2) assess various strategies in conducting translocations of desert bighorn in Texas using a system modeling approach, and (3) model potential desert bighorn habitat via a landscape scale probability occurrence distribution model within the Trans-Pecos region of Texas. Understanding survival post-release, population dynamics, habitat utilization by desert bighorn could allow researchers and managers the ability to discern, and therefore delineate, desert bighorn resource requirements in different mountain ranges.

CHAPTER II

SURVIVAL AND CAUSE-SPECIFIC MORTALTITY OF TRANSLOCATED DESERT BIGHORN IN TEXAS

Introduction

Biologists and resource managers have been constrained in terms of available management options, when addressing population declines and localized extinctions in the last several decades; consequently new forms of conservation interventions are being explored (Seddon et al. 2012). Translocation of animals, the movement of animals from one area with free release in another area (IUCN 1987), has been essential to many wildlife management programs such as the stocking of game species and furbearers, the reintroduction of extirpated species, and the management of endangered and threatened species (Craven et al. 1998). In general, the objective of translocations is to establish a self-sustained population of a species within its historic range (Griffith et al. 1989).

In Texas, historically abundant desert bighorn (*Ovis canadensis* spp.) populations were extirpated due to a combination of issues (Douglas and Leslie 1999, Toweill and Geist 1999, Serrano et al. 2006), with livestock presence, disease, predation, and dispersal largely being attributed to the decline of these populations (Rominger et al. 2004). There have been widespread efforts to restore desert bighorn throughout their range since the 1950s (Krausman and Shackleton 2000), and translocations have been used as the primary restoration tool. In Texas, managers have used translocations to expand and reestablish populations in historic ranges of desert bighorn since the inception of restoration efforts (Kilpatric 1990), and many attribute translocations as the primary reason for ongoing populations in the western United States (Krausman and Shackleton 2000).

Survival of translocated individuals can represent a critical challenge for restoration efforts, especially within small populations during the initial reestablishment period. Predation, hunting, disease, weather, population density, and food supply influence survival, and thus subsequent population growth or establishment (Davis and Taylor 1939, Buechner 1960). A cost-effective means of estimating survival is through the monitoring of translocated desert bighorn populations by direct observation (Bleich and Taylor 1998), the accuracy of which can be increased with the use of radio-collared individuals (Locke 2003).

In the last 20 years, substantial improvements have been made, in capture, marking, and monitoring techniques, to facilitate survival estimates for free-ranging animals (Krebs 1999, Kenward 2001, Millspaugh and Marzluff 2001). Development of software allowing for the analysis of complex survival parameter estimation also emerged during that time (Williams et al. 2002). Concurrently, the increasing availability of Global Positioning System (GPS) location data for wildlife populations has provided opportunities to investigate concerns (Clapp et al. 2014), such as the circumstances influencing survival. The use of this technology permits not only finding cause-specific mortalities, but it also provides much more detailed information about the time and location of mortality.

Despite past desert bighorn restoration efforts, the relative success of these restoration programs has not been sufficiently evaluated (Janke 2015). Survival estimates help biologists understand wildlife population declines, the understanding of which is essential for improvement of restoration activities and ultimately in the management and conservation of any species. Additionally, conservation biologists have given considerable attention to survival estimation in order to better understand changes in population dynamics (Murray and Patterson 2006) and population viability. Because understanding the root cause of population decline or increase is central to any attempt at population restoration (Jones and Merton 2012), the objective of this study was to analyze post-release survival and sources of mortality for desert bighorn using GPS telemetry data from translocation efforts in the Trans-Pecos ecoregion of Texas.

Study Area

Located in the Chihuahuan Desert, the Trans-Pecos (Figure 2.1) is bordered to the east by the Pecos River, to the west and south by the Rio Grande River, and by New Mexico to the north (Hatch et al. 1990). The region includes Brewster, El Paso, Culberson, Hudspeth, Jeff Davis, Pecos, Reeves, Presidio, and Terrell counties. Mountain ranges including Baylor, Beach, Christmas, Chinati, Chisos, Davis, Eagle, Franklin, Glass, Guadalupe, Santiago, Sierra Diablo, Sierra Vieja, Van Horn, and Wiley reside in the Trans-Pecos (Powell 1998). Elevations range from 762–2,667 m and annual precipitation varies from 200–460 mm, the majority of which occurs as monsoonal thunderstorms in the months of July, August, and September (Powell 1998). Higher elevations receive more rainfall (300–460 mm) than do the lowlands and basins (200–300 mm; Powell 1998). Climate varies as a result of topography, with precipitation increasing and temperature decreasing with elevation (Turner 1977). As a result, spatial variation in vegetation is tied to topography across this region.

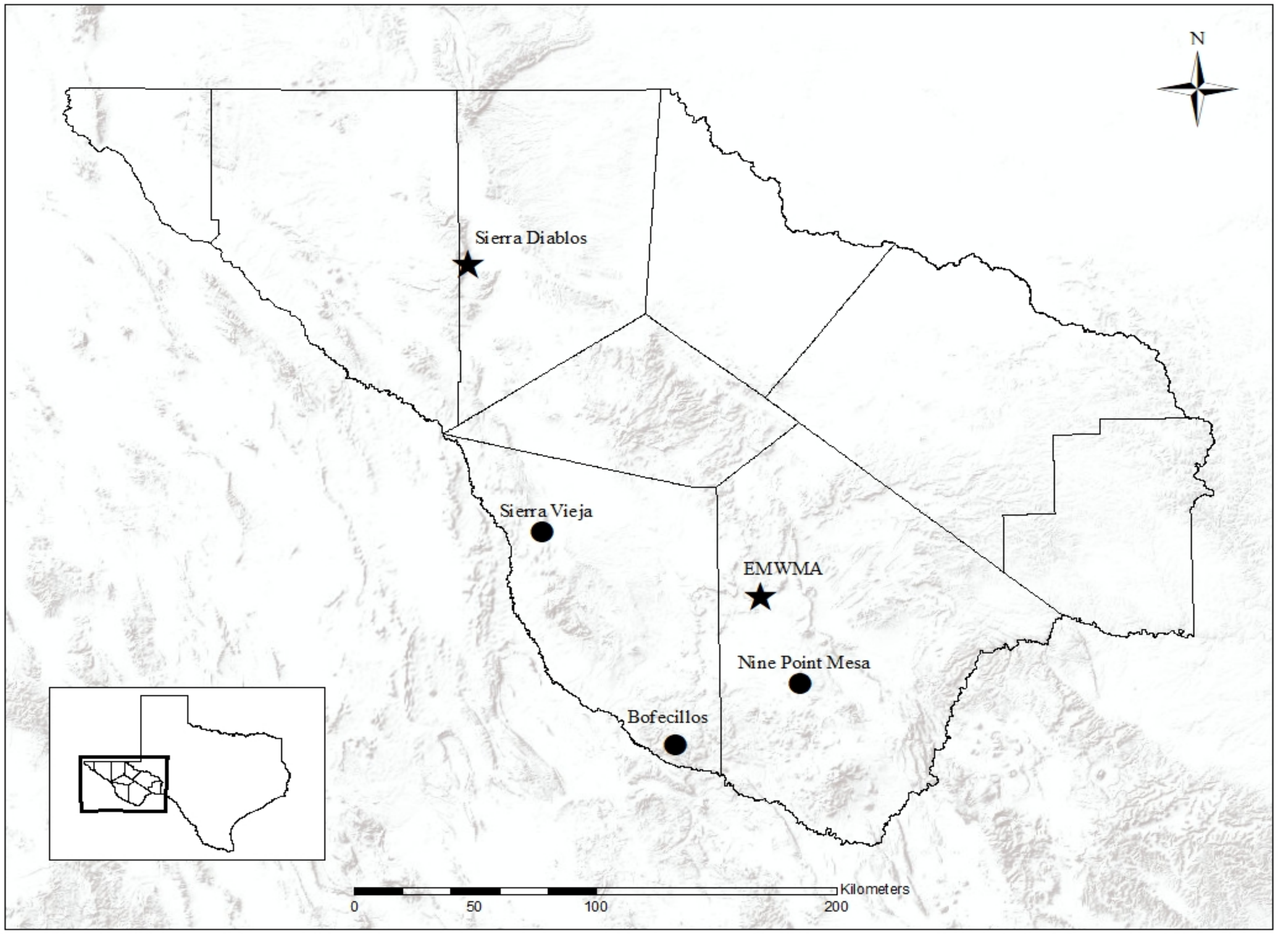

Figure 2.1. The Trans-Pecos Texas, USA with the locations of study sites indicated by circles and capture locations indicated by stars.

Vegetation varies with creosote bush (*Larrea tridentata)* and tarbush *(Flourensia cernua*) communities in the lower elevations, to grasslands and a mix of juniper *(Juniperus* spp.), oak (*Quercus* spp.), and pinyon pine (*Pinus edulis*) forest (Hatch et al. 1990).

Methods

A combined total of 246 desert bighorn (101 M, 240 F) was captured in the winters of 2010, 2011, 2012, and 2014, using the helicopter net-gun method (Krausman et al. 1985). Captures occurred at Elephant Mountain Wildlife Management Area (EMWMA; 2010, 2012, and 2014; *n* = 46, 44, and 64, respectively) and the Sierra Diablo Mountains (Sierra Diablos Meta Population; SDMP; 2011 *n* = 95). Once captured, individuals were hobbled, blindfolded, and aerially transported to a staging area. Personnel then collected data including sex, age, body condition, fecal and hair samples, nasal and ear swabs, and whether ewes were lactating. Age was determined by horn growth rings (Geist 1966, Hansen and Deming 1980) and tooth wear and replacement (Hansen and Deming 1980). After data were collected and a collar fitted on select individuals, ewes were placed in modified livestock trailers and rams were placed into modified crates for transportation. Once captures were completed for the day, all desert bighorn were transported to the restocking site and released the same day.

Due to lack of funding, only a total of 172 GPS collars was attached to individual animals. GPS collars were programmed to record locations at intervals between 1 and 8 hours, depending on the objectives of other studies planned for each study site. GPS collars were programmed to remain on the individual for 300 days

(n = 8), 12 months (n = 3), and up to 25 months (n = 161). In 2010, 35 desert bighorn (10 M, 25 F) were collared with Lotek GPS 3300 collars programmed to collect GPS locations every 3 hours for 25 months. In 2011, 35 desert bighorn (10 M, 25 F) were fitted with Advanced Telemetry Systems (ATS; Advanced Telemetry Systems, Isanti, MI, USA) G2110D GPS collars and 8 individuals (4 M, 4 F) were equipped with North Star NSG-D1 satellite collars programmed to collect GPS locations every 5 hours for 25 months. For 2012, 27 collars were ATS G2110D GPS (12 M, 15 F), 8 of the collars were ATS G2110E2 Iridium collars (1 M, 7 F), and 5 of the collars were North Star satellite collars (5 M). All collars were programmed to save location data every 5 hours with mortality sensors set at 8 hours. In 2014, 31 desert bighorn (9 M, 22 F) were radio-collared with ATS G2110D GPS and 5 (4 M, 1 F) were collared with ATS G2110E2 Iridium satellite collars. Location intervals were standardized to 5 hours for this study and all mortality sensors were set to 8 hours. Variation in equipment and settings was due to the "trial and error" nature of method development among studies during this period.

Very high frequency (VHF) beacons were programmed to transmit from 0800–2000 hours and did not transmit on Sundays to increase duration of collar battery. Weekly VHF telemetry was conducted from the ground using a receiver and antenna (Yagi, folding directional antenna). Aerial telemetry was conducted as needed using fixed-wing aircraft with an H-antenna mounted to each wing. Collars switched to mortality mode (80 beeps/minute [bpm] from a base rate of 40 bpm) when collars were stationary for ≥8 hours. When a mortality signal was acquired from a collar, the location of the animal and the collar status was recorded. Mortalities signals were investigated as soon as possible after detection. When mortalities were found, an investigation was conducted to determine the cause of death and to estimate the rates of specific mortality causes based on criteria from Janke (2015). Once a collar was recovered, either after the animal died or the collar successfully dropped off, records from the collars were saved as text files and imported into Microsoft Excel.

When a mortality signal was acquired from a collar, an investigation would be conducted regarding the site and carcass. Different features of the mortality site were noticed when conducting an investigation such as: general description of location and physical characteristics of the carcass (e.g., broken bones, visible indication of disease presence). Attributed cause of death was classified as mountain lion (*Puma concolor*) predation if ≥2 of the following were observed: bite marks in either the neck or skull, signs of struggle or chase, drag trail from the kill site to the cache site, cache site(s) of remains, rumen eviscerated from the carcass, broken or chewed bones, fresh mountain lion tracks or scat at kill or cache site, or mountain lion scrapes at or near the cache site.

I analyzed data using Program MARK (Program MARK, version 8.0; Cooch and White 2015) to evaluate survival. Parameters analyzed included sex (M or F), year following release (Year1 or Year2), season (not separated by year), and specific season (season separated by year). Seasons were delineated as gestation (15 Nov-14 Feb), lambing (15 Feb-14 May), lactation (15 May-14 Aug), and breeding (15 Aug-14 Nov).

Results

Survival Models

The top model ($Season_{FR}$+Locations*Sex) produced an AIC_C weight of 0.29770 of the available model weights (Table 2.1). The 2 next closest models, $Season_{FR}$+Sex had a ΔAIC_C weight of 0.28653 and $Season_{FR}$+Source had AIC_C weight of 0.28338.

Survival Estimates

Annual survival of desert bighorn released in the Bofecillos Mountains in 2010 was 0.80 (SE = 0.06) in the first year following release and 0.80 (SE = 0.09) in the second year. Annual survival of desert bighorn released in the Bofecillos Mountains in 2011 was 0.46 (SE = 0.07) and 0.94 (SE = 0.05) in the respective years following release. Although final post-release survival after 2 years was greater for the Bofecillos Mountains in 2010 (0.63, SE = 0.13) than in 2011 (0.44, SE = 0.10), most Bofecillos 2011 desert bighorn mortalities occurred within the first year following release. For desert bighorn translocated in 2012 at Nine Point Mesa, annual survival was 0.85 (SE = 0.01) and 0.69 (SE = 0.03) in the first and second year following release, respectively. Desert bighorn released in the Sierra Vieja mountains in 2014 experienced annual survival rates of 0.70 (SE = 0.01) for the first year and 0.76 (SE = 0.05) for the following year. Annual survival of males released in the Bofecillos Mountains 2010 were 0.90 (SE = 0.09) and 0.83 (SE = 0.15) for each year of monitoring, respectively. The males released there in 2011 displayed survival of 0.44 (SE = 0.08) and 1.00 (SE = 0.00) for each year following translocation. Annual survival of males from the Nine Point Mesa translocation was 0.75 (SE = 0.06) and 0.58 (SE = 0.03). Males released in the Sierra Vieja Mountains showed annual survival of 0.77 (SE = 0.01) and 0.76 (SE =

Table 2.1. Model selection results, based on Akaike's Information Criterion with small size correlations (AIC_c), for analyses examining translocated desert bighorn survival as a function of population source, sex, season, and time following release in the Trans-Pecos, Texas, USA, 20 models were considered.

Model[a]	No. Parameters	AIC_c	ΔAIC_c[b]	AIC_cWeight
$Season_{FR}$+Locations*Sex	24	388.9490	0.0000	0.29770
$Season_{FR}$*Locations+Sex	24	389.0255	0.0765	0.28653
Locations*$Seasons_{FR}$+Sex+Source	25	389.0476	0.0986	0.28338
$Season_{FR}$+Sex	21	391.8173	2.8683	0.07095
Location*$Season_{FR}$	23	394.4714	5.5224	0.01882
Location*$Season_{FR}$+Source	24	394.4934	5.5444	0.01861
$Season_{FR}$+Source	21	395.4498	6.5008	0.01154
$Season_{FR}$	20	396.2609	7.3119	0.00769
Source+Sex	3	398.8697	9.9207	0.00209
Source	2	401.4130	12.4640	0.00059
Sex	2	401.7625	12.8135	0.00049
$Season_{FR}$+Locations*Source	24	401.8138	12.8648	0.00048
Year	6	401.9690	13.0200	0.00044
Location*$Season_{GEN}$+Sex	9	402.2795	13.3305	0.00038
Locations	4	404.4609	15.5119	0.00013
Locations*$Season_{GEN}$	8	405.5466	16.5976	0.00007
All Equal	1	406.8067	17.8577	0.00004
$Season_{GEN*}$Sex	9	407.1953	18.2463	0.00003
Locations*$Season_{GEN}$+Source	9	407.5928	18.6438	0.00003
$Season_{GEN}$	5	408.1340	19.1850	0.00002

[a]Source = translocation population source, $Seasons_{GEN}$ = Seasons combined for year 1 & 2 post-release. (i.e., gestating, lambing, lactating, and breeding in general), $Seasons_{FR}$ = Season by year post-release. (i.e., gestating 1 and 2, lambing 1 and 2, lactating 1 and 2, and breeding 1 and 2), and All Equal = there was no difference between any of the functions. [b]ΔAIC_C refers to difference in AIC_c between the most supported and given model.

0.04). Ewes released in the Bofecillos Mountains in 2010 showed survival of 0.75 (SE =0.08) for the first year and 0.78 (SE = 0.11) for the following year. Ewes released in the Bofecillos Mountains in 2011 had survival of 0.75 (SE= 0.08) in the first year and 0.78 (SE = 0.11) in the second. Desert bighorn males released in 2011 at Bofecillos Mountains annual survival was 0.48 (SE = 0.09) and 0.93 (SE = 0.06) the second year. Nine Point Mesa females presented annual survival of 0.95 (SE = 0.01) and 1.00 (SE = 0.00) subsequent to translocation. Sierra Vieja Mountains female annual survival were 0.67 (SE = 0.07) for the first year and 0.84 (SE = 0.01) for the fallowing year (Table 2.2)

Collectively (males and females combined) annual survival estimates for desert bighorn released in Bofecillos Mountains 2010 displayed survival of 0.80 (SE = 0.20) up to 1.00 (SE = 0.00), having lowest survival during the first 3 seasons following the translocation. Bofecillos 2011 translocation exhibited more constant survival across seasons having the lowest survival during the lambing season in the first year post-release with 0.85 (SE = 0.15). Nine Point Mesa displayed the lowest season survival compared to all study sites with a 0.77 (SE = 0.23) in the second lactating season. All mortalities seen in this study site during that season were males associated with sore mouth disease. However, this site also had the highest survival > 0.93 in every other season. Lastly, Sierra Vieja displayed a constant survival ranging from 0.87 (SE = 0.13) up to 1.00 (0.00) (Table 2.3).

Male seasonal survival for Bofecillos 2010 ranged from 1.00 (SE = 0.00) down to 0.80 (SE = 0.18) with the lowest survival occurring during lactation season of the second year post-release. Bofecillos 2011 males had 0.64 (SE = 0.15) as their lowest

Table 2.2. Annual sex and combined survival estimates ($\hat{S}$) and adjusted standard errors (SE) for the 2 years following release of translocated desert bighorn in the Trans-Pecos, Texas, USA.

	Bofecillos 2010[d]			Bofecillos 2011[e]			Nine Point Mesa[f]			Sierra Vieja[g]		
	n^a	$\hat{S}$	SE	n^a	$\hat{S}$	SE	n^a	$\hat{S}$	SE	n^a	$\hat{S}$	SE
Males												
Year 1[b]	10	0.90	0.09	14	0.44	0.08	18	0.75	0.06	13	0.77	0.01
Year 2[c]	9	0.83	0.15	5	1.00	0.00	11	0.58	0.03	9	0.76	0.04
Average	10	0.72	0.17	14	0.42	0.13	18	0.44	0.05	13	0.84	0.02
Females												
Year 1	25	0.75	0.08	29	0.48	0.09	22	0.95	0.01	23	0.67	0.07
Year 2	18	0.78	0.11	15	0.93	0.06	16	1.00	0.00	7	0.84	0.01
Average	25	0.59	0.11	29	0.48	0.09	22	0.95	0.01	23	0.57	0.05
Combined[h]												
Year 1	35	0.80	0.06	43	0.46	0.07	40	0.85	0.01	36	0.70	0.01
Year 2	27	0.80	0.09	20	0.94	0.05	27	0.69	0.03	14	0.76	0.05
Average	35	0.64	0.07	43	0.46	0.06	40	0.72	0.02	36	0.530	0.04

[a] Number of desert bighorn radio-collared during time period.
[b] First year post-release for each translocation.
[c] Second year post-release for each translocation.
[d] Translocation conducted in year 2010 into Bofecillos Mountains.
[e] Translocation conducted in year 2011 into Bofecillos Mountains.
[f] Translocation conducted in year 2012 into Nine Point Mesa.
[g] Translocation conducted in year 2014 into Sierra Vieja Mountains.
[h] Combined translocation conducted for all years.

Table 2.3. Collective seasonal survival estimates (Ŝ) and adjusted standard errors (SE) for the 2 years following release of translocated desert bighorn in the Trans-Pecos, Texas, USA.

	Bofecillos 2010[c]			Bofecillos 2011[d]			Nine Point Mesa[e]			Sierra Vieja[f]			Combined[g]		
Season[b]	n[a]	Ŝ	SE	n[a]	Ŝ	SE	n[a]	Ŝ	SE	n[a]	Ŝ	SE	n[a]	Ŝ	SE
Year 1															
Gestating	35	0.80	0.20	43	0.93	0.07	40	1.00	0.00	36	1.00	0.00	154	0.90	0.03
Lambing	33	0.82	0.18	34	0.85	0.15	38	0.95	0.05	35	0.97	0.03	140	0.86	0.02
Lactating	31	0.83	0.17	26	0.94	0.06	33	0.94	0.06	31	0.89	0.11	121	0.87	0.05
Breeding	30	1.00	0.00	19	0.97	0.03	30	0.93	0.07	27	0.86	0.14	106	0.91	0.01
Year 2															
Gestating	27	1.00	0.00	19	1.00	0.00	27	1.00	0.00	24	0.88	0.12	97	0.95	0.02
Lambing	25	1.00	0.00	19	0.91	0.09	27	1.00	0.00	22	0.87	0.13	93	0.82	0.08
Lactating	18	0.93	0.07	18	0.94	0.06	22	0.77	0.23	19	0.92	0.08	77	0.74	0.07
Breeding	8	1.00	0.00	13	0.91	0.09	17	1.00	0.00	19	1.00	0.00	57	0.75	0.04

[a]Number of sheep radio-collared during time period.
[b]Biological seasons delineated as gestating (15 Nov-14 Feb), lambing (15 Feb-14 May), lactating (15 May-14 Aug), and breeding (15 Aug-14 Nov).
[c]Translocation conducted in year 2010 into Bofecillos Mountains.
[d]Translocation conducted in year 2011 into Bofecillos Mountains.
[e]Translocation conducted in year 2012 into Nine Point Mesa.
[f]Translocation conducted in year 2014 into Sierra Vieja Mountains.
[g]Combined translocation conducted for all years.

survival during the first year in the lambing season. Nine Point Mesa had the highest mortality, 0.50 (SE = 0.16) shown in the second season of lactating. The Sierra Vieja translocation exhibited lowest survival of males occurring in the season of lactating in both years post-release having 0.67 (SE = 0.16) and 0.75 (SE = 0.22), respectively (Table 2.4).

Seasonal female survival for Bofecillos 2010 fluctuated from 1.00 (SE = 0.00) down to 0.84 (SE = 0.07), lowest survival was displayed during lambing season of the first and second year post-release. Bofecillos 2011 females had 0.73 (SE = 0.08) and their lowest survival was seen to occur during the first 3 seasons post-release. Nine Point Mesa had the highest survival with only the first lactating season post-release showing survival rate of 0.94 (SE = 0.06). The Sierra Vieja translocation exhibited lowest survival of males occurring in the season of lactating in both years post-release having 0.67 (SE = 0.16) and 0.75 (SE = 0.22), respectively (Table 2.5).

I recorded 58 mortalities throughout the study period (24 M, 34 F) (Figure 2.2). Cause of mortality for 27 desert bighorn was undeterminable due to lack of evidence, 20 mortalities were determined to be mountain lion predation, 5 were related to contagious ecthyma (parapoxorf virus), 1 poached in Mexico, 1 birth complication, 1 was attributed as an infection due to a broken jaw, 1 was due to the ingestion of toxic vegetation (cloakfern, *Astrolepis sinuate*), and 1 fell from a cliff.

Discussion

Top models from the study found seasons to have important survival influence on desert bighorn, suggesting it is not only important where individuals are translocated to or where are they translocated from, but also the time of year. Though

Table 2.4. Seasonal survival estimates (Ŝ) and adjusted standard errors (SE) for males for the 2 years following release of translocated desert bighorn in the Trans-Pecos, Texas, USA.

	Bofecillos 2010[c]			Bofecillos 2011[d]			Nine Point Mesa[e]			Sierra Vieja[f]			Combined[g]		
Season[b]	*n*[a]	Ŝ	SE	*n*[a]	Ŝ	SE	*n*[a]	Ŝ	SE	*n*[a]	Ŝ	SE	*n*[a]	Ŝ	SE
Year 1															
Gestating	10	1.00	0.00	14	0.85	0.01	18	1.00	0.00	13	1.00	0.03	55	0.96	0.01
Lambing	10	1.00	0.00	12	0.64	0.15	18	0.89	0.07	13	1.00	0.00	53	0.79	0.00
Lactating	9	0.90	0.09	8	0.83	0.15	16	0.94	0.06	9	0.67	0.16	42	0.77	0.08
Breeding	9	1.00	0.00	4	1.00	0.00	14	0.86	0.09	8	0.83	0.15	35	0.88	0.12
Year 2															
Gestating	9	1.00	0.00	4	1.00	0.00	11	1.00	0.00	7	0.80	0.18	31	1.00	0.10
Lambing	9	1.00	0.00	4	1.00	0.00	11	1.00	0.00	7	1.00	0.00	31	0.87	0.07
Lactating	7	0.80	0.18	4	1.00	0.00	10	0.50	0.16	6	0.75	0.22	27	0.70	0.09
Breeding	2	1.00	0.00	2	1.00	0.00	5	1.00	0.00	6	1.00	0.00	15	0.85	0.09

[a]Number of sheep radio-collared during time period.
[b]Biological seasons delineated as gestating (15 Nov-14 Feb), lambing (15 Feb-14 May), lactating (15 May-14 Aug), and breeding (15 Aug-14 Nov).
[c]Translocation conducted in year 2010 into Bofecillos Mountains.
[d]Translocation conducted in year 2011 into Bofecillos Mountains.
[e]Translocation conducted in year 2012 into Nine Point Mesa.
[f]Translocation conducted in year 2014 into Sierra Vieja Mountains.
[g]Combined translocation conducted e for all years.

Table 2.5. Seasonal survival estimates (Ŝ) and adjusted standard errors (SE) for females the 2 years following release of translocated desert bighorn in the Trans-Pecos, Texas, USA.

	Bofecillos 2010[c]			Bofecillos 2011[d]			Nine Point Mesa[e]			Sierra Vieja[f]			Combined[g]		
Season[b]	n^a	Ŝ	SE	n^a	Ŝ	SE	n^a	Ŝ	SE	n^a	Ŝ	SE	n^a	Ŝ	SE
Year 1															
Gestating	25	1.00	0.00	29	0.73	0.08	22	1.00	0.00	23	1.00	0.00	99	0.85	0.01
Lambing	23	0.84	0.07	22	0.86	0.07	20	1.00	0.00	22	0.95	0.05	87	0.90	0.05
Lactating	22	0.95	0.05	18	0.84	0.08	17	0.94	0.06	22	1.00	0.00	79	0.87	0.04
Breeding	21	0.95	0.05	15	1.00	0.00	16	1.00	0.00	19	0.88	0.08	71	0.86	0.09
Year 2															
Gestating	18	1.00	0.00	15	1.00	0.00	16	1.00	0.00	17	0.92	0.08	66	0.90	0.10
Lambing	16	0.87	0.09	15	1.00	0.00	16	1.00	0.00	15	0.82	0.12	62	0.75	0.00
Lactating	11	1.00	0.00	14	0.93	0.07	12	1.00	0.00	13	1.00	0.00	50	0.80	0.00
Breeding	6	0.88	0.12	11	1.00	0.00	12	1.00	0.00	13	1.00	0.00	42	0.92	0.03

[a]Number of sheep radio-collared during time period.
[b]Biological seasons delineated as gestating (15 Nov-14 Feb), lambing (15 Feb-14 May), lactating (15 May-14 Aug), and breeding (15 Aug-14 Nov).
[c]Translocation conducted in year 2010 into Bofecillos Mountains.
[d]Translocation conducted in year 2011 into Bofecillos Mountains.
[e]Translocation conducted in year 2012 into Nine Point Mesa.
[f]Translocation conducted in year 2014 into Sierra Vieja Mountains.
[g]Combined translocation conducted for all years.

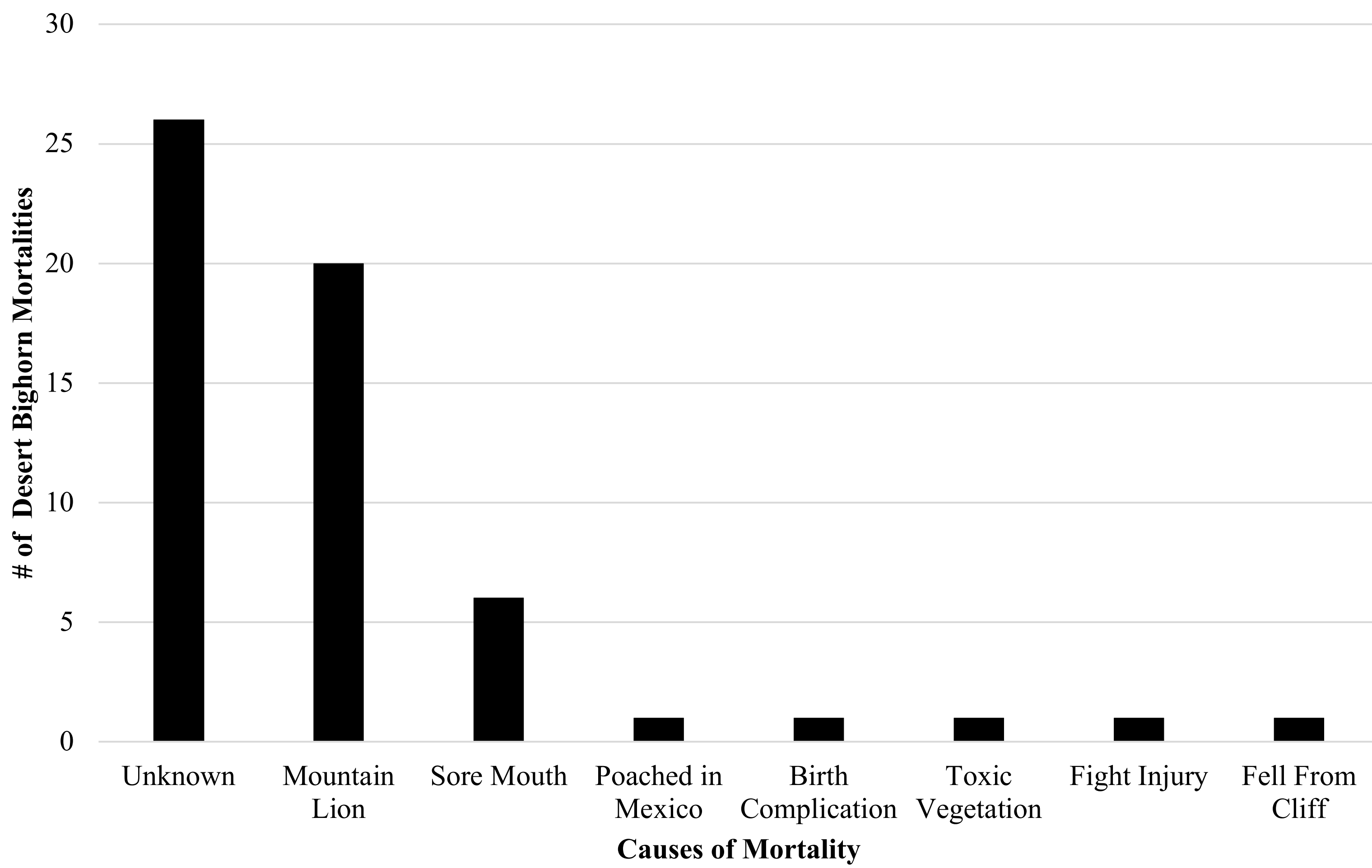

Figure 2.2. Causes of mortality of radio-collared desert bighorn translocated within the Trans-Pecos, Texas, USA.

habitat was not monitored through the duration of the study, it is possible that the temporal effect seen could have been due to changes in habitat. Because survival and mortalities varied across sites, it is possible habitat quantity, quality, and distribution across the landscape may have a greater effect on translocations than previously assumed (Le Gouar et al. 2012). Future research should include assessment of habitat use in conjunction with translocations to understand survival across different temporal and spatial scales.

Specific Mortality Causes

Despite desert bighorn being considered alternative prey for mountain lions (*Puma concolor*) when mule deer abundance is low (Anderson 1983), categorization of mountain lion kills on desert bighorn have been reported in multiple studies (Hayes et al. 2000, Kamler et al. 2002, Rominger et al. 2004). Using established mortality site investigation criteria, certain features were considered (e.g., caching, drag trails, and identification of canine marks) as most diagnostic for mountain lions. Of the 20 kills attributed to mountain lion, 10 (50%) were cached, 11 (55%) had distinguishable drag trails, and 14 (70%) had canine marks indicative of mountain lions. It is not uncommon for high numbers of mortality of translocated desert bighorn to be attributed to mountain lions (Wehausen 1996). Estimates of mountain lion predation may be conservative due the extent of time required to find mortalities and diagnostic methodology. Depending on the magnitude and duration of the predation, some bighorn sheep populations, especially small herds, could be extirpated. Under some circumstances, predation on bighorn sheep may need to be mitigated through short-term predator removal (McKinney et al. 2006).

Collectively, mountain lions were deemed to be the main cause of mortality. However, the majority of mortalities attributed to mountain lions occurred in Bofecillos Mountains (representing 18 of 20) whereas other sites observed only 1 mountain lion mortality each representing 9% total mortalities for Nine Point Mesa and 6% for Sierra Vieja. Unfortunately, for a great proportion of mortalities I was unable to determine cause of mortality. Many of the carcasses were degraded or had no signs of a conclusive cause of mortality suggesting other causes of mortality could be underrepresented. Since translocations were done at different periods, at separate locations, and with dissimilar sources of stock, it is possible that mountain lion predation could be a cyclic event subject to habitat and availability of alternative prey (McKinney et al. 2006).

Desert bighorn commonly are exposed to a number of diseases associated with domestic livestock and wild ungulates (Prestwood et al. 1974, Stauber et al. 1977, Jessup 1985, deVos 1989, Elliott et al. 1994), and disease outbreaks may contribute to desert bighorn population declines (Sandoval 1980, DeForge and Scott 1982, Cassirer et al. 1998, Monello et al. 2001). The parapox orf virus is the cause of contagious ecthyma in wild and domestic sheep (Robinson and Kerr 2001). This virus has been documented in bighorn sheep populations from Alaska to California, in addition to many other artiodactyls. Contagious ecthyma is characterized by the presence of lesions on the lips and udders of animals. Lesions can vary from small hardly-detectable crusts to thick, hard scabs that cover the entire face (Robinson and Kerr 2001). The lesions may persist for months on the animal (McKeever 1984) and can prove fatal as the animals starve due to their inability to forage. It is transmitted by direct contact with an infected individual or a detached scab containing the parapox orf virus. The prevalence of contagious ecthyma in a wild population depends on the number of individuals immune to the virus. Generally, animals can recover from the disease as immunity is established and lesions wear off, but severe outbreaks can result in mortalities (Robinson and Kerr 2001).

Unpredictable events such as drought could be related to ungulate population declines and could pose a challenge to conservation strategies (Bleich and Taylor 1998). Mortality of desert bighorn may be influenced by multiple variables acting within the same time frame (DeForge and Scott 1982, McNamara and Houston 1987, Gaillard et al. 1998).

Prolonged drought may correspond with downward trends in desert bighorn abundance associated with relatively poor forage production and quality that could lead to lower productivity and survival. Information regarding translocation, regional, and temporal trends in survival and cause-specific mortality rates are needed to guide sound management for restoration efforts. Predation, disease, weather, population density and food supply all play roles in limiting survival. Despite the value in this information, a difficulty in reintroduction programs is the identification of which variables influence survival and population growth.

Over the last decade, wildlife survival estimation has improved through the use of software designed for modeling complex survival functions (Harrell and Goldstein 1997, White and Burnham 1999, Williams et al. 2002). Program MARK (White and Burnham 1999) is the standard for wildlife survival estimation from capture-recapture and band-recovery studies. It is particularly powerful because it allows for construction and evaluation of a variety of survival models based on a large number of independent variables. Prior to Program MARK, no program easily combined the estimation of survival from both live and dead re-encounters (White and Burnham 1999). Survival of reintroduced populations can be modeled to make predictions that can be adjusted to allow for management decisions, such as predator control, disease management, and when to make releases for restoration. These simulations can assist with evaluating objectives, uncertainty, and effects of management decisions. However, I suggest model results from this study to be taken cautiously because of the sources covariate being skewed due to most translocations occurred from the EMWMA population. As well, translocations occurred at different years, which makes it challenging for comparison of seasons. Nonetheless, despite the difficulties for the construction of the models, outcomes displayed temporal scale to be an important influence to take into account for future restoration efforts. Further research with a meticulous study design and taking temporal scale into account should be able to produce reliable analysis for the creation of models.

In recent decades, survival estimates have had considerable focus on the development of reliable methods for application with free-ranging animals (Brownie et al. 1985, Burnham et al. 1987, Pollock et al. 1990, Lereton et al. 1992). Survival estimates derived from this study are based on free-ranging animals, which is crucial for addressing basic questions in desert bighorn population growth and in the development of population dynamics models, species conservation, and management programs (e.g., Lebreton et al. 1992, McCallum 2000). Annual survival of rams during the study were similar to those seen in other translocation efforts (Creeden and Graham

1997, Kamler et al. 2002). Annual survival of ewes during the study period was lower (0.71 and 0.88) when compared to similar studies that reported survival rates of 0.96 and 0.93 (Locke 2003, Rominger et al. 2004).

Management Implications

Managing mountain lions and prey are complex issues (Ross et al. 1996, Douglas and Leslie 1999, Ballard et al. 2001, Casey et al. 2005). Past studies (Douglas and Leslie 1999, Kamler et al. 2002, Rominger et al. 2004, McKinney et al. 2006) suggest management of mountain lion on a case-by-case basis could help translocations in areas where habitat might be marginal, or where populations of alternative prey are low. Localized removal of mountain lions may be integrated into management practices during early stages of restoration in order to allow the desert bighorn population to grow to a sustainable population size (McKinney et al. 2006). While managing large carnivores can be controversial, it is necessary to continue research for gathering of data to aid wildlife managers by reducing speculations with information from studies. Results from this study demonstrate mortality percentages attributed to mountain lions to be different at each site. Future studies with the use of GPS radio-collared should study predators (e.g., mountain lions, bobcats (*Lynx rufus*), golden eagles (*Aquila chrysaetos*) and desert bighorn for a better understanding of community dynamics affecting survival.

Translocations carry the risk of negative effects, and it is the responsibility of the managers to minimize this risk (Ewen et al. 2012), and such is the case of animals that may carry diseases and pathogens. Desert bighorn from herds infected with particular pathogens, such as sore mouth, should not be introduced into herds suspected to be free of a pathogen. Translocations of animals carrying a disease should be avoided if possible to prevent potential infections of other herds. Research to determine prevalence and spatial distribution of desert bighorn diseases in Texas are needed to minimize the risk of infecting populations when translocations occur.

CHAPTER III

EVALUATION OF TRANSLOCATION ALTERNATIVES FOR DESERT BIGHORN SHEEP

Introduction

Desert bighorn sheep (*Ovis canadensis*) occur throughout much of northern Mexico and the southwestern United States, distributed in naturally fragmented populations, often as small, isolated demes as well as metapopulations (Krausman and Leopold 1986, Bleich et al. 1990, Andrew et al. 1999). Carson (1941) believed there may have been as many as 1,000-1,500 desert bighorn living in Texas during the late 1800s. However, Texas was believed to have lost the last of its native desert bighorn by the 1960s (Kilpatric 1990). This population decline was believed to be due to a combination of unregulated hunting, competition and disease transmission from domestic sheep and goats, habitat fragmentation, and other unknown causes (Davis and Taylor 1939, Buechner 1960).

One successful approach for conservation of large mammals has been the translocation of animals into their former habitats (Krausman 2002). However, translocations of large ungulates or carnivores can be expensive and challenging (Beck et al. 1994, Biggins and Thorne 1994, Wolf et al. 1996, Fritts et al. 1997). Beginning in 1957, the Texas Game and Fish Commission (now Texas Parks and Wildlife Department [TPWD]) translocated desert bighorn from Arizona to a breeding facility to initiate the restoration process (Kilpatric 1990). Over the next 4 decades, 146 desert bighorn were transplanted to Texas facilities from Nevada (n = 107), Arizona (n = 31), Mexico (n = 6), and Utah (n = 2) (Brewer and Hobson 2000). By 1997, 237 desert bighorn had been captured from re-established Texas populations and released into other Texas mountain ranges (Cook 1994, Brewer and Hobson 2000).

Survival and population growth are ways of measuring translocation success (Singer et al. 2000), and both should occur if all resource needs of the individuals are fulfilled (Janke 2015). However, a long-standing problem in population biology has been understanding aspects and organismal traits that affect the stability and growth of populations (Muller and Huynh 1994). To better understand the influence of these variables, simulation models that mimic dynamics of relocated populations could become useful for the restoration of species (Lopez et al. 2000).

Theoretical models are used to aid the understanding of population dynamics (Grant et al. 1997), and are particularly useful for assessing demographics within heterogeneous and fragmented landscapes (Pulliam 1988, Akçakaya 2000, Morris 2003). Similarly, simulation-learning environments are having a profound impact on the way we learn and teach about complex problems in the natural sciences (Grant et al. 1997, Repenning et al. 1999). Since the late 1980s, scientists, educators, and engineers have suggested they could develop a greater understanding of phenomena if they could build and manipulate models of these occurrences (Bransford et al. 1999). Our understanding of any system constitutes a model of that system, because models are abstractions of reality (Grant et al. 1997, Armstrong and Reynolds 2012). Complex domains, such as population dynamics, can be depicted as a collection of inter-related items (e.g., stocks and flows in system dynamics) characterized by internal feedback mechanisms, delays, and uncertainties (Sterman 1994). System dynamics modeling tools, such as Stella (High Performance Systems Inc., Hanover, New Hampshire), enable users to experiment with complex systems and to develop perceptions of the mechanisms that govern dynamic interactions (Milrad 2002). Because the objective of reintroductions should be to establish self-sustained populations (meaning the population should be stable or growing in size), reintroduction success comes from population growth and persistence, which depend on demographics, specifically births, immigration, mortality and emigration (Converse and Armstrong 2016).

The purpose of this study was to assess different strategies for conducting translocations of desert bighorn in Texas. Specific objectives were to assess: (1) population growth by comparing the initial number of desert bighorn translocated; (2) initial sex ratios translocated; and (3) repeated translocations of desert bighorn into the same area.

Study Area

Located in the Chihuahuan Desert, the study site was situated at the southern point of the Trans-Pecos region of Texas (Figure 3.1). Elephant Mountain Wildlife Management Area (EMWMA) is bordered by private lands in Brewster County nearly 42 km south of Alpine, Texas. Within this 93-km^2 property, 20 desert bighorn were reintroduced in 1987 (TPWD 1998) and currently the population has grown to over 160 individuals (TPWD unpublished data).

Elevation of the site is 1,896 m above sea level and the top of the mountain rises 610 m from the surrounding areas. Annual mean precipitation is 33 cm, mostly in the form of monsoonal thunderstorms in the months of July, August, and September. Main vegetation types include desert scrub, desert grasslands, and oak-pinyon-juniper woodlands in much of the higher elevations. Vegetation in EMWMA is highly diverse

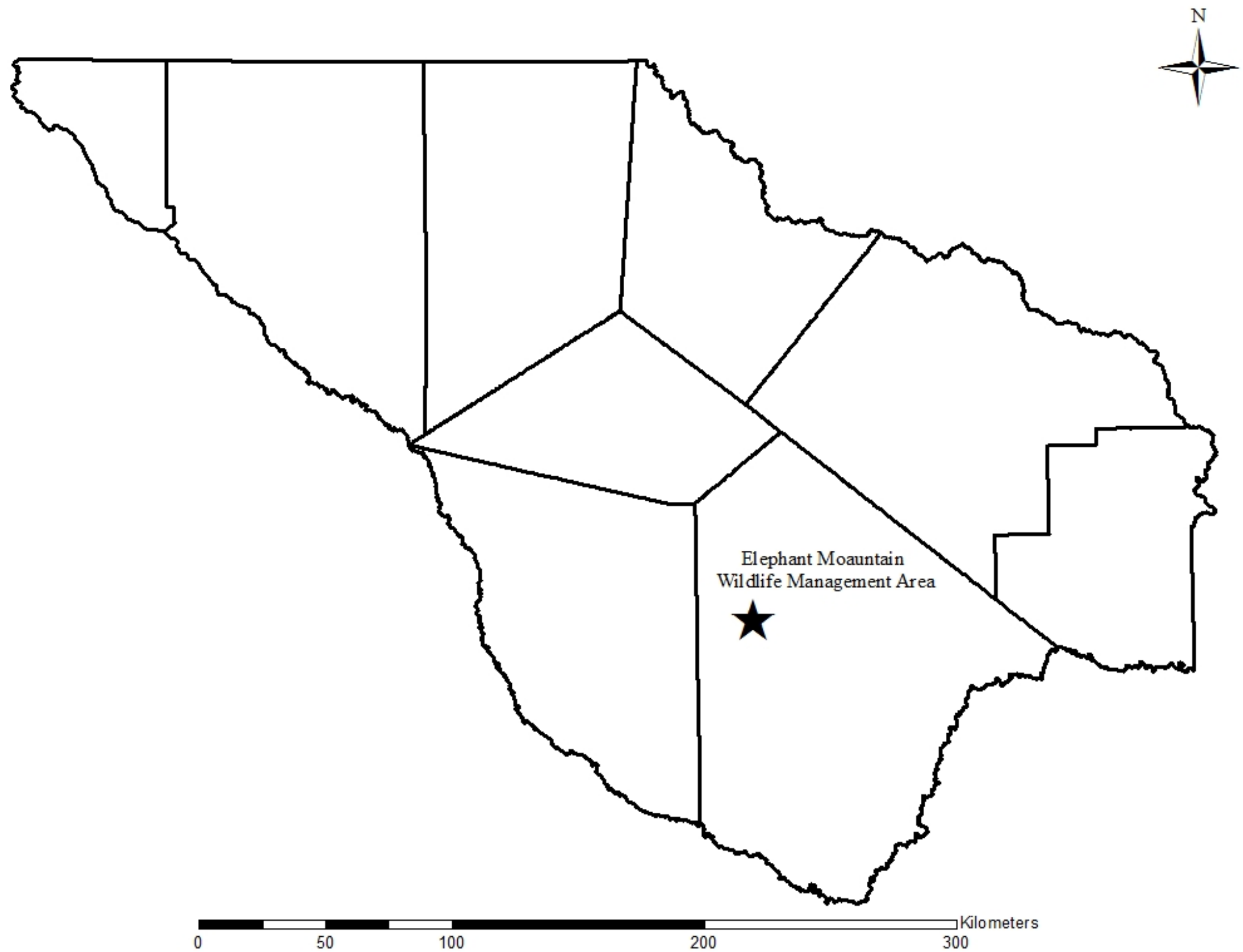

Figure 3.1. Location of Elephant Mountain Wildlife Management Area across the Trans-Pecos Texas, USA.

with typical plant species including creosote *(Larrea tridentata),* lechuguilla *(Agave lechuguilla),* acacia *(Acacia* spp.), ocotillo *(Fouquieria splendes),* prickly pear (*Opuntia* spp*),* sotol *(Dasylirion* spp.), mesquite *(Prosopis* spp), yucca (*Yucca sp*.), catclaw mimosa (*Mimosa biunicifera*), and mariola *(Parthenuem incanum).* Common grasses included black grama *(Bouteloa eriopoda),* blue grama *(Bouteloa gracilis),* chino grama *(Bouteloua ramosa),* Lehman lovegrass *(Eragrostis lehmanniana),* tobosa *(Pleuraphis mutica),* threeawns *(Atristida* spp.), sacaton *(Sporobulus* spp.), and tridens *(Tridens* spp.). Typical forbs include borage (*Borage* spp.), slender janusia (*Janusia gracilis*), goosefoot (*Chenopodium incanum*), and globemallow (S*phaeralcea* spp.) (Brewer and Harveson 2007).

Model Description

Overview

I developed a sex- and age-structured compartment model based on difference equations ($\Delta t = 1$ year) to simulate demographic outcomes of desert bighorn reintroductions at EMWMA in the Trans-Pecos, Texas, USA under different management practices. The model, which is programmed in STELLA Architect 1.7.1®, represents recruitment of annual cohorts of individuals as lambs (L, young-of-the-year), their subsequent survival as yearlings (Y) and as adult males (AM) and adult females (AF), and also represents the reintroduction of adult males and adult females via translocations (TM and TF, respectively) (Figure 3.2). Recruitment (R) is a function of the number of AF and a density-dependent per capita birth rate (BR) which depends on total population size (P). Lamb deaths (LD) yearling deaths (YD), adult male deaths (AMD), and adult female deaths (AFD) are functions of density-dependent

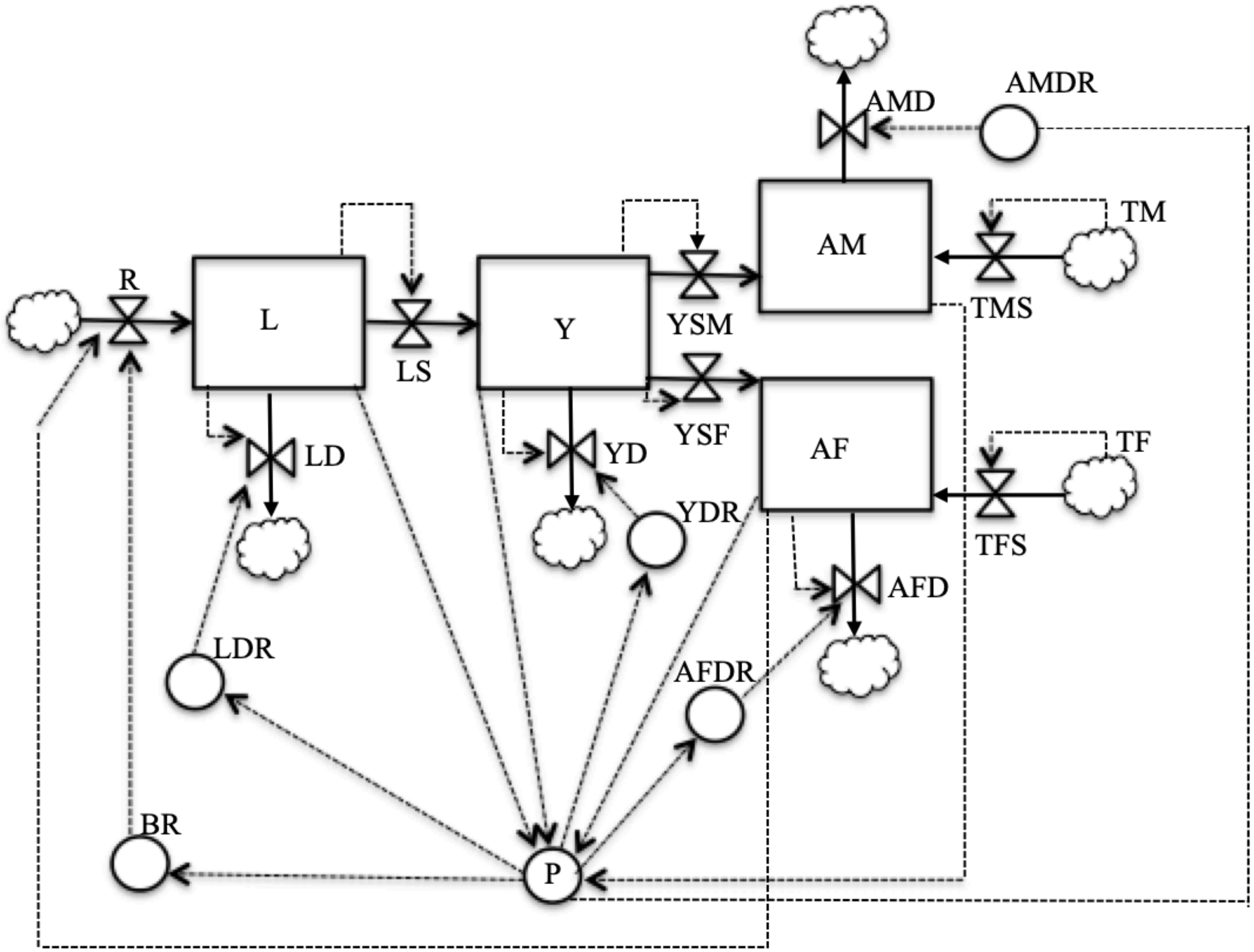

Figure 3.2. Conceptual model representing population growth of desert bighorn in Trans-Pecos, Texas.

per capita death rates (LDR, YDR, AMDR, and AFDR, respectively) which depend on P. L survive (LS) to become Y, and Y survive (YSM or YSF) to become either AM or AF. Timing of entry of translocated adult males (TMS) and translocated adult females (TFS) depends on the scenario being simulated. LDR, YDR, AMDR, AFDR, TMS, and TFS are stochastic variables, R is deterministic.

Parameterization, Calibration, and Verification

I parameterized YDR, AMDR, AFDR, TMS, and TFS based on previous studies done in the region (Locke 2002, Janke 2015, Cross 2016). I parameterized BR and LDR, as well as adult sex ratios (40% males, 60% females), based on TPWD surveys conducted over a 30-year period (1987-2016, TPWD unpublished data). The model consists of the following equations.

$P_t = (L_t + Y_t + AM_t + AF_t)$ (eq. 3.1)

$R_t = BR_t * AF_t$ (eq. 3.2)

$BR_t = 1.933 - 0.0083 * P_t$ if $124 \leq P_t \leq 160$ (eq. 3.3)

$BR_t = 0.9$ if $P_t \leq 124$ (eq. 3.4)

$BR_t = 0.6$ if $P_t > 160$ (eq. 3.5)

$L_{t+1} = L_t + (R_t - DL_t)\ \Delta t$ (eq. 3.6)

$LD_t = LDR_t * L_t$ (eq. 3.7)

$LDR_t = -2.311 + 0.0194 * P_t$ if $124 \leq P_t \leq 160$ (eq. 3.8)

$LDR_t = 0.1$ if $P_t \leq 124$ (eq. 3.9)

$LDR_t = 0.8$ if $P_t > 160$ (eq. 3.10)

$LS_t = L_t - LD_t$ (eq. 3.11)

$Y_{t+1} = Y_t + (LS_t - YD_t)\ \Delta t$ (eq. 3.12)

$YD_t = YDR_t * YD_t$ (eq. 3.13)

$YDR_t =$ if $P_t \leq 160$ (RANDOM (0.01, 0.1)) else
if $P_t \geq 161$ (RANDOM (0.05, 0.15)) else 0.15 (eq. 3.14)

$YSM_t = (Y_t - YD_t) * 0.4$ (eq. 3.15)

$YSF_t = (Y_t - YD_t) * 0.6$ (eq. 3.16)

$AM_{t+1} = AM_t + (YSM_t - AMD_t)\ \Delta t$ (eq. 3.17)

$AMD_t = AMDR_t * AMD_t$ (eq. 3.18)

$AMDR_t =$ if $P_t \leq 160$ RANDOM (0.01, 0.1) else
if $P_t \geq 161$ RANDOM (0.05, 0.15)
else 0.15 (eq. 3.19)

$AF_{t+1} = AF_t + (YSF_t - AFD_t)\ \Delta t$ (eq. 3.20)

$AFD_t = AFDR_t = AFD_t$ (eq. 3.21)

$AFDR_t =$ if $P_t \leq 160$ RANDOM (0.01, 0.1) else
if $P_t \geq 161$ RANDOM (0.05, 0.15)
else 0.15 (eq. 3.22)

$TM_t =$ Number of translocated males (eq. 3.23)

$TMS_t = TM_t *$ RANDOM (0.49, 0.90) (eq. 3.24)

$TF_t =$ Number of translocated females (eq. 3.25)

$TFS_t = TF_t *$ RANDOM (0.49, 0.90) (eq. 3.26)

I calibrated the density-dependent relationship of BR to P (eqs. 3.3-3.5) and of LDR to P (eqs. 3.8-3.10) such that the model was capable of simulating the historical population trend at the EMWMA, assuming that carrying capacity of the Area was 160 individuals

(1987-2016, TPWD unpublished data). I verified that the calibrated model was capable of mimicking the historical trend by running 50 replicate stochastic simulations, each representing the initial historical translocation translocation of 10 adult males and 10 adult females, and comparing simulated population dynamics with the historical trend (Figure 3.3). I assumed a closed population, since there has been no documented emigration or immigration from or to the EMWMA mountain range, even after carrying capacity was presumed to have been reached.

Projections

I simulated the potential demographic outcomes of desert bighorn reintroductions at the EMWMA under several hypothetical management scenarios in which I varied (1) sex ratio of reintroduced animals, (2) timing of a second reintroduction, and (3) number of animals reintroduced. I simulated a sex ratios of (1) ≈1M:3F (13M:37F), (2) ≈1M:2F (17M:33F), (3) 1M:1F (25M:25F), (4) ≈2M:1F (33M:17F), and (5) ≈3M:1F (37M:13F), assuming a single reintroduction of 50 adults; second reintroductions which occurred 1, 2, 3, 4, and 5 years after the initial reintroduction, assuming that both the initial reintroduction and the second reintroduction consisted of 25 adults (12M:13F); and single reintroductions of (1) 16, (2) 32, (3) 48, (4) 64, and (5) 80 adults, which represented 10, 20, 30, 40, and 50% of the assumed carrying capacity of the Area (160 individuals), assuming a 1:1 sex ratio. I ran 50, 30-year, replicate stochastic simulations of each scenario, and monitored the number of years required for the population to reach the assumed carrying capacity.

Simulation results indicated that the number of years required for the population to reach carrying capacity (1) was reduced when proportionally more females than

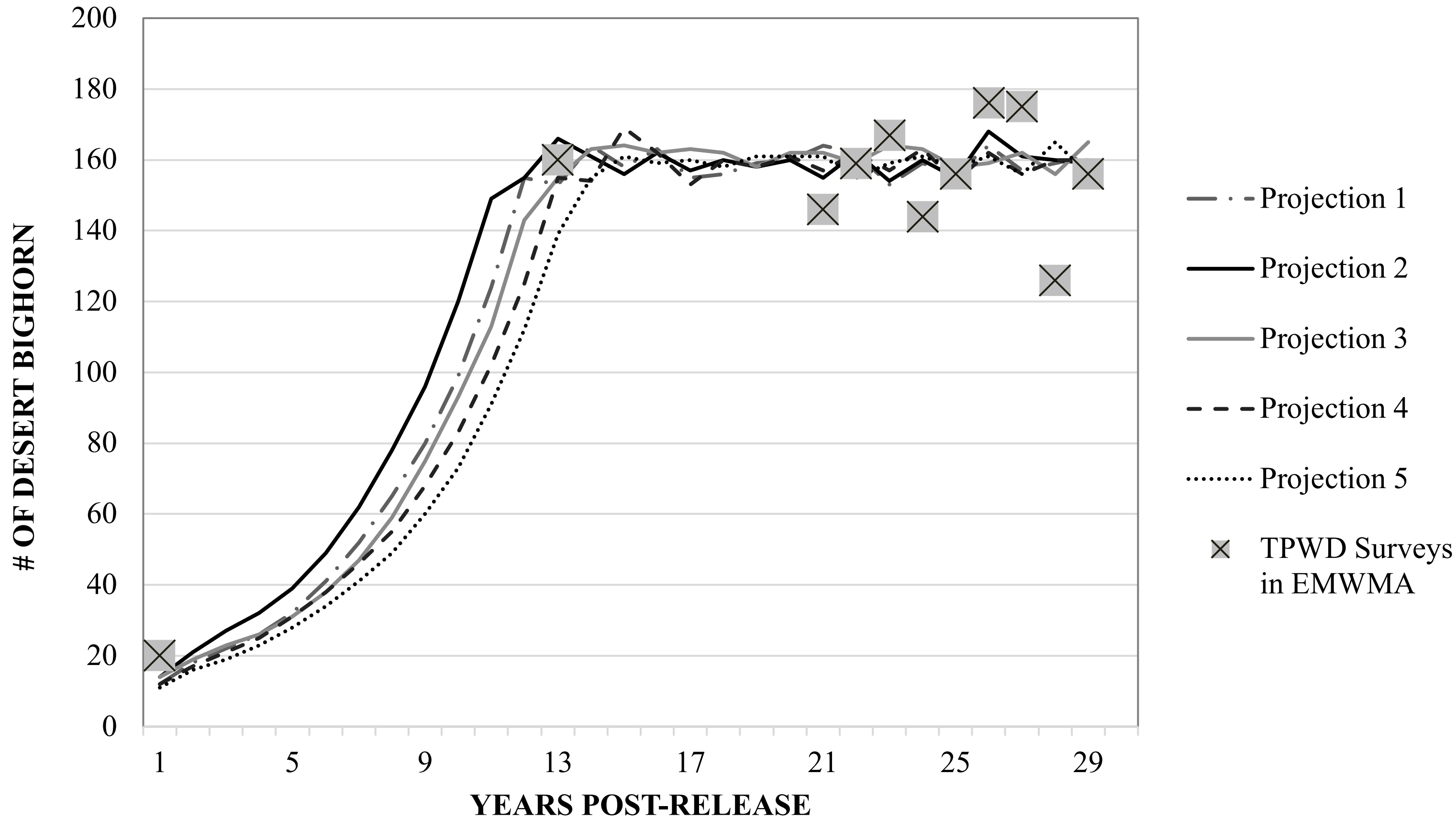

Figure 3.3. Comparison of simulated population dynamics of desert bighorn (lines) with the historical population trend at the Elephant Mountain Wildlife Management Area (crosses) based on survey results from the Texas Parks and Wildlife Department (1987-2016 TPWD unpublished data). Lines represent results from 5 simulations, randomly selected from 50 replicate stochastic simulations.

males were reintroduced (Figure 3.4), (2) was reduced slightly more by shorter than by longer time lags between the initial and the second reintroduction, although differences were negligible (Figure 3.5), and (3) was reduced when a larger number of animals (representing a larger proportion of carrying capacity) was reintroduced (Figure 3.6).

Discussion

Forming criteria is important for evaluating if a model is acceptable for its intended use (Rykiel 1996). The importance of a model depends on how useful it is for selecting management actions that can fulfill management objectives (Converse and Armstrong 2016). While validation of the model is at the center of much debate, it has been suggested that it be referred to as "model evaluation" based on how potentially useful the model is (Grant et al. 1997). No model will produce fully accurate projections, as all models have known or unknown inaccuracies (Converse and Armstrong 2016). However, based on rationality of model structure, interpretability of functional relationships, model behavior, and demographic parameter projections that fit rates reported at EMWMA, I consider this model useful for desert bighorn translocation efforts.

Natural systems are complex, vary with location, and are therefore understudied, particularly as it pertains to management decisions, due to lack of knowledge regarding how the system will respond to management alternatives (Runge et al. 2011). While there is a great deal to learn about translocation of endangered or sensitive species, success should increase as we gain knowledge with regards to the needs of the species and as appropriate

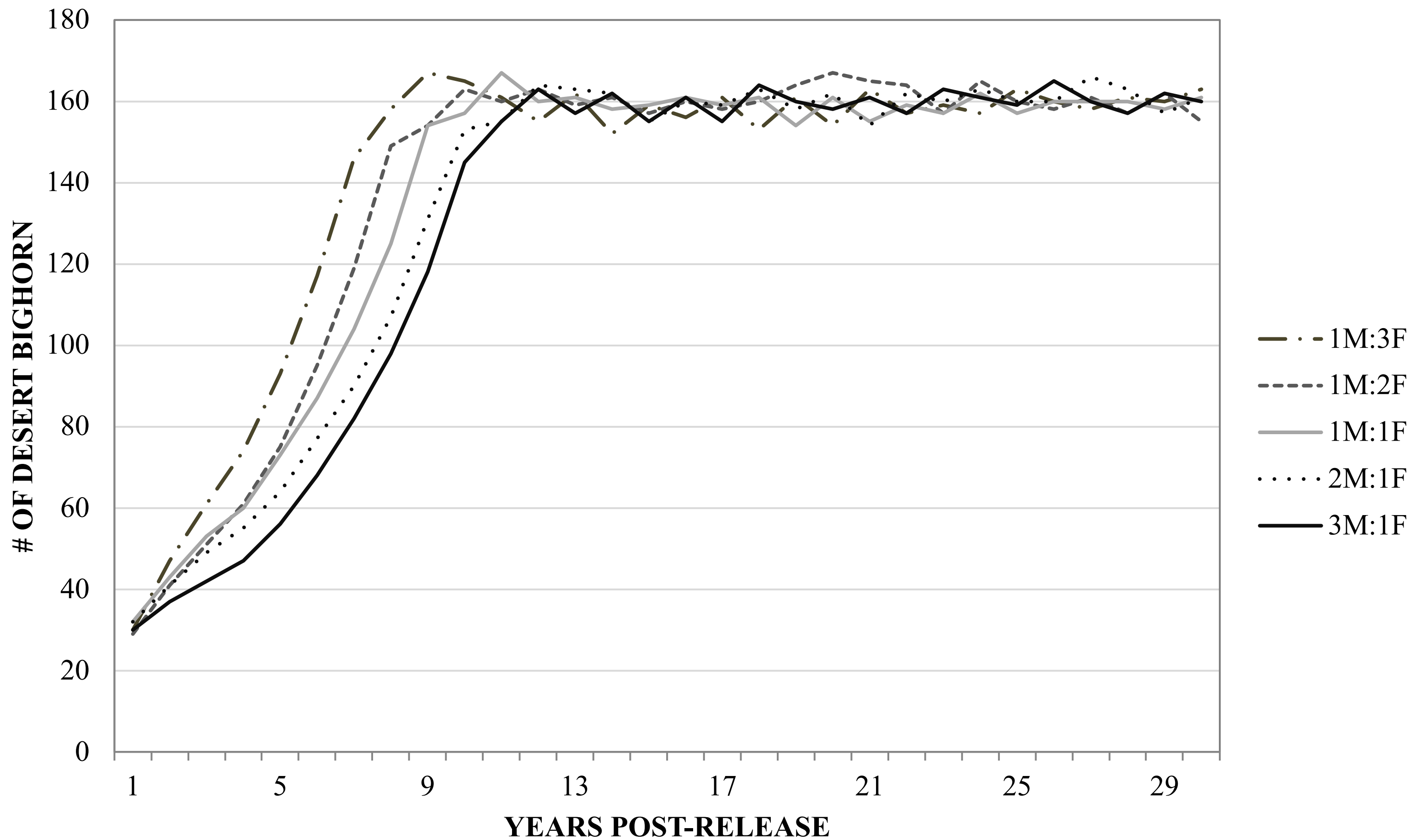

Figure 3.4. Comparison of simulated population dynamics of desert bighorn the Elephant Mountain Wildlife Management Area, assuming different sex ratios. Lines represent results from 1 simulation from each scenario, randomly selected from 50 replicate stochastic simulations of each scenario. See text for details.

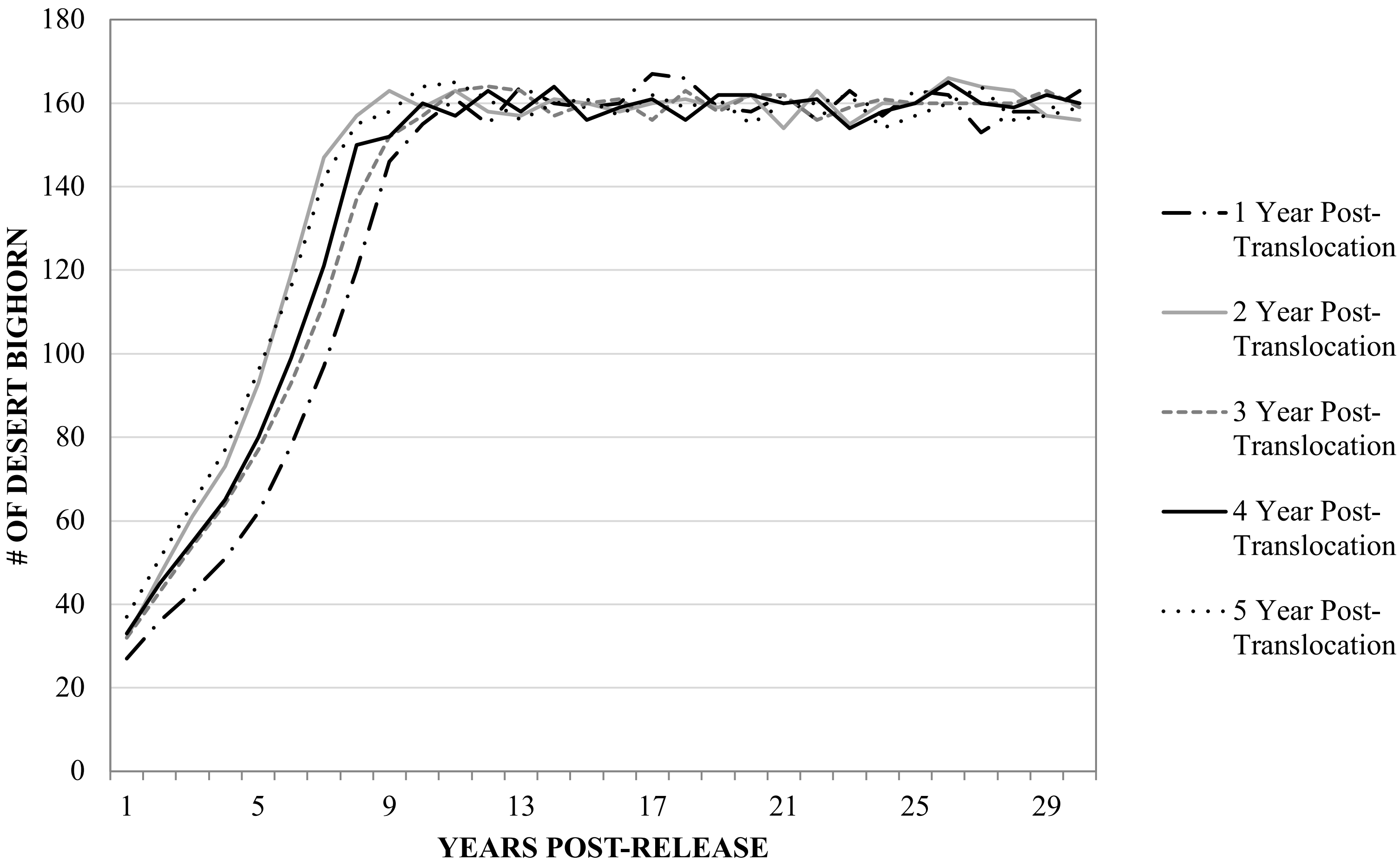

Figure 3.5. Comparison of simulated population dynamics of desert bighorn the Elephant Mountain Wildlife Management Area, assuming different timing of a second reintroduction. Lines represent results from 1 simulation from each scenario, randomly selected from 50 replicate stochastic simulations of each scenario. See text for details.

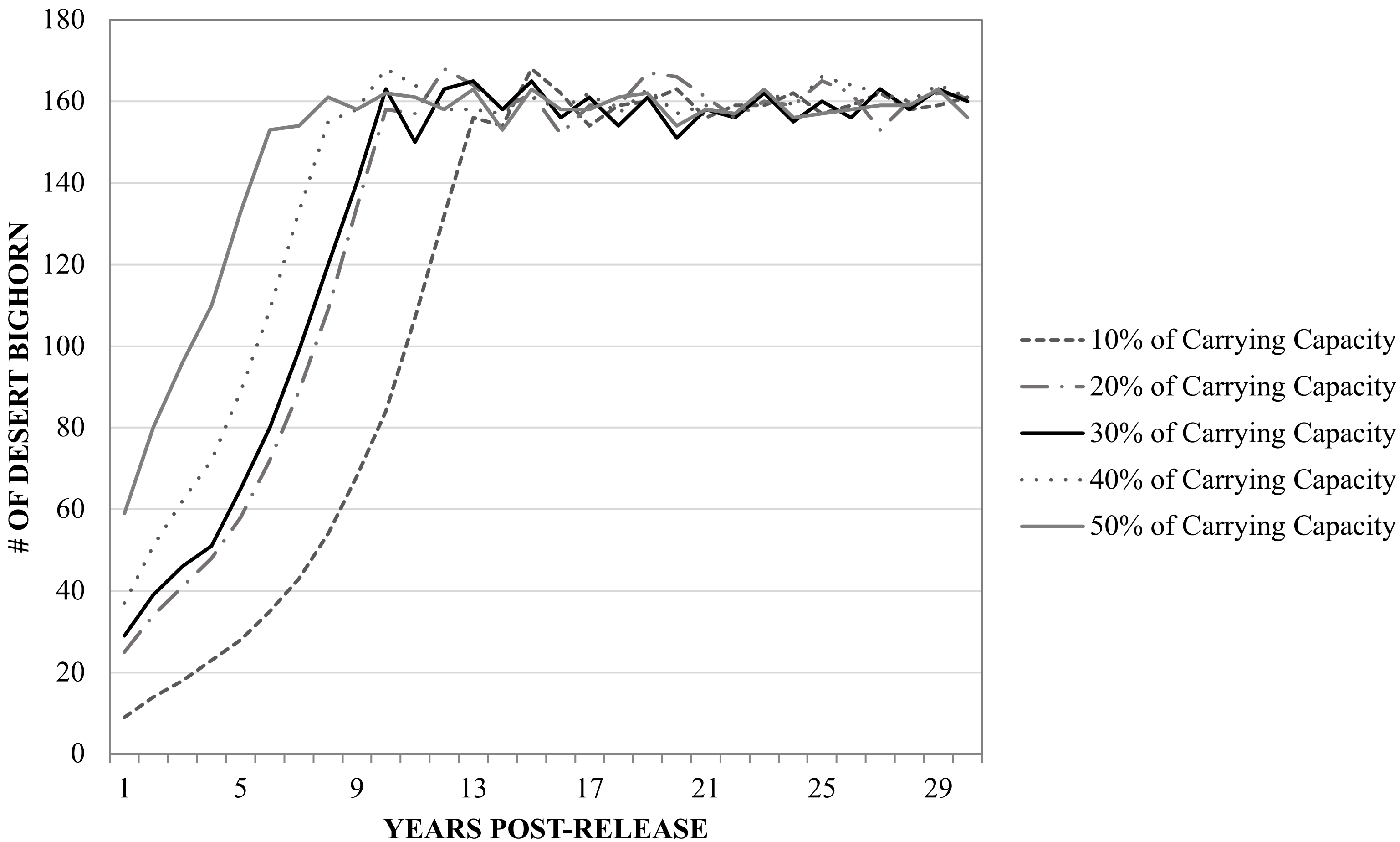

Figure 3.6. Comparison of simulated population dynamics of desert bighorn the Elephant Mountain Wildlife Management Area, assuming different number of animals reintroduced. Lines represent results from 1 simulation from each scenario, randomly selected from 50 replicate stochastic simulations of each scenario. See text for details.

management techniques continue to be developed (Seddon et al. 2007). In a decision-making setting, population models are valuable for estimating the influences of alternative management activities on populations of interest, such as the effects of translocation population demographics on population growth rate. Comparing projections and recognizing sensitivities can aid in determining monitoring priorities by identifying observation-worthy attributes (McCarthy et al. 2012). Simulation outcomes from this study revealed important demographic outcomes, which will allow for development of appropriate management techniques, such as where additional attention to demographic rates could be implemented to promote a greater influence on population growth. These projections showed population growth to be mostly affected by alterations in survival of lambs, birth rates, and adult female survival.

A subject for management strategies that includes reintroductions is whether to do a single translocation or follow-up translocations to the same site (Armstrong and Ewen 2001). This is possibly due to expected higher probability of establishment if release is staggered into 2 or more translocations (Griffith et al. 1989) or logistical reasons such as the release of animals as they become available (Armstrong and Ewen 2001). Projections from simulations showed the use of follow-up translocations were more effective the sooner they were conducted after the first translocation. A longer period of time between initial and follow-up translocations correlated with a longer time to reach biological carrying capacity for the EMWMA population. This is perhaps due to the population nearing biological carrying capacity as translocation of new individuals is postponed, which according to the model assumptions would lower reproduction and survival of lambs as the population nears the assumed capacity.

Subsequent restocking with low numbers of females showed to have the lowest impact on population growth. Translocations with a disproportionate number of males many years after initial translocation efforts should be avoided if possible because it would be likely to have the least effect on population growth.

Population growth could be restricted by unfavorable environment circumstances and competition for limited resources (Grant et al. 1997). Precipitation has important effects on the nutritional status of individual herbivores, and nutritional status has been demonstrated to be fundamental for the health, survival, and reproductive processes of wild and domestic herbivores (Sams et al. 1996, Keech et al. 2000, Cook et al. 2004). Precipitation is possibly related to population growth, birth rates, and survival of lambs, and it is likely that the relationship of precipitation on forage quantity and quality would allow for increased survival, nutrition, and population growth. The model did not include stochastic environmental effects that could have affected the population growth. Assuming that all suitable habitats in the Trans-Pecos would have similar results as those seen in EMWMA may not be entirely representative due to differences in plant community, spatial distribution of habitat, competition for resources with other species, different predation rates, and precipitation across other mountains. Therefore, future analyses should try to incorporate climate, particularly precipitation, competition for resources, and predation effects into population growth modeling of desert bighorn at each particular site.

Restoration programs tend to be expensive to initiate and sustain (Kleiman 1989). Models simulating dynamics of relocated populations can be valuable for restoration by evaluating restocking strategies prior to initiating costly restoration programs (Lopez et al. 2000). It would be wasteful to continue releasing animals beyond the level needed to achieve a self-sustaining population (Armstrong and Ewen 2001, Schaub et al. 2009, Wakamiya and Roy 2009), when a population is not increasing due to unknown causes, or when the population is incapable of self-sustenance despite future translocations (i.e., limited resources). Given that there may be no cost to delaying the decision for translocations, research can give a stronger indication of whether a restocking translocation is justified (Armstrong and Ewen 2001).

Management Implications

Uncertainty is present in reintroduction programs (Armstrong and Seddon 2008) and the long-recognized challenge is to manage wisely when confronted with doubt (Holling 1978, Walters 1986). Because of this, models have been used to predict the potential fate of certain populations, and their main strength is in their ability to evaluate different management alternatives in small populations (Bustamante 1998). With the knowledge gained from this study, simulations of different release methods and demographics, reintroductions of desert bighorn in Texas could use valuable alternatives to select the timing and demographics of individuals to be released. Through the use of this model, reintroductions should focus on maximizing habitat resources in order to increase the probability of successful translocations. Forthcoming models should include structured decision frameworks that explicitly define objectives, describe uncertainties and assumptions, and weigh costs and benefits of possible outcomes (Armstrong and Reynolds 2012).

Because, reintroductions may be regarded as a series of challenging decisions made under uncertainty (Converse and Armstrong 2016), with limited amount of resources, ambiguities in population trends, environmental effects, and absence of information with respect to common problems. Results from this study exemplified the need to reduce the uncertainties for decision-making when translocations occur. However, the model also displayed the need for additional studies to assess lamb and adult survival of resident populations in comparison to translocated desert bighorn. These 2 variables were seen to be key mechanisms for population growth and there is little knowledge of resident populations. Assumptions made in the creation of the model should be tested to better assess management alternatives and reliability of the model. By evaluating how accurate the assumptions made for the model are, the model will increase strength and be able to reduce uncertainties of generated projections for different management alternatives.

Another challenge in restoration programs is that objectives may differ among the manager of the area animals are released or captured, agencies involved, and the entity paying for the translocation (Converse and Armstrong 2016). Model projections, such as the ones from this study, can provide estimates of the time necessary to reach objectives, cost of different management actions, and a quantitative evaluation of the different alternatives (Maguire and Lacy 1990, Lindenmayer and Possingham 1996, Gaona et al. 1998). Additionally, simulations can be used to determine whether current practices are still adequate, if management techniques could be improved, or if reintroductions should continue to occur (Bustamante 1998, Green et al. 1996, Nolet and Baveco 1996). Model outcomes from this study can assist in building an assessment of desert bighorn reintroductions prior to taking action by projecting management alternatives and identifying where efforts for conservation should be concentrated. Results from the analysis demonstrated that the greatest effect on population growth was seen when a greater number adult females were translocated and when survival of lambs was increased. As well, re-stocking of self-sustained populations appears to have a small influence on population growth the longer time intervals are delayed. If re-stocking occurs, it should be done as soon as possible since a larger initial population appeared to have a more rapid growth than smaller populations. Additionally, upcoming research focused in the establishment of new populations should emphasize efforts in translocating large number of desert bighorn with high ratios of females to males and aid in lamb survival.

Translocations of animals are becoming more numerous, but detailed studies of reintroductions are rarely available (Scott and Carpenter 1987, Griffith et al. 1989, Hodder and Bullock 1997) despite the fact that they are essential for conservation biology (Wolf et al. 1996, Griffin et al. 2000). While there are some early reintroduction success stories (Stanley Price 1989, Cade and Burnham 2003), the failure of other reintroductions meant overall low success rates (Griffith et al. 1989), and the failed situations were not improved by absence of monitoring, which meant timing and cause of failure were not documented (Seddon et al. 2007), increasing doubt of effectiveness of management practices.

Upcoming reintroductions of desert bighorn should continue to monitor the animals post-release to gather data that could allow for future assessment and improvement of models. Simulations with data generated from small sample sizes can lead to uncertainty in projections by not considering in long-term environmental variability and effects of the translocation itself (Armstrong and Ewen 2001). Without monitoring reintroductions and implementing studies of vital demographics (e.g., birth, lamb survival, yearling and adult female survival), programs are guided blindly without proper assessment of goals, understanding of what contributed to accomplishing the goals, or what prevented the goals from being achieved. Additionally, future effort to aid the model should incorporate species interactions, allocation of resources, and monitoring of habitat.

CHAPTER IV

HABITAT-BASED PROBABILITY DISTRIBUTION MODEL FOR DESERT BIGHORN SHEEP

Introduction

Restoration efforts for desert bighorn (*Ovis canadensis*) have relied heavily on translocations (Bailey 1990), with over 2,000 individuals being translocated since 1978 (Krausman 2000). Presently >50% of bighorn populations are from the result of translocations (Bailey 1990), however, the success rate of 6 western states was only 41% between 1923–1997 (Singer et al. 2000). A frequently cited reason for unsuccessful translocation efforts has been insufficient knowledge of what determines habitat (Wolf et al. 1988, Griffith et al. 1989). Without a delineation of where current suitable ranges occur and the environmental characteristics of these ranges, translocations have a low chance of success regardless of the number of translocated animals (Griffith et al. 1989). The objective of reintroductions is to increase the viability and survival of a species (Burgman et al. 1993). Therefore, understanding use of habitat and distribution of the species across a landscape level is vital for conservation efforts.

The environmental requirements of desert bighorn are important factors that could influence population fluctuations and determine what habitat characteristic are sought by the species (Miller and Gaud 1989). McCarty and Bailey (1994) suggested that visibility, lack of competition for water, exclusion of domestic sheep, and protection from human intrusion were essential factors to consider when designating optimal habitat. Topography has also been acknowledged as an important habitat variable and could provide important sources of cover for desert bighorn when bedding, lambing, and seeking escape cover (Geist 1971, McQuivey 1978, Van Dyke 1978, Hansen and Deming 1980, McCarty and Bailey 1994). Elevation classes utilized by desert bighorn have a documented range from 78 m below sea level in Death Valley, California (Welles and Welles 1961) to 4,267 m above sea level in the White Mountains of California (Kovach 1979). Any preference, or lack of preference, for specific elevations by desert bighorn could signify that their presence is more correlated to the proximity of other habitat variables such as distance to water and escape terrain (Krausman et al. 1999). However, despite desert bighorn being considered habitat specialists (Geist 1971), the relationships within habitat variables and their effect on habitat selection by desert bighorn is complex and poorly understood (Krausman and Leopold 1986). A general lack of knowledge regarding environmental mechanisms and the roles they play for desert bighorn in Texas could have detrimental consequences on restoration and management decisions.

To aid reintroduction efforts and increase their rate of success, several studies have explored the use of analytical methods to assist in the identification of suitable ranges before translocation occurs (Cook et al. 2009). Although a number of qualitative habitat rating procedures have been developed to evaluate desert bighorn habitat (Hansen and Deming 1980, Holl 1982), recent Geographic Information System (GIS) and a landscape approaches have increase the success of several restoration programs (Johnson and Swift 2000, Singer et al. 2000, Zeigenfuss et al. 2000, Locke et al. 2005). Additionally, recent advances in global positioning system (GPS) telemetry (Haller et al. 2001, Hulbert and French 2001, Cagnacci et al. 2010) and spatial technologies (e.g., ArcGIS) have provided opportunities for both more accurate and detailed information to be collected in regard to animal ecology.

Species distribution modeling first commenced during the 1990s (Osborne and Tigar 1992, Buckland and Elston 1993, Franklin 1995) and since then has continued to evolve due to new technological advancements (Osborne and Seddon 2012). These types of models are becoming increasingly important as conservationists attempt to comprehend species distributions while confronted with changing environments, invasive species, and other challenges (Yackulic et al. 2013). These models allow researchers to display quantitative relationships between the probability of occurrence of a species with one or more characteristics of their environment (Dorazio 2012). The results from these models can be useful for restoration ecology as they provide a predictive measure of occurrence for a species over its potential geographic range (Scott et al. 2002).

Models can be valuable in understanding the ecology of a species and are essential when making management decisions for the recovery of an endangered species (Turner et al. 2004). Therefore, the objectives of this study were to (1) quantify the relation between habitat variables used by desert bighorn, and (2) identify the distribution of such habitat across a landscape for desert bighorn within the Trans-Pecos region of Texas.

Study Area

The Trans-Pecos region of Texas encompasses 9 counties (Brewster, El Paso, Culberson, Hudspeth, Jeff Davis, Pecos, Reeves, Presidio, and Terrell), which are located at the western edge of the state. Within the larger Chihuahuan Desert

Ecoregion, the Trans-Pecos region is bordered to the east by the Pecos River, to the west and south by the Rio Grande River, and to the north by the New Mexico state line (Hatch et al. 1990) (Figure 4.1). Elevations within this area range from 762 m to 2,667 m and include mountain ranges such as Baylor, Beach, Christmas, Chinati, Chisos, Sierra Diablo, Sierra Vieja, and Van Horn, (Powell 1998). Annual precipitation varies from 200 to 460 mm, as compared to the Texas average of 700 mm, and accumulates mostly in the form of monsoonal thunderstorms during July, August, and September. Higher elevations receive more rainfall (300-460 mm) than do the lowlands and basins (200–300 mm).

Vegetation across the Trans-Pecos is vastly diverse. In much of the higher elevations, the main vegetation types include desert scrub, desert grasslands, and oak (*Quercus spp.*) pinyon-juniper woodlands. Typical plant species included junipers (*Juniperus spp*.), creosote (*Larrea tridentata*), lechuguilla (*Agave lechuguilla*), acacia (*Acacia spp*.), ocotillo (*Fouquieria splendes*), prickly pear (*Opuntia spp.*), sotol (*Dasylirion spp.*) mesquite (*Prosopis glandulosa*), and mariola (*Parthenuem incanum*). Common grasses included black grama (*Bouteloa eriopoda*), blue grama (*Bouteloa gracilis*), chino grama (*Bouteloua ramosa*), Lehman lovegrass (*Eragrostis lehmanniana*), tobosa (*Pleuraphis mutica*), threeawns (*Atristida spp.*), sacaton (*Sporobulus airoides*), and tridens (*Tridens spp.*). Most relevant forbs include basketflower (*Centaurea americana*), buffalobur (*Solanum rostratum*), common broomweed (*Xanthocephalum dracunculoides*), doveweed (*Croton* spp.), erect dayflower (*Commelina erecta*), pigweed (*Amaranthus* spp.), snakeweed (*Guiterrezia*

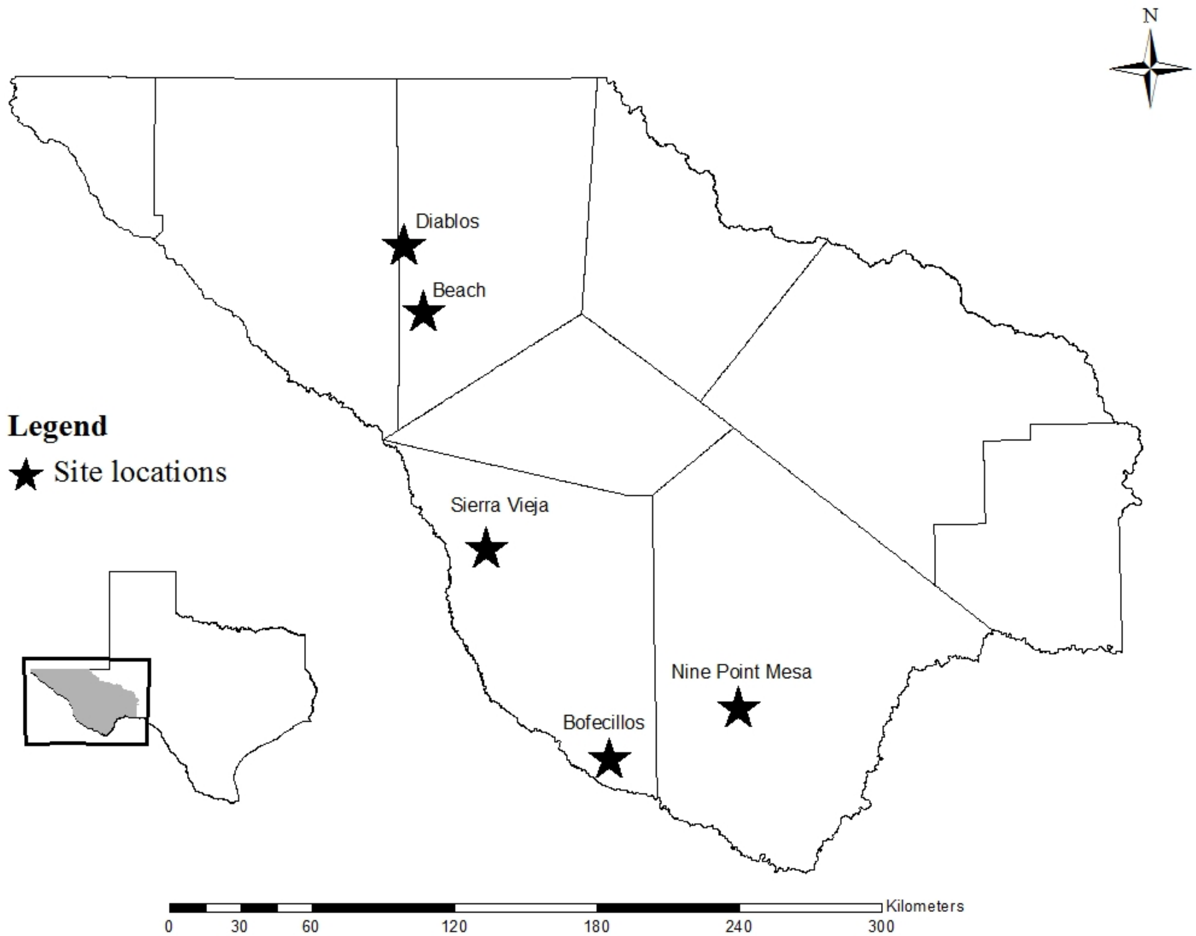

Figure 4.1. Site locations within the Trans-Pecos Texas, USA where desert bighorn were outfitted with GPS collars.

sarothare), sunflower (*Helianthus* spp.), and western ragweed (*Ambrosia cumanensis*) (Brewer and Harveson 2007).

Methods

Desert bighorn was captured from Elephant Mountain Wildlife Management Area (EMWMA) in December 2010 (12 M, 34 F), Sierra Diablos Meta Population (SDMP) in December 2011 (19 M, 76 F), EMWMA in December 2012 (20 M, 20 F), EMWMA in January 2014 (16 M, 30 F), and SDMP in January 2015 (8 M, 10 F) using a net gun fired from a helicopter (deVos et al. 1984, Krausman et al. 1985). Upon capture, each individual was blindfolded, hobbled, and transported to a central staging area where they were fitted with GPS collars. Sex, age, and physical condition were recorded for each individual. A veterinarian inspected each animal, administered antibiotics, and took blood samples. Once all data was collected and all individuals were equipped with collars, the ewes were placed in modified livestock trailers and rams were placed into modified crates for transportation. Individuals from captures that occurred in 2010 and 2011 were translocated to Big Bend Ranch State Park (BBRSP), those from 2012 were translocated to Nine Point Mesa, 2014 individuals were translocated to Sierra Viejas Mountains, and 2015 individuals were released in the same site they were captured (SDMP).

Global positioning system (n = 172) collars was allocated on desert bighorn. GPS collars were programmed to record locations at intervals between 1– 8 hours and to stay on the individual for 300 days (n = 8), 12 months (n = 3), or 25 months (n = 161). In 2010, 35 individuals (10 M, 25 F) were released with Lotek GPS 3300 collars programmed to gather GPS locations every 3 hours for a total deployment period of 25 months. In 2011, 35 (10 M, 25 F) GPS Advanced Telemetry Systems (ATS) G2110D GPS collars and 8 (4 M, 4 F) North Star NSG-D1 satellite collars were deployed on desert bighorn and were programmed to collect GPS locations every 5 hours for 25 months. For 2012, 27 ATS G2110D GPS (12 M, 15 F) collars, 8 ATS G2110E2 Iridium collars (1 M, 7 F), and 5 North Star satellite collars (5 M) were deployed on desert bighorn. Each collar was set to record locations every 5 hours. All collars utilized for this study were had mortality sensors set at 8 hours. In 2014, 31 desert bighorn (9 M, 22 F) were radio-collared with ATS G2110D GPS and 5 (4 M, 1 F) were collared with ATS G2110E2 Iridium satellite collars. All collars were programmed to acquire locations every 5 hours and had their mortality sensors established at 8 hours.

Very high frequency (VHF) beacons were on from 0800–2000 hours and did not transmit VHF on Sundays to increase duration of collar battery. Collars switched to mortality mode (80 beeps/minute [bpm] instead of 40 bpm) when collars were inactive for ≥8 hours. Once a mortality signal was acquired from a collar, the final location of the sheep and the collar status was recorded. Telemetry was then conducted using a receiver and antenna (Yagi, folding directional antenna) to locate the collar. If needed, aerial telemetry was utilized using fixed-wing aircrafts with 2 H-antennas mounted to the wing struts. When mortalities were found, an investigation was conducted to determine the cause of death and to later facilitate survival analysis.

Once collars were recovered, either after the individual died or the collar successfully dropped off, the records were saved as text files and imported in Microsoft Excel. Because collars were programmed to acquire locations at different time interval, I used a 5-hour interval as a standardized minimum time interval between documented locations. This time period was selected as the majority of collars were programed to record locations every 5 hours. Consecutive locations which were recorded more frequently than 5 hours were not used for analysis. A column was created to include date and time (yyyy-mm-dd hh:mm:ss) for each location. A change in time column (hh:mm:ss) was created based on difference in times between the current and previous row of data. By doing this, I was able to screen locations with a lower time interval than 5 hours.

In order to standardize sampling efforts across study sites, the lowest number of recorded locations at a site (SDMP) was determined and then utilized as the universal sample size for each location (6,591 locations/site). A random selection of 6,591 locations at each site was then conducted. A column labeled “days-post release” was created based on time differences in the current and first row’s date values. Desert bighorn having <30 days of data were then excluded from analyses. Also, several desert bighorn crossed international borders into Mexico, and thus locations outside the USA were dropped from analysis. A total of 234,947 locations was acquired between 2010 and 2016. The implementation of the standardized interval (5 hours) between individual locations for each collar reduced this total to 26,364 locations (Figure 4.2) that were utilized for analyses.

Principal component analysis (PCA) was used to evaluate differences within environmental factors used in a maximum likelihood test. This facilitated a selection of characteristics that explained the majority of variability within the environmental variables (elevation, terrain ruggedness index [TRI], percent slope [Slope], canopy cover [CC], canopy cover error of canopy cover [CCE], and aspect) selected for the maximum likelihood test. This method allows for the reduction of strongly correlated data groups and only utilizes the factors that explain the most variance and are not related to each other (Janžekovič and Novak, 2012).

To assess occupancy, a presence-only maximum likelihood approach was used (Royle et al. 2012). For this methodology, I used the recorded locations of desert bighorn on the landscape (GPS locations) and the associated environmental variables to predict species occurrence throughout an area. Once results were finalized, files were loaded into ArcGIS® 10.1 (Environmental Systems Research Institute [ESRI], Redlands, CA) for better visualization in the North American Datum 1983, Universal Transverse Mercator (UTM) Zone 13 N coordinate system. After the completion of the above analysis, to assess the validity of the model, I utilized the data from 4 GPS collars and compared the model results. These collars were assumed to be independent locations because the 4 GPS collars (1,120 locations/collar) were recovered from SDMP after the model was created.

Results

Results from the PCA (Figure 4.3) showed Slope (49.74%) to have the greatest variability explanation followed by Elevation (21.26%). The scree plot results (Figure 4.4) exemplified appropriate components by allowing visual aid for determining an appropriate number of principal components to be used for the maximum likelihood model. The curve in the scree plot showed lower variability explained by the CCE (2.24%) and Aspect (2.01%). The model was able to explain 95.73% of variability (Table 4.1) by using the remaining 4 variables.

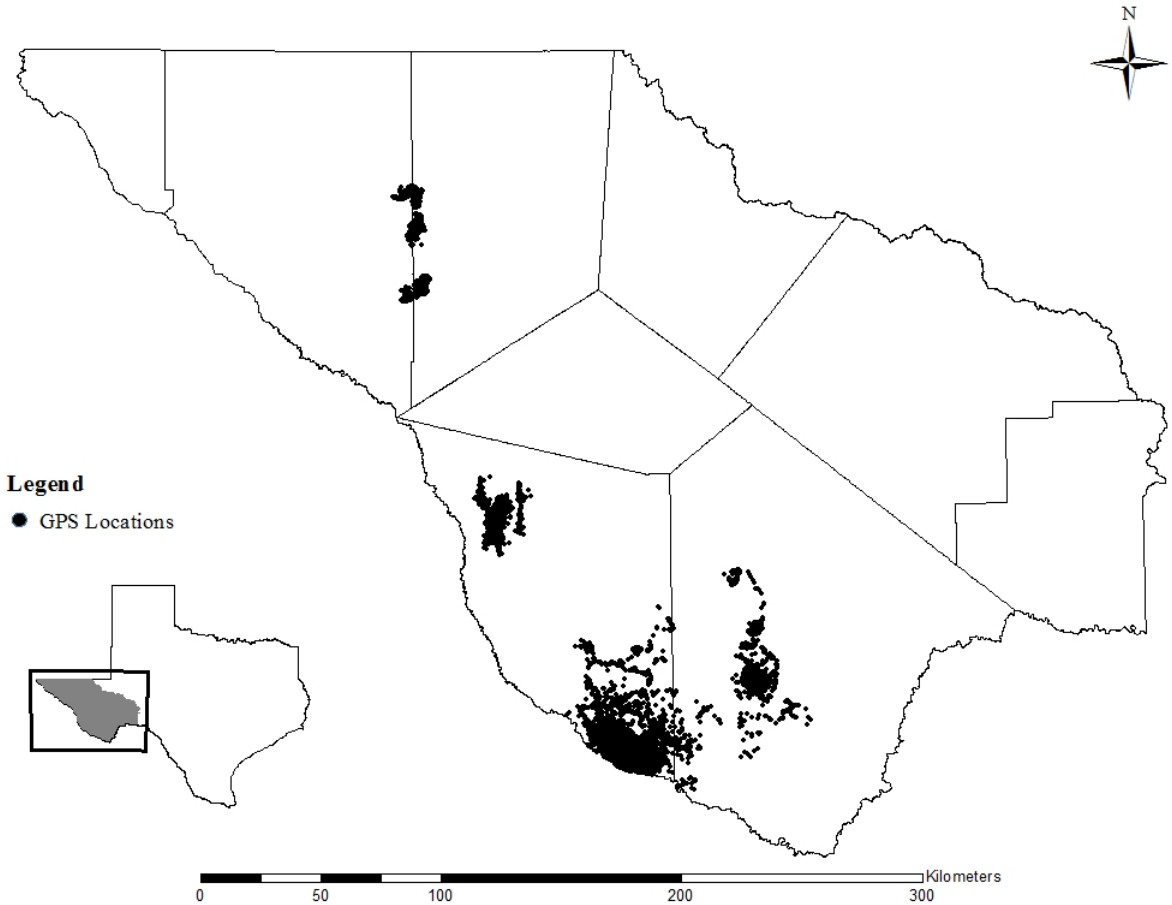

Figure 4.2. The distribution of 26,364 GPS locations acquired from collared desert bighorn in Trans-Pecos Texas, USA.

56

Components used for the model were taken to be at the point at which the remaining eigenvalues are relatively small and all near the same size. Results showed high correlation between CC and CCE, and low explanation of data variability from Aspect, the PCA allowed to reduce the number of variables used for the creation of the Maximum likelihood model (Figure 4.5). The factors Slope, Elevation, TRI, and CC were used in the Maximum likelihood model, and aspect and CCE were excluded. Maps for each mountain in Texas with higher resolution were created (Appendix A) for better visualization of results.

Distribution values for Slope (Figure 4.6) demonstrated selection values ranging from 0.09 to 314, having a median of 56.6 with a lower quartile of 38.2 and upper quartile of 76.3. Elevation values (Figure 4.7) showed greater selection for elevations between 1,200 m and 1,600 m having the median of 1,459 m. Elevation values ranged from 721 m to 2,024 m. The 4 collar data sets used as a comparison between the model and samples showed a relationship between habitat selectivity and habitat prediction from the model. In a combined average, 92.4% of the documented locations occurred in areas with a 50% or higher occurrence probability. Similarly, 71.25% of the documented locations for desert bighorn occurred in areas with a 75% predictability or higher.

Discussion

Scientists and biologists have a basic understanding of how natural or environmental systems are structured, the aspects that drive fluctuations in resources within an ecosystem, how systems respond to management actions, and potential uncertainties Millspaugh et al. 2009). However, many individuals might not realize that

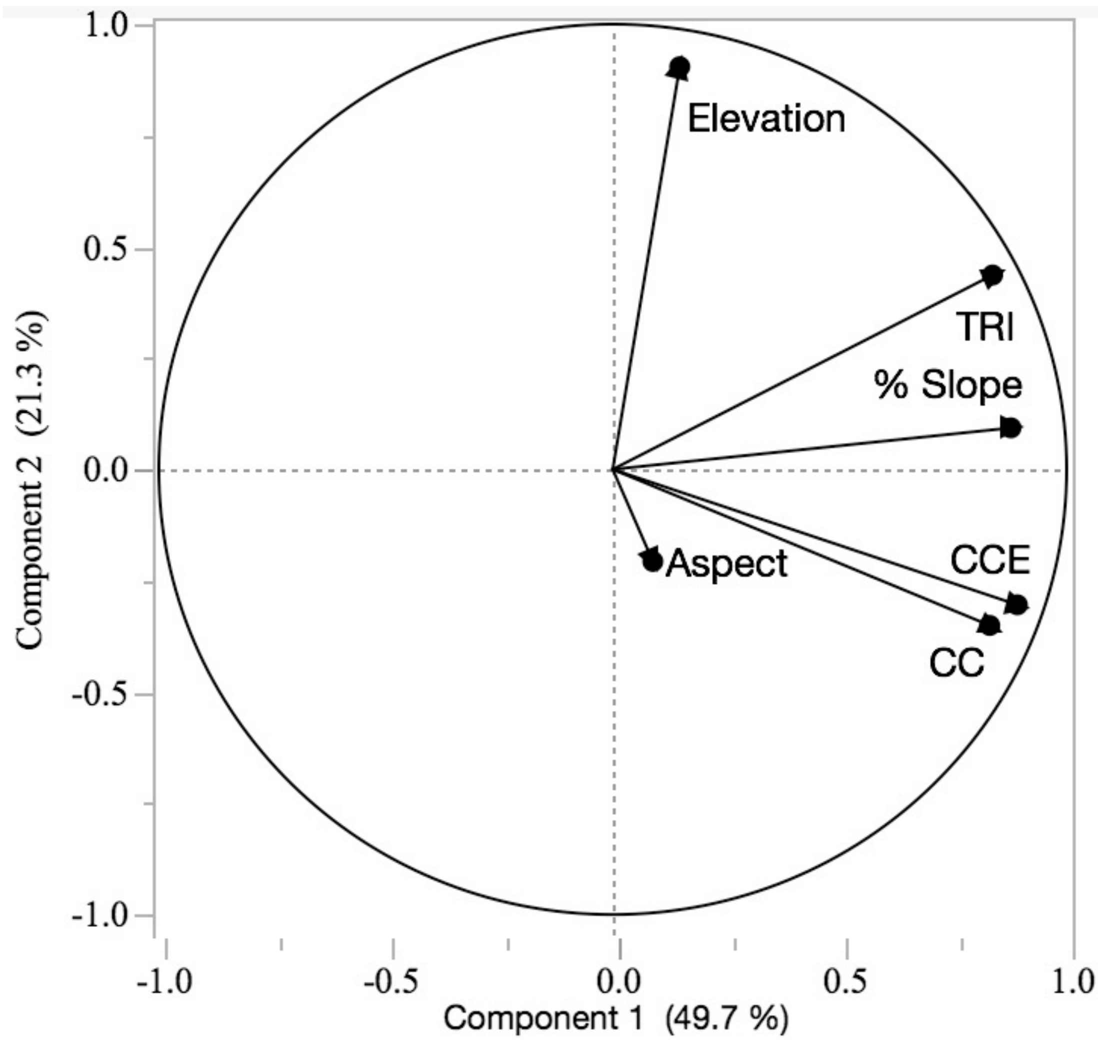

Figure 4.3. Results from principal component analysis performed for comparison of colinearity of environmental variables selected.

[a]TRI:Terrain Ruggedness Index
[b]CC:Canopy Cover
[c]CCE:Canopy Cover Error

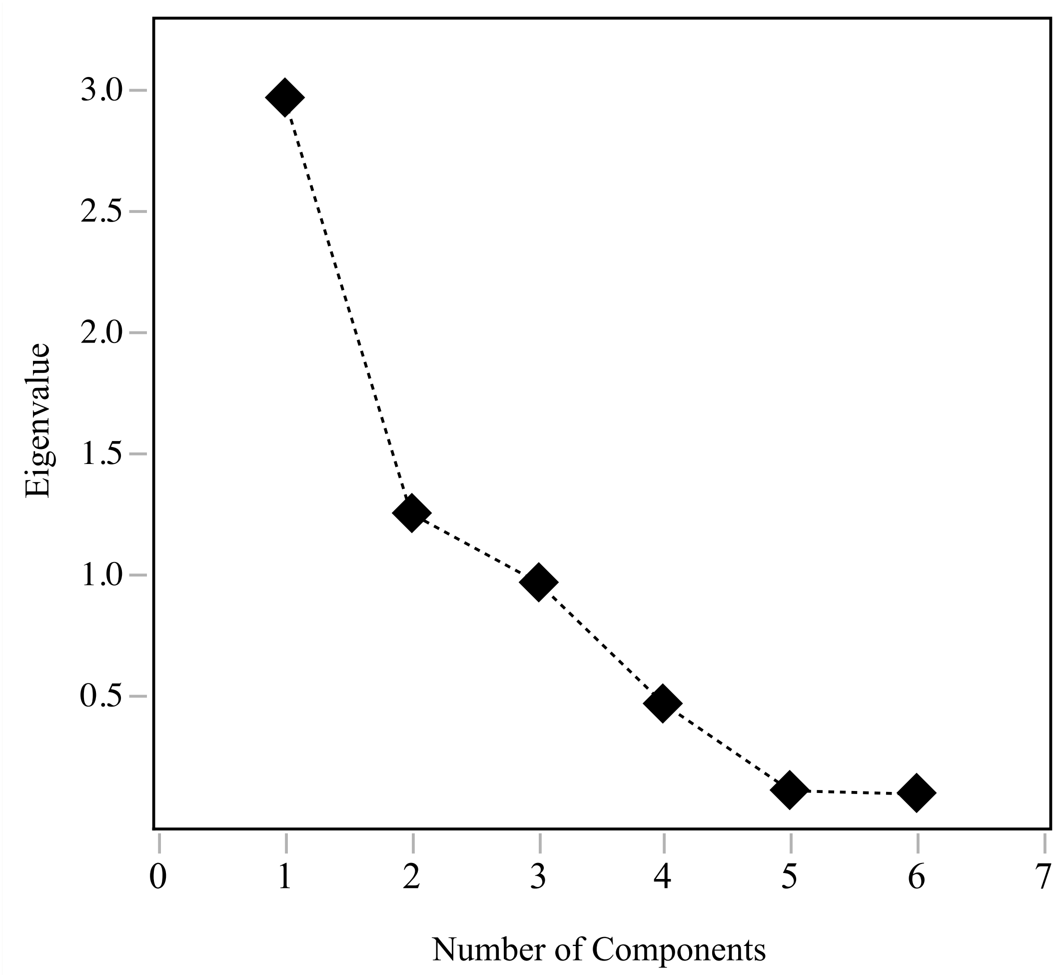

Figure 4.4. Scree plot graph of eigenvalue against component numbers used to determine appropriate components for creation of habitat model. The curve in component 5 and 6 explains lower variability (Canopy Cover Error and Aspect).

Table 4.1. Proportion of variance explained individually and cumulative as variables are added to the model.

Component	Eigenvalue	Percent	Cumulative Percent
Slope	2.98	49.73	49.73
Elevation	1.27	21.26	70.99
[a]TRI	0.99	16.55	87.54
[b]CC	0.49	8.18	95.73
[c]CCE	0.13	2.24	97.98
Aspect	0.12	2.01	100.00

[a]TRI:Terrain Ruggedness Index
[b]CC:Canopy Cover
[c]CCE:Canopy Cover Error

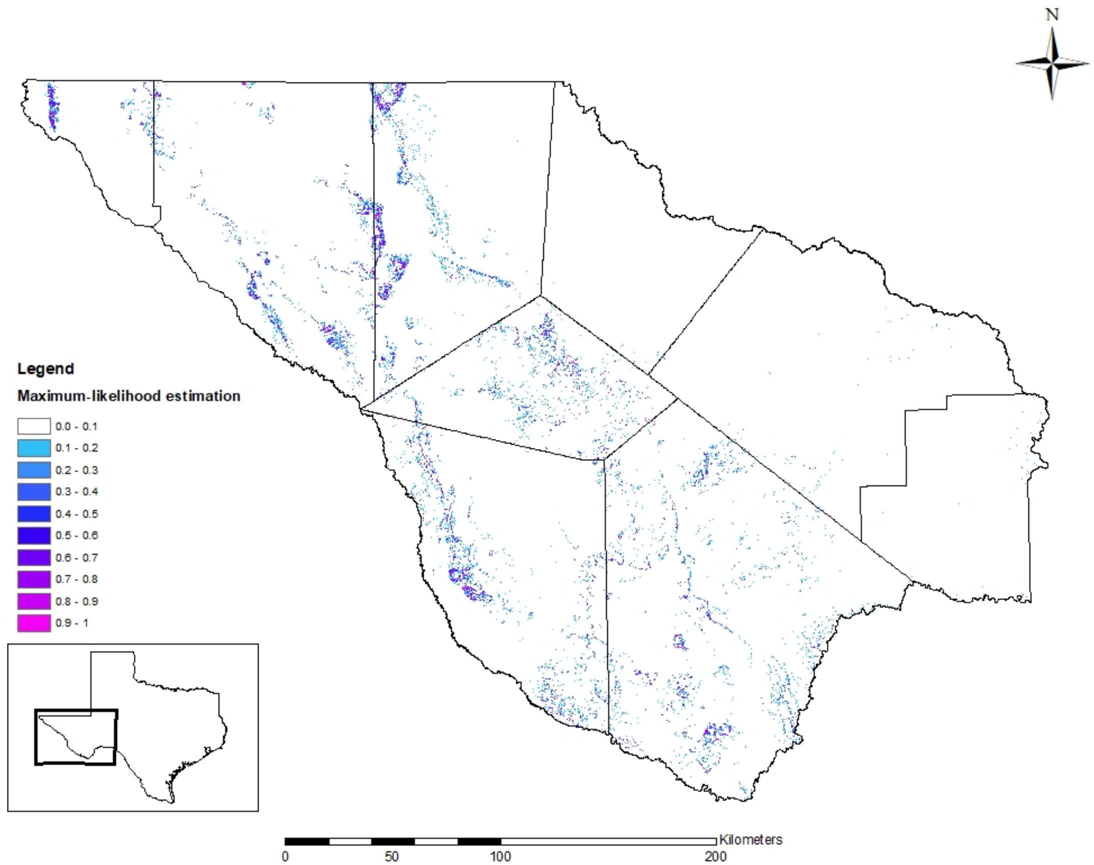

Figure 4.5. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Trans-Pecos Texas, USA. Light blue areas represent lower probability of occurrence and pink areas represent higher probability of occurrence.

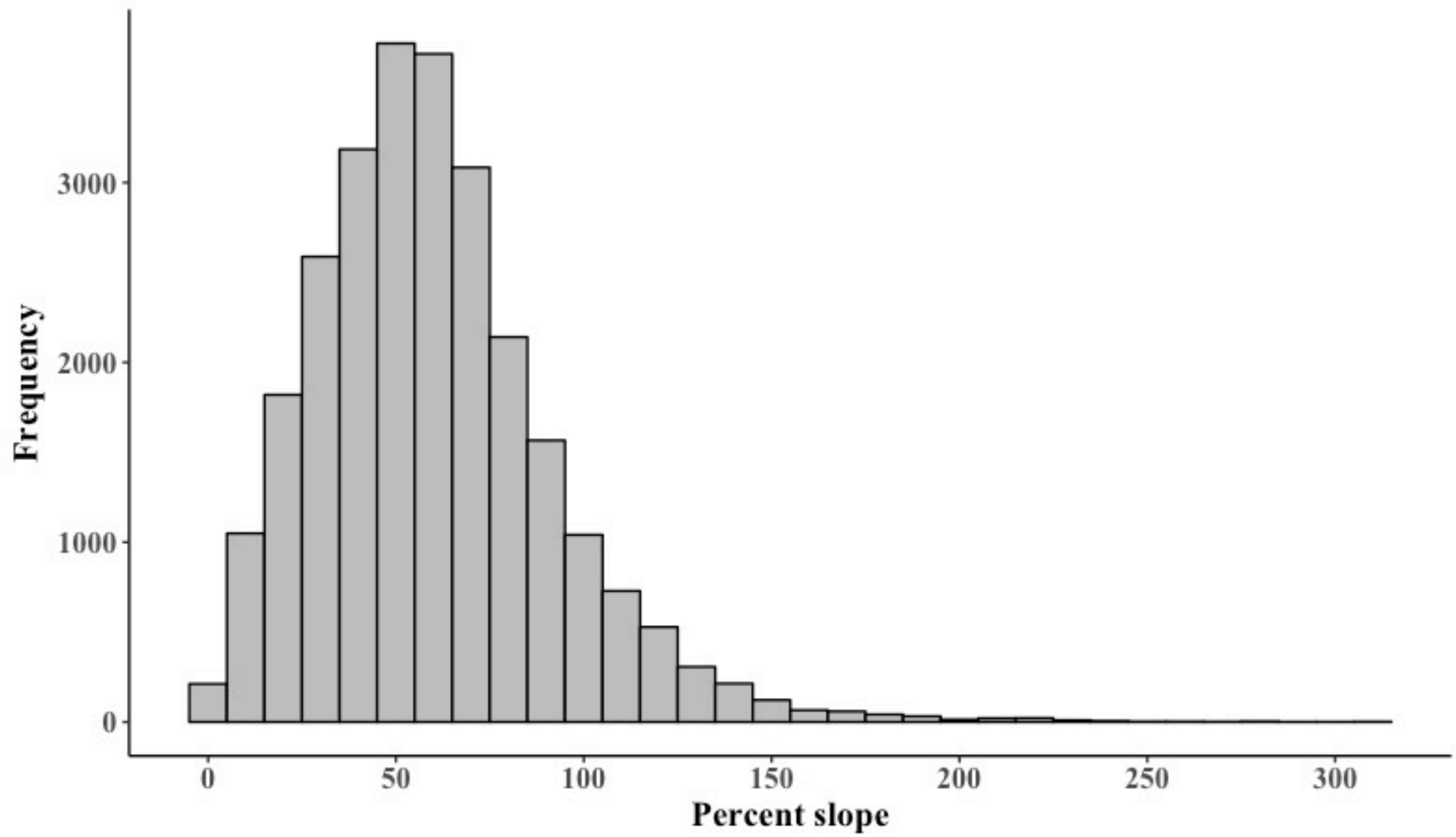

Figure 4.6. Percent slope used by desert bighorn in the Trans-Pecos Texas, USA.

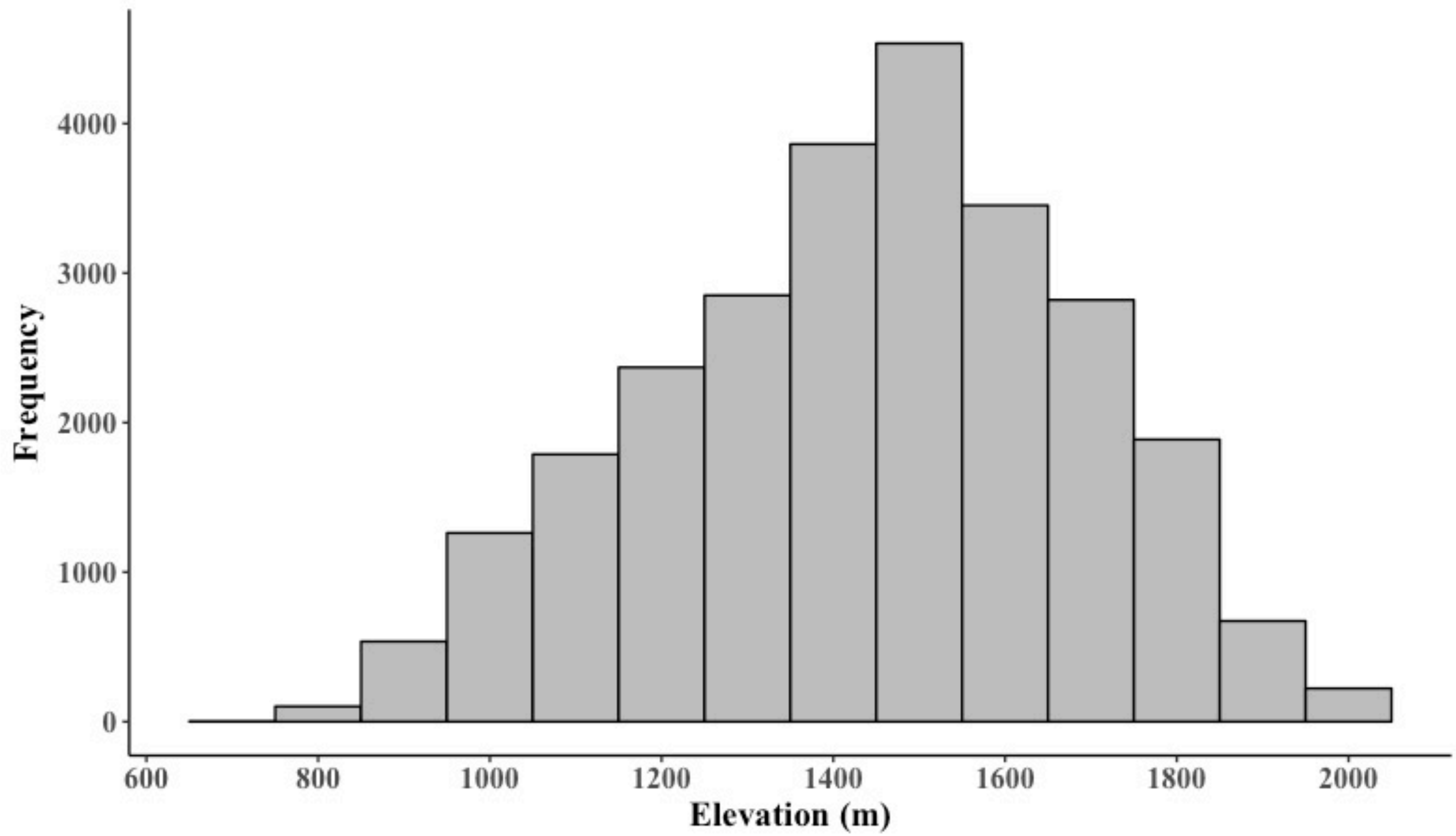

Figure 4.7. Elevation distribution graph used by desert bighorn in the Trans-Pecos Texas, USA.

the framework is the basis for conceptual modeling that can be expressed into diagrams and equations (Millspaugh et al. 2009). Models can help researchers understand the ecology of species and are becoming a fundamental tool for making management decisions directed towards the recovery of endangered species (Turner et al. 2004). Although managers have to make decisions confronted with uncertainty, these models serve as a tool to both evaluate this uncertainty and facilitate its reduction while guiding management decisions in the right direction. Therefore, these ever-improving management models can become a means for decision-making and help prioritize efforts needed to address gaps in our understanding (Millspaugh et al. 2009).

A common theme in wildlife research is determining parameters of habitat selection by evaluating the association between habitat and wildlife (Alvarez-Cardenas et al. 2001). Due to seasonal variability in food resources, predation, and weather, wildlife may not be able to adequately assess the advantageous and detrimental resources that are potentially present at a given location. Instead, wildlife must rely on habitat features that are stable measures of these resources (Smith and Shugart 1987). Although, several studies have been done concerning desert bighorn habitat (Turner et al. 2004, Jansen et al. 2007, Sappington et al. 2007, Rubin et al. 2009, Hoglander et al. 2014), none of these studies used a landscape approach or ordination techniques to determine relations between habitat components. Despite being considered a habitat specialist (Geist 1971), little is known about relations between desert bighorns and their selected environmental habitat factors (Krausman and Leopold 1986). The lack of information on how environmental components play a role in habitat selection for desert bighorn in Texas could have negative consequences on both restoration efforts and management decisions.

Analyses of the relationship between individuals of a species and habitat variables have frequently been used to characterize vital habitat for species (Horne 2002). If correlation are done between the occurrence of an organism with a series of environmental variables in a given site, descriptions of the niche of the organism, and consequently, predict its pattern of habitat occupancy in other areas (Horne 2002). A principal component analysis allows understanding the difference in a set of variables in terms of a smaller number of independent linear combinations (principal components) of those variables. Because representation of results is important in visualizing multivariate data, by decreasing it to graph dimensions a principal component is a way to picture the independent structure of the data by using as few variables as possible. Results from this study demonstrate the need for evaluating multiple habitat variables and understating which have a greater impact on habitat selection, as was the case for slope, elevation, TRI, and CCE for desert bighorn.

The maximum likelihood method is an encouraged approach because it allows for the estimation of absolute occupancy quantities and only requires presence locations for a species (Royle et al. 2012, Fitzpatrick et al. 2013, Merow and Silander 2014). Despite the ability of maximum likelihood to predict occupancy based upon a small sample of locations, it has been controversial due to the original model description suggesting that small sample sizes might increase model uncertainty (Royle et al. 2012). However, Fitzpatrick et al. (2013) found that maximum likelihood performed well for models with dozens or as few as seven presence locations. In Texas, this study represents the largest environmental assessment done at a landscape level, and the analysis is the first attempt probability occurrence model for desert bighorn (Figure 4.3, Appendix A). Results from the model represent possible delineation and distributions of habitat for current desert bighorn populations in Texas.

Habitat variables such as vegetation, disease, and water availability could be argued to be important factors for desert bighorn distributions and were not included in the creation of the model. The model does not necessarily represent delineation of habitat quality, rather only the probability of presence under the variables used in the creation of the model. Because

Desert bighorn are flexible in regard to vegetation preferences, which may be a function of vegetation structure rather than composition. Preferred habitats are generally devoid of thick vegetation (Hansen 1980) because desert bighorn are dependent on keen eyesight and acute agility in rugged terrain to avoid predators (Geist 1971), and it is reported that communities where shrub cover is >30% are avoided (Holl 1982). Wilson et al. (1980) suggested desert bighorn avoid areas where vegetation exceeds 76 cm in height, although rams may seek densely vegetated areas for thermal protection. Despite I did not include information on vegetation (other than canopy cover). However, most distribution models represent snapshots assessing habitat suitability at a single time (Bartel and Sexton 2009). By using non-stochastic environmental components results can be extrapolated into a larger scale and have a better representation at a landscape approach.

Possible diseases caused by proximity to livestock was also not included in the model, despite the evidence of a strong relationship between diseases in wild bighorn sheep and domestic sheep that has been seen in the past (Goodson 1982, Spraker and Adrian 1990). Epizootic pathogens have been documented to cause devastating reductions of desert bighorn populations in the United States (Spraker 1977, Monson 1980, Onderka and Wishart 1984, Onderka et al. 1988). Despite this, distance from domestic sheep was not included in the model due to lack of information in private lands regarding domestic sheep being present or absent. Also, there is no evidence of pneumonia being problematic in the state of Texas as no cases have been documented (Froylan Hernández, TPWD, personal communication).

The relevance of water for desert bighorn has been studied in the past (Turner 1970, Leslie and Douglas 1979), and it has been assumed that free-standing water is critical for desert bighorn because of its usage when available (Graves 1961, Blong and Pollard 1968, Turner 1970). However, desert bighorn are present on ranges without free-standing water and may obtain moisture through succulent plants. As well, there is evidence of desert bighorn being able to survive from water present in their food and metabolic water formed from oxidative metabolism (Krausman et al. 1999). But, locating all permanent water sources in the Trans-Pecos for data to be included would have represented a challenging task and could have resulted in misinterpretation of the model due to potential miss-sampling or false absences. Because of the lack of conclusive information on relevance of permanent water and the lack of data on where water is available year round across the Trans-Pecos, this variable was not used.

Important considerations for desert bighorn include food, visibility, competition for water, exclusion of domestic sheep, and protection from human intrusion (McCarty and Bailey 1994). Latitude, precipitation, space, exposure, and land use practices determine habitat communities (Monson and Sumner 1980, Krausman et al. 1999). Future studies should incorporate biotic environmental components as well as other meaningful factors that could affect the distribution of desert bighorn to improve the current model.

Our knowledge and the practice of animal reintroductions have increased rapidly with the use of occurrence models, which are quickly becoming a necessary tool for management, particularly when large landscapes are considered (Millspaugh et al. 2009). A key challenge for future reintroductions is to have results evaluated and provided in a way that is available to all potential decision-making personnel, practitioners, land managers, and the public as they develop restoration programs while they address the fundamental questions of why they translocate and where to translocate desert bighorn (Jachowski et al. 2016). Because reintroductions are motivated from a range of intrinsic and utilitarian values, which drive what animal species are reintroduced and often how success should be defined (Jachowski et al. 2016). Knowledge gained from this study could be used as a key tool to assess suitability of areas for restoration, private land managers, policy-makers, and the general public involved in reintroductions must be able to access the information for decision-making.

Management Implications

Habitat can be the most influential factor for determining success in the translocations of animals undergoing reintroductions (Osborne and Seddon 2012). Despite the fact that assessment of habitat is an important component of species restoration efforts, guidelines on how to proceed are lacking (Osborne and Seddon

2012). Results represent the first suitability model of potential habitat for desert bighorn in Texas that can be used as a basis for making decisions for future translocation efforts.

It is now possible to evaluate not only where suitable habitat may be present, but also locations of marginal habitat. Desert bighorn should not be reintroduced into historical habitat solely on the basis that they once occupied that range, as historical locations might no longer indicate current viable habitat. Studies have shown that the ranges of species are historically dynamic, expanding and contracting regionally over time (Hengeveld 1990). This aspect reinforces the value of conducting habitat assessment at landscape approach with non-stochastic variables. Evaluation of habitat should not be optional, as it has been proposed at landscape management (Lindenmeyer et al. 2008). Translocating into poor and fragmented habitat may increase mortality and cause increased movements (Osborne and Seddon 2012). Results from this study could be used as a tool for estimating the potential for desert bighorn to occur in areas not previously surveyed.

Models can be useful tools for management decisions and prioritizing efforts for future research in areas poorly understood (Millspaugh et al. 2009). Upcoming translocations should assess plant species composition and structure available in new release areas in comparison to those from desert bighorn capture sites. Proper habitat evaluation should not be discretionary prior to translocations, and should be done in an experimental outline (Lindenmeyer et al. 2008). The information gained by conducting translocations within this framework can improve future restoration and conservation by refining our knowledge and understanding of desert bighorn ecology.

Wildlife management demands periodic monitoring to uphold educated decision-making (Walters 1986, Possingham et al. 2001). As well, monitoring provides estimates needed for making decisions and assessing how objectives were or were not met (Nichols and Armstrong 2012). However, many reintroduction efforts have been critiqued for failure either to conduct suitable monitoring or to not account results of monitoring (Lyles and May 1987, Griffith et al. 1989). Based on usefulness for management decisions shown by this study, I recommend research should continue documenting information on how environmental factors influence desert bighorn survival, reproduction, identification of international travel corridors between Texas and Mexico, and movements for future restoration and management efforts.

CHAPTER V

CONCLUSIONS

Reintroduction efforts for multiple species have occurred for at least 100 years (Kleiman 1989), but the field of reintroduction biology initiated later because of poor success of reintroduction programs. Although there have been success stories (Butler and Merton 1992), it became clear during the 1980s that most reintroduction efforts were inadequate and little was being learned (Lyles and May 1987, Scott and Carpenter 1987). Because of this, this study evaluated important aspects of reintroduction biology of desert bighorn in the Trans-Pecos region of Texas.

First, survival estimates are an important population parameter in the recovery and conservation of endangered populations (Harveson 2005). Survival of desert bighorn can represent a critical challenge for restoration efforts, especially within small populations. A cost-effective method of estimating survival is monitoring translocated populations by direct observation (Bleich 1998) and can be increased in accuracy with the use of radio-collared individuals (Locke 2003). To evaluate the survival, this study evaluated (1) survival of translocated desert bighorn and (2) causes of mortality. From the 172 collared individuals a total of 58 mortalities was recorded (24 M, 34 F). Causes of mortality were: 27 undeterminable, 20 by mountain lion predation (*Puma concolor*), 5 were attributed to contagious ecthyma (parapox orf virus), 1 poached in Mexico, 1 birth complication, 1 infection due to a broken jaw, 1 ingestion of toxic vegetation (cloakfern, *Astrolepis sinuate*), and 1 fell from a cliff.

Second, with the objective being to evaluate alternative strategies for improving desert bighorn translocations. An evaluation in population growth through the use of a systems analysis model was created to assist alternatives and understand population dynamics. Nature is complex and variable, having decisions surrounding such system can be overwhelmed with uncertainty of how the system will respond to management alternatives (Runge et al. 2011). While there is a great deal to learn about translocation of species, success can increase as we gain knowledge with regards to the needs of the species and appropriate management techniques continue to be developed (Seddon et al. 2007). In decision-making sceneries, models as the one created in this study, can estimate influences of management alternatives on populations. With knowledge gained from simulations new understanding of demographics, reintroductions could select the timing and demographics of individuals to be released. With the use of this model, reintroductions could focus on maximizing resources and increasing probabilities of successful translocations.

Third, the goal was to do an assessment of potential desert bighorn release sites through a habitat-based model. Occurrence models can be valuable in understanding the ecology of a species and are essential for management decisions for restoration of desert bighorn. Results from the PCA showed Slope (49.74%) to have the greatest variability explanation followed by Elevation (21.26%). The scree plot results exemplified appropriate components by allowing visual aid for determining an appropriate number of principal components to be used for the maximum likelihood model. The curve in the scree plot showed lower variability explained by the CCE (2.24%) and Aspect (2.01%). The model was able to explain 95.73% of variability by using the remaining 4 variables.

In conclusion, reintroductions are increasingly used to re-establish populations of threatened species within their historical ranges (Sarrazin and Barbault 1996, Seddon et al. 2007). However, many reintroduction attempts of multiple species have been unsuccessful (Griffith et al. 1989, Wolf et al. 1996) and the main reasons of failure are seldom understood (Fischer and Lindenmayer 2000, Letty et al. 2007). Increasing pressure on research and conservation with limited budgets demands that investments in monitoring, research, and management practices to be as rigorously justified as possible (Wintle et al. 2010). Therefore, analysis of factors influencing reintroduction outcomes is important to improve the success of future reintroduction programs (Sarrazin and Barbault 1996, Ewen and Armstrong 2007, Sutherland et al. 2010, Le Gouar et al. 2012). Monitoring and research is central for the management of natural resources because it provides the primary tool by which we learn about the success or failure of conservation investments (Wintle et al. 2010).

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APPENDIX

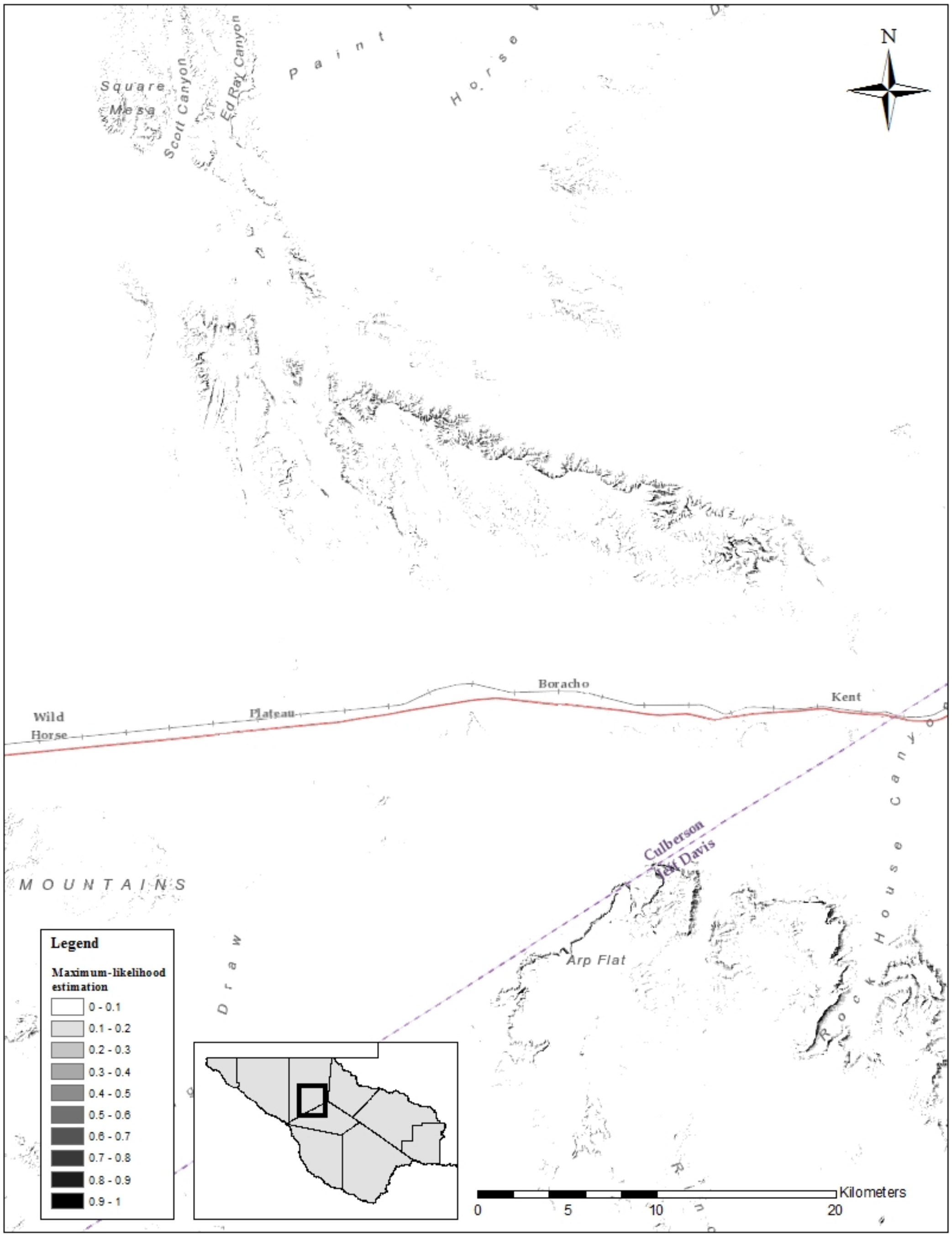

Figure A.1. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Apache Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

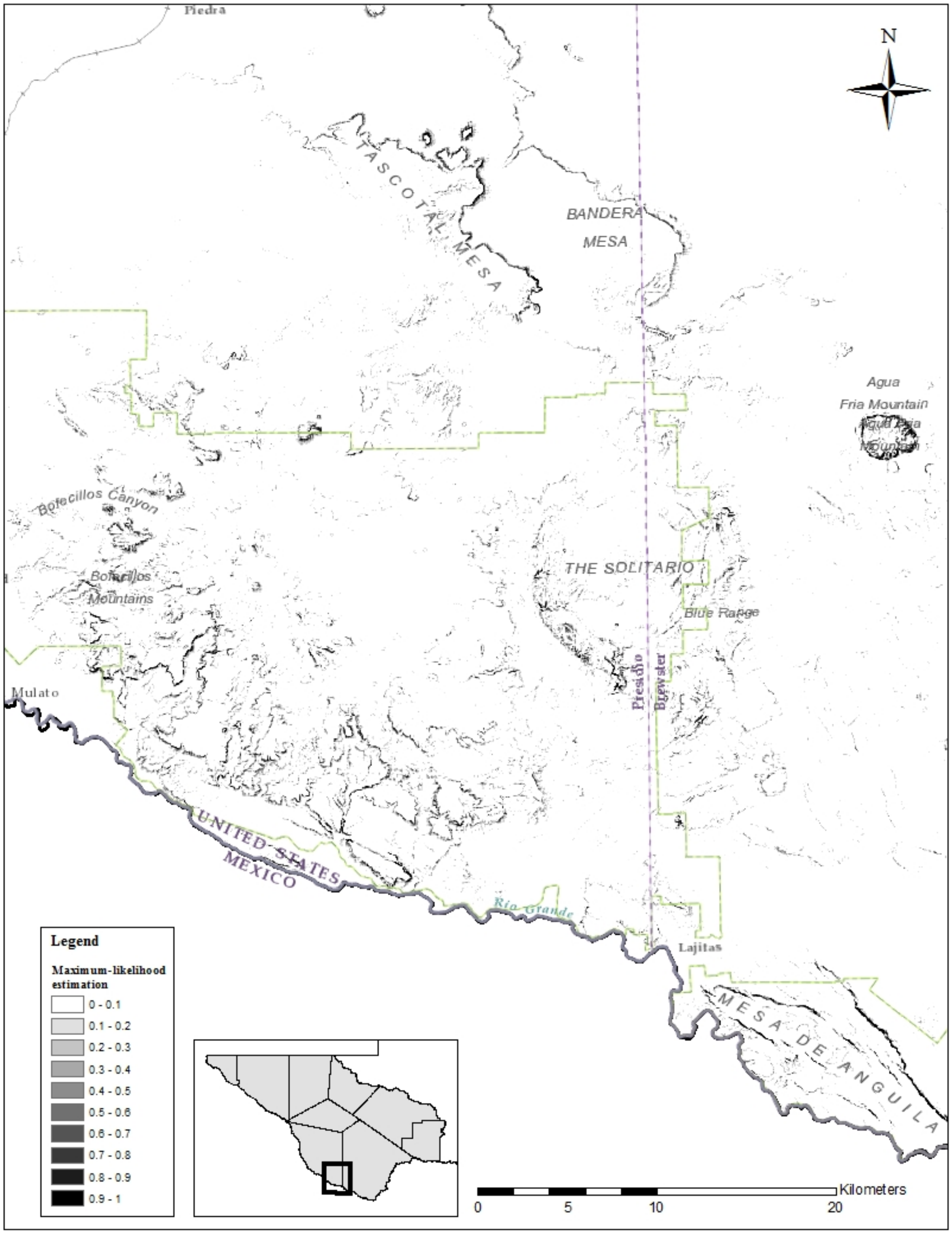

Figure A.2. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Bofecillos Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

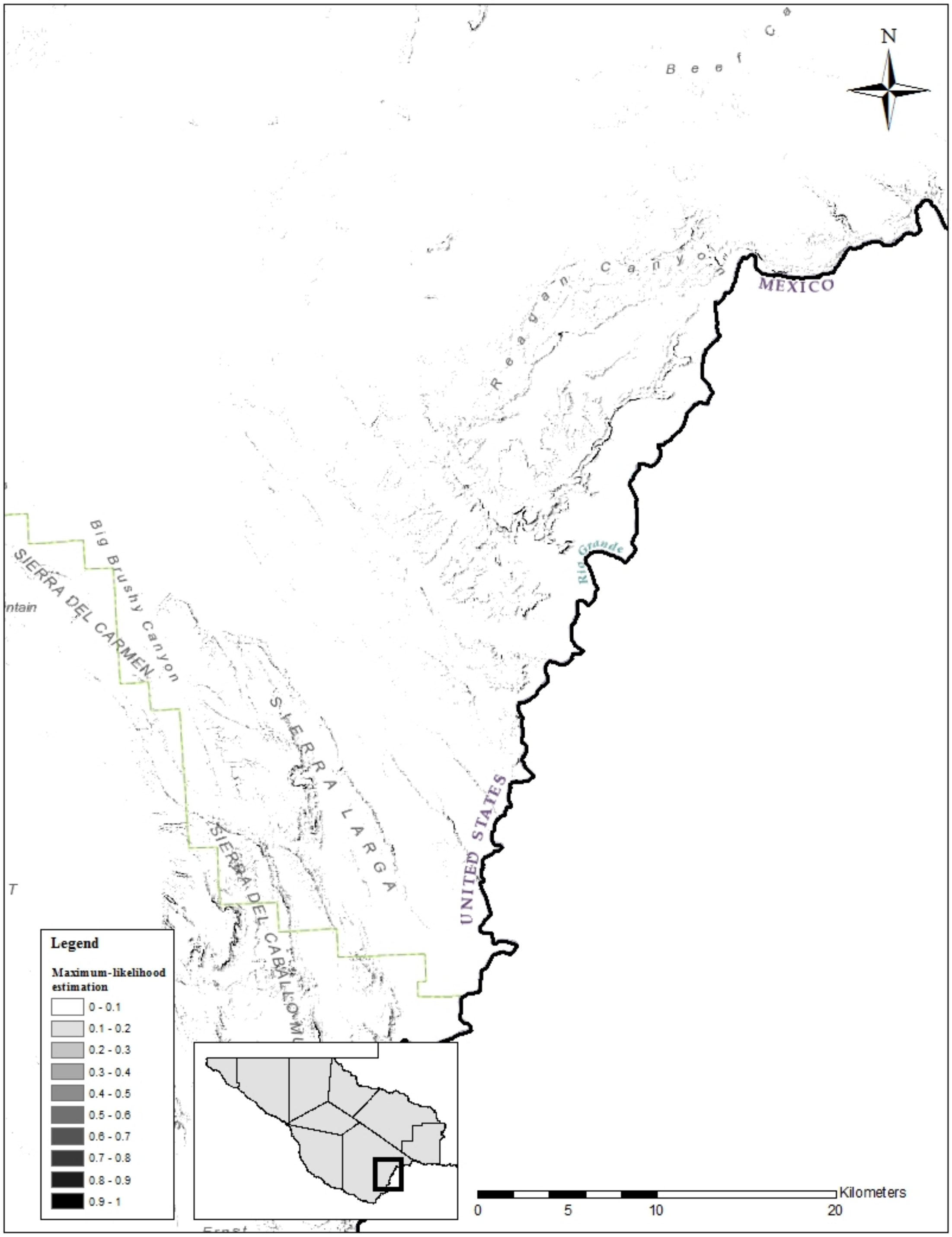

Figure A.3. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Black Gap area, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

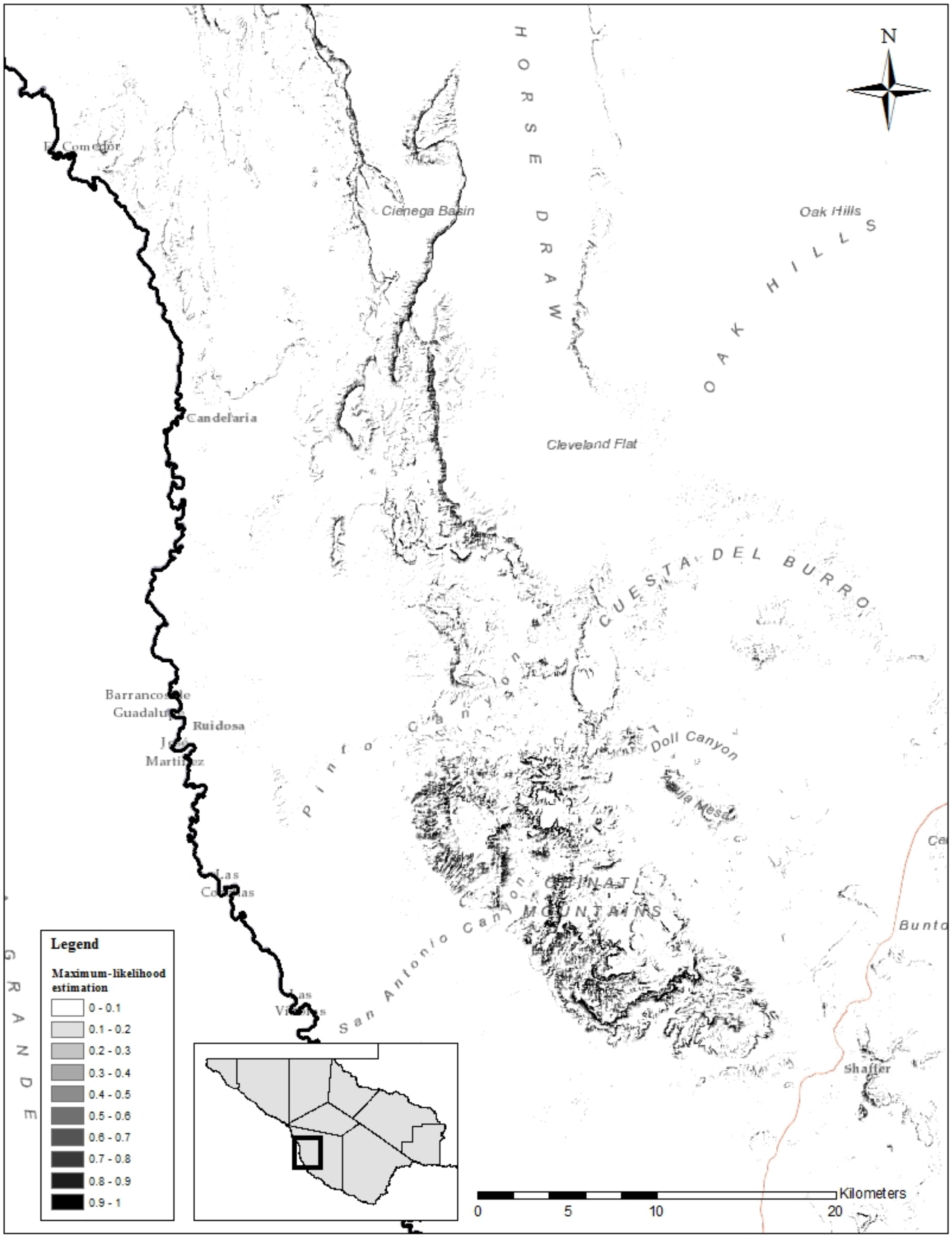

Figure A.4. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Chinati and Capote Peak Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

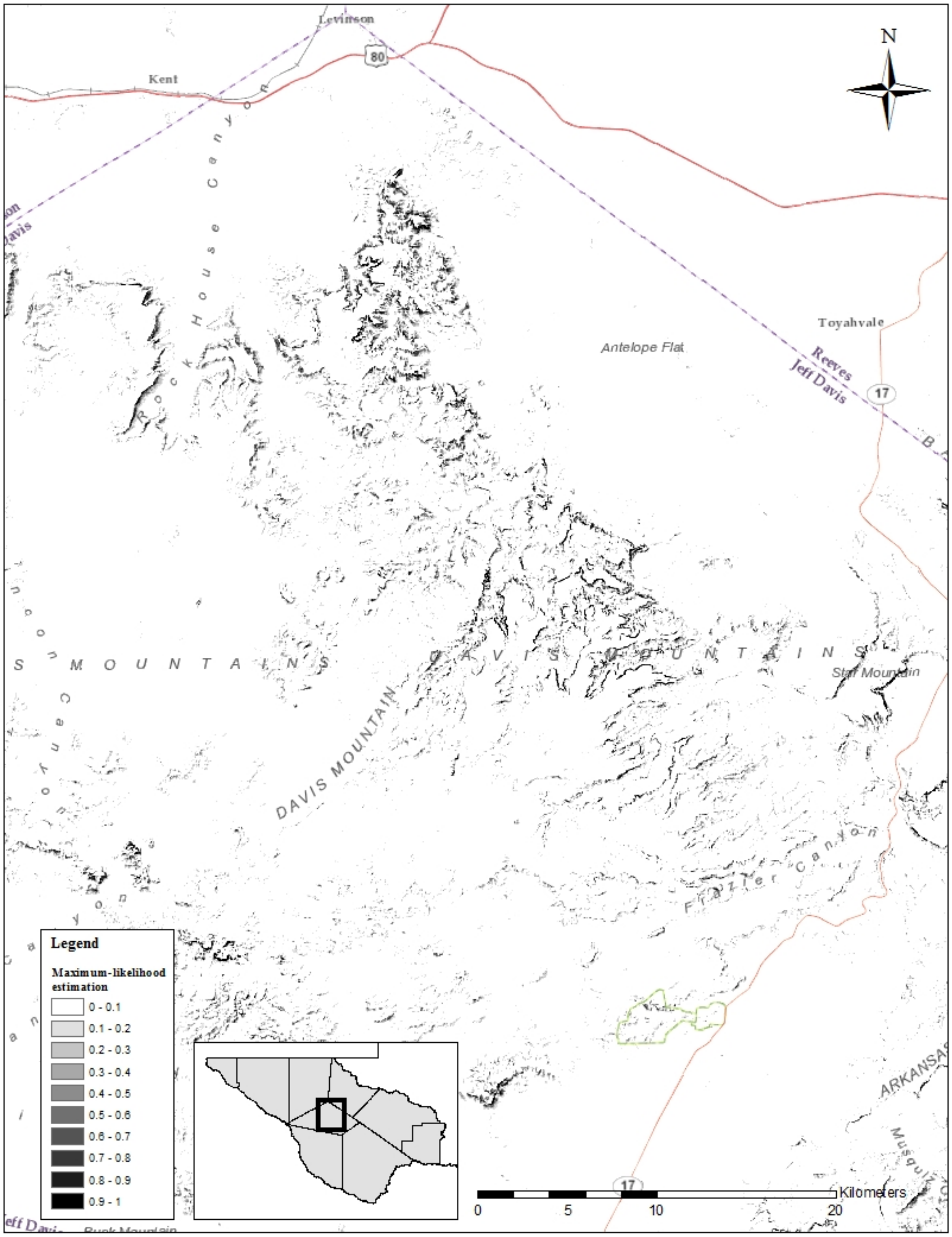

Figure A.5. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Davis Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

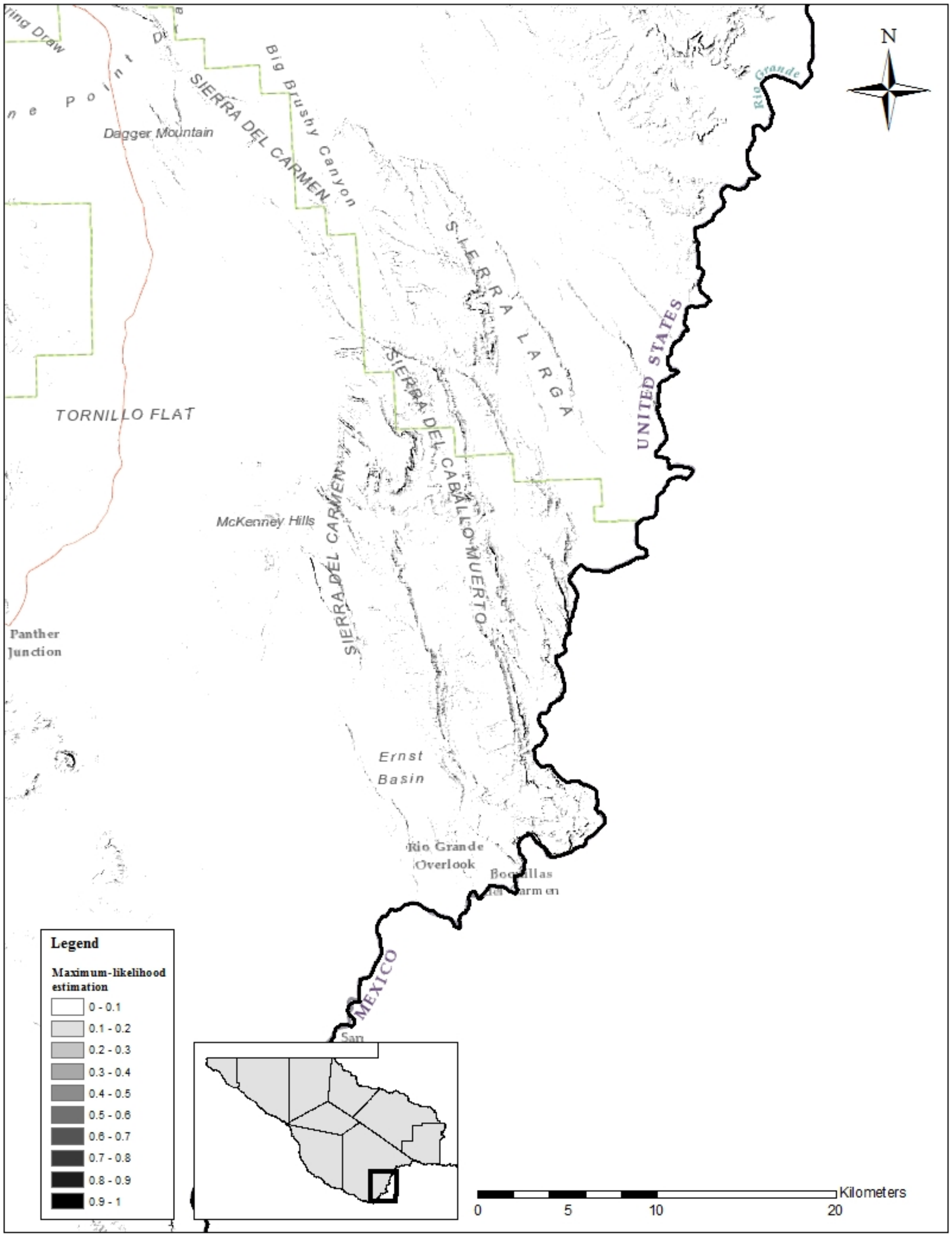

Figure A.6. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Dead Horse Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

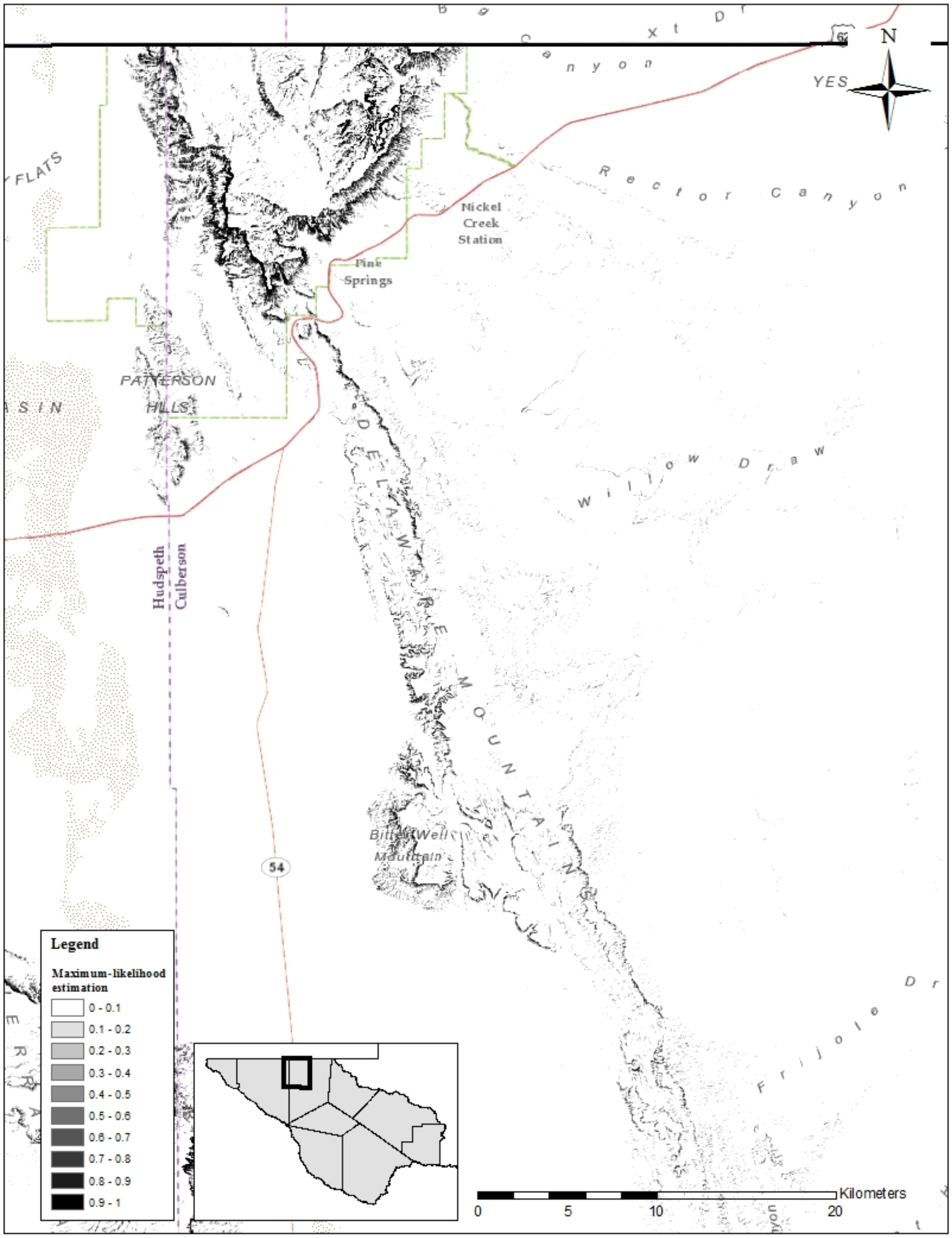

Figure A.7. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Delaware and Guadalupe Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

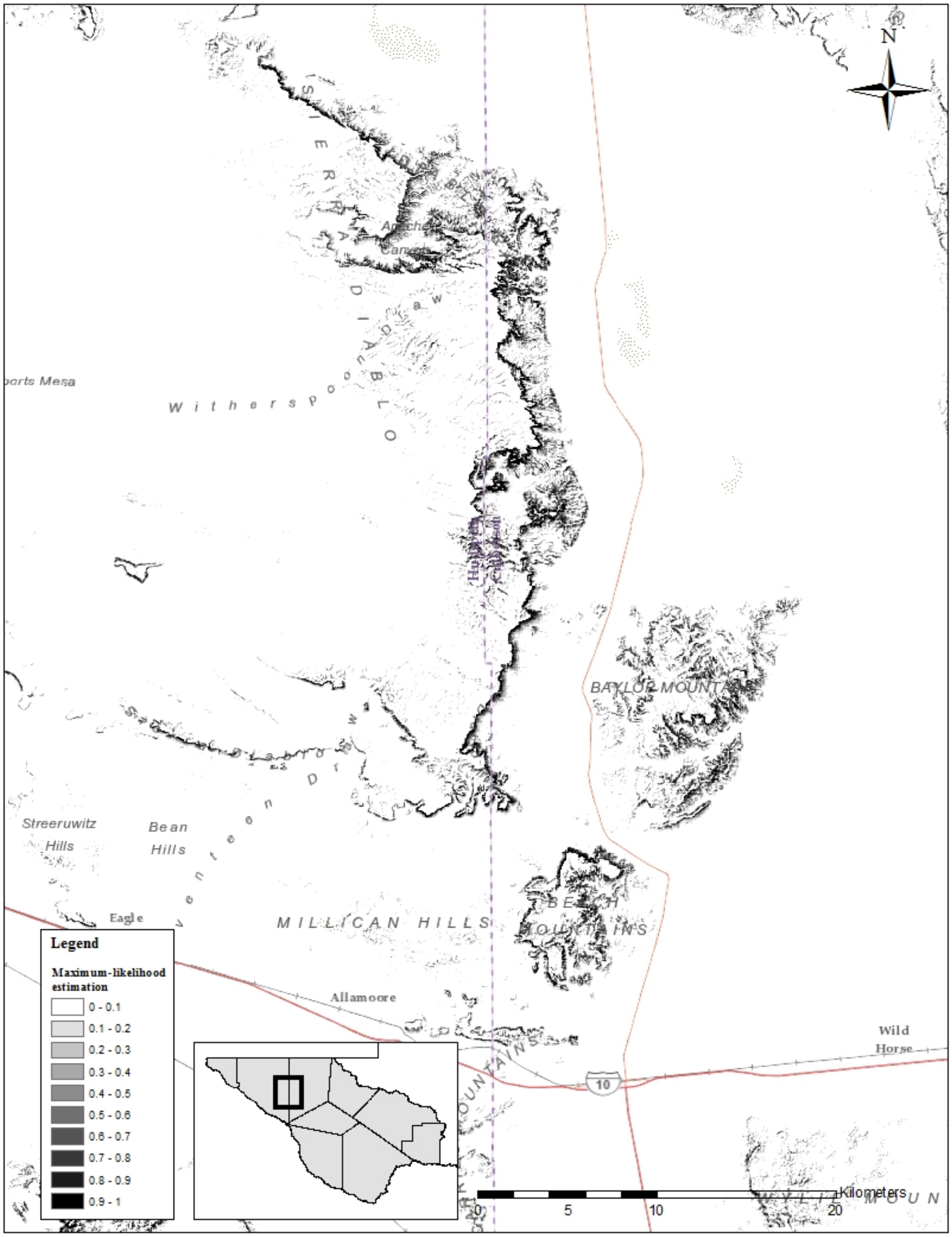

Figure A.8. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Diablos, Beach, and Baylor Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

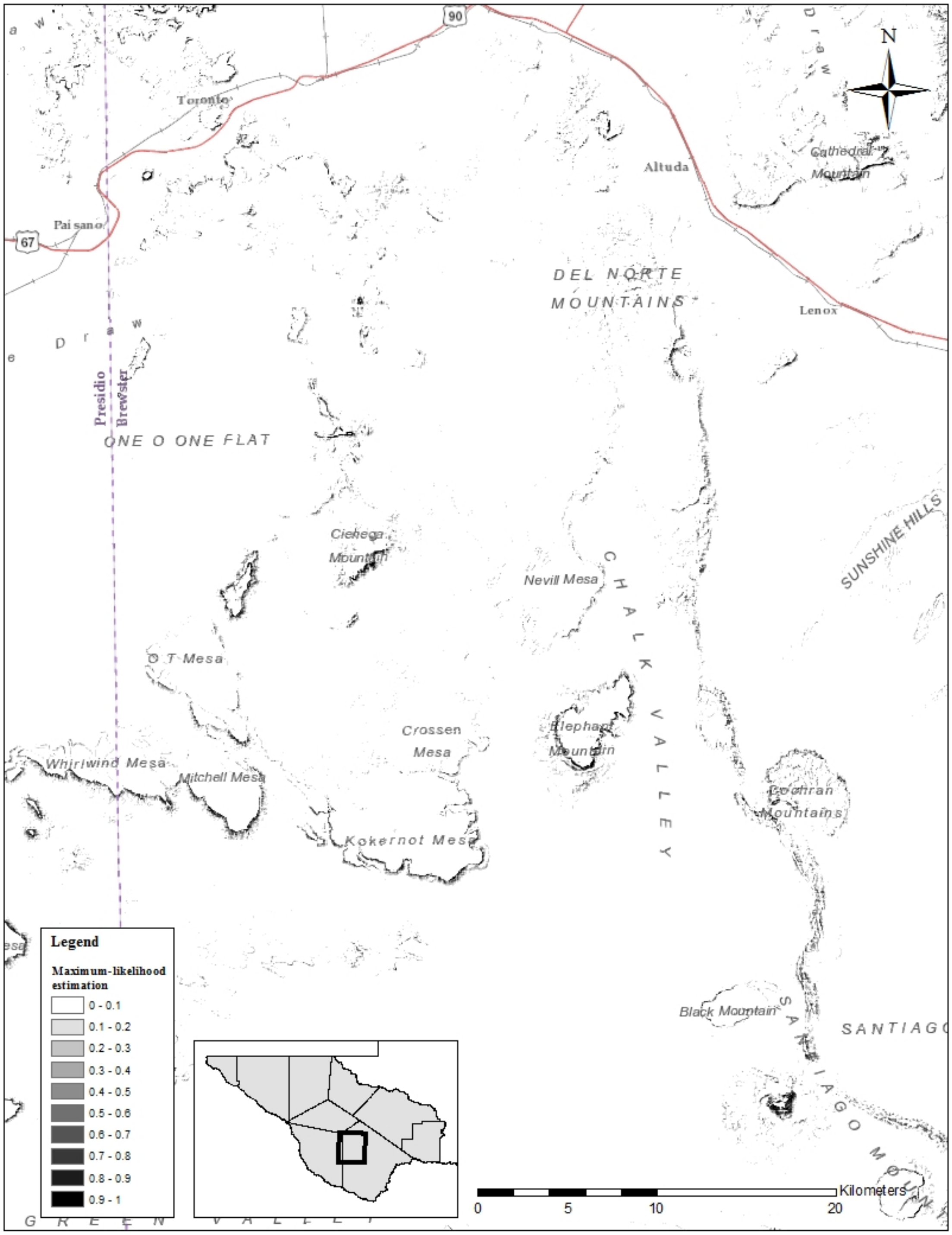

Figure A.9. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the EMWMA, Cienega, Goat, and Del Norte Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

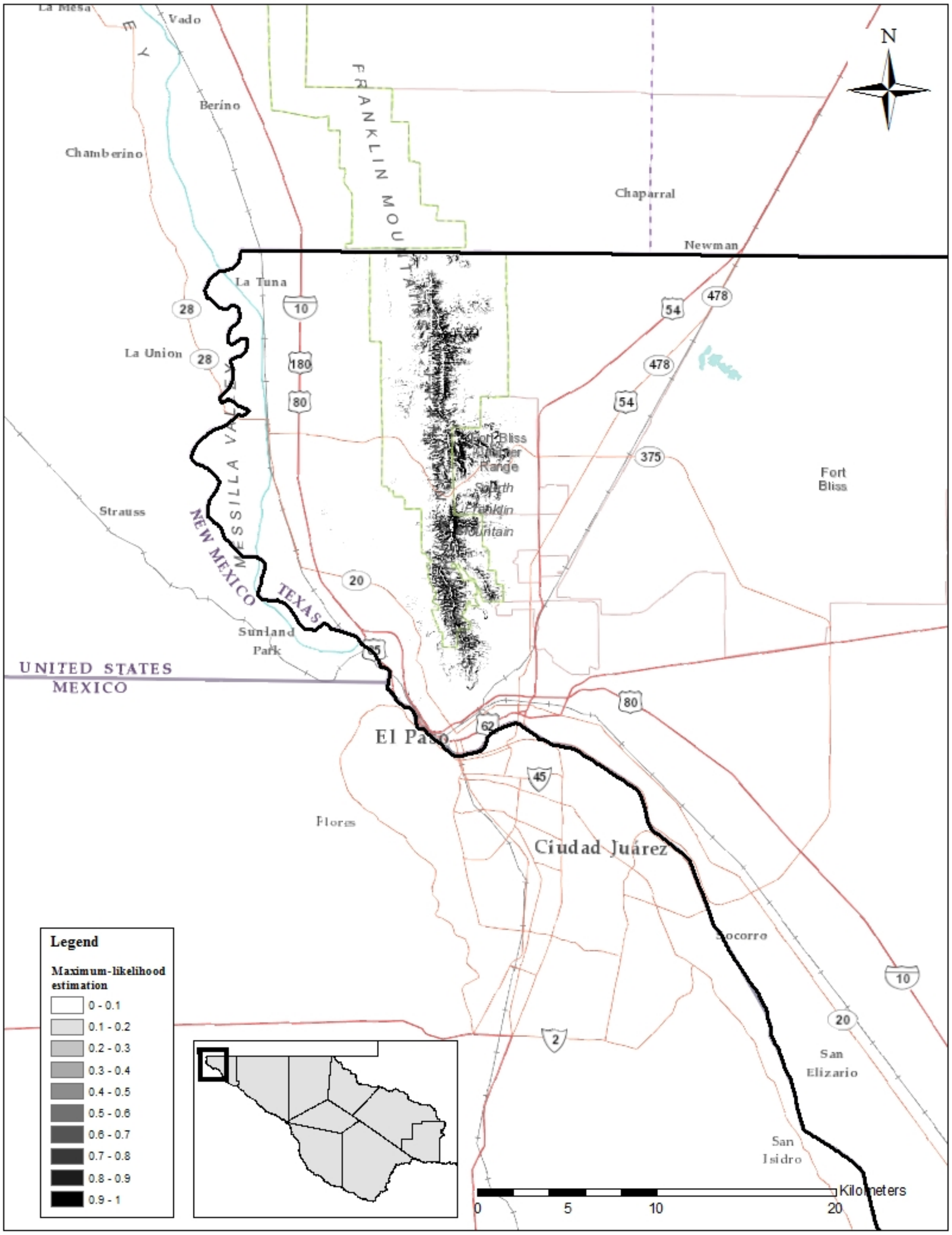

Figure A.10. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Franklin Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

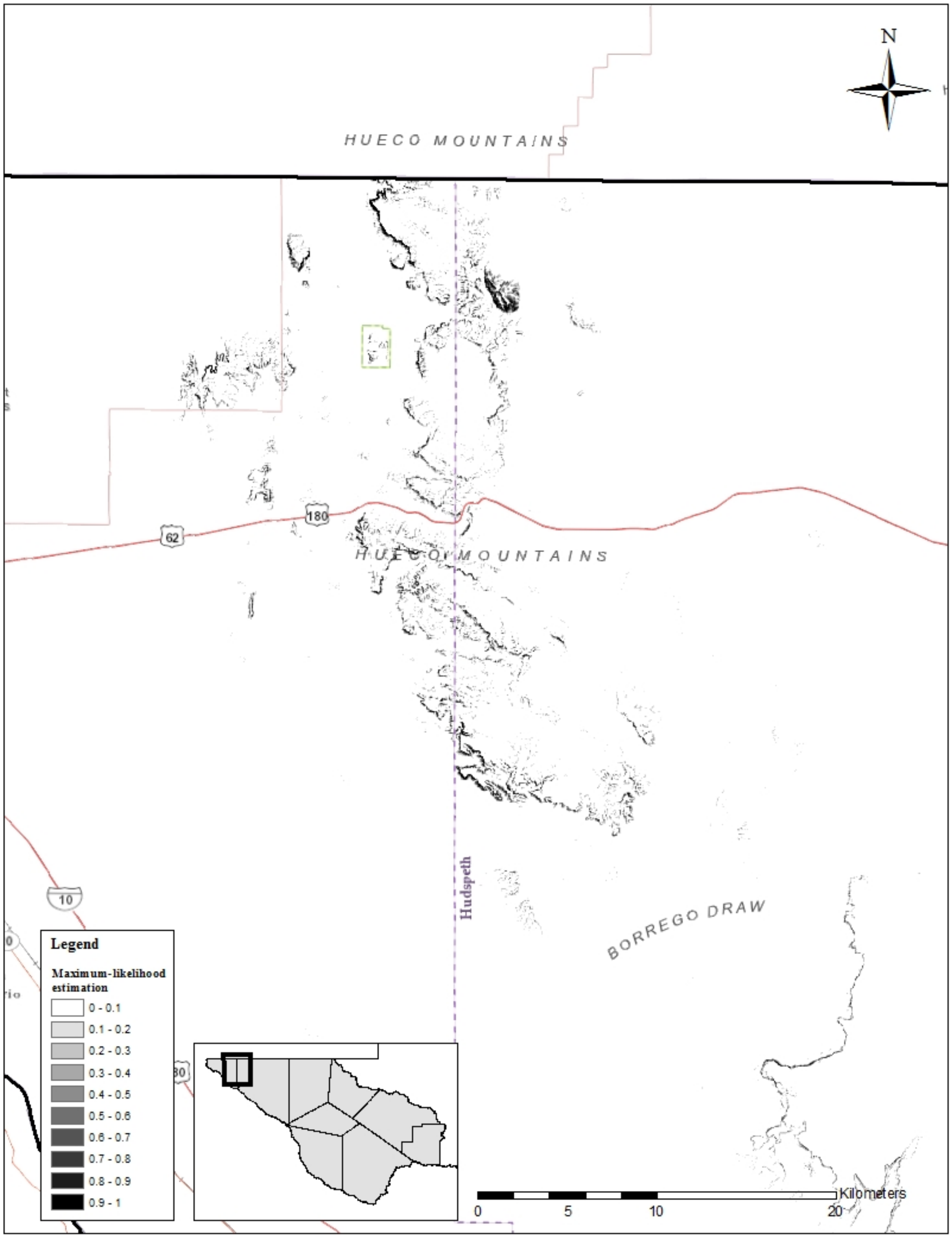

Figure A.11. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Hueco Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

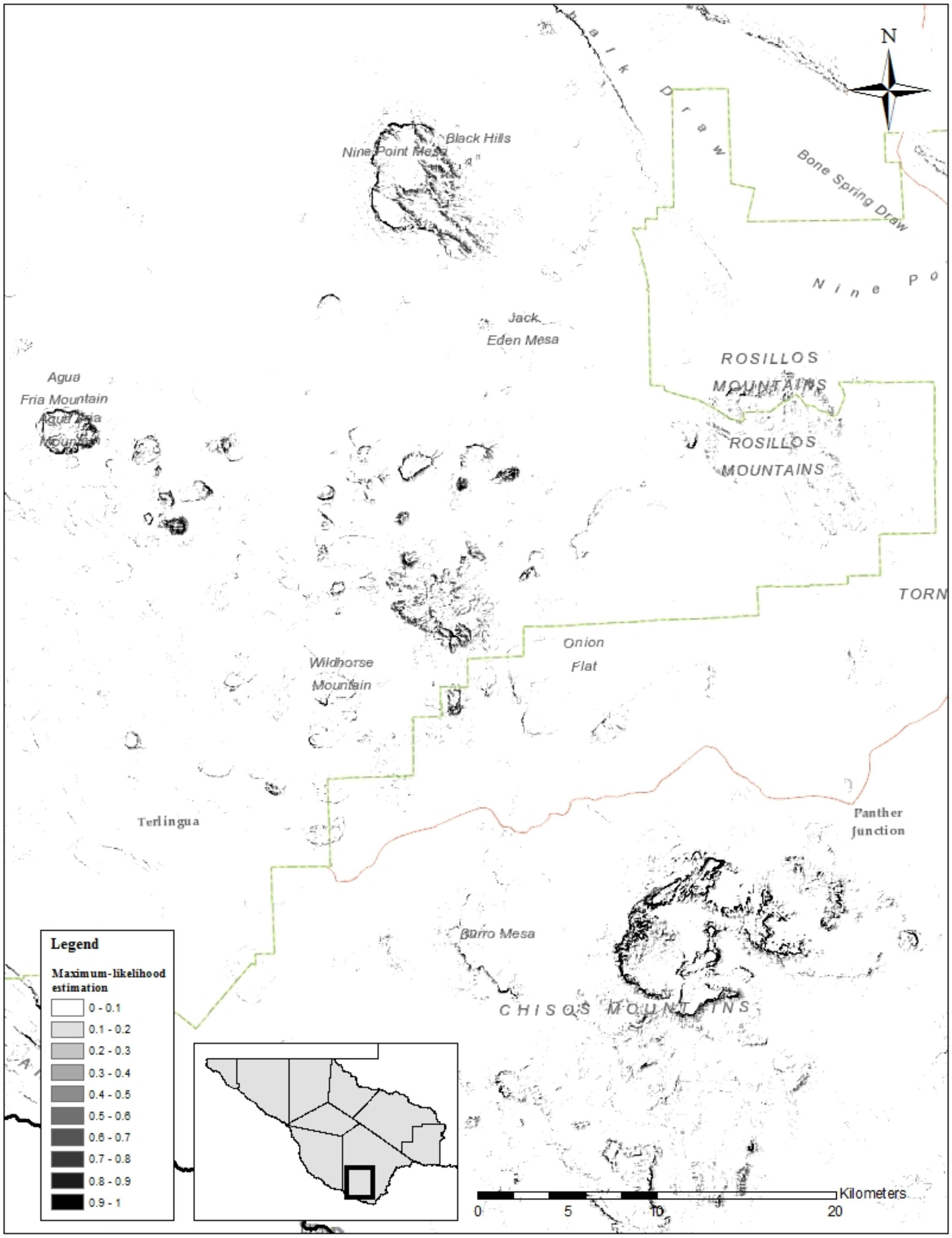

Figure A.12. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Nine Point Mesa, Chisos, Christmas, and Rosillos Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

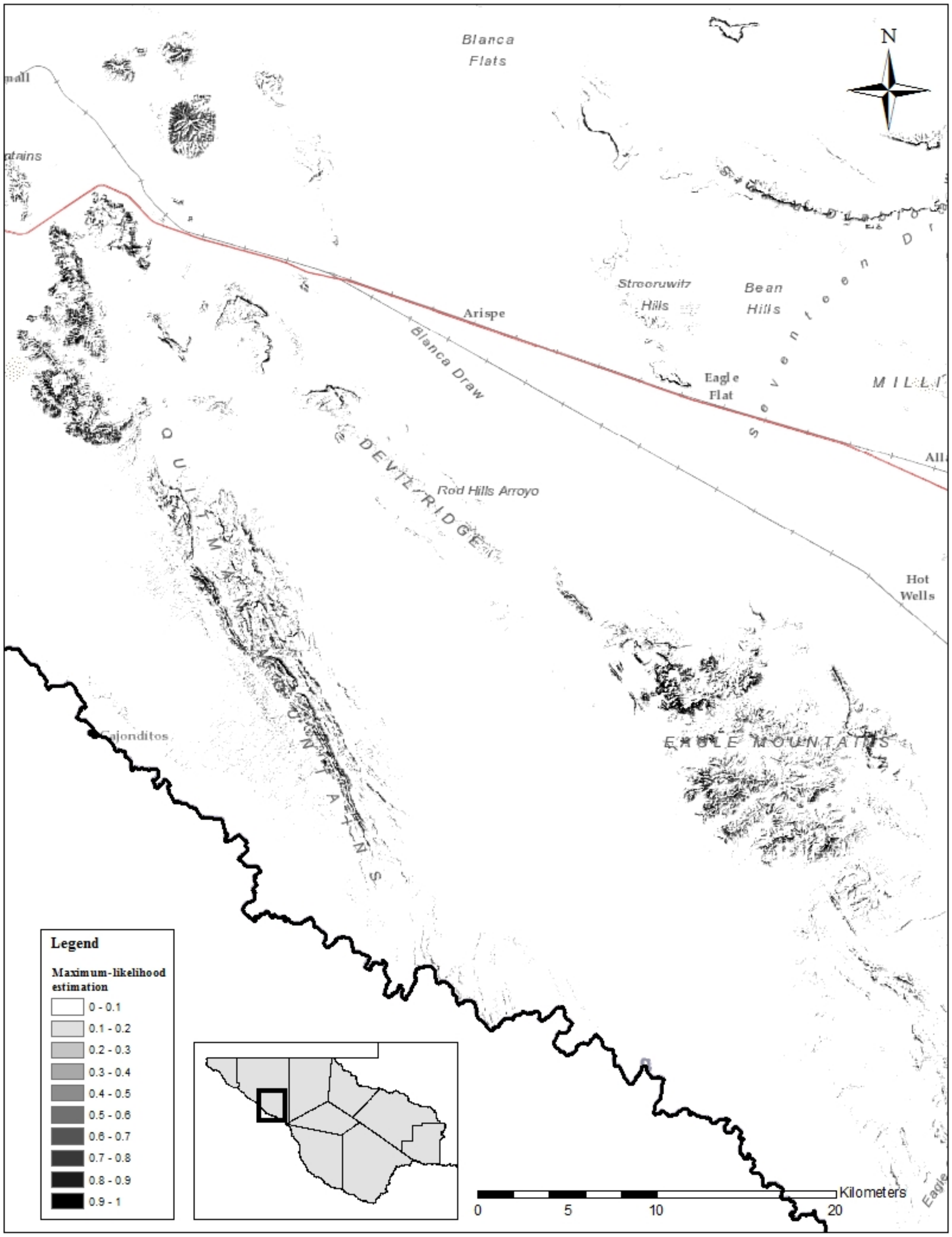

Figure A.13. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Quitman and Eagles Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

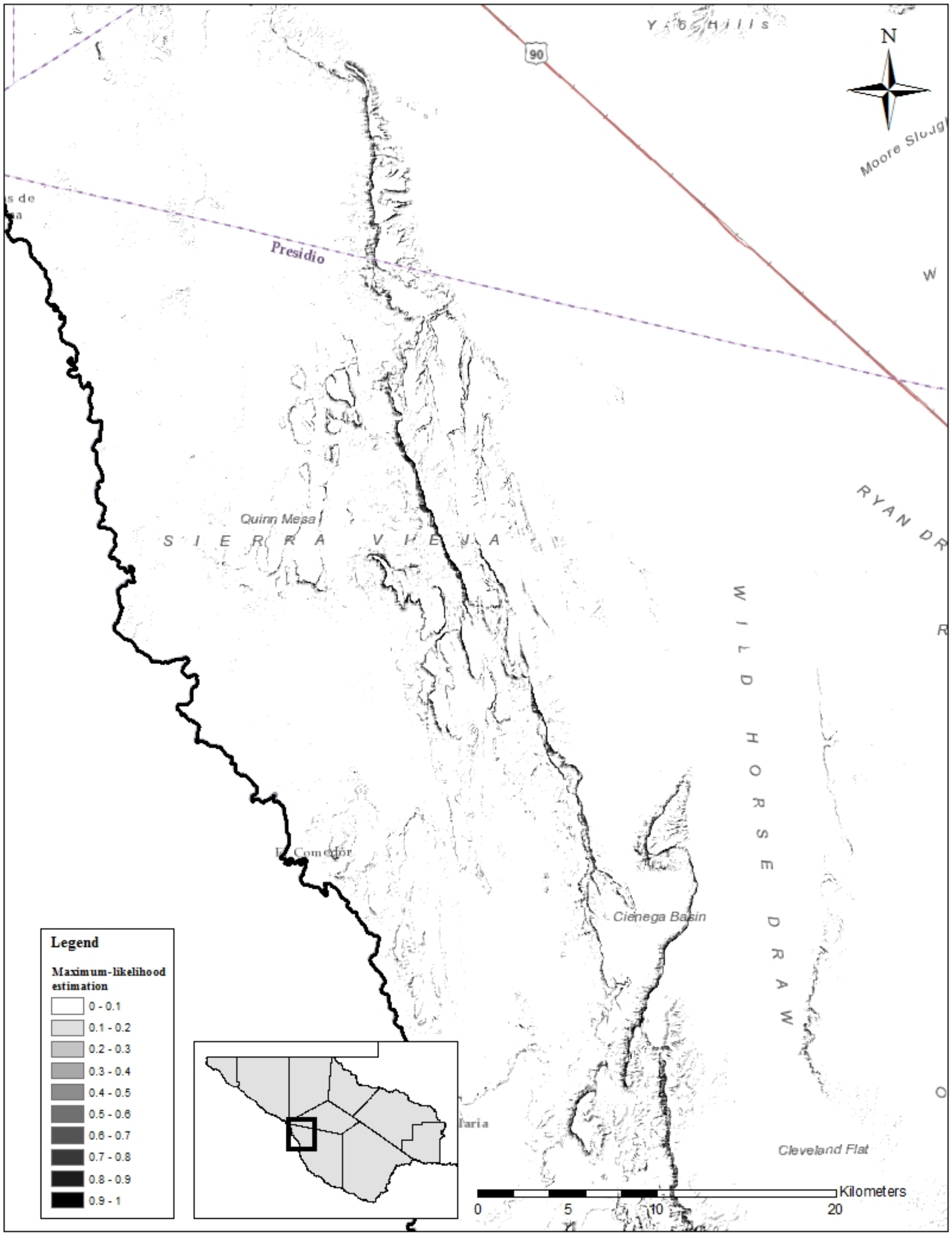

Figure A.14. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Sierra Vieja Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.

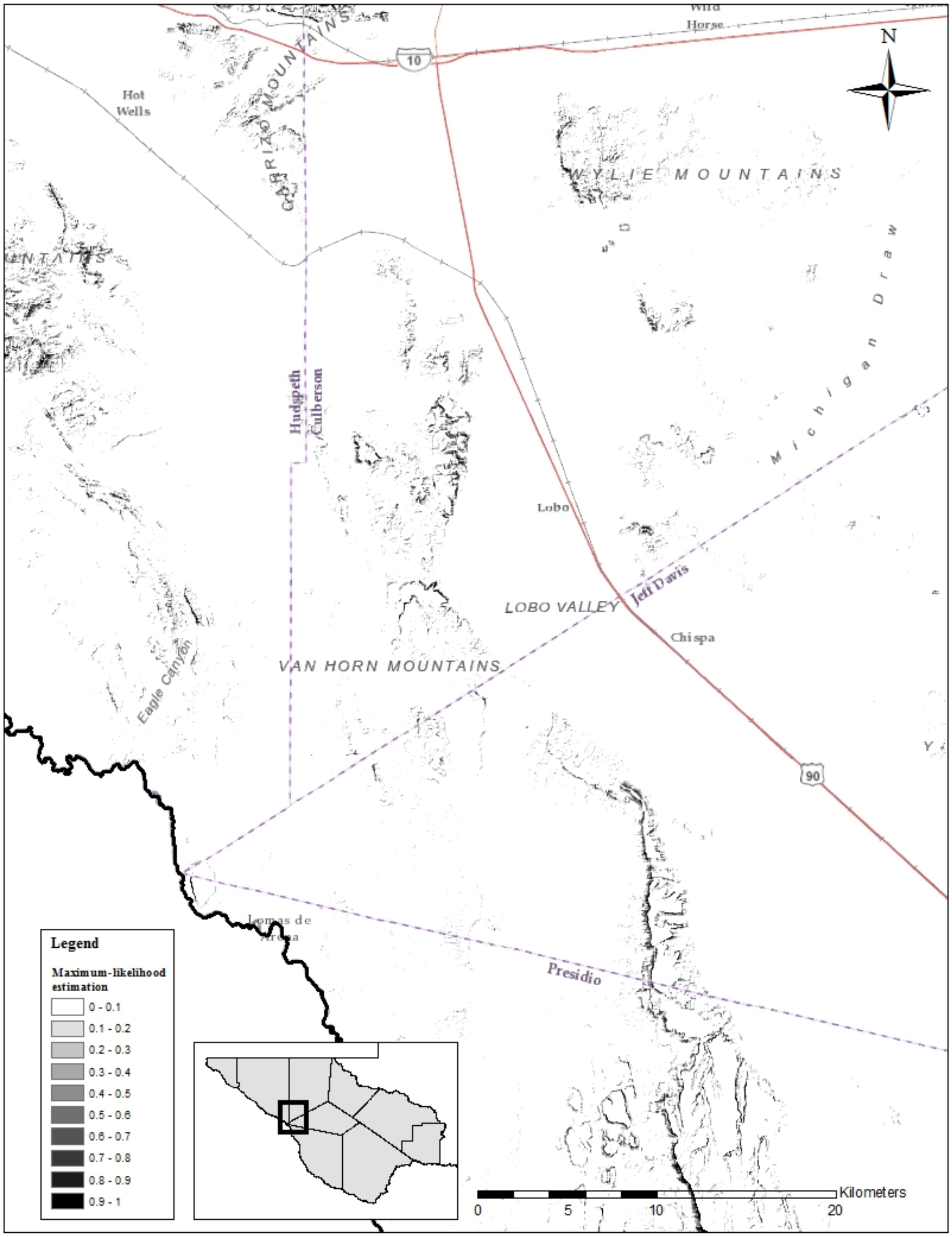

Figure A.15. The probability of occurrence for desert bighorn (*Ovis canadensis* spp.) in the Van Horn Mountains, Trans-Pecos, Texas, USA. Darker areas represent higher probability of occurrence.