

AN ASSESSMENT OF HABITAT SUITABILITY FOR PRONGHORN POPULATIONS  
OF THE CENTRAL VALLEY REGION OF CALIFORNIA

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## ABSTRACT

An assessment of habitat suitability for pronghorn populations of the Central Valley region of California

Virginia Burroughs

Efforts to reintroduce and maintain populations of pronghorn (*Antilocapra americana*) to the California Central Valley, specifically the Carrizo Plain National Monument (CPNM) and the Mojave Desert (Antelope Valley) portion of Tejon Ranch, have largely been unsuccessful due to dwindling numbers of translocated animals. The objective of this study was to improve upon previous models for the CPNM using aerial survey data and then apply the model to the Tejon Ranch. Aerial survey data collected from 2000-2010 on the CPNM was used to establish “use” and “non-use” areas in the model. Model variables included vegetation type (forest, shrub, grassland, semi-desert scrub, crops, and bare areas), slope, and road density. Vegetation and road density variables were treated categorically and slope as a continuous variable. Kernel density estimation (KDE) was used to estimate utilization distributions and home ranges (Fieberg 2007). An 80% isopleth was used to define “used” and “unused” habitat areas within the study site. Binary logistic regression was used to detect correlations between habitat variables and habitat use by pronghorn. Results of the regression analysis indicated overall significance with a p-value of  $< 0.0001$  (testing that all slopes = 0). Each habitat variable comparison was made after adjusting for the other variables (e.g., slope effects were evaluated after adjusting for road density and vegetation type) and was found to be significant. Each variable coefficient was then included in a predictive equation and entered into GIS to generate a map to predict where pronghorn would likely be observed. Similar layers were created for the Tejon Ranch and the predictive equation was run with the CPNM statistical analysis. Limited conclusions about habitat suitability on the CPNM or the Tejon Ranch can be made based on the habitat data available for this model. While slope, road density, and vegetation type are all significant habitat variables influencing pronghorn habitat use, further study is needed to understand the mechanisms driving these relationships. With additional data expansion of the current habitat suitability model would help to further define pronghorn habitat use, specifically the creation of a focused model of a particular season, life history period, or individual animal use to identify more detailed habitat use patterns.

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## INTRODUCTION

Pronghorn (*Antilocapra americana*) are a unique species native to North America. Historically their distribution was widespread across the western United States, including the California Central Valley. However, due to overhunting and habitat loss, pronghorn were extirpated from the Central Valley by the 1940's (Longshore and Lowrey 2008). Efforts to restore populations of pronghorn to portions of their native range resulted in translocation efforts in the late 1980's-early 90's to the Carrizo Plain National Monument (CPNM) and the Tejon Ranch. Originally over 200 animals were translocated to the Carrizo Plain area and roughly 100 to the southern portion of the Tejon Ranch. Since the time of release, both populations have declined. Total numbers are estimated to be approximately 30-35 animals at both locations (Bob Stafford, Mike White, personal comm.).

Local biologists, including the California Department of Fish & Game (now Wildlife; CDFW), have made several recent efforts to better understand why the CPNM pronghorn population is declining. One of those efforts has been to try to better understand the habitat conditions. To assist with that the CDFW commissioned two efforts to model and map pronghorn habitat characteristics and quality in the CPNM. The first of these models restricted its analysis to 46 pastures within the CPNM boundary and found that overall pronghorn habitat suitability in these areas ranked moderate to low (Longshore and Lowrey 2008). The second model extended its evaluation outside of the CPNM boundary and included the northern end of the Carrizo Plain. This model suggested that overall habitat suitability for pronghorn was high throughout the majority of the Carrizo Plain (Penrod, et al. 2010). One of the primary differences between these two models was the scale at which vegetation was characterized, with the first model using finer-scale vegetation data for its analysis. The conflicting results of these two models reflect the need for additional study of pronghorn habitat suitability and the factors that contribute to quality habitat on the CPNM.

The objective of this study was to create a predictive model of suitable habitat on the CPNM using aerial survey GPS data of pronghorn locations as the basis for establishing used and unused habitat areas. Three explanatory habitat variables were included in the final working model: slope, road density, and vegetation type. This model was then applied to the Tejon Ranch in an effort to identify suitable habitat areas for pronghorn on the Ranch and assess the applicability of the model to other habitat areas.

## **LITERATURE REVIEW**

Currently, the scientific literature offers some evidence regarding the interactions of pronghorn with certain habitat types and characteristics. My purpose in this literature review is to summarize some of the more pertinent investigations of how pronghorn populations are impacted by habitat conditions, and the design of habitat suitability models for pronghorn and other wild ungulates.

Studies of pronghorn habitat and range varied in their focus and conclusions. Based on Deblinger and Alldredge's (1991) study, water needs within pronghorn habitats remain unclear, and availability may have little influence on pronghorn distribution in normal precipitation years. Clemente, et al. (1995) estimated home range sizes to average  $2,259 \pm 656$ ha and found that adults have smaller home range sizes than yearlings due to familiarity with the habitat. There was also some agreement from the literature regarding use of habitat for fawn bed sites. Fawn bed site selection is believed to be based on predator avoidance; therefore bed sites are selected that provide suitable concealment while the fawn is bedded down, but increased visibility prior to and after bedding (Canon and Bryant 1997). Additionally, the general habitat containing the bed site was found to be influenced by the dam, but the fawn independently selects its bed site (Alldredge, et al. 1991). Beyond that, identification of suitable habitat and range components becomes more varied and influences of pronghorn distribution require continued research. The one area of accepted consensus in studies of pronghorn habitat and range characteristics is that pronghorn prefer habitats characterized by gentle slopes that are made up of open grassland or shrub-grasslands.

Dietary studies for pronghorn tend to describe them as opportunistic herbivores that rely primarily on forbs or shrubs rather than grasses (Smith, et al. 1998). Jacques, et al. (2006) determined that when the forbs are not the primary component of the diet, then availability of

forbs is likely limited due to low habitat quality, which may also be a contributing factor to a population decline. Studies of dietary overlap tended to agree that the greatest incidence of dietary overlap between pronghorn and livestock or other wild ungulates (such as feral horses) tends to occur in years of drought or lower precipitation when there is less availability of preferred forage so that pronghorn make necessary dietary adjustments (Stephenson, et al. 1985 and McInnis and Vavra 1987).

Mortality and recruitment factors are varied and complex and are primarily attributed to predation, nutrition, and climate factors. These variables manifest themselves in a variety of ways depending on the region of the United States being studied. Drought/low precipitation, as in indirect influence, and predation are generally considered to be the two primary factors in mortality and low recruitment, based on the available literature. Pronghorn exhibit strategies for avoiding predation, such as birth synchrony, “hiding” their young, and birth-site selection and fidelity (Gregg, et al. 2001 and Barnowe-Meyer, et al. 2010). However, these strategies begin to fail in times of drought when less nutritious forage is available. Thus, recommendations to reduce mortality and increase recruitment tend to center around habitat improvement and management, particularly for small populations that are particularly vulnerable to stochastic events (Dunn and Byers 2008). Predator control is another recommended management strategy, although its effectiveness is still unclear and may only be useful during certain seasons and in conjunction with habitat management. Further research of mortality and recruitment factors is advised.

Tracking and monitoring techniques primarily consist of aerial surveys and the use of GPS collars. Firchow, et al. (1990) concluded that more study is needed to determine the observability of pronghorn during strip surveys in different habitats in order to evaluate the usefulness of this technique for management purposes. Allen and Samuelson (1987) determined that refined census procedures are needed and are especially important in areas of marginal pronghorn range that

typically have more intensive land use and limited pronghorn numbers. And Ransom (2012) recommended techniques for reducing bias that may produce better population estimates for use in decision making and management. Brief consideration of the use GPS units addressed the need for establishing precision, accounting for effects of cover and topography, and the potential effects on wildlife (Hansen and Riggs 2008 and McMahon, et al. 2011).

Lastly, analyses of habitat suitability modeling and kernel density estimation methods generally concluded that these are valid techniques for assessing pronghorn habitat suitability and probability of use. The usefulness of habitat suitability models is often debated, however Roloff and Kernohan (1999) suggest that when properly developed and tested, habitat suitability index models should satisfy the requirements of being repeatable, scientifically credible, and legally defensible. And the use of kernel density estimation, provided a representative sample of locations, is a reasonable method for obtaining useful estimates of home-range size and relative space use (Fieberg 2007b). Also, models may be improved by creating focused models of particular seasons, life history periods, or individual animal use rather than year-round models (O'Brien, et al. 2005).

With the following compilation I have attempted to synthesize the current literature on pronghorn habitat and range characteristics, diet, mortality and recruitment factors, tracking and monitoring methods, habitat suitability modeling, and the use of kernel density estimation to infer population characteristics from. I have also suggested some specific management strategies based on this review, and included ideas for additional research drawn from this literature that might be helpful for improving management of pronghorn in this area.

### ***General Ecology and Life History of Pronghorn***

I will begin with a brief synthesis of general pronghorn biology and ecology, some aspects of which will be addressed in more detail throughout the review. Pronghorn (*Antilocapra*

*americana*) are a species native only to North America with five recognized subspecies: *A.a. americana*, *A.a. oregona*, *A.a. mexicana*, *A.a. peninsularis*, and *A.a. sonoriensis* (O’Gara and Yoakum 2004). Pronghorn are also one of the smaller ungulate species found in North America with a body length of 40-60 inches and weighing just 90-120 pounds. The typical lifespan of the pronghorn is 5-9 years, although it is possible for pronghorn to live up to 14 years (O’Gara and Yoakum 2004).

Pronghorn are considered opportunistic browsers and exhibit a preference for palatable and succulent forage. During the spring and summer they graze primarily on grasses and forbs and switch to shrubs during the fall and winter when grasses and forbs are harder to come by. Forbs are the preferred forage and pronghorn population dynamics may be related to the availability of forbs for many rangelands (O’Gara and Yoakum 2004).

Pronghorn habitat can be generally divided into three categories: grassland, shrub-grassland, and hot deserts. Within these types of habitats, pronghorn use a wide variety of ecosystems that include: steppes, grasslands, meadows, forb patches, dry lake beds, and recent wild burns (O’Gara and Yoakum 2004). Population densities of pronghorn vary and are directly related to habitat characteristics, particularly precipitation and subsequent water availability and, most importantly, vegetation. In addition to requiring certain vegetation types for forage, pronghorn prefer habitats characterized by a mixture of grasses, forbs, and shrubs that are generally no higher than 24 inches and preferably 15 inches high. In the winter when preferred forage plants are most difficult to find, survival of pronghorn is typically the lowest. The combination of adverse weather conditions and lack of forage often contributes to high losses of pronghorn in many rangelands (O’Gara and Yoakum 2004).

The structure and group size of pronghorn varies depending on the season. Typically, group sizes are larger towards the end of breeding season and are made up of mixed ages and sexes that stay

together through the winter months (Rolling Hills 2005). These large groups break up into smaller groupings come spring that are segregated by sex as fawning season approaches. Does form small groups of generally 12 or fewer members and young males form small bachelor groups. Older males (>3 years) compete for and defend territories (Rolling Hills 2005).

The pronghorn breeding season typically takes place during September or October over a 2-3 week period, although it may begin in August for southern populations. Pronghorn reach sexual maturity at 15-17 months of age, at which time does typically mate. Males, however, generally do not breed until at least 3 years of age, when they have established dominance and territory (Rolling Hills 2005). Fawning generally takes place from mid-May to mid-June after a 250 day gestation period, and does typically give birth to twins (although it is common for first-time mothers to have a single fawn) (O’Gara and Yoakum 2004, Rolling Hills 2005). Does that are ready to fawn will seclude themselves from the rest of the group and fawn in an isolated location. Fawns are large in size in proportion to their mothers and pronghorn does exhibit the highest known rate of maternal investment among ungulates, which apparently makes does and fawns particularly sensitive to forage conditions. Newborn fawns will lie quietly and “hide” in this various bed-site locations in between feedings for about a week while it gathers its strength (O’Gara and Yoakum 2004). Even so, at two days of age a pronghorn fawn can outrun a human and at four days it can outrun the average horse (Rolling Hills 2005). At three weeks of age fawns begin feeding on vegetation and will stop “hiding” and join the group several weeks after birth. During this period, fawns are most susceptible to high mortality due to inclement weather, predation, disease, and lack of quality forage (O’Gara and Yoakum 2004). Fawns are not weaned until 4-5 months of age, at which time it becomes difficult to distinguish them from adults, so rapid is their growth rate (O’Gara and Yoakum 2004, Rolling Hills 2005)

### ***Habitat and Range***

Factors affecting pronghorn selection of habitat and range are varied and not particularly well understood, including selection of fawning sites. Studies of pronghorn habitat identify some general common characteristics, but are also largely site-specific in their descriptions of optimal pronghorn habitat, making it necessary to determine habitat requirements based on each unique, individual ecosystem. No detailed assessment of pronghorn habitat characteristics on the Carrizo Plain National Monument has been done to date. This makes comparisons of habitat characteristics on the CPNM to features described in the literature difficult.

### ***Pronghorn and Free Water***

Pronghorn habitats are typically characterized by areas of grasslands of low to medium height, mixed grass-shrub, and desert habitats. Pronghorn population numbers are identified as highest within these habitats when annual precipitation ranges from 25-38cm (Clemente et al. 1995). This connection suggests that water may be a limiting factor in determining distribution in habitats with low annual precipitation. One suggestion regarding distances to water is that water must be available within 1.6-6.4km of the home range, while others suggest available water must be no farther than 3.2km; clearly, the availability of water on pronghorn habitat is not completely understood. (Clemente, et al. 1995, Deblinger and Alldredge 1991)

Deblinger and Alldredge sought to determine if the availability of free water could influence pronghorn distribution in a portion of Red Desert, Wyoming. The study habitat was characterized by very little water availability by midsummer such that pronghorn relied on plant-moisture and rain puddles for water (Deblinger and Alldredge 1991). The general indication from the study was that more pronghorn were found in locations with free water; however when free water was unavailable, spatial distributions did not change. Summer weather during the course of the study was considered to be normal without any unseasonably hot and/or dry conditions, thus, these

results may not be applicable during extremely hot or dry conditions (Deblinger and Alldredge 1991). Additionally, other studies have found pronghorn numbers to be greatest in areas with abundant free water, but distribution is not limited to those areas and habitats with large pronghorn numbers also contained a large forb component. Lastly, Deblinger and Alldredge found that distances traveled by pronghorn during summer were minimal compared to other times of the year, and this was likely based on environmental variables such as water availability. Management suggestions based on this study suggested that water development as a management strategy should be carefully evaluated prior to implementation as a sole means of influencing pronghorn distribution or density, or for providing a seemingly limited resource. In habitats similar to the study area, water availability may actually have little influence on pronghorn distribution during most years (Deblinger and Alldredge 1991).

A similar study on the effects of water on pronghorn habitat and distribution conducted in the Chihuahuan Desert of southcentral New Mexico evaluated the influence of water distribution on pronghorn movements by determining their closest distance to permanent water, their home range size, and habitat use (Clemente, et al. 1995). The study found that pronghorn distance to closest water source for all seasons averaged  $2,667 \pm 164\text{m}$ . Also, home range sizes averaged  $2,259 \pm 656\text{ha}$  and ranged from 938 to 3,773ha. Pronghorn core habitat areas had two things in common: mid-points of the core areas were no further than 3km from the nearest water source, and the core area was larger and incorporated a higher proportion of the home range when the distance from the nearest permanent water source to other water sources was closer than 6km (Clemente, et al. 1995). These authors found that adults ranged farther from water than yearlings, but distances to water did not differ seasonally. The authors speculate that yearlings may have greater water needs than adults, and this might have been why they were commonly found closer to water sources than adults. Adults also made shorter daily distance movements and had smaller home ranges than yearlings. The authors speculated that adult familiarity with an area gained through

established travel routes and home ranges allowed them to meet their minimum daily nutritional requirements within a smaller area and less daily travel than less experienced yearlings (Clemente, et al. 1995).

Pronghorn water needs are poorly understood, however prior published reports have stated that pronghorn will not use freestanding water sources, so the point of this next study was to determine whether this was in fact true due to the beneficial role free-standing water plays in habitat improvement for other desert ungulates (Morgart et al. 2005). The Sonoran pronghorn range is <10% of its suspected historical habitat and the subspecies was on the verge of extirpation. The habitat occupied by Sonoran pronghorn experiences a bimodal precipitation pattern characterized by widely scattered thunderstorms in the summer and a winter rainy season. Much of the precipitation during the summer thunderstorms is lost to runoff and evaporation, making it unavailable for plant growth ( $\frac{1}{3}$  to  $\frac{1}{2}$  of the annual rainfall can occur from July to early September). During the winter rainy season (December-March), the rains are gentler, of longer duration, and more widely distributed, making it more available for plant growth (Morgart, et al. 2005). Results of this study found that Sonoran pronghorn do indeed use freestanding water sources, whether natural or anthropogenic, and in particular during the hottest and/or driest times of the year (typically May-August). It was also found that there was a continual use of water sources by the same animals. Thus, the authors recommend the use of freestanding water sources as a management strategy for improving habitat conditions for Sonoran pronghorn. Further study is needed to determine timing and use of water sources relative to reproductive success, fawn survival, and recruitment (Morgart, et al. 2005).

#### *Estimating Pronghorn Home Range Sizes*

Reynolds and Laundré (1990) suggest that it is important to consider when looking at home-range estimates that accuracy is often biased by sample size and the time interval between recorded

locations and sought to determine the optimal interval between locations of radio-tagged animals by analyzing estimates of daily movements and home range sizes of pronghorn and coyote that would not sacrifice the utility or compromise the reliability of the data set for either spatial or behavioral analysis. They found that daily distance traveled estimates for both coyotes and pronghorns were sensitive to the time interval between locations (Reynolds and Laundre' 1990). The longest location interval for pronghorn for which reliable and useful movement data could be obtained appeared to be four hours. Sampling intervals based on statistically independent data provided underestimated home range size (calculated by either a minimum area method or linked-cell grid method) and daily distance traveled. Better estimates of true home-range sizes were obtained by autocorrelated data than by independent data (Reynolds and Laundre' 1990). Therefore, the authors concluded that to maximize available information from radio-telemetry studies, data should be collected at short time intervals, even though the premise of statistical independence may be violated. Information of biological significance may be sacrificed if sampling efforts are restricted to intervals exhibiting statistical independence (Reynolds and Laundre' 1990).

#### *Fawning Habitat Characteristics*

Fawning habitat characteristics are believed to be important to fawn survival, but the available literature is limited in its descriptions of what actually makes up preferred fawning habitat for does or fawn bed-site selection (Canon and Bryant 1997). A study by Canon and Bryant (1997) in Texas attempted to quantify relations among vegetation and topography of fawn bed sites based on the hypothesis that bed-site selection affects fawn survival. The study found that fawn activity periods appeared to be initiated by mothers, but fawns independently selected bed sites. High variation in bed site selection indicated that fawns selected for, or at least initially imprinted on, an optimal combination of local habitat features necessary to avoid predators (Canon and Bryant 1997). Characteristics of bed site selection indicated that important factors of selection

are elements of both concealment and visual awareness. Fawns' age also influenced changes in microhabitat characteristics associated with bed sites. Older fawns tended to select less concealing features, indicating that younger fawns that are less mobile need more concealment from predators while older fawns with more developed motor skills need more visibility so they can flee from predators (Canon and Bryant 1997). Overall this study determined that fawns seem to select for bed sites that provide suitable concealment while bedded down but increased visibility prior to and after bedding. These selection preferences appeared to be related to predator avoidance strategies, which evolved as the fawns developed motor skills necessary for fleeing predators. The authors recommended that it may be possible to provide adequate cover for fawns and forage for dams without hampering long-range visibility by utilizing livestock stocking rates that encourage increased grass and forb production (Canon and Bryant 1997).

A second study on fawning habitat sought to ascertain characteristics of vegetation at pronghorn birth and fawn bedding sites and compared those to nearby areas (Alldredge, et al. 1991). The authors stipulated here that ungulates demonstrate two distinct forms of maternal-infant behavior: "hiding" and "following," both of which are thought to be influenced by predation. Neonates that hide seem to behave independently in their selection of hiding or bedding sites, but appear to be dependent on their mothers to initiate activity periods. Hiding by pronghorns is a successful strategy likely due to four intrinsic features of the young: 1) lack of early scent gland development, 2) cryptic coloration, 3) the ability to lie motionless for long periods of time, and 4) the ability to select proper concealment; with the last three being largely influenced by habitat (Alldredge, et al. 1991). This study found that birth sites were observed to have greater shrub canopy cover compared to random low use area sites, but that parturient does never used patches of dense, tall (>1.5m) big sagebrush and rubber rabbitbrush likely due to the fact that these areas are usually associated with draws, which are frequented by predators. Similar canopy cover was found between birth sites and fawn bed sites, and there were a couple likely reasons for this.

First, fawns may “imprint” on vegetation at their birth site and thus continue to seek similar cover, and second, dams may directly influence bed site selection. Fawn bed site canopy cover was also significantly different from random sites within the fawn home range. Fawns did not use dense, taller shrub areas and it has been suggested that bed site selection by fawns is primarily a behavioral response to predation (Alldredge, et al. 1991). Additionally, the authors concluded that the general habitat containing the bed site was influenced by the dam, but the fawn independently selected its bed site. Observations of fawns revealed that post-nursing, fawns would move away from the dam and select their own bed site. Also telemetered fawns never used the same bed site twice and fawns may change bed sites as much as 10 times a day. Lastly, fawns tended to seek seclusion in vegetation similar to that in which they were born until about 3 weeks of age (Alldredge, et al. 1991).

#### *Birth Site Fidelity*

Wiseman, et al. (2006) looked at pronghorn birth-site fidelity. In many mammal species, the mother’s choice of birth site is a potentially important contributor to infant survival, particularly in mobile species. This may be even more critical in hider species compared to follower species as hider neonates typically stay in a restricted area after birth until after the hiding period ends (Wiseman, et al. 2006). This study was conducted at the National Bison Range where pronghorn does appear to be highly variable in their birth-site location choice and birth-site fidelity may vary from year to year for individuals. Objectives of the study were to determine whether female pronghorn practice birth-site fidelity and with what frequency, to identify factors potentially associated with fidelity to a specific site, and determine whether number of fawns weaned, maternal age, or precipitation in the previous year predicted birth-site fidelity using observational data (Wiseman, et al. 2006). Overall, birth-site fidelity of female pronghorns was low with 18% exhibiting fidelity and 82% moving to new birth sites from year to year. Some females were found to use the same birth-site fidelity tactic from year to year while others used different tactics

over consecutive years. Selection of birth-site by does may be a trade-off between nutritional requirements of the mother and necessity of lower predation risk to the fawn. Birth-site fidelity patterns may be a result of females tracking habitat parameters that vary from year to year (Wiseman, et al. 2006). Birth-site selection may also be based on predation risk since some birth-sites are used repeatedly by females (not necessarily the same individuals), indicating the possibility of the site allowing better detection of predators and defense of fawns. With this in mind, predator distribution and concentration changes may cause a female to abandon a previously successful site. Ultimately, It was unclear whether female pronghorn that successfully raised twins change sites based on changes in environmental conditions or if it was simply a random stay-move strategy (Wiseman, et al. 2006). Does did seem particularly sensitive to fawn mortality and were equally likely to change birth-sites in the subsequent year if either one or both fawns died. The findings here demonstrate the complexity of birth-site selection and fidelity and indicate the need for further study of other populations, particularly whether pronghorn subjected to lower levels of fawn mortality would exhibit higher birth-site fidelity (Wiseman, et al. 2006).

### ***Pronghorn Diet***

It is generally agreed upon that pronghorn primarily prefer forbs as the majority of their diet composition, but there are a number of factors that may influence pronghorn forage selection, including availability of preferred forage, forage quality and nutritional needs, and interspecific competition, among others. Insufficient quantity and quality of forage has been considered a factor in pronghorn population declines and research efforts have been made in an attempt to corroborate this hypothesis (Jacques, et al. 2006). Additional studies have been conducted in various habitat conditions to determine the effects of competition with livestock and other wildlife for available forage and how large a role this competition may play in pronghorn population decline.

### *Diet Composition*

A study by Jacques, et al. (2006) attempted to assess pronghorn diet composition and forage selection in Wind Cave National Park (WCNP), South Dakota due to a decline in population numbers to about 30 animals from a reintroduced pronghorn population and based on the assertion that the quality and quantity of forage consumed by pronghorn influences production and survival. Pronghorn exhibit selective foraging strategies compared to larger North American herbivores however little is understood about food selection by pronghorn in similar habitats to the WCNP (i.e., grassland-dominated habitats) where sagebrush distribution is limited. The authors found that pronghorn diets contained high amounts of blue grama throughout the year, shrubs were frequently consumed during winter months and forbs were used during summer months (Jacques, et al. 2006). However, dietary composition of the WCNP pronghorn differed from other pronghorn populations located in western North America. This was based on information from studies that found that pronghorn consume large amounts of sagebrush year-round, particularly during fall and winter due to the increased availability of high protein content in shrubs relative to forbs and grasses which indicates the necessity of shrubs in the diet for the overall health and survival of pronghorn (Jacques, et al. 2006). However, pronghorn of the WCNP encountered a limited distribution and availability of shrubs, particularly sagebrush. In contrast, the role of grasses in pronghorn diets is less clear and has been identified as being a total use of just 10% in annual diets in many habitats (Jacques, et al. 2006). For WCNP pronghorn, preferred types of shrubs and forbs were limited in availability and therefore the pronghorn had to make greater use of grass than is typical. The literature has identified forbs as the preferred forage of pronghorn and consumption of forbs exceeds consumption of grasses and shrubs across all habitat types and seasons (Jacques, et al. 2006). However, this study found that pronghorn did not consume a large variety of forbs, nor did forbs make up the largest percent of their diet during the summer months, when forbs would be more readily available. The fact that the WCNP

pronghorn diets included a small proportion of plants normally common in pronghorn diets elsewhere suggested that habitat quality contributed partly to the population decline (Jacques, et al. 2006). Additionally, drought conditions during the study may have contributed to low production of preferred forages and poor habitat quality. The relationship between drought and forage selection by pronghorn is unclear, yet Jacques, et al. (2006) hypothesized that the long-term drought conditions affected habitat quality which resulted in reduced distribution and diversity of optimal forage and in turn contributed to population decline. Additional study is needed to quantify forage availability and determine the quality of preferred forage during normal precipitation years in the WCNP (Jacques, et al. 2006).

#### *Diet Composition and Overlap*

A northcentral New Mexico study looked at the possibility of expanding pronghorn numbers if forage quantity was increased (Stephenson, et al. 1985). Specifically, the study investigated the composition of pronghorn diets yearly and seasonally, identified dietary overlap between pronghorn, cattle, and sheep, and, due to severe drought during one growing season of the study, an assessment was made of the influence of drought on pronghorn diets. The study area was dominated by a low shrub-short grass vegetation type (Stephenson, et al. 1985). Findings indicated that pronghorn diets contained the greatest number of plant species in the spring and the least during the winter. Forbs were used in the diet more than grasses or shrubs. Pronghorn used shrubs the most of the three functional groups during the winter, forbs the most during the summer, and grass during the spring (Stephenson, et al. 1985). When forbs were unavailable, pronghorn selected shrubs over grasses when both shrubs and grasses were available and winter was the highest period of browse consumption while summer was the lowest. Winter diets were influenced the most by drought conditions and spring diets were influenced the least. The primary effect of drought on winter diets was the reduction of forb consumption and increased shrub consumption. Also, comparisons of pronghorn diets across years indicated that diets can

vary drastically between years on the same range (Stephenson, et al. 1985). In the analysis of dietary overlap, a high proportion of both forbs and shrubs were consumed by both cattle and sheep when green grass was unavailable. Moderate grazing intensity by cattle during the summer can result in complete utilization of many shrub and forb species important for wild ungulate diets. Thus, it would appear that dry winters and springs may have more adverse effects on pronghorn and lead to higher dietary overlap in northcentral New Mexico than dry summers (Stephenson, et al. 1985).

Further study into the effects of dietary overlap and interspecific competition are analyzed in McInnis and Vavra (1987) and Smith, et al. (1998). McInnis and Vavra (1987) postulated that high rates of feral horse population growth in areas where horses, cattle, and pronghorn coexist may lead to competition for resources, particularly dietary resources. Therefore, it is necessary to understand how species use and partition the resources available to them if they are to be managed in such a way as to achieve ecological balance (McInnis and Vavra 1987). For this study, seasonal food habits, dietary overlap, forage quality, and dietary quality for horses, cattle, and pronghorn in southeastern Oregon were examined. Results of this study indicated that pronghorn consumed principally forbs and shrubs throughout the year, unlike horses and cattle. Availability of these forage classes influenced their use, which varied widely; forb use peaked in summer months and declined through the fall and winter when forbs became less available and pronghorn shifted to a diet dominated by shrubs (McInnis and Vavra 1987). Dietary overlap between pronghorn and horses averaged 16% on an annual basis and ranged from 7-26% with grasses making up 70% of the total overlap. Pronghorn dietary overlap with cattle was not high with an average overlap of 14% annually and grasses comprising approximately 50% of the overlap (McInnis and Vavra 1987). In the case of dietary quality, all three species consumed diets containing approximately 16% crude protein (CP) in the spring, but pronghorn selected plants with consistently higher CP levels than cattle or horses in every other season (McInnis and

Vavra 1987). Based on these results, the authors concluded that cattle and horses primarily used grasses for forage throughout the year, while pronghorn used grasses sparsely. Other studies have confirmed little dietary overlap between pronghorn and horses or cattle. However, these results do not rule out the possibility of exploitative competition, when the consequences of dietary overlap partially depend on resource availability. And while this possibility could not be evaluated in this study, the data does reveal the potential for exploitative interactions among these three species (McInnis and Vavra 1987). The low levels of dietary overlap between pronghorn and cattle or horses suggests a wide buffer between noncompetitive coexistence and exploitative competition. However, studies have shown that drought conditions contribute to pronghorn dietary adjustments that may lead to higher dietary similarity with cattle and horses such that competition for forage between browsers and grazers is more likely to occur on depleted ranges where food habits of these ungulates converge (McInnis and Vavra 1987).

The last dietary study reviewed here took place within the White Sands Missile Range (WSMR) in south-central New Mexico. This area is home to approximately 1,800 gemsbok (introduced African antelope), 1,400 feral horses, and 300-350 native pronghorn (Smith, et al. 1998). This study set out to determine diets, diet overlap, and key forage species (meaning  $\geq 5\%$  of the seasonal diet) of these three ungulate species on the WSMR. Identifying possible dietary competition in multispecies grazing systems through the use of diet studies can be crucial for determining impacts on native ungulates by free-roaming exotic ungulates (Smith, et al. 1998). There has been little evaluation of pronghorn diets in desert environments, but the literature describes pronghorn as being opportunistic herbivores that rely primarily on forbs or shrubs rather than grasses (Smith, et al. 1998). Literature sources used for this study indicated that grasses are ingested in early spring by pronghorn in northern New Mexico, with forbs and browse dominating the dietary intake throughout the year, particularly common winterfat. When grasses are selected by pronghorn, they are typically comprised of fine textured species, such as blue

grama. This study found that, overall, pronghorn diets exhibited little overlap with horse or gemsbok diets, likely due to their consumption of forbs and greater diet diversity (Smith, et al. 1998). Specific questions about dietary overlap between these species could not be addressed by this short-term study. Food habit studies should be conducted long-term to determine dietary needs under various climate systems and differences in dietary needs based on age and sex. These dietary differences based on sex and age could then influence conclusions about diet composition and overlap (Smith, et al. 1998) .

### ***Pronghorn Mortality and Recruitment***

Factors contributing to mortality and recruitment are varied, but often interrelated. They include predation, climate/precipitation, vegetation/forage quantity and quality, disease, human interference, etc. (Brown and Conover 2011). Regardless of the factor(s) involved, mortality generally affects young animals more often than adults, which in turn affects recruitment. This phenomenon is succinctly described by Tanner (1966). In his paper, Effects of population density on growth rates of animal populations, he explains that under some conditions, predation is a mortality effect that may be an increasing function of density and in others it may be a density-independent or even decreasing function. For example, at low prey densities, predation may be an increasing function of density and at high prey densities, predation may be a decreasing function. Tanner (1966) postulates that this scenario appears to be more complex in vertebrate species. For example, deer are non-territorial species that were originally controlled by predators, but with the disappearance of natural predators many deer populations became overpopulated. Population size in vertebrates is controlled by different processes for territorial versus nonterritorial species. In non-territorial species, population size is usually regulated by predation on juveniles (Tanner 1966). When this fails (such as when predators are removed by man), regulation shifts to competition for food which generally results in starvation of younger individuals. The significance to pronghorn specifically is that these processes that regulate

vertebrate populations affect either reproduction or juvenile survival. Therefore, in general, adults are exempt from the process and instead the production (and survival) of young is regulated (Tanner 1966). In the following studies, multiple factors of mortality and recruitment will be examined, keeping in mind Tanner's description of population density growth rates. It is interesting to observe that regardless of the factor described, the ultimate cause of mortality or low recruitment tends to result from predation or lack of adequate nutrition, which are the two regulators of population size identified by Tanner (1966). And while each of the following studies generally focuses on a primary mortality or recruitment factor, as stated above, these factors are typically interrelated so that it is uncommon for just one factor to be identified as the only cause of pronghorn mortality or low recruitment.

#### *Predation Effects*

One factor of significance and intense focus in pronghorn studies is predation. A study of pronghorn fawn survival on the Carrizo Plain National Monument (CPNM) attempted to identify a relationship between fawn survival and factors that affect survival in an effort to form an understanding of pronghorn population dynamics on the CPNM (Johnson, et al. 2010). Survival and recruitment rates of pronghorn fawns on the CPNM are poorly understood and population numbers have fluctuated and remain below the original number translocated to the area between 1978 and 1990 (Johnson, et al. 2010). A combination of variables are believed to contribute to fawn mortality, including predation, poor habitat conditions, disease, and adverse weather conditions during fawning. The majority of fawn mortality occurs within the first 17 days after birth. The primary cause of fawn mortality is often predation, however the importance of predation as a limiting factor for pronghorn populations covaries with habitat quality such that predation tends to increase with poor habitat quality and pronghorn habitat on the CPNM has been characterized as moderate to low quality (Johnson, et al. 2010). Another factor related to fawn survival is density dependence. Fawn survival is more often correlated with population size

and summer precipitation than coyote abundance in populations regulated by density dependence. Pronghorn exhibit birth synchrony, which is where most females in a population give birth in a relatively short period of time together, producing a high number of young that may serve to overwhelm the local predator population. This may reduce fawn mortality where pronghorn populations are large and vigorous, but in smaller populations such as the one inhabiting the CPNM, too few fawns may be born to “swamp” predators with prey. In this case, birth synchrony may instead lead to an Allee effect (Johnson, et al. 2010). Results of the study found that fawns on the CPNM are typically born late April-late May and births appeared to be synchronized; however too few fawns were born to saturate predators with prey. For this study, four fawns of the nine born in 2010 were collared; three of the collared fawns died before 20 days of age; two from predation and one from infection (Johnson, et al, 2010). Evidence from the sites suggested that one of the predation occurrences was from a coyote(s) and the other by an eagle. Fawning areas consisted of flat, open grassland habitat containing relatively little shrub cover and located adjacent to salt brush habitat. Due to the small sample size and the low probability of detecting a statistically significant result, the authors recommended continuation of this project before drawing any significant conclusions (Johnson, et al. 2010).

Gregg et al. (2001) investigated birth synchrony and its role in the survival of pronghorn fawns. As stated previously, predation is often a primary mortality factor of fawns and some ungulate species utilize birth synchronization as a strategy to reduce predation on young. The objectives of this study were to identify fawn mortality causes and compare survival of fawns born during non-peak vs. peak fawning periods in two fawning areas in southcentral Oregon. Results of this study found a relationship between birth date and fawn survival such that those fawns born during peak periods had greater rates of survival than those born during non-peak periods. This was the first time any research demonstrated the importance of birth synchrony in pronghorn and these results indicated that birth synchrony may be an important factor influencing neonatal pronghorn

survival (Gregg, et al. 2001). Typically, birth synchrony is not advantageous for species that hide their young but this does not appear to be the case for pronghorn. Pronghorn females tend to choose birth sites spaced as widely as possible, thereby lessening fawn concentration and predator encounter rates (Gregg, et al. 2001). Thus it appeared that birth synchrony coupled with wide spacing between fawns is an additional adaptation against high predation pressure on fawns. Female body condition and social status may also be factors that affect birth synchrony as does in good condition may come into estrus earlier, have shorter gestation periods, and experience more synchronized births than those in poor condition. Due to the possibility of a relationship between female body condition and birth synchrony, the authors recommended that management efforts should focus on providing high quality habitat in summer ranges (i.e., adequate available forage), or restoring habitat if necessary (Gregg, et al. 2001). It would also be prudent to avoid excessive disturbance during the breeding period, as excessive disturbance may disrupt the social breeding structure and thereby lengthen the breeding period and reduce the occurrence of birth synchrony. Results of the study also supported the suggestion that coyote predation may be limiting pronghorn population growth in the study area. However further research is needed in the following areas: annual variation of synchrony and problem associated with reduced synchrony, female nutrition during the summer (and how it affects birth synchrony), and female social structure during the breeding season (and effects on birth synchrony) (Gregg, et al. 2001).

Assessments of the effects of predation on pronghorn populations have led to studies that evaluate population response to predator, and specifically coyote, control. Brown and Conover (2011) looked at large-scale coyote removal and its effects on pronghorn productivity and abundance. It is unclear whether coyote predation alone actually limits pronghorn recruitment, however some data suggest that when coupled with inclement weather, disease, or habitat change, predation can have a significant effect on ungulate abundance. The study took place in southwest Wyoming and northeast Utah (Brown and Conover 2011). Many studies have produced mixed results as to

whether coyote removal actually decreases coyote abundance. This study, however, found that coyote densities were significantly lower in removal sites, likely due to the fact that removals took place over larger areas than are typical of most studies. Determinations of the impacts of coyote removal on pronghorn abundance and productivity showed that pronghorn productivity was higher within removal sites than non-removal sites and was positively correlated with the removal effort. Pronghorn abundance was also positively correlated with both the number of coyotes removed and the removal effort (Brown and Conover 2011). Thus, the resulting differences in pronghorn numbers between the removal and non-removal sites may have been a result of both an increase in pronghorn productivity in removal sites and the immigration of animals into the removal sites. In looking at the impact of timing on the effectiveness of coyote removal, there was a stronger positive correlation with pronghorn productivity and abundance when coyote removal occurred during the 5-month period prior to parturition as opposed to removal during intermediate (8 months) or long periods (10 months). This suggests that coyote removals occurring during winter and spring increased pronghorn productivity and abundance more than removals occurring the previous fall or summer (Brown and Conover 2011). Overall, the authors concluded that coyote removal may be used as a tool to enhance early pronghorn survival. However, the timing of the coyote removal is important; winter and spring removals appeared to benefit pronghorn abundance and productivity more than prior summer or fall removals. Thus it is important to concentrate removal efforts in the winter and spring to maximize benefits to pronghorn (Brown and Conover 2011).

A similar study looked at the removal of coyotes for livestock protection and how this may benefit free-ranging ungulates. Coyote predation has been identified as the greatest source of pronghorn fawn mortality in some parts of the western United States (Harrington and Conover 2007). Coyote control has produced an increase in offspring survival and pronghorn densities in areas where population densities are low and beneath carrying capacity. However, it is still

unclear if pronghorn densities or fawn survival would increase as a result of coyote control due to a lack of well-designed experiments implemented in areas greater than 1,000km<sup>2</sup>. Thus the objective of this study was to conduct a large-scale study covering a total of >1,900km<sup>2</sup> to evaluate whether pronghorn densities and offspring:female ratios are related to the level of predator control conducted for livestock protection. The study area included parts of northeastern Utah and northwestern Colorado (Harrington and Conover 2007). In this study, a positive effect was found for pronghorn densities based on hours spent aerial gunning and the number of coyotes removed. Particularly, preventive coyote population reduction during winter and spring appeared to be most effective for protecting livestock as territorial coyotes are most vulnerable during this time. And even though vacant territories will be reoccupied by new coyotes, they do not have the time to establish a territory and breed in the same year. As a result, coyote control during winter and spring may reduce predation on livestock and pronghorn due to a lack of pairs with pups, as opposed to actually reducing coyote densities (Harrington and Conover 2007). Offspring survival based on offspring:female ratios did not increase with predator control, but pronghorn densities were higher where coyote control had taken place. The authors hypothesized that ultimately pronghorn moved away from high coyote density areas to areas with few or no coyotes. This explanation is feasible, since it has been observed in other studies that ungulates will change their location, behavior, or habitat to avoid predation risk (Harrington and Conover 2007). Pronghorn may have learned to move into areas where predator control has taken place if it is conducted in the same place each year in order to avoid coyotes. If so, then the authors may not have found an actual increase in pronghorn population densities over a large area, but rather movement by pronghorn into areas where coyote control had taken place and lowered the predation risk. Since preventive winter-spring coyote removal for livestock was positively correlated with increased pronghorn densities, this would suggest a removal program implemented during the winter-spring seasons. However, the benefits of coyote removal only last as long as the predator program continues (Harrington and Conover 2007).

Next, Phillips and White (2003) suggested that stochastic population modeling is a powerful management tool that is increasingly relevant for evaluating possible outcomes of controversial management actions, such as predator control. Their study presents the results of a population modeling effort to help resolve management controversy regarding the use of lethal coyote control to benefit a free-ranging pronghorn population. The study site was the Hart Mountain National Antelope Refuge (HMNAR) in southeastern Oregon. Objectives of the study were to independently evaluate potential effects of coyote control on pronghorn and use existing population data for pronghorn on the HMNAR and surrounding areas and results of a coyote control study near the refuge for deterministic and stochastic modeling to explore potential outcomes of focused, intensive coyote control. The modeling exercise was presented as an example of how modeling can be used in evaluating options for management plan development (Phillips and White 2003). The authors began with an initial population size of 1400 because this number was equivalent to the estimated population size of the HMNAR herd and it would be useful to evaluate how the “present” population would respond to coyote control. The model showed that intensive coyote control may boost the pronghorn herd at HMNAR substantially, particularly if it is implemented before the herd size decreases much below the threshold of 1250 individuals. Thus, three years of coyote control at the time of the study may have been effective at keeping numbers above this threshold (to about 10 years after initiating control), but this was not guaranteed. In the absence of coyote control the median simulated population size always decreased over time. However, management changes at HMNAR to enhance habitat (e.g., feral horse removal, cattle exclusion, and greater prescribed burning use) may also enhance fawn neonate and overwinter survival (Phillips and White 2003). Coyote control has often been dismissed as an ineffective tool for increasing pronghorn populations due to the coyotes’ ability to quickly rebound from removal efforts. However, the results of this study encourage cautious optimism regarding the use of coyote control to increase pronghorn numbers. At HMNAR, habitat improvements would ideally be sufficient to allow for increase in pronghorn population

numbers so as to increase resiliency to habitat variation. However, if that is not enough, then periodic application of spatially and temporally focused intensive coyote control could be utilized to supplement habitat enhancements (Phillips and White 2003).

### *Climate and Precipitation Effects*

Another oft-studied pronghorn mortality and recruitment factor involves climate and precipitation. This next study discusses these effects, and illustrates the interaction between climate and predation. The Sonoran pronghorn (*Antilocapra americana sonoriensis*) range from the plains of west-central Sonora, Mexico to north and southwestern Arizona (Bright and Hervert 2005). A severe drought in 2002 resulted in 80% mortality, leaving the total population at less than 30 animals. An understanding of predation and other mortality factors was deemed essential for developing management strategies that would promote the long term survival of the Sonoran pronghorn. The primary habitat of the Sonoran pronghorn within the study area consisted of valleys and bajadas, with some use of foothills and lower drainages of mountain ranges. Water sources were uncommon and the area is characterized by a bimodal rainfall pattern (Bright and Hervert 2005). Based on the study results the researchers concluded that drought is a significant indirect factor in pronghorn mortality by decreasing the quality and quantity of forage and water availability. Adult mortality appeared to result most often from increased predation, as adults moved to areas with more diverse and succulent forage that was found in areas with higher predator densities and poorer visibility/rugged terrain. Fawn mortality was found to be directly related to winter rains and the amount of time between winter and summer rains since poor winter rainfall led to insufficient forage to support lactation. The highest fawn mortality rates occurred during or following drought periods when forage was in poor condition (Bright and Hervert 2005). Resulting management suggestions for reducing mortality included providing water sources and increasing forage in areas with low/less predator density. This strategy could be particularly useful in early spring and summer to reduce fawn mortality. Limited, localized

coyote control could be a short term option for promoting fawn survival, but would likely only be effective during years when there is adequate nutritious forage available. Lastly, disease could also be a significant mortality factor, but its effects have been largely uninvestigated (Bright and Hervert 2005).

Another study of pronghorn populations in Arizona asserted that pronghorn abundance, distribution, recruitment, and persistence are believed to be influenced strongly by fragmentation, loss, and quality of habitat in Arizona (McKinney, et al. 2008). Chronic low survival of fawns has been a major management problem in Arizona where fawn survival is believed to be a key element in pronghorn population viability. However, mortality causes are poorly understood; thus this study was conducted to determine whether the recruitment of pronghorn through late summer was associated with landscape scale winter precipitation in Arizona (McKinney, et al. 2008). The authors drew three main conclusions from the results of their study. First of all, winter rainfall preceding fawning season was positively correlated with fawn:female (ratio of fawns:100 females) ratios through late summer while summer precipitation effects appeared negligible. Secondly, the mean recruitment of 60% of the study populations fell below the statewide mean recruitment of 27 fawns:100 females for the study period. It has been reported that populations may be expected to increase in size given recruitment levels at or greater than 40 fawns:100 females, but population size or trend changes depend on differences between recruitment and mortality, not simply level or recruitment (McKinney, et al. 2008). Lastly, more than 75% of pronghorn habitat in Arizona is rated as poor or unsuitable. Drought conditions in these areas affect primary production which may lead to adult mortality and in turn, declines in population numbers. Drought frequency, however, did not appear to explain the difference in recruitment among populations. In summary, the authors' hypothesis that winter precipitation was a factor influencing pronghorn recruitment through late summer in Arizona and insufficient winter precipitation was a potential factor for chronically lower recruitment was supported by

their findings. However, winter precipitation was not the only factor that could explain high annual recruitment variability among and within populations, thus other factors need further research (McKinney, et al. 2008).

A study of the influence of precipitation on pronghorn demography in Texas was done by Simpson et al. (2007). The authors asserted that pronghorn populations in the southwestern US appear to be regulated more by density-independent factors than density dependent factors. There had been an overall 70% decline in pronghorn in the Trans-Pecos region of Texas over the decade prior to the study period. Thus the goal of this study was to evaluate the relationship between precipitation and pronghorn demography in Trans-Pecos, Texas. Within the study area pronghorn habitat was characterized by open, low-rolling grasslands or shrub steppes and much of the rangelands were used for agricultural purposes (Simpson, et al. 2007). Results of the study indicated that, based on the relationship between pronghorn abundance and precipitation measures, the Trans-Pecos pronghorn population appeared to be closely related to long-term moisture conditions and that the high variation in precipitation characteristic of the area indicated that pronghorn demographics were more susceptible to drought conditions than other pronghorn populations in the US (Simpson, et al. 2007). These results suggested that long-term and short-term drought affect pronghorn populations and fawn production and precipitation levels may be more of a limiting factor in more extreme reaches of pronghorn ranges than in more moderate ranges. Overall, the authors concluded that precipitation influences pronghorn habitat quality which in turn influences pronghorn production and abundance in the Trans-Pecos (Simpson, et al. 2007).

Drought may also have mortality effects on adult pronghorn, as the next study examined. In an arid region of southwestern New Mexico, fawn:doe ratios have been correlated with the amount of precipitation from the previous winter (Brown, et al. 2006). The authors hypothesized that midsummer drought measurements could be used to predict doe pronghorn mortality and index

population trends as indicated by annual survey data. No predator control efforts were used during the period of study. In the southwestern U.S., midsummer drought impacts on the reduction of doe numbers might be more significant in determining pronghorn population trends than winter precipitation effects on fawn recruitment (Brown, et al. 2006). The scarcity of nutritious forage was seen to affect pronghorn numbers observed in the study areas. Based on this assessment the authors hypothesized that an increase in stress and thus an increase in mortality resulted from the reduced availability of winter-spring forbs during a spring-summer drought. Doe mortality increased and competition intensified with inadequate forage availability which can cause pronghorn numbers to become increasingly density-dependent (Brown, et al. 2006). Translocation of 173 does during part of the study period greatly increased numbers for the relatively small population, however releases were preceded by severe drought and numbers of does observed on subsequent surveys remained at relatively low levels through the drought of 2000. This led to the conclusion that the release of additional animals failed to significantly bolster the population. Based on results of the study, the authors concluded that it is important to manage doe numbers to maintain population size and to utilize different strategies in years of normal precipitation than during times of drought, when less nutritious forage is available (Brown et al. 2006). Thus, in years of insufficient winter-spring rains and severe summer drought seems likely, increasing carrying capacity by reducing interspecific competition from livestock and big game, and avoiding releases of additional animals would be advantageous. During dry/drought periods, maintaining pronghorn populations at or below carrying capacity would help minimize intraspecific competition and enhance doe survival. And lastly, attempts to increase pronghorn survival, such as predator control, during drought years would likely be counter-productive as there would be insufficient forage available to support additional animals regardless (Brown, et al. 2006).

Severe climate conditions can have particularly drastic effects on small pronghorn populations. As Dunn and Byers (2008) observed, the long-term viability of small, isolated populations is a major concern in conservation biology. The impact of stochasticity on these populations is of particular interest as stochastic variation in environmental conditions can lead to population bottlenecks and extinctions. Much is known about the mechanisms that contribute to survival and fecundity under normal environmental conditions, but little is known about their effects during a bottleneck due to a lack of long-term data sets that encompass times before and after a bottleneck (Dunn and Byers 2008). On the National Bison Range (NBR) pronghorn populations have been studied extensively, so a severe drought in the summer of 2003 that resulted in high overwinter mortality and widespread reproductive failure that led to a bottleneck allowed for testing of mechanisms leading to reduced survival and fecundity during a stochastic environmental event. The study found that mortality was distributed across all age classes for females, but in males it occurred primarily in the youngest and oldest age classes (Dunn and Byers 2008). Also, in considering mate-choice strategy, female mate sampling energy costs, in some circumstances, were large enough to create a fitness cost. Findings also suggested that female pronghorn can demonstrate climate-induced reproductive failure. In measures of maternal energy-expenditure, pronghorn are on the high end of this spectrum, thus the energy cost was evident in both the analysis of overwinter survival of females and the analysis of fecundity. The fact that none of the females that weaned fawns in 2003 gave birth in 2004 further confirmed the significance of this (Dunn and Byers 2008). Based on this study, some considerations for managing small populations include, 1) addressing the significance of fences and other barriers that prevent the natural migration of pronghorn populations, 2) the importance of striving to maximize genetic variation and minimize inbreeding in populations; particularly in small, isolated populations, and 3) factors such as age, sex, genetic variation, and prior energy expenditure may further influence survival and reproduction under extreme environmental conditions, so managers need to be aware

that stochastic environmental events, such as droughts or severe winters, could significantly affect or eliminate a population that may otherwise seem stable (Dunn and Byers 2008).

### *Forage Effects*

A look at forage effects on maternal investment and reproductive success was the topic for the study done by Barnowe-Meyer et al. (2011). Reproductive aspects in many large herbivore species are determined by spring and summer-autumn nutrition and maternal investment in pronghorn has been determined to be the highest among ungulates. In Yellowstone National Park (YNP), some pronghorn populations are partially migratory with some individuals remaining year-round on winter range while others move to higher elevations during the summer. This study found that fawn birth weights and survival to August for the YNP pronghorn was normal relative to other populations. Causal links connecting habitat conditions, maternal investment, or perinatal condition and eventual survival to weaning were neither supported nor disproved by the data (Barnowe-Meyer et al. 2011). Thus future work should explore seasonal diet quality, habitat structure, and fawn condition at death within migrant and non-migrant population segments. Overall, the pronghorn population of YNP appeared relatively healthy. At birth fawn mass was higher for migrants than non-migrants indicating that spring/summer nutrition may have been better for migrants. Winter nutrition did not appear to have a great effect on reproductive investment. However, improving winter nutrition could promote higher baseline condition for females entering late gestation which could translate into improved perinatal condition of non-migrant fawns (Barnowe-Meyer et al. 2011).

Another vegetation study looked at the implications of cattle exclusion for pronghorn as they relate to vegetation cover and forb responses. Due to the implication that cattle grazing often reduces vegetative cover and forage abundance for wildlife and contributes to declines in northern Arizona pronghorn populations, this study was aimed at determining effects of cattle

grazing on pronghorn population viability (Loeser, et al. 2005). The study found that vegetative cover in areas of cattle exclusion increased significantly at target (i.e., fawn sized target representations) distances of 5m, but showed no significant difference at distances of 10 m or 25 m. It is difficult to define the critical components necessary for adequate pronghorn fawn cover, however these results make it doubtful that cattle exclusion as a management strategy would produce meaningful short-term benefits in hiding cover for fawns on Anderson Mesa (study location). Cattle exclusion also did not appear to increase forb richness and cover. Overall, this study showed that cattle removal was unlikely to yield significant, short-term improvements to pronghorn habitat parameters (Loeser et al. 2005). It is possible that the grasslands required more recovery time than the 5 years allowed for the study period, such that longer cattle removal may lead to long-term improvement, but this is unknown. For a pronghorn population that needs immediate management strategies, cattle removal alone is unlikely to have a strong influence on increased recruitment. In fact, appropriate cattle grazing management may actually benefit pronghorn habitat, although additional study is needed to confirm this (Loeser, et al. 2005).

### ***Tracking and Monitoring***

Primary tracking and monitoring techniques for large ungulate species such as pronghorn include aerial surveys and the use of GPS (global positioning system) telemetry collars. Depending on desired research outcomes, time, and funding factors, one or both methods may be implemented for use in determining population size and habitat use. Aerial surveys are often used to overcome the obstacles associated with counting animals from the ground and can be effective for estimating a population size over large areas (Ransom 2012). They can also be used to attain herd composition counts and density estimates (Firchow, et al. 1990). GPS collar use can be advantageous for providing more accurate position locations and tracking of animals and their use has increased as they have become smaller, more lightweight, and more accurate (Hansen and Riggs 2008). The primary interest in these methods in the context of this paper to identify survey

methods that are efficient and minimize data collection bias for the purpose of obtaining data for GIS habitat suitability model building.

### *Aerial Survey Techniques*

Aerial surveys are often considered to be a useful census technique, particularly in areas where certain species are limited in number and distribution (Allen and Samuelson 1987). A number of studies have been conducted to determine the best survey methods and identify sources of bias. One such study sought to compare aerial strip and aerial quadrat survey techniques to determine the most appropriate method for monitoring population trends and obtaining age and sex ratios (Firchow, et al. 1990). To do this, strip surveys were established as east to west transects located 1.6km apart over the entire study area (100% coverage) and quadrats were selected at random with two constraints: none would be adjacent to each other, and none would touch corners. These constraints reduced the chance of error associated with double counting animals or flushing animals away from adjacent quadrats. Population estimates were then based on the number counted (strip surveys) and extrapolation (quadrats) (Firchow, et al. 1990). The study found that the 1.6km strip average observability estimate was only 50%, indicating that an adjustment may be needed for observer efficiency or strip width reduction to accurately estimate pronghorn density in shortgrass prairie characteristic of the study area. To determine observer efficiency for other habitat types, it may be necessary to test the relationship between visibility and strip width. Additionally, a disparity arose between the two population estimates from quadrat surveys (the latter averaged only 56% of the former). Possible causes for this included: marked individuals were counted more than once, quadrat selection was not truly random, color-marked animals were not randomly distributed in the populations, and the incorrect assumption that collared animals behaved similarly to radio-marked animals in regard to mortality, emigration, and collar retention (Firchow, et al. 1990). Ultimately, the aerial survey mark-reobservation data indicated the observation of only 55-65% of animals within a 1.6km strip in shortgrass prairie, which may

lead to an underestimation of the population. On the other hand quadrat observability was presumed to be high, although standard errors were also high. Both types of surveys were conducted on similar dates, had similar age ratios in all comparisons, and similar sex ratio data in 3 of 5 comparisons. In terms of practicality, strip surveys are relatively simple and efficient to conduct whereas quadrat surveys are more difficult to navigate, inefficient in terms of actual flight time per area sampled, and the initial time required for locating and marking corners is high. However, more study is needed to determine observability of pronghorn during strip surveys in different habitats to evaluate usefulness of this technique for management purposes (Firchow, et al. 1990).

Another study looked at precision and bias of total pronghorn population estimates calculated from four levels of increasing numbers of transects selected systematically and randomly, and then compared those to the total pronghorn population within two management units (Allen and Samuelson 1987). The study area was located in northwestern North Dakota where a count was taken via complete aerial coverage in two management units. Censuses were conducted by flying transects 0.8km apart with poor visibility areas searched more intensely. Observations of pronghorn by transect were then converted to pronghorn/km<sup>2</sup> by transect to account for variable transect length. Results of this study led to several implications. First of all, sampling in high population densities provides more reliable total population estimates than low densities, so more effort needs to be made in areas with low densities and in fact, a complete census may be necessary. To achieve reliable population estimates, larger sampling areas may be needed (Allen and Samuelson 1987). This may be especially true for low densities where an annual census of units less than 1280km<sup>2</sup> may be somewhat unreliable. It still may be possible, however, with wide confidence intervals to establish reasonably sound management practices; for example, one may use lower confidence figures to establish harvest rates if the desire is to increase numbers of animals. When considering seasonal census taking, a winter pronghorn census is less desirable

for two reasons: 1) distribution of pronghorn during the winter becomes more clumped which would cause larger coefficients of variation and less precise population estimates with more bias, and 2) inability to include doe:fawn ratios (which can provide estimates of reproductive performance and fawn survival) (Allen and Samuelson 1987). The advantages of a summer census include acquiring doe:fawn ratios (so an estimate of reproductive performance and fawn survival are both obtained) and ability to obtain population estimates prior to the annual harvest and winter mortality rather than after. Overall, the authors concluded that refined census procedures are needed and are especially important in areas of marginal pronghorn range that typically have more intensive land use and limited pronghorn numbers (Allen and Samuelson 1987).

Ransom (2012) takes another look at bias when designing aerial surveys. Observation bias may arise from internal factors, external factors, and environmental factors. These potential influences are often ignored when estimating animal abundance, but the numerous biases may often be addressed by using aerial survey methods that utilize statistical sampling theory to make corrections to the actual number of animals numbers observed. The five main categories of sampling techniques include double-sampling, mark-resight, mark-recapture, sightability bias correction methods, and distance sampling (Ransom 2012). A census method is often the most commonly used and assumes all animals are seen, but can lead up to  $\frac{1}{3}$  of large mammals being unaccounted for (in aerial surveys). Therefore, better estimates may be obtained when sources of bias and the magnitude of effects influencing aerial detection of wildlife are understood that lead to better project planning and more informed statistical analysis. In this study that involved aerial surveys of feral horses, internal, external, and environmental factors influenced detection probability of feral horses in surveys across the western US. In both fixed wing and helicopter surveys, group size and sun effect were the most important influences on the visibility of horse groups, but observer variation also produced an observer effect that influenced detection

probability (Ransom 2012). In gregarious animals such as feral horses, large groups are easier to detect than small groups which results in negatively skewed population estimates, so the author found a strong relationship between detection probability and group size, for both types of aircraft. This effect of group size cannot be moderated simply by altering survey design, but it can be accounted for with statistical corrections. Overall, census aerial surveys can lead to inaccurate and negatively biased estimates of abundance when heterogeneous observation conditions influence detection of animals on the landscape (Ransom 2010). Factors such as sun effect, percent snow cover, and observer fatigue can be minimized through careful planning. Factors such as group size and topography cannot be completely minimized through planning, but increasing sample size (i.e., flying more transects) and using aerial survey techniques that apply statistical sampling techniques are critical for accounting for negative biases created by these factors. Use of these techniques may produce better population estimates for use in decision making and management (Ransom 2012).

#### *Using GPS Collars*

Another option for tracking wildlife and developing an understanding of habitat use and home ranges is the use of GPS collars. Beyond the advantage of GPS collars over aerial surveys for obtaining more reliable determinations of habitat use, concerns regarding collar accuracy and observation rates under various habitat conditions have arisen with the increased use of GPS collars to document habitat relationships (Hansen and Riggs 2008). To ensure that data for all study animals is consistent, researchers should use GPS collars that are the same model and are installed with the same version of the manufacturer's internal location calculation firmware. Firmware adjustments need to be considered depending on the type of data desired, as these adjustments can allow for greater accuracy or longer battery life. Firmware information should be provided by the GPS manufacturer so that informed decisions may be made regarding trade-offs between location precision and quantity of data (Hansen and Riggs 2008). Initial tests at

fixed reference locations are necessary to establish that precision and observation rates are similar before placing the units in the field. Researchers must also be aware of the effects of forest cover and topography on the variation in precision in these locations. Lastly, locational precision and observation-rate measurements for the GPS collar models used in topography and/or cover conditions should be included in any manuscripts presented for publication (Hansen and Riggs 2008).

When considering the use of GPS telemetry collars, it is also important to consider the direct effects of the devices on wildlife. Advances in technology have made it possible to monitor animal behavior at unprecedented levels with relatively easy-to-use technology and software applications (McMahon et al. 2011). As a result, remote monitoring of wildlife has become a popular research area of concentrated investigation. However, there is limited research on the effects these devices have on the animals themselves, which is a cause for concern. Although data may be limited on the effects of telemetry devices on wildlife, it is important to consider potential negative effects. These may be split into four categories: 1) capture effects, 2) the type of device, including shape, size, and coloration, 3) attachment method, and 4) device attachment timing and duration (McMahon, et al. 2011). While each of these brings with it unique animal welfare concerns, each may impact survival and reproduction. Thus, it is vitally important to take each of these into account when designing studies involving the use of such devices, and in particular, the importance of capture effects should be addressed. McMahon et al. (2011) suggests some basic measures that should be taken prior using telemetry devices for a research project. First of all, it is important to establish methods that are the least invasive and minimize the effects of the capture process. Secondly, researchers should determine how to minimize impacts to animal welfare and identify the type of attachment best suited to the animal. Third, identify the best attachment methods and how to minimize the effects of these methods. Fourth, a minimum time duration should be established for attaching the device to the animal and explore

alternative methods (McMahon, et al. 2011). An additional consideration for fitting pronghorn with GPS collars is the relatively high rate of capture myopathy that can occur with pronghorn. Pronghorn are particularly sensitive to capture myopathy and even with well-planned capture procedures, mortality rates can reach 30% ( O’Gara and Yoakum 2004). This factor should be a crucial consideration in determining whether or not to use GPS collars on pronghorn and may, in fact make it a less appealing option for collecting habitat use data in areas, like the Carrizo, where population numbers are limited.

### ***Habitat Suitability Modeling***

Habitat suitability models are tools used by biologists and wildlife managers for evaluation wildlife habitat quality, to predict wildlife responses to habitat changes, and as a basis for wildlife management decisions (Cook and Irwin 1985). Their validity and subsequent use has been criticized by some and field-testing habitat suitability models is necessary for determining if they indeed incorporate appropriate habitat variables (Irwin and Cook 1985). Two habitat suitability models for pronghorn were previously constructed for the Carrizo Plain National Monument and will be described here, followed by another example of habitat suitability modeling for pronghorn, as well as a discussion of design, validation, and evaluation of habitat suitability models and their application in management decisions for wildlife.

### ***Examples of Pronghorn Habitat Suitability Models***

The first CPNM habitat suitability model was developed by Longshore and Lowrey in 2008 to investigate potential causes of pronghorn population decline on the Carrizo Plain National Monument (CPNM) as they are related to habitat suitability. While a number of studies have been conducted on pronghorn habitat for perennial grassland and desert shrubland communities, this is a unique study of habitat and diet of pronghorn on the CPNM (Longshore and Lowrey 2008). According to the available literature, pronghorn exhibit fairly specific habitat preferences,

the following of which formed the basis for the habitat suitability model development. Slope of 30% or less and vegetation structure averaging 15-24 in (38-61 cm) in height and tall enough to conceal fawns during fawning season is necessary for avoiding predators. A higher amount of herbaceous and grass cover and less shrub cover is also important for predator avoidance and optimal foraging availability. (Longshore and Lowrey 2008) Forbs are the preferred forage so a diversity of forbs is also important, although shrubs are more important for forage than grasses when forbs are not available. Lastly, water availability is critical, particularly during dry months and pronghorn are generally found within 8.0 km of water sources (Longshore and Lowrey 2008). Thus the variables used for evaluating the CPNM habitat suitability model included: area (sq. km), slope %, herb cover %, grass cover %, shrub cover %, bare ground cover %, vegetation height (cm), distance to water (m), species diversity (herbs/forbs), species diversity (grass), and species diversity (shrubs). Using GIS and the selected criteria, the quantity and quality of pronghorn habitat was evaluated in 46 pastures encompassing over 490 km<sup>2</sup> of relatively low elevation areas. Longshore and Lowrey's (2008) habitat evaluation indicated that overall pronghorn habitat suitability on the CPNM ranked moderate to low. High quality habitat was indicated by proximity to water and low slope values, however adequate shrub cover and diversity ranked consistently low. This variable was deemed important on the CPNM due to pronghorn reliance on shrubs for forage during fall months when forb and grass availability is low. Also, dry years may lead to low carrying capacity as a result of the combination of low shrub diversity and cover with low herbaceous vegetative production (Longshore and Lowrey 2008). This may necessitate supplemental feeding in order support populations through periods of reduced carrying capacity. Animals located outside of the monument on private lands consisting of fallow agricultural areas may be an indication of these low carrying capacity conditions as they may previously have been part of the monument population. It is unknown whether or not these animals move back to the CPNM. Historically, the CPNM was considered pronghorn range, however due to overgrazing, farming, and the introduction of exotic species, the

composition of the region has changed dramatically. As a result of these changes and anthropogenic effects on the range, it may be that perennial shrubs are an important component of sustainable habitat on the CPNM, indicating a need for habitat rehabilitation in order to support a viable pronghorn population (Longshore and Lowrey 2008). Management recommendations based on these results included conducting long-term monitoring of CPNM pronghorn populations in conjunction with habitat and environmental conditions that would be useful considering the low pronghorn population numbers. Developing a knowledge of pronghorn movements to and from nearby agricultural areas would aid in improving populations size estimates as well (Longshore and Lowrey 2008). Recommendations for improving habitat quality included seeding with drought tolerant perennial forage species to increase vegetative cover and density and to aid in improving fawn survival, it may be beneficial to seed areas that meet the determined bed site criteria with relatively tall perennial grass species. It was also recommended that long-term monitoring of available biomass during summer, fall, and winter to provide information about forage availability under varying environmental conditions be conducted and further study of the effects of eating potentially toxic plants is warranted (Longshore and Lowrey 2008).

The second Carrizo study created habitat suitability models (HSMs) to predict habitat use by focal species (pronghorn, tule elk, and San Joaquin kit fox) on the CPNM because of proposals for solar energy production that may impact these species, with the purpose of developing mitigation options and strategies to reduce these impacts (Penrod, et al. 2010). The HSMs were developed using variables, valuations, and weighting selected in consultation with biologists who had expert knowledge of the species. These variables included vegetation (type and density), road density, slope, and terrain ruggedness (terrain ruggedness was not used for the pronghorn model). Core and patch areas of habitat were identified for each species where cores were defined as contiguous areas with higher suitability scores that could sustain at least 50 individuals

and patches were the same but contained less than 50 individuals with at least one breeding pair. Results of this study found that open, forb-rich communities on gentle terrain in the study area compose suitable habitat for pronghorn on the CPNM (Penrod, et al. 2010). Abundant medium to high suitable habitat was identified by the model (somewhat contradicting the conclusion of the previous study) on both sides of the Tumbler Range and open grasslands and alkali desert scrub habitats on the floor of the Carrizo Plain and San Joaquin Valley and the Antelope Plain offered the most extensive areas of highly suitable habitat. These findings corresponded well with the distribution of pronghorn sightings. Areas of medium to high habitat suitability also consisted of some agricultural lands. The model also found that core areas and habitat patches all occurred within dispersal distance, although some movement barriers may exist between suitable habitat areas (Penrod, et al. 2010). As for landscape permeability, model results indicated that the least-cost corridor between northern and southern Target Zones for the study is approximately 70 km long and from 5 to 20 km wide using the most permeable 3% portion of the landscape. This corridor is much broader in the southern half and more restricted by topography in the northern half. Additionally, the most permeable path corresponds to an area containing many fences; however, it is difficult to ascertain fence barrier effects due to the variation of fence types and number of fence breaks (Penrod, et al. 2010). Overall, the model found highly suitable pronghorn habitat throughout the majority of the CPNM, as well as in the Cholame Valley and portions of the San Joaquin Valley. Each area was delineated as a core area, with all but the San Joaquin Valley currently occupied by pronghorn. (Penrod, et al. 2010) The study Target Zones and least-cost corridor overlap the three known pronghorn subherds, suggesting potential connectivity between these groups. Connectivity between these subherds and pronghorn in the San Joaquin Valley would require further study, should future re-colonization of pronghorn occur there (Penrod, et al. 2010).

Another study involved modeling pronghorn habitat suitability for an endangered pronghorn population in the Sonoran desert of Arizona (O'Brien, et al. 2005). The authors' objectives were to create landscape-level predictive models of pronghorn habitat use specific to the Sonoran Desertscrub biome in Arizona and then use those models to evaluate potential pronghorn habitat in southwestern Arizona. To do this, the authors modeled existing locations of Sonoran pronghorn and the southwest corner of Arizona was evaluated as potential habitat. Due to the size of the study area, two modeling approaches were chosen that included used and unused locations by Sonoran pronghorn (O'Brien, et al. 2005). A two-step modeling approach was used that consisted of model creation and model application. Five explanatory variables were selected for use in the model: aspect, biome, distance to washes, slope, and soil association. Aspect was used because of rainshadow effects and frost deposition that could affect vegetation growth. Biomes were selected because they offered the best available coverage of plant communities and they reflected the effects of dominant climatic variables. The selection of the distance to washes variable was due to the fact that Sonoran pronghorn tend to select areas close to washes because they use thermal cover and forage found there. Slope was used because there was a suspected differential use of slopes  $\leq 20\%$  by pronghorns, as demonstrated by observed bajada and mountain slope usage. And lastly, soil association was included because soil attributes affecting moisture retention and vegetation growth may have an influence on pronghorn habitat use (O'Brien, et al. 2005). Results of these models did not imply causal interpretation between explanatory variables and habitat use by Sonoran pronghorn. The two methods of analysis (Classification and Regression Tree, or CART, and logistic regression) used in this study each had benefits associated with their use. The CART model was better at identifying used and unused points and it provided a simple dichotomous key for intuitive use. It was more conservative in that it identified less habitat than the regression model. The logistic regression model required fewer variables, it provided probability of pronghorn occupation, and it tended to include a larger area of current range as habitat. Suggestions for model improvement based on this study included

creating focused models of particular seasons, life history periods, or individual animal use level rather than a year-round model and by using finer-scale vegetation characteristics (O'Brien, et al. 2005). When used for guiding management actions, using landscape-level models for preliminary evaluation of potential translocation sites is recommended as an approach that allows for potential sites to be identified so that further evaluation using ground-based assessments may be conducted. However, it is important to recognize that these models classified habitat and probability of use, but they did not address habitat quality. Additional influences that may affect quality include: levels of disturbance and/or development, barriers (roads, canals, highways, etc.), water availability, and forage availability and quality. The primary purpose of these models was to provide an initial step in identifying translocation sites. Additional steps to be taken may include public input, review of predator presence and density, fencing, presence of preferred forage and water, support of pronghorn recovery by land managers, and limiting factors on the current range not present in the translocation area (O'Brien, et al. 2005).

#### *Evaluating Habitat Suitability Models*

There are a number of factors to consider in the creation of HSMs. An apparent but important consideration is the determination of appropriate variables for the model. Integral to Habitat Evaluation Procedures developed by the U.S. Fish and Wildlife Service, habitat suitability index (HSI) models typically developed from literature reviews have often been criticized as a method for determining suitability of habitats for species due to the generalist nature of some species and lack of information available for others (Irwin and Cook 1985). It is important, therefore, to field-test these models to determine if they incorporate appropriate habitat variables. An HSI model was developed to rate habitat quality of pronghorn winter ranges. Statistical evaluations of relationships between population and habitat variables on 29 winter ranges were reported in order to determine if the model incorporated appropriate habitat variables. It was assumed by the authors that long-term pronghorn densities and reproductive indices were limited by winter range

physical factors and habitat conditions, essentially following the concept of carrying capacity. The results of a direct field test by comparing HSI model outputs with population densities on the same study areas were included in the study as a source for analysis (Irwin and Cook 1985). To determine if the draft model incorporated important pronghorn habitat variables, simple correlation and multiple regression were used to identify relationships of habitat and non-habitat variables with pronghorn densities on winter ranges and with fawn:doe ratios for herds occupying the study sites. The draft HSI model assumed the importance of 5 variables on pronghorn winter range: shrub canopy cover, shrub height, topographic cover, shrub diversity, and the availability of winter wheat; this study supported the inclusion of the first three. The most important variable correlated with both pronghorn density and fawn:doe ratios was shrub cover, and this correlation was supported by the literature (Irwin and Cook 1985). Topographic diversity was also found to be important, although its effect is more dependent on region. It would appear that pronghorn use higher slopes in areas with harsher winters due to forage availability, so this factor may not be as significant in areas with less harsh winter weather. Shrub height was considered important because it appeared to contribute to greater fawn production. The significance of the shrub diversity and use of winter wheat variables could not be determined, but due to small sample size, the authors suggested retention of these variables in the model. It is important that users of the model adapt it to satisfy local, unique conditions; stringent testing is necessary to provide reliable habitat models that are useful for management or mitigation purposes. In summary, nearly 25 environmental variables were quantified that potentially reflected various aspects of: vegetation, climate, topography, livestock grazing, development activities, and pronghorn harvest intensities (Irwin and Cook 1985). Simple and multiple regressions were used to determine if the draft HSI model included variables that influenced pronghorn populations. Dependent variables for the model were pronghorn densities and fawn:doe ratios. Results of the analysis strongly supported the inclusion of 2 of the HSI model variables: shrub canopy cover and topographic diversity, and weakly supported the inclusion of shrub height. The HSI model was deemed useful for winter

pronghorn habitat management and mitigation due to the relative ease with which the HSI variables correlated with pronghorn population characteristics that may be managed or impacted (Irwin and Cook 1985).

Another consideration for habitat suitability models is their validation and modification. Some tests of HSI or similar models have produced negative or inconclusive results, suggesting a need for an objective, critical field testing of the draft pronghorn HSI model to evaluate its accuracy and use (Cook and Irwin 1985). As a continuation of the previous study, the authors present results from a direct field test of the pronghorn model by correlating model outputs with estimates of wintering pronghorn densities that are assumed to reflect habitat quality. The validity of the pronghorn HSI model was supported by both indirect and direct model assessment, although the evaluations were largely dependent on the assumption that winter pronghorn densities reflect habitat quality which was supported by the indication that pronghorn segregate by hierarchy during winter and data collected for this study was taken over several years during the winter. Also, winter habitats are critical to pronghorn on northern ranges (Cook and Irwin 1985). The most important variables in this model were determined to be shrub cover and topographic diversity. However, it was not recommended that the other variables be eliminated because their apparent lack of significance may have been a result of sampling limitations. Inclusion of these variables rather than a simplified version of the model including only shrub cover and topographic diversity will also help ensure the use of the model in a greater variety of conditions. It is important to recognize that the findings of this model, that it produces ratings of habitat quality that correlate well with population indicators of habitat quality over a wide region, contradict several previous tests of HSI or similar models (Cook and Irwin 1985). However, many of those tests appear unable to withstand critical scrutiny due to small sample sizes and/or incomplete statistical analysis. These authors concluded that their model combined limiting factors more realistically and accurately and could better assess habitat quality when combined

with knowledge of local conditions. The authors further stated that their model could be of particular use in assessing development impacts that alter substantial pronghorn habitat areas but it would be less useful for evaluating developments that primarily impact populations through direct disturbance such as harassment, poaching, etc. (Cook and Irwin 1985). Suitable applications of their HSI model appear to include evaluating the effects of various habitat management options on habitat potential, and identifying winter ranges for pronghorn transplants, so long as snow depths are not prohibitive. Ultimately, the model seemed most applicable in areas where severe winters often influence pronghorn population. The model may need to be modified to be applicable in other areas; for example, topographic diversity was found to be significant in this instance but may not necessarily be so in areas with less severe winters where pronghorn tend to use flat areas year-round (Cook and Irwin 1985). The authors also suggested some precautions for its use. First, they did not recommend using it to rate winter ranges based on snow accumulation. Second, they did not recommend using it to predict actual pronghorn numbers, as it was only designed to describe habitat potential. The authors did not have opportunity to test the model through severe winters, which occur infrequently, so this would not be an appropriate use without testing under those conditions. Overall, the authors felt their HSI model was a potentially valuable tool for evaluating pronghorn winter habitat quality with modifications as needed for specific areas (Cook and Irwin 1985).

An evaluation of the reliability of HSIs was performed by Roloff and Kernohan (1999). They assert that HSI models are often criticized for unreliable performance in the absence of validation frameworks. The purpose of HSI models is to quantify an organism's life requirements using structure, composition, and spatial components of its habitat. Prior validations of HSI models have produced inconsistent results, likely due to a variety of factors that include inadequate population sampling, sampling in a limited range of habitats, use of model equations that are not representative of actual wildlife-habitat relationships, misinterpretation of results, model

application to inappropriate spatial scales, and inadequate consideration of variability of data (Roloff and Kernohan 1999). This study used models that followed the US Fish and Wildlife Service's HSI "blue book" format and identified seven criteria that could account for model validation error. It then evaluated the performance of validation studies using criteria categorized as follows: 1) model components evaluated, 2) input data variability, 3) validity of comparative test(s) used, 4) scale, 5) range of HSIs, 6) population index, and 7) duration of population data collection. The authors found that the weakest components of all validation studies were inadequate consideration of input data variability and how variability affects final HSI output interpretation. No assessment of how input data variability influences HSI scores were done by the evaluated studies, so inferences of model performance were uncertain (Roloff and Kernohan 1999). Selecting the appropriate spatial scale is critical for proper model application due to the integration of multiple limiting factors into overall ratings of habitat quality. Model application of geographic scale should reflect the size of the species' home range, its degree of habitat specialization, habitat heterogeneity, and life history period under analysis. Validation of the habitat model at a spatial scale can be defined by either the available energy expenditure of the species or the minimum documented home range size. Theoretically, at optimum habitat conditions species will use their energetically defined spatial scale; as habitat quality decreases, larger scales are required. Since optimum habitat conditions are generally rare, models must be adjusted accordingly to represent the rarity of optimal scores. Thus, using allometric relationships is one approach for defining the appropriate energetic scale (Roloff and Kernohan 1999). Prior to HSI model implementation, sensitivity tests should be conducted to ensure model output is responsive to changes in input data. Additionally, land classification systems and associated attribute data should be of sufficient resolution to allow quantification of habitat quality differences. The use of a combination of relative abundance and fitness indicators to aid in understanding causal mechanism(s) behind an observed population response is recommended. Also, a combination of short duration studies, relative abundance indices, and species with

dramatic population fluctuations can cause poor model performance. (Roloff and Kernohan 1999) Statistical replicates were another area identified as a model weakness due primarily to typically insufficient sample sizes. To avoid this error, the following was recommended: clearly identify replicates, make sure that replicates are aligned with model parameterization (e.g., areas fall within the geographic range of model application, telemetered animals are not all the same age or sex), present variability displayed within (e.g., an area) or by (e.g., telemetered animal) each replicate, and ensure that the complete range of habitat conditions is sampled by selecting enough replicates. Habitat suitability modeling is an evolving process and the consideration of the seven criteria established by this study is encouraged for future validation studies. The usefulness of HSI models is often debated. However, if properly developed and tested, HSI models should satisfy the requirements of being repeatable, scientifically credible, and legally defensible (Roloff and Kernohan 1999).

The last suitability model consideration to be discussed here had to do with confidence intervals. According to Bender et al. (1996) it is best to view HSI models as hypotheses of species-habitat relationships based on the assertion that habitat suitability can be linked to habitat attributes by some functional relationship. A common problem associated with the validation and application of HSI models is failure to account for variability in model parameters (habitat variables). Model parameters are generally approximated from sample means based on field measurements of habitat attributes (Bender, et al. 1996). These values are often used as model inputs, but the variance associated with these types of mean values is often ignored; thus this variance represents sampling error and the natural heterogeneity of the replicate unit, which are both important considerations in habitat assessments. Thus, many HSI scores do not include statistical estimates of dispersion, without which it is impossible to determine if HSI scores actually differ; if these differences cannot be demonstrated it is impossible to deduce model validation or application where different HSI scores are assumed to reflect difference in habitat quality. Bender et al.'s

(1996) study used two techniques to account for variance associated with habitat model inputs: Monte Carlo simulations and bootstrapping. Using a forest stand-based grey squirrel model that consisted of 2 sub-models, Monte Carlo and bootstrapping approaches were used to calculate approximate confidence intervals around the mean HSI scores for each sub-model.

Nonparametric bootstrapping was used because the frequency distribution of habitat variables was not always known, or the sample sizes were too small to infer distribution. Confidence intervals were determined at approximately 90% (Bender, et al. 1996). The bootstrap confidence intervals were always tighter than the Monte Carlo intervals which resulted in differing statistical interpretations. Both methods used exhibited overlapping confidence intervals and consequently a lack of statistical differences between mean HSI scores for the majority of the variables tested. The authors concluded that the assumption that different HSI scores represent actual habitat quality differences is implied in HSI or similar habitat model validation attempts, but these scores may in fact not represent actual differences (Bender, et al. 1996). It is important that variability in HSI scores and animal response indicators be accounted for in HSI model validation attempts; when this variability is ignored validation attempts will likely produce mixed results without making any determination as to whether the faulty relationships are due to the model or because of animal or habitat variation. Instead, validation attempts that associate statistically significant HSI groupings with statistically significant animal-indicator groupings may offer clearer insight into the ability of HSI models to assess habitat quality accurately. Using only the final HSI score for model verification may fail because the assumed differences in habitat quality (differing HSI scores) do not exist (Bender, et al. 1996). The techniques used in this study for determining variability associated with HSI scores were more versatile than other techniques for the following reasons: (1) all habitat variables do not have to be collected from the same plot, (2) statistical significance comparisons are possible and their power can be increased by simply increasing the number of iterations, and (3) Monte Carlo or bootstrapping simulations allow confidence interval calculations not only for regression format models, but traditional HEP (habitat evaluations

procedures) HSI models as well (Bender, et al. 1996). Either method can be used, but in this study Monte Carlo confidence intervals were consistently larger than bootstrap CI's due to small sample sizes. And while nonparametric bootstrapping is robust to small sample sizes and unknown data distributions, underestimation of the variability in habitat characteristics may result if only the range in data found in sampling is used. Thus, Monte Carlo or parametric bootstrapping may provide better assessment of overall variability in sampled habitat parameters and therefore HSI scores if the minimum sample size is adequate to confidently determine data distribution (Bender, et al. 1996).

### ***Kernel Density Estimation***

Many studies of wildlife resource selection compare used to available resources, however there is the potential for error in resource use determination based on the misclassification of habitat patch use by radio telemetry and mapping systems, which occurs more frequently in areas with rugged topography and patchy habitats (Rittenhouse, et al. 2008). Additionally, it is necessary to understand how an animal perceives its environment in order to adequately define available resources. Likewise, due to the large spatial scale definition of resource availability in most studies, some supposed resources may not actually be available to the animal. Animals also cannot be monitored constantly, so interpreting results of use-availability analysis is difficult and thus the perceived importance of certain resources may be incorrect. An alternative strategy to the approach of use versus availability is the consideration of resource use intensity within an animal's home range (Rittenhouse et al. 2008). An example of this is to use the percentage of total telemetry locations contained within grid cells to model resource selection using categorized relative intensity use such as low, medium, and high. Ordinal polytomous regression can then be used to connect patch resource characteristics to the different use-intensity levels. Resource use should be defined as a continuum of intensity throughout the entire animal use area. The animal's home range, defined as the extent of area with a distinct use-intensity during a specified

time period, is a logical scale for delineating areas potentially used by the animal (Rittenhouse, et al. 2008).

An increasingly popular recent method for estimating utilization distributions (UDs) and home ranges that summarizes habitat use along a continuum throughout the animal's range is the kernel density estimator (KDE) method (Fieberg, 2007a and Rittenhouse, et al. 2008). Advantages of UD approaches such as kernel-based estimators include 1) the reduction of misclassifications due to radiotelemetry or mapping errors because the UD is a function of all telemetry locations and it is not required that individual locations be placed in specific habitat patches, and 2) rather than a binary classification system of used or not used, the estimated probability density function can be used to compute a refined measure of use along a continuum throughout the animal's home range. With this method, the comparison of characteristics of resources used at different intensities is used to model resource selection (Rittenhouse, et al. 2008).

Numerous methods have been developed for estimating UD, but kernel approaches have been identified as being superior to previously used methods in most contexts (Fieberg 2007a).

Typically in wildlife space use studies, the adaptive kernel approach exhibits higher bias than fixed kernel approaches and thus, fixed kernel approaches are generally favored. It is generally recommended that an automatic bandwidth selection approach is used and most often the reference or normal method (REF) and least squares cross-validation (LSCV) have been used in ecological studies (Fieberg 2007a). The best recommendation of sample size is a minimum of 30 locations when estimating home range size for fixed kernel methods using LSCV. Performance improves for any bandwidth method as sample size increases, up to some threshold (Rittenhouse, et al. 2008). Also, location data should provide a representative sample of the relative amount of time spent in different parts of the landscape, regardless of the UD or home range size estimation method used, which provides another indication of the importance of taking a large sample using a random sampling design (Fieberg 2007a). This leads to another consideration for KDE

development that the construction of kernel density estimators of space use generally takes place under the assumption that data were obtained from a simple random sample, when in reality the data are often collected through either a systematic or a haphazard design. It may be difficult to implement a simple random design in these types of studies, thus a stratified design offers greater flexibility and can ensure that sampling takes place evenly over the time period of interest for measuring space use and that could be used to determine a self-weighting design (Fieberg 2007a). For designs that are not self-weighting, weighted kernel density estimators (WKDEs) can prove useful (Fieberg 2007b).

The primary drawback in using KDE methods is their sensitivity to the choice of smoothing parameters. This has led to a number of simulation studies conducted for the purpose of comparing objective, data-based methods for selecting optimal smoothing parameters in the framework of home range and UD estimation (Fieberg 2007a). However, the general role of smoothing in data analysis, which is primarily to increase precision at the cost of increased bias, is often overlooked in evaluations of optimal choice of smoothing parameters. KDEs that use smoothing methods improve precision for estimating home ranges and UDs by using information from “like” individuals or sample units (e.g., observations that are “near” each other). Smoothing methods are similar to pooling and reduce variability at the cost of increasing bias (Fieberg 2007a). All smoothing methods, including KDEs of home range and habitat use, contain this bias-variance trade-off and it is important to understand the effect of smoothing on bias and variance for the interpretation of research results. Additionally, it is important to recognize that unbiased results do not necessarily imply that individual estimates will be optimal or look like the true distribution (Fieberg 2007a).

Sample size effects on smoothing are relatively easy to understand. The variance of KDEs will decrease and bias will primarily influence the mean integrated square error (MISE) as sample size increases. Therefore it is better to smooth less for larger sample sizes in order to reduce bias

(Fieberg 2007a). Autocorrelation is another problem often associated with KDEs and other home range estimators. However, for studies of space use during a fixed study period, it can be argued that it is more important to focus on obtaining a representative sample of locations than autocorrelation (Fieberg 2007a).

Further research is needed to address spatial autocorrelation and build spatial models for ordered categorical variables. Spatial models have not yet been employed, even though they are a natural choice for resource selection studies (Rittenhouse et al. 2008). Additionally, certain assumptions must be met to appropriately model resource selection: 1) it is assumed that use is accurately represented by the UD where animals were not detected, 2) enough data must be available to compute an unbiased estimate of UD, 3) habitat patches are assumed to be biologically relevant to the species being studied, and 4) the intensity of use is assumed to signify the relative importance of habitat patches to the study animal (Rittenhouse et al. 2008). Overall however, KDEs, provided a representative sample of locations, are a reasonable method for obtaining useful estimates of home-range size and relative space use (Fieberg 2007b). The use of UD methods such as KDEs, instead of using binary approaches that define some patches as used and some as available, emphasize which patches receive more or less use and the spatial boundary of use is defined by the home-range boundary, eliminating the need to define locations that are assumed to be available (Rittenhouse, et al. 2008).

### ***Conclusions***

Understanding habitat requirements of pronghorn is a logical first step in establishing management strategies that promote population growth and recruitment. To do so, however, requires a basic understanding of pronghorn biology and ecology so that management practices may be focused on habitat components integral to pronghorn recruitment and survival. Habitat suitability modeling has become a recognized tool for making determinations of wildlife habitat

needs and identifying probability of habitat usage (Cook and Irwin 1985). By using tracking and monitoring techniques, it is possible to validate these models and begin to understand habitat characteristics that define quality habitat and range for pronghorn. In the absence of tracking and monitoring data, validation of the model becomes difficult. However, HSMs may still be useful in identifying a threshold of suitable habitat for pronghorn that can be used to establish initial management strategies.

## METHODS

### *Study Areas*

Figure 3<sup>1</sup> depicts a map of the study locations. The Carrizo Plain National Monument (CPNM) is located on the southwestern edge of the California Central Valley. It is managed by the Bureau of Land Management in partnership with the CDFW and encompasses a total of 253,628 acres with an average elevation of 2017 feet. The topography along the valley is relatively flat with rolling hills extending to the Caliente Range in the southwest and the Temblor Ranges in the northeast (Longshore and Lowrey 2008). The CPNM is a semi-arid environment with an average rainfall of 15cm, although this can be highly variable (Longshore and Lowrey 2008, Penrod, et al. 2010). Vegetation on the CPNM is a complex of barrens, grasslands, and scrublands dominated by spiny saltbush (*Atriplex spinifera*), iodine bush (*Allenrolfea occidentalis*), annual grasses such as brome (*Bromus* spp) and wild oats (*Avena* spp), and Juniper-oak cismontane woodland and cismontane juniper woodland and scrub at higher elevations (Longshore and Lowrey 2008). There are also some dryland grain crops, fallow grain fields, one-time areas of wheat cultivation, and some orchards and vineyards; the seasonally fallow agriculture fields are known to be frequented by both tule elk (*Cervus canadensis*) and pronghorn (Penrod, et al 2010). Much of the CPNM has also been divided into fenced pastures, although many fences have been removed (Longshore and Lowrey 2008).

The Tejon Ranch is the largest contiguous expanse of private land in the state of California and encompasses nearly 270,000 acres. It is located in the Tehachapi Mountains, between Bakersfield and Los Angeles (<http://tejonranch.com/about-us/>). Owned by the Tejon Ranch Company, in 2008 it executed an agreement with five environmental organizations that would conserve up to 240,000 acres of the Ranch. In addition, the Tejon Ranch Conservancy was

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<sup>1</sup> Figures 3-21 can be found in Appendix I

formed and charged with developing and implementing a Ranch-Wide Management Plan that would establish science-based conservation and management practices for preserving the natural heritage and biodiversity of the ranch (Applebaum, et al. 2010). Elevation ranges from 400 to 6,800 feet on this topographically and environmentally diverse ranch that is divided into four ecoregions: the San Joaquin Valley, the Tehachapi Range, the Mojave Desert, and the southern California Coastal Ranges. Pronghorn inhabit the Mojave Desert section of the Ranch which encompasses roughly 48,800 acres. Land-use in this section of the Ranch includes limited agriculture and cattle grazing. Vegetation includes Antelope Valley grassland, desert scrub, desert/wash/riparian seeps, developed land, foothill woodland, Joshua tree woodland, mixed oak, non-native grassland, and scrub oak chaparral. Climate and precipitation vary widely across the ranch (Applebaum, et al. 2010).

### ***Carrizo Plain National Monument***

#### ***Input Data***

The pronghorn data used for this project consisted of aerial survey data of observation locations collected by CDFW staff from 2000-2010 on the CPNM. Global positioning system (GPS) coordinates (geographic coordinates in WGS84) were recorded for every incidence of at least one sighted pronghorn with an estimated 500 m radius positional accuracy (Bob Stafford, pers. comm.). Surveys were conducted from single engine, fixed-wing aircraft. The number and month of surveys varied from year to year and month to month such that some months and/or years had considerably more datapoints than others, with each datapoint representing the number of pronghorn seen at that location (e.g., some points represent just one animal and others represent 20 animals). For each GPS location, counts of pronghorn were recorded, by sex and age when possible. The only exception to this was the data collected in 2000; no animal count totals were recorded so these locations were treated as one animal sighted for the purposes of the habitat use analysis. All survey years were treated as a single time unit to determine an overall

use pattern in the study area for the purpose of developing a landscape-level model of predicted pronghorn habitat use. O'Brien, et al. (2005) recommend that creating focused models of particular seasons, life history periods, or individual animal use may be more useful than a landscape-level model, however due to constraints of the available data, this was not possible. The inadequate distribution of observations across times further justified the use of a single time frame for analysis. This was accomplished by treating the 326 observation points as a single collection and when we set up the model variables, time was not included.

A spreadsheet containing the aerial observations of pronghorn and geographic coordinates (latitude and longitude, referenced to WGS 84) was added into ESRI® ArcGIS™ 10.1 software as a csv file, and the 'Display XY' command was used to plot the recorded pronghorn locations. In order to compute further density metrics, pronghorn points were projected into the coordinate system NAD 83 UTM zone 10 with units in meters. A rectangular project boundary extent was then generated to encompass the Carrizo Plain and surrounding terrain. Within this boundary a restricted modeling extent was generated using a 100% MCP of observation points, which was then extended by a 2km buffer (Figure 4).

#### *Development of GIS Habitat Layers*

A variety of habitat variables that the literature suggested may influence pronghorn habitat selection were initially considered for use in the model. These included vegetation (type, density, cover, etc.), road density, slope, distance to water, area, aspect, distance to washes, soil association, barriers, precipitation/climate, and livestock interactions (Longshore and Lowrey 2008, Penrod et al 2010, O'Brien et al 2005, and Irwin and Cook 1985). Some of these variables were eliminated because no data were available. For example, no GIS datasets with vegetative density or cover metrics are presently available for these areas, therefore the influence of those parameters could not be tested. Because our objective was to test the modeling approach over two sites, only parameters for which GIS layers were available at both sites could be used.

Initially the variables selected for use in the model included vegetation, slope, aspect, road density and distance to water. Distance to water was subsequently eliminated when it became clear that the available GIS water layer for the CPNM was not a reliable map of water availability across seasons. And aspect was later eliminated after it exhibited no effect in the statistical analysis. Ultimately the variables used in the model were vegetation type, slope, and road density. Selection of these variables was partially based on their use in the previous model constructed for the CPNM by Penrod, et al. 2010, due to the similar scope of evaluation area. Road density was also selected as a quantifiable measure of human activity and/or disturbance on the landscape. Heavily traveled roads can affect pronghorn mortality and act as a pronghorn movement barrier, seasonal or otherwise (O’Gara and Yoakum 2004).

The vegetation layer was limited by data availability. The 30m USGS GAP Land Cover layer (<http://gapanalysis.usgs.gov/data/>) was best suited for our modeling approach. Vegetation categories from this layer follow the USGS National Vegetation Classification System (NVCS) and for this study, vegetation classes were reclassified to the relevant cover types present on both the Carrizo and Tejon Ranch study sites: forest, shrub, grass, semi-desert scrub, crops and bare (including urban, open water, recently burned, etc.). A detailed breakdown of the vegetation reclassification can be seen in Appendix II. The Carrizo vegetation layer was then created from this (Figure 5). A layer with finer-scale vegetation characteristics is recommended for better model output (O’Brien, et al. 2005). However, this was not available.

Slope was treated as a continuous variable. Rather than use slope as a categorical variable with pre-determined breaks (e.g., limiting pronghorn use to  $\leq 30\%$  slope), I decided to allow the data to identify a cutoff point based on the pronghorn data analysis. The 10m resolution Digital Elevation Model from the USHS National Map (<http://viewer.nationalmap.gov/viewer/>) was used and the slope raster (percent slope) was derived from the 10m USGS DEM (Figure 6).

Road density was treated categorically. A pre-determined density cutoff was determined based on the elimination of single-lane road occurrences as non-use areas since the majority of roads within both locations were unpaved and low-traffic roads. Road features were downloaded from the US Census Tiger Roads database 2010 and clipped to the extent of the study area (<http://www.census.gov/cgi-bin/geo/shapefiles2010/layers.cgi>). This dataset contains major highways and minor roads and streets, but does not contain road surface material such as paved versus unpaved. Road types were not differentiated in the analysis.

To compute road density, a string of points was created at 30m intervals along all road segments. The 'Point Statistics' tool was used to create a 10m raster which summed the number of points along a 250 m circular moving window. A series of Raster Calculator steps was used to convert these values from 'Sum of Points' to 'Sum of Road length' to Road Length per unit, represented in Km/Km<sup>2</sup> (total km of road/total km area of road). To limit the inclusion of single roads, a density cutoff of 3km/km<sup>2</sup> was used (Figure 7).

#### *Reproduction of Previous Carrizo HSM*

Once I had compiled the layers my initial objective was to reproduce the HSM methods and results previously applied to the Carrizo Plain by Penrod et al. (2010). The methods described by Penrod were applied to the raster layers on both the Carrizo Plain and Tejon Ranch using the general procedures described in Appendix II. Suitability maps for the three input layers were then generated based on Penrod's parameters. Examples for Carrizo can be seen in Figure 8, Figure 9, and Figure 10.

The methods described by Penrod et al. (2010) did not include a specific description of their calculation of road density. Differences in the calculation method can influence the results of the HSM model, but due to the generalized nature of the method, large effects were not expected (for example the influence of roads in the weighted HSM model is 0.1).

Using the Weighted Geometric (Multiplicative) Mean equation where the variables were scaled to 1, I was able to practically duplicate Penrod's habitat suitability model which indicated a high level of pronghorn habitat suitability throughout the Carrizo Plain (Figure 11). Application of the model to Tejon Ranch yielded similar results of high habitat suitability throughout much of the Antelope Valley portion of the Ranch (Figure 12) and will be discussed later in more detail.

#### *Defining "Use" Areas*

I used kernel density estimation (KDE) to estimate utilization distributions and home ranges (Fieberg 2007). Positional uncertainty of point observations (due to moving pronghorn, moving aircraft, etc.) made kernel density estimation a desirable method due to its generalization of the use-area. Rather than a minimum convex polygon (MCP) that establishes habitat use boundaries to the exact point location, KDEs produce a generalized area of use taking into account a distance effect around each point (Fieberg 2007). The use of the KDE would also account for the variable number of animals at each point location, which is another limitation of MCPs that do not offer weighting of points. Geospatial Modeling Environment software version 0.7.2.1 and the KDE function was used to generate the KDE. The KDE does require more input parameters, such as the bandwidth,  $h$ , which is the distance effect around each point. In this study, Least Squares Cross-Validation (LSCV) was used as an automated method to choose  $h$  (which remains a popular method to try as a first attempt) (Rittenhouse, et al. 2008). It does this by attempting several different bandwidths while withholding some of the sample data and comparing against the full dataset to establish a "best fit line." The results of the automated methods were evaluated visually and LSCV produced the best fit. A weighted KDE was generated using the actual pronghorn points. The resulting map identified pronghorn habitat use trends and "hotspot" areas of use based on number of animals located in those areas. From this, I generated an 80% Isopleth habitat use area based on the pronghorn observations that generalized areas of use versus non-use and excluded individual or outlier point locations (Figure 13). Again, I tried a few

options until deciding on the 80% Isopleth. A 90% Isopleth appeared to be too large and included too many individual or outlier point locations that did not particularly define regular habitat use. It also included a fairly large amount of high slope areas as a result of the bandwidth effect, so that there was a possibility of losing the slope effect.

The 80% Isopleth boundary was used to identify “used” and “unused” habitat areas within the study site. For the model analysis, I randomly generated 1000 sample points so that those occurring within a “yes” or “used” area were identified as successes and those not occurring within a “yes” area (or “unused” area) were identified as failures (Figure 14). This analysis was based on the assumption that the aerial surveyors did indeed view all possible pronghorn location points in the study area, and areas without observation points were actual “no’s.” The GIS variables (slope, road density, and vegetation type) were then extracted at the location of each sample point from the underlying 30m-resolution rasters.

#### *Statistical Analysis*

Binary logistic regression was used to identify correlation between habitat variables and habitat use by pronghorn, based on its recurrent use for model building in the literature (O’Brien, et al. 2005, Irwin and Cook, 1985). Minitab version 16 software was used to conduct binary logistic regression using presence within the 80% isopleth as the binary (y/n) response, and the extracted GIS variables as the predictors for the map output of the variable extraction at the 1000 sample points. Vegetation was treated as a categorical variable with six categories, as was road density (with two categories) and slope was treated as quantitatively. The “bare” vegetation category was eliminated following the first round of statistical analysis due to non-performance in the model. A new set of 1000 sample points was then generated that excluded the “bare” vegetation category so that all 1000 points would be included in the analysis. Success or failure (or yes/no) in the analysis was based on the 80% Isopleth boundary. The variable coefficients from the statistical output were then used in the following equation:

Equation 1:

$$p = \frac{\exp(\beta_0 + \beta_1 x_1 + \beta_2 x_2 + \cdots \beta_k x_k)}{1 + \exp(\beta_0 + \beta_1 x_1 + \beta_2 x_2 + \cdots \beta_k x_k)}$$

Where:

$p$  = predicted probability of occurrence

$\beta_k$  = regression coefficient

$x_k$  = predictor variable value

Analysis of prediction accuracy of our model was done using the area under the curve (AUC) of the receiver-operating characteristic (ROC) plot, which is a frequently used threshold-independent metric for measuring species distribution model performance (Franklin, 2009). The AUC, or area under the curve, describes the overall ability of the model to distinguish between the two conditions being evaluated, species presence or absence. Our AUC value was 0.76220 (see Appendix II), which, according to Franklin (2009) falls in the moderate category (0.7-0.9), indicating that the model performance is moderate. The ROC table also generated a binary cutoff value of 0.2197 based on the input of the 1000 sample points used to build the model.

The resulting map of continuous use enabled visualization of the model results and evaluation of regression cut-off values. The binary regression predicts a binary response (yes or no) and a cutoff value is used to separate the continuous response into either of the predicted “yes” or “no” categories. In addition to the 1000 points used to generate the “continuous use” area, I generated an additional 500 validation points and used the Kappa statistic, a commonly used measure of agreement or accuracy where 1 indicates perfect agreement between an estimator and a reference and 0 indicates agreement purely by chance (meaning that no level of correspondence between the estimator and the reference could be detected), to validate the model and verify the cut-off value for probability of success (Congalton and Green 1999, Franklin 2009, Viera and Garrett 2005). Initial results from the Kappa equation suggested high Type I error in the model, indicating the need to adjust the cut-off to balance error. Using multiple probability of use cut-off

values, a graph of the resulting Kappa statistic was generated in order to find the optimal value of the statistic that balanced Type I and Type II error. A probability of use cut-off based on the Kappa statistic results and controlling for error was determined to be 0.30 (Figure 1). This was higher than the 0.2197 cut-off generated by the ROC table, but because it better minimized the Type I and Type II errors, I felt justified in using it. Using this cut-off value, the predictive map was finalized.

### ***Tejon Ranch***

Methods for completing the Tejon Ranch habitat suitability map were similar to the steps taken for developing the CPNM map, except no Tejon Ranch pronghorn observations were available. Therefore the methods differed in that aspect and use/non-use areas were based on those developed for the Carrizo Plain. The area for analysis was approximately delineated to include much of the Antelope Valley, which extends beyond the southern boundary of the Tejon Ranch, and north to just below the mountain ridgeline. A slope layer for Tejon Ranch was developed from the 10m USGS DEM and measured in percent slope (Figure 15). The USGS GAP Land Cover was again used for the vegetation layer and applied as had been established for the Carrizo Plain (Figure 16). The roads layer used was the same 2010 US Census Tiger Roads layer used in the CPNM model for construction of the road density variable in  $\text{km}/\text{km}^2$  (Figure 17), thus standardizing this variable to both study locations.

For actual application of the CPNM model, the statistical output from the CPNM analysis (based on the CPNM KDE and 80% isopleth) was used in the predictive equation (Equation 1) and run for the Tejon GIS map layers. Due to the categorical nature of the vegetation data, the equation was run individually for each vegetation category and then merged into one and applied to the map analysis area. From this, I generated a predictive map of pronghorn habitat suitability for the Tejon Ranch using the 0.30 cutoff value validated by the Kappa statistic for the CPNM analysis.

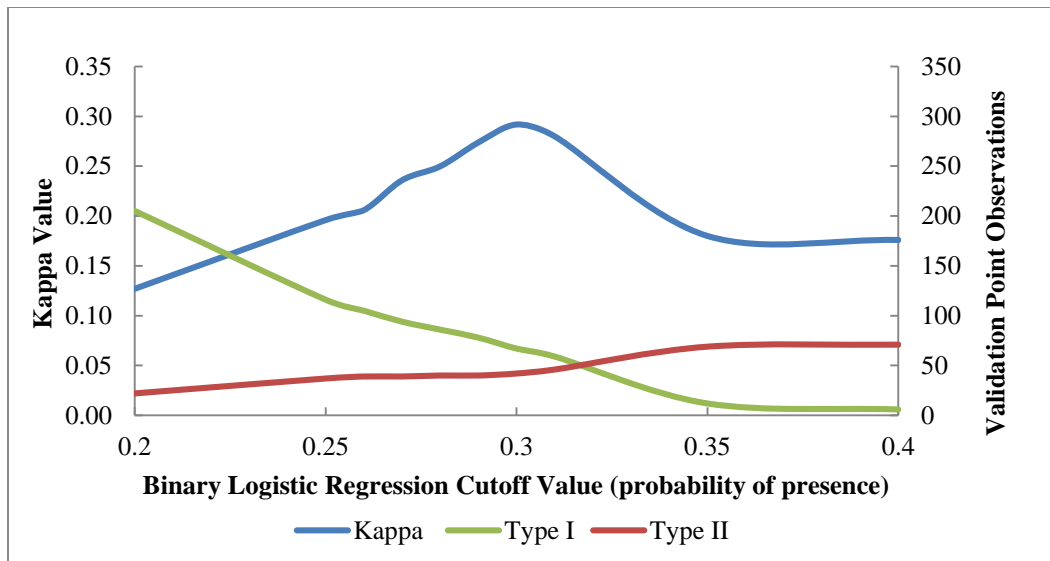


Figure 1: Evaluation of the regression prediction cutoff using Kappa and validation point observations while accounting for changes in Type I and Type II error.

## RESULTS

### *Carrizo Plain National Monument*

Results of the binary logistic regression analysis are summarized here. The analysis was significant with an overall p-value of  $< 0.0001$ . Significant effects of slope, road density, and the fallow cropland and grassland vegetation layers were detected using an  $\alpha$  of 0.05. Shrub and semi-desert did not differ from forested areas. Each variable effect was analyzed after the other variables were adjusted, e.g., road density effect was evaluated for constant slope and vegetation values. The Minitab output including the logistic regression table, goodness-of-fit tests, and measures of association can be seen in Appendix II.

I used the Hosmer-Lemeshow Goodness-of-Fit test because of its grouping based on values of the estimated probabilities to evaluate model fitness (Hosmer et al, 2013). The p-value (0.082) indicated that binary logistic regression was a reasonably valid approach for this model (I do not reject the null hypothesis of model fitness). The concordant pairs measure of association between outcomes and predictions rating was 76.0%, meaning that the model gave a higher predicted probability to where there was actual success in pairs with known and unknown sample points 76.0% of the time, or, every pair of (yes, no) was looked at and 76.0% of the time the yes outcome had a higher predicted probability of success.

Results of the regression analysis indicated overall significance ( $p < 0.0001$ ; testing that all slopes = 0). Slope is significant ( $p < 0.0001$ ) and the slope odds ratio of 0.95 means that with each 1 unit (degree) increase in slope, the odds of having a pronghorn present are multiplied by 0.95, assuming road density and vegetation type are held constant. The slope effect is not large, however, due to how far the odds ratio confidence interval is from one (0.93, 0.96) and the odds decrease with increasing slope.

Road density is also significant ( $p=0.041$ ). This odds ratio of 0.59 means that, with slope and vegetation held constant, the odds of having a pronghorn present are reduced by 59% for every one unit increase in road density ( $\text{km}/\text{km}^2$ ). The confidence interval (0.35, 0.98) shows an effect of higher magnitude than slope because it is not as close to one and the wider CI indicates less precision in the road effect.

The vegetation categorical comparisons were made after the other variables (slope and road density) had been adjusted for presence. Shrub, grassland, semi-desert, and crops categories were compared back to the forestland category. The probabilities of pronghorn occurring in either grasslands or croplands were different from forested areas ( $p=0.045$  and  $p<0.0001$ , respectively). The likelihood of having an animal in grasslands was 2.99 times higher than for forested areas (again, after adjusting for the other habitat variables). The odds of pronghorn being detected in a cropland was 12.32 times higher than for forested areas. The wide confidence interval with values far from one (3.50, 43.40) indicates that croplands have a large, but variable effect. Neither the shrub or semi-desert categories were different from forest areas ( $p=0.406$  and  $p=0.324$ , respectively).

When univariate logistic regression (see Appendix II) was run for each model variable to examine the explanatory power of each on its own, all vegetation categories were found to be significant predictors of presence, as was slope. Road density was not found to be a significant predictor of presence ( $p=0.423$ ), indicating that it is not a reliable descriptor of pronghorn presence on its own. Correlation among all data layers was 25.08% which indicates some relationship between variables. Pairwise correlation values (see Appendix II) were not additive, however. This suggests the parameters are relatively free of multicollinearity. Additionally, the binary logistic regression analysis ran each variable while adjusting for the other two, so that these correlation effects were controlled for by the multiple logistic regression process.

Each variable coefficient was included in Equation 1 and entered into GIS to generate a predictive map. The first map based on the logistic regression output displays the continuous probability of pronghorn presence, ranging from 0-0.7 (Figure 18). Cropland, as suggested by the statistical analysis, appears to have the highest probability of usage, with varying probabilities of usage throughout the rest of the valley and low usage in high road density and slope areas. For the binary logistic regression output, a probability of presence cutoff was needed. This was generated using the Kappa statistic equation. Initial results indicated a high level of Type I error, so to adjust for this and balance the error, the optimum Kappa statistic value was found and the corresponding cut-off value of 0.30 was implemented in the final predictive map (Figure 19).

### ***Tejon Ranch***

The input results of the prediction equation produced a continuous probability predictive map of pronghorn habitat suitability (Figure 20). To this map I applied the validated Kappa statistic probability of use cutoff of 0.30 to create the final predictive map depicting areas of suitable pronghorn habitat based on the model variables (Figure 21). The final predictive map identified 103 km<sup>2</sup> of suitable pronghorn habitat within the study area, 70.58 km<sup>2</sup> located within the Tejon Ranch boundary itself (within the delineated study area, not the entire ranch).

Table 1: Binary logistic regression output for model habitat variables. Vegetation category significance values are based on comparison to forestland. CI's are based on odds ratios of the variables.

Predictor	P-value	Odds Ratio	95% CI
Constant	0.002		
Veg: Shrub	0.406	1.61	(0.52, 4.98)
Veg: Grassland	0.045	2.99	(1.02, 8.73)
Veg: Semi-desert	0.324	1.85	(0.55, 6.23)
Veg: Crops	<0.0001	12.32	(3.50, 43.40)
Slope	<0.0001	0.95	(0.93, 0.96)
Road Density	0.041	0.59	(0.35, 0.98)

## DISCUSSION

### *Carrizo Plain National Monument*

Development of a habitat suitability model is a long and complex process. A common dilemma with all habitat suitability models is the selection of variables thought to influence habitat use, which are often limited by available data or vary between models because of geographical differences. For example, a habitat suitability model for Sonoran pronghorn included a distance to washes variable (due to provision of thermal coverage and forage availability) whereas a model developed for pronghorn living in cold winter climates included the availability of winter wheat as an important winter forage source (O'Brien, et al. 2005, Irwin and Cook 1985). Approaches to model development also vary and may or may not use existing location data of species. The two previous models created for the Carrizo Plain did not incorporate existing location data of pronghorn into their model development while the Sonoran model did (Longshore and Lowrey 2008, Penrod, et al. 2010, O'Brien, et al. 2005). Our modeling approach drew from a combination of methods found in the literature and essentially took the Penrod et al. 2010 model a step further by incorporating pronghorn location data into the development of the model to establish a data-based pattern of use areas that would enable model validation. I also refined the input variables and use parameters of those variables to make the process repeatable.

Each of the habitat variables included in the bivariate model were significant predictors of habitat use, but the mechanisms driving this are uncertain. Percent slope has long been recognized for its influence on pronghorn habitat use and is probably the best understood habitat variable of my three, although the literature is inconsistent regarding which slopes are preferred by pronghorn (O'Gara and Yoakum 2004). Based on probability of use, my data suggest that for the CPNM pronghorn population(s), slopes of 30% or less constitute suitable habitat. Literature sources such as Penrod, et al. 2010 suggest 20% slopes as the maximum slopes pronghorn will use, and this

may be an accurate assessment as quality habitat appears to vary depending on location, i.e., characteristics that define suitable pronghorn habitat in Montana vary significantly from those that define suitable pronghorn habitat in Arizona. Pronghorn habitat suitability cannot be generalized across these areas such that pronghorn may use higher slopes in certain locations that they would not use in others, for example, populations of Pronghorn in Yellowstone National Park have been documented as dispersing to higher elevations prior to fawning in what appears to be a predator avoidance strategy (Barnowe-Meyer et al. 2010). The effect of road density on pronghorn habitat use is less well understood and in the case of the Carrizo Plain was used as a quantifiable measure of human activity and/or disturbance on the landscape. Regardless, my results suggest that there is indeed a road density effect on pronghorn habitat use when combined with vegetation and slope, particularly in the high road density area that separates the southern portion of the Carrizo Plain from the northern portion. Whether this effect is due to human activity, disturbance on the landscape, presence of fences along these roads, some other mechanism, or a combination of these needs further study. Type of road (e.g., major highway, minor road or street, etc.) may also influence this effect, but this was not considered here.

The influence of vegetation on the use of habitats by pronghorn is well documented and has been modeled in several different ways, including by vegetative cover, vegetation height, vegetation type, vegetation density, and so on (O'Brien et al 2005, Irwin and Cook 1985). My analysis of the effect of vegetation on habitat use was fairly general due to a lack of fine-scale plant composition and distribution data. Thus, any conclusions drawn from this study relating to vegetation are quite broad – principally that pronghorn are more likely to be observed in cropland or grassland areas than in forests, shrublands, or semi-desert areas. It is important to note that current plant distribution maps fail to resolve vegetative differences across the Carrizo Plain, even though there is an obvious pattern of habitat use by pronghorn that seems to exclude certain portions of the Plain as illustrated in the KDE in Figure 13. A fine-scale study of the quality,

quantity, and composition of vegetation on the Carrizo Plain as it relates to pronghorn needs and preferences is recommended as a first step toward differentiating between the use and non-use vegetation areas indicated by the pronghorn datapoints used for this study. Overall, the model suggests that pronghorn prefer habitat composed of slopes  $\leq 30\%$ , road density approximately  $\leq 3$  km/km<sup>2</sup>, and a high preference for cropland areas. Grassland areas also represent suitable habitat components. More data is needed to differentiate use both within plant community types and between shrublands and semi-desert areas.

The application of the term “habitat *suitability* model” must be made with some reservation in this case. The literature contains fairly consistent recommendations of suitable pronghorn habitat elements. O’Gara and Yoakum (2004) summarize essential pronghorn habitat as short, mixed, and tall grasslands, shrubsteppe biomes, or deserts. Within these habitats, suitable characteristics include flat to rolling terrain (less than 10% slopes), rangelands that receive 20.3-38.1 cm of annual precipitation, year-round drinking water, snow depths that do not exceed 30 cm, and a living vegetation composition of 40% or more with a high diversity of forage species (forbs, grasses, and shrubs). Also vegetation height should average 38.1 cm and not exceed 76.2 cm (O’Gara and Yoakum 2004). In the case of the Carrizo pronghorn, and by extension the Tejon Ranch pronghorn, it is unlikely that all of these habitat suitability requirements are being met, thus the pronghorn populations may be *selecting* the available optimal habitat within these regions that may not actually be considered biologically *suitable* habitat for the pronghorn species in general. Due to the limited scope and data availability for this model, this distinction is not addressed within the model itself but should be considered when evaluating predictions of pronghorn habitat use on the Carrizo Plain and Tejon Ranch.

Furthermore, habitat characteristics and vegetation composition of the two study sites vary considerably. While the Carrizo Plain is characterized as a fairly flat, barren, grassland, and scrubland community the Tejon Ranch consists of more rolling hills and variable vegetation

ranging from grassland to desert communities (Longshore and Lowrey 2008, Applebaum, et al. 2010). Figure 2 gives a visual representation of the vegetation composition of both the Carrizo Plain and Tejon Ranch based on the vegetative categories used for my analysis (a more detailed breakdown of vegetative communities for both sites can be seen in Table 2 and 3 of Appendix II.) Some striking differences exist between the two study sites, particularly in the semi-desert and scrub categories. And while neither of these categories exhibited significant influence on pronghorn presence in my statistical model, it would be interesting to see if that would change were the analysis to be run with pronghorn location data collected from the Tejon Ranch study site. At first glance, both sites appear to contain proper vegetative communities for suitable pronghorn habitat based on O’Gara and Yoakum’s (2004) characterization, however the failures of pronghorn populations at both sites suggests otherwise. Additionally, access to all habitat areas may well be limited by terrain, fences, human influence and other factors so that some suitable habitat areas represented in this model may not in fact be *available* habitat areas. Ultimately, the true measure of habitat suitability for a species is fitness of that species – its ability to survive *and* reproduce. Some habitat variables within both the Carrizo Plain and the Tejon Ranch are limiting pronghorn fitness. It is still unclear exactly what these are, but with this habitat suitability model I have attempted to narrow down the possibilities to direct future research in a more useful manner so that appropriate management strategies may be implemented that will promote the survival and success of the Carrizo Plain and Tejon Ranch pronghorn populations.

### ***Tejon Ranch***

The predictive map (Figure 21) developed for the Tejon Ranch identified much of the suitable pronghorn habitat areas being located primarily in lower slope grassland regions, with some roads present. The semi-desert areas of the study site are depicted as non-suitable habitat areas, however this is likely a misrepresentation of actual pronghorn use on the Tejon Ranch since

pronghorn have been seen in the semi-desert regions. The extent of their use of these areas is unknown and warrants further investigation. However, this discrepancy illustrates the restrictive nature of a habitat suitability model and its application to sites beyond the one for which it was developed. Parameters established for the CPNM do not necessarily transfer directly or apply to the same degree in another location (i.e., Tejon Ranch), regardless of similarities between the two areas. Thus, for a habitat suitability model to be truly useful, it must be modified to the specific site being evaluated. The application of this model to the Tejon Ranch study area provides very limited information regarding suitable habitat for pronghorn and requires additional data and validation before any management decisions should be made based on these results.

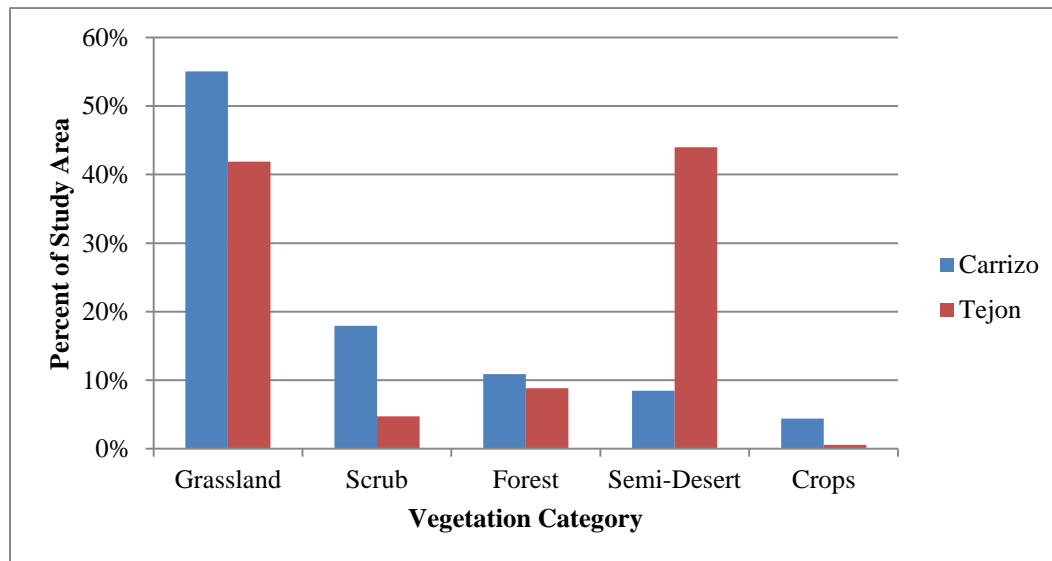


Figure 2: Comparison of vegetation composition between the Carrizo Plain and Tejon Ranch study areas based on the vegetative categories used in the HSM.

## RECOMMENDATIONS

### *Carrizo Plain National Monument*

Based on my results and what is known about pronghorn ecology and suitable habitat components, I make the following recommendations for furthering pronghorn habitat research on the CPNM that will work toward positive management solutions for the pronghorn population(s). As described previously, some habitat elements on the Carrizo Plain are apparently contributing to a lack of fitness in the pronghorn population. Determining exactly what these are is an enormous undertaking so the problem must be approached through a process of elimination. For example, based on my model results, an improved understanding of the vegetative communities (composition and quality) on the Carrizo Plain is of more importance than further investigation of road density effects (as a proxy for human activity and/or disturbance on the landscape). Vegetation surveys to evaluate species composition, forage quality and quantity, vegetation height and cover, etc. of the Carrizo Plain at varying seasons would greatly improve the data available for further evaluation of vegetative differences across the landscape and identification of actual vegetation use by pronghorn. In the last few years an ongoing survey of vegetation on the Carrizo Plain by the California Native Plant Society has been implemented that could provide some of these data in the near future ([http://www.cnps.org/cnps/vegetation/pdf/carrizo-vegetation\\_rpt2011.pdf](http://www.cnps.org/cnps/vegetation/pdf/carrizo-vegetation_rpt2011.pdf)). Tissue samples from species identified by CNPS from core areas might improve the accuracy of dietary estimations made from fecal sample microhistological analyses (Henley et al., 2001). Comparisons of pronghorn forage consumption to available vegetation types, quantity, and quality are essential for establishing management strategies that will promote population growth (e.g., supplemental feeding, seeding, etc.). In addition to evaluating vegetation communities for forage availability, it would be useful to consider the other habitat components described by O’Gara and Yoakum (2004) and determine what components the Carrizo Plain habitat is lacking. In addition to evaluating vegetation in terms of forage availability, vegetation

surveys should also be conducted to evaluate cover availability, particularly for fawning purposes.

Another important habitat component that was not evaluated in my model is the effects of precipitation and water availability. O’Gara and Yoakum (2004) indicated that annual precipitation should measure 20.3-38.1 cm in areas of pronghorn habitat suitability. Timing and amount of precipitation, combined with climate, is essential for production of certain forages preferred by pronghorn. However, The CPNM is characterized as a semi-arid environment with an average annual rainfall of just 15cm, although this can be highly variable (Longshore and Lowrey 2008, Penrod, et al. 2010). More significantly, summers on the Carrizo are arid with very little rainfall, suggesting a potential lack of available forage during critical life history periods (i.e., post-fawning season and breeding season). Tracking of annual precipitation levels along with pronghorn population numbers could provide some evidence of precipitation influence/effects on pronghorn success. Also, ensuring that water is available on a year-round basis every 1.6-3.2 km (O’Gara and Yoakum 2004) is a relatively simple step toward improving pronghorn habitat on the Carrizo Plain.

With these additional data, expansion of the current habitat suitability model with additional variables would help to further define pronghorn habitat use. The current literature recommends multiple variables for use in HSMs including (but not limited to) aspects of vegetation (type, density, diversity, height and cover, etc.), slope, precipitation, distance to water, aspect, barriers, and soils, but very little data is presently available for the CPNM (O’Brien, et al. 2005, Irwin and Cook, 1985, Longshore and Lowrey, 2008, & Penrod, et al. 2010). Additionally, refining the scope my current model by creating a focused model of a particular season, life history period, or individual animal use to identify more detailed habitat use patterns (O’Brien, et al. 2005) seems advisable due to annual and seasonal differences in pronghorn habitat use. To do this, it would be necessary to conduct aerial surveys in such a way as to focus the data collection according to the

model focus, e.g., for a focused model of fawning season habitat surveys should be conducted from April-June over several years. Observations for this study were collected across all seasons but not all months, and the number of observations across seasons varied considerably. Data were collected over a period of 10 years and yet some months were not represented at all, and others were disproportionately represented. Due to the haphazard nature of the current data, my analysis was limited to the creation of a general habitat suitability model that spans all seasons. Future surveys should be conducted the same number of times, with similar time intervals between them, and during the same months and time of day over multiple years for refining the scope of the model. I recommend that the number and frequency of surveys be determined based on resource availability. Surveys conducted over many years allow for variations in climate and precipitation to be included in the analysis. However, if surveys are limited to 2-3 years, for example, then it would be prudent to conduct as many as possible within the determined time frame. Lastly, placement of GPS collars on a number of animals would support the recommendations above by providing more detailed information of habitat use. Collars would also be a better tool (than aerial surveys) for defining populations and distribution extent and could be used in conjunction with aerial surveys to create an overall picture of population size, distribution, and patterns of habitat use. Tracking actual animal movements over time and seasons would further enhance the habitat suitability model analysis.

### ***Tejon Ranch***

In addition to the recommendations made for the Carrizo Plain, I have two priority recommendations for the Tejon Ranch pronghorn. First of all, I recommend collection of habitat use data for the pronghorn population, by whatever means possible (aerial surveys, telemetry collars, or possibly even game cameras). Without some sort of validation data, the usefulness of the model as a tool is limited. As was discussed earlier, the habitat types available on the Tejon Ranch are considerably different from the Carrizo Plain, particularly in the classification of the

semi-desert areas. An understanding of actual pronghorn use of these areas on the Ranch is essential for characterizing actual use versus non-use areas specific to the Ranch. These data, as well as other recommended data, can then be used to modify the model specifically for the Tejon Ranch so that a more useful predictive map of habitat use may be generated that will promote sound management strategies for improving pronghorn habitat on the Tejon Ranch and increasing the success of the pronghorn population(s) there.

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## APPENDIX I: GIS MAP IMAGES

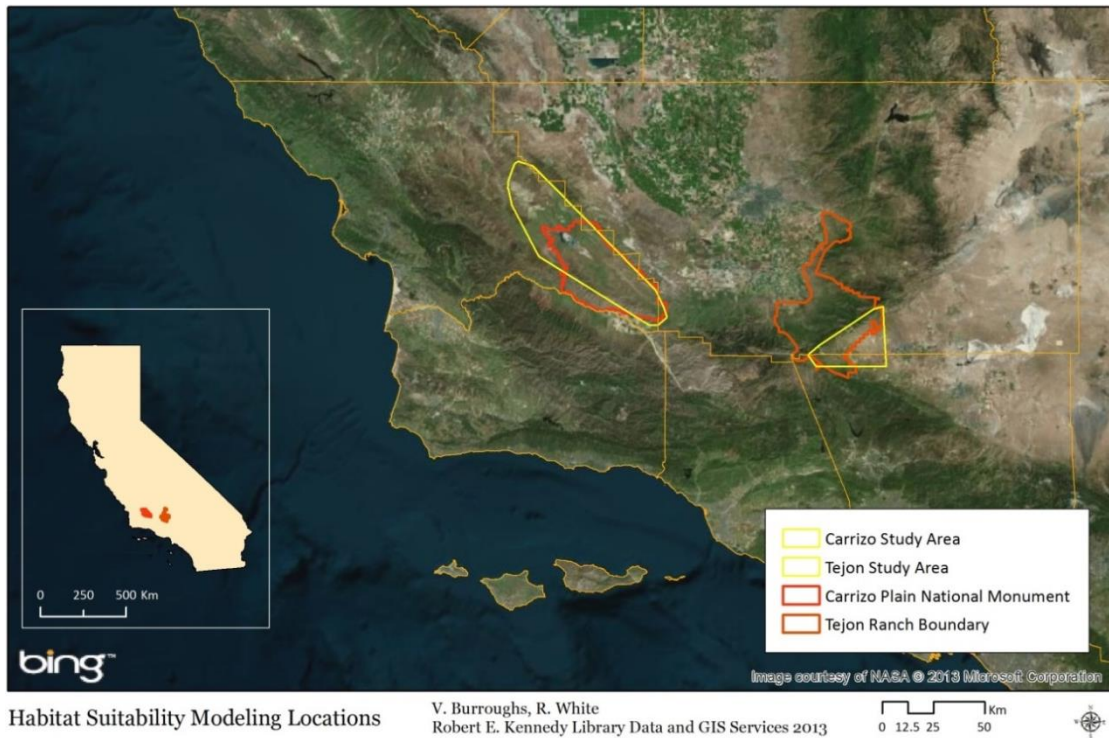


Figure 3: Overview map of the two habitat suitability modeling study areas in California.

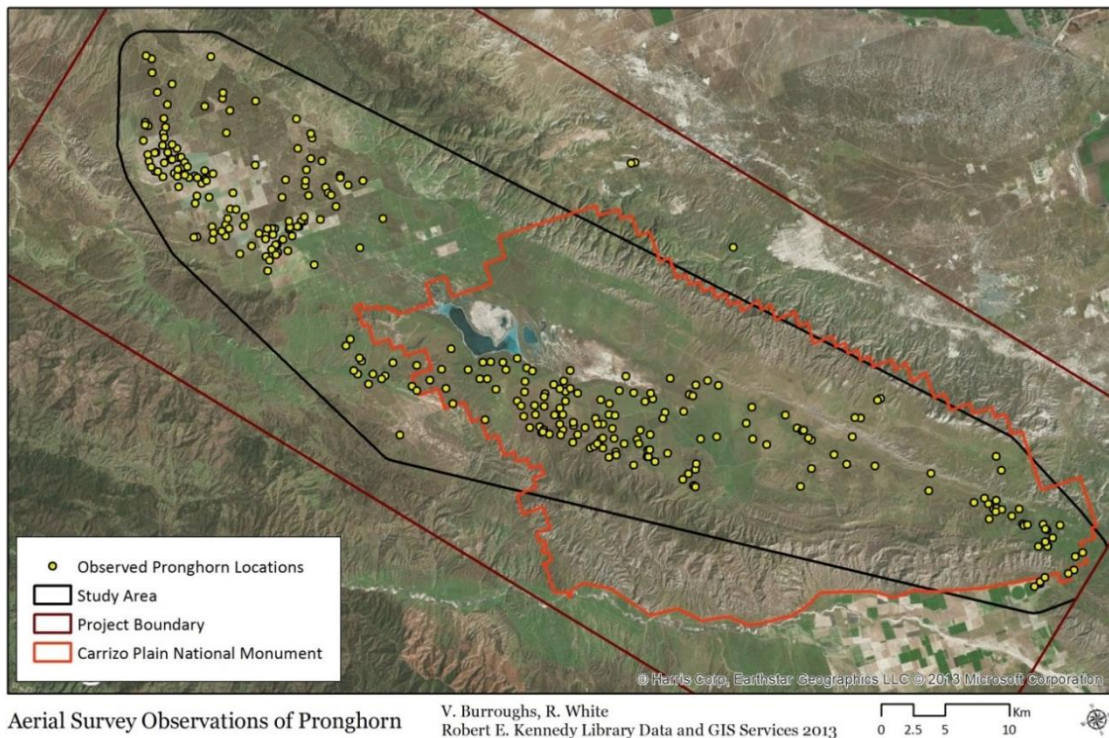


Figure 4: Aerial observation points of pronghorn within the Carrizo study area (2000-10).

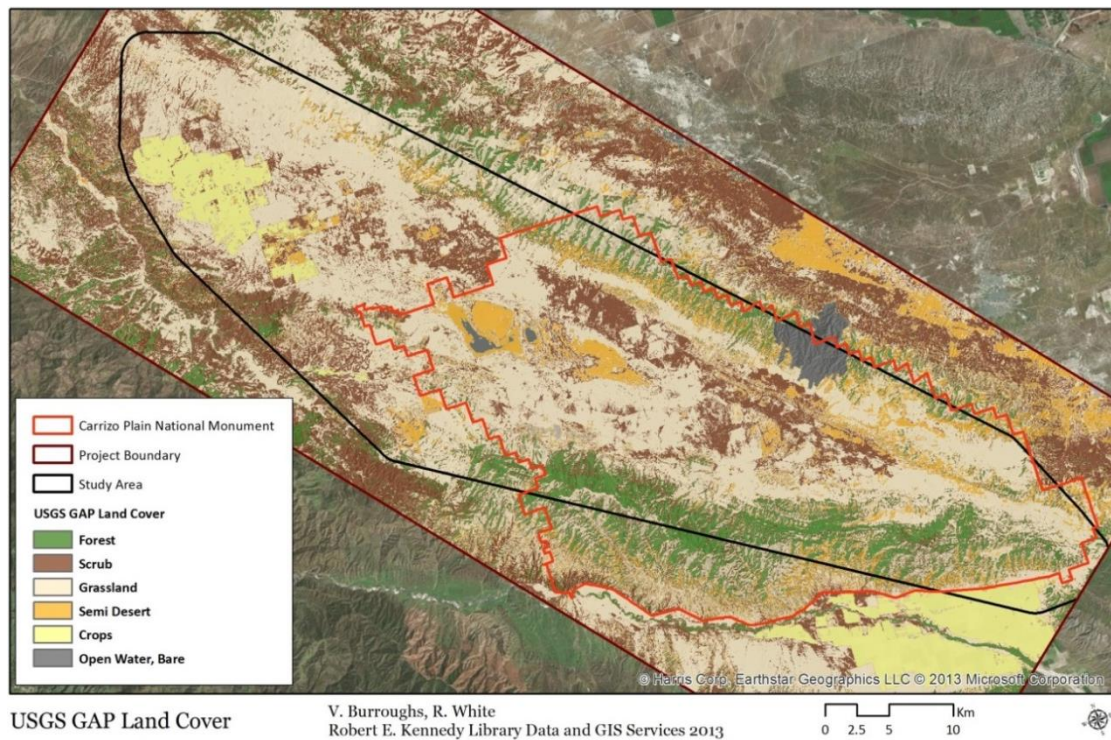


Figure 5: Vegetation composition categories for the Carrizo Plain HSM.

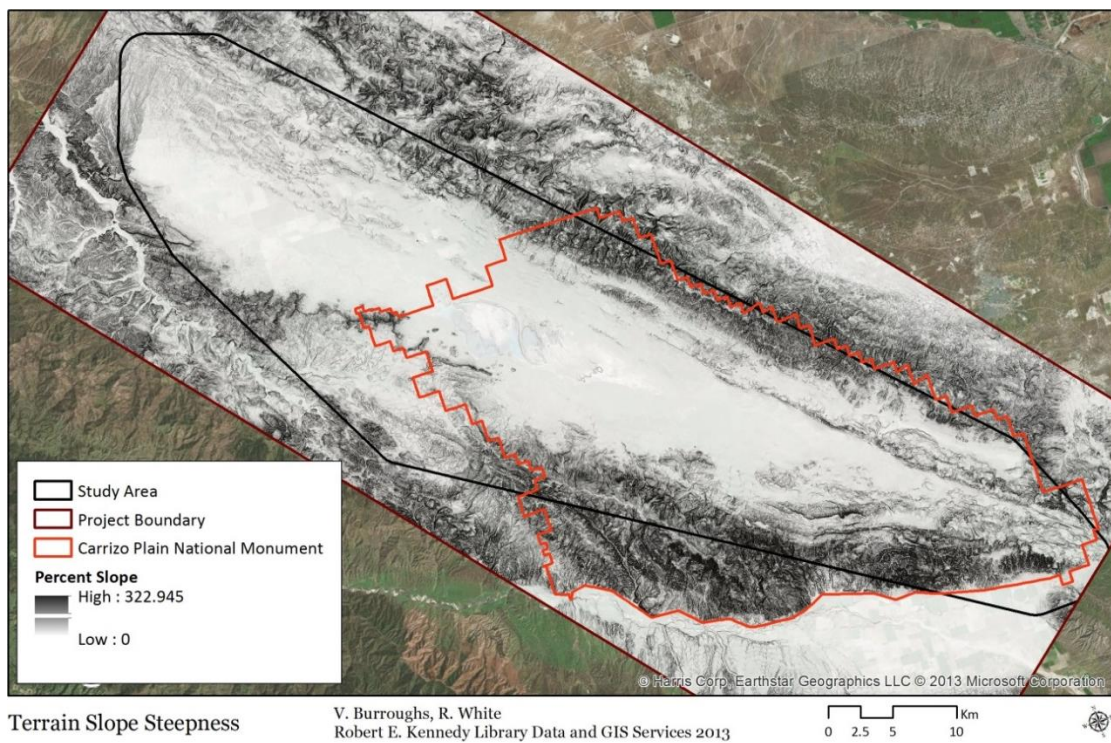


Figure 6: Terrain slope steepness for the Carrizo Plain HSM.

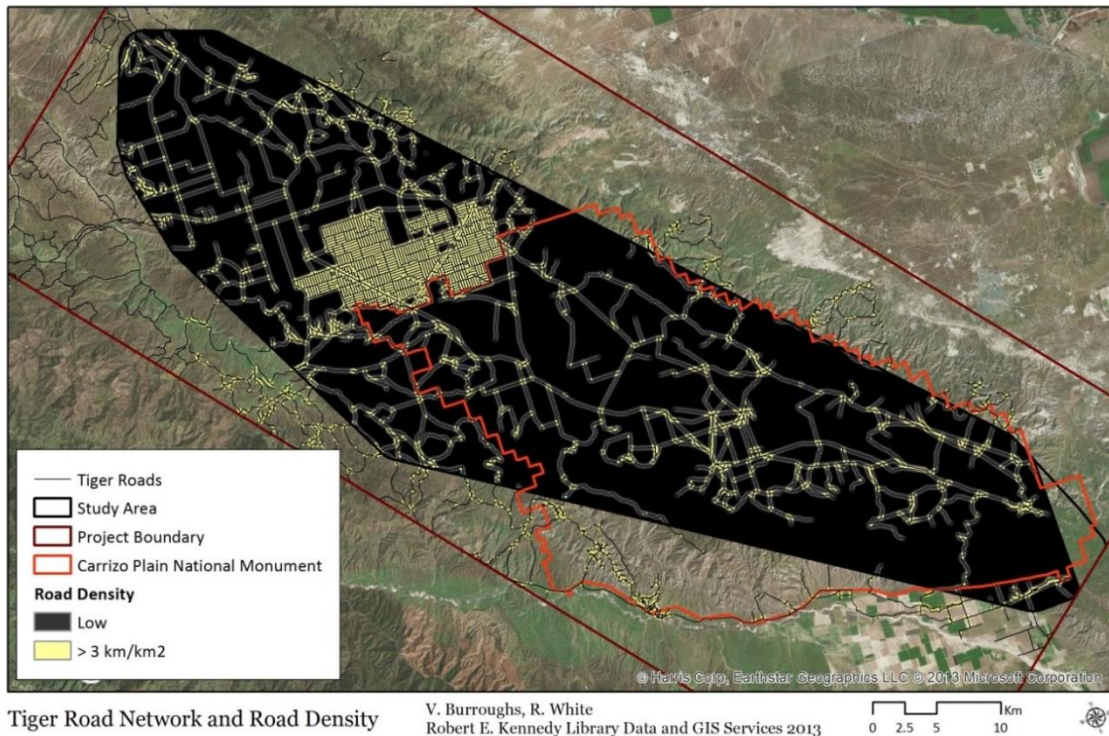


Figure 7: Roads and road density measurements for the Carrizo Plain HSM.

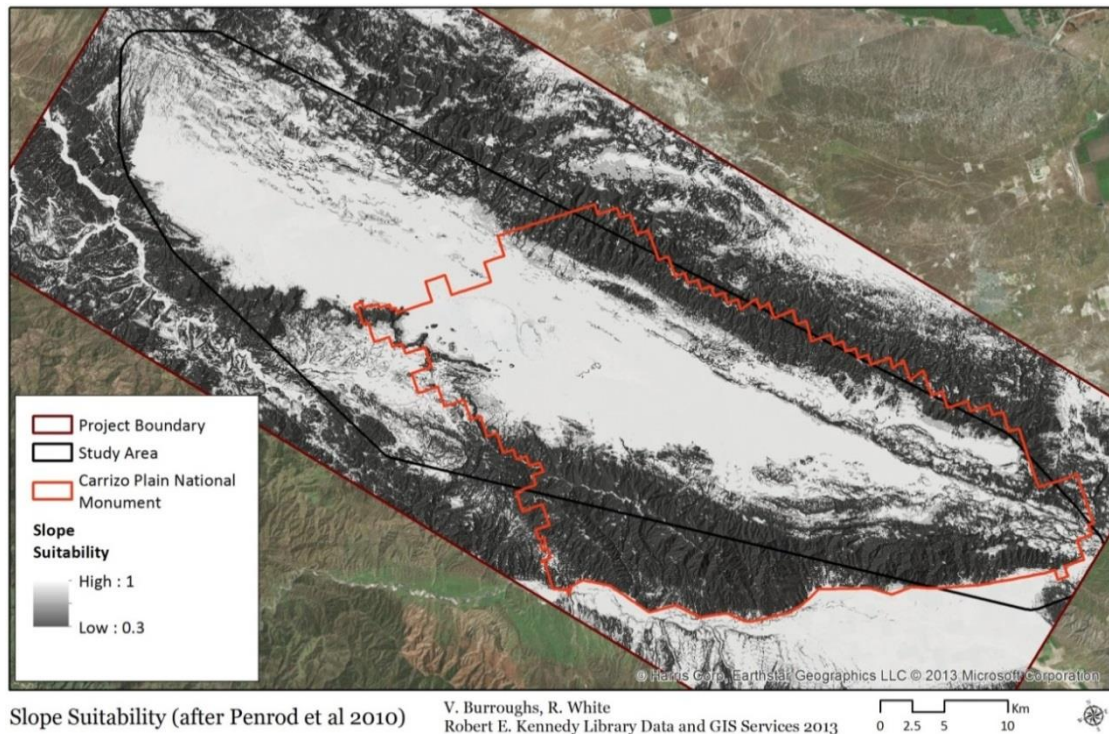


Figure 8: Slope suitability map for Carrizo based on parameters from Penrod et al. 2010.

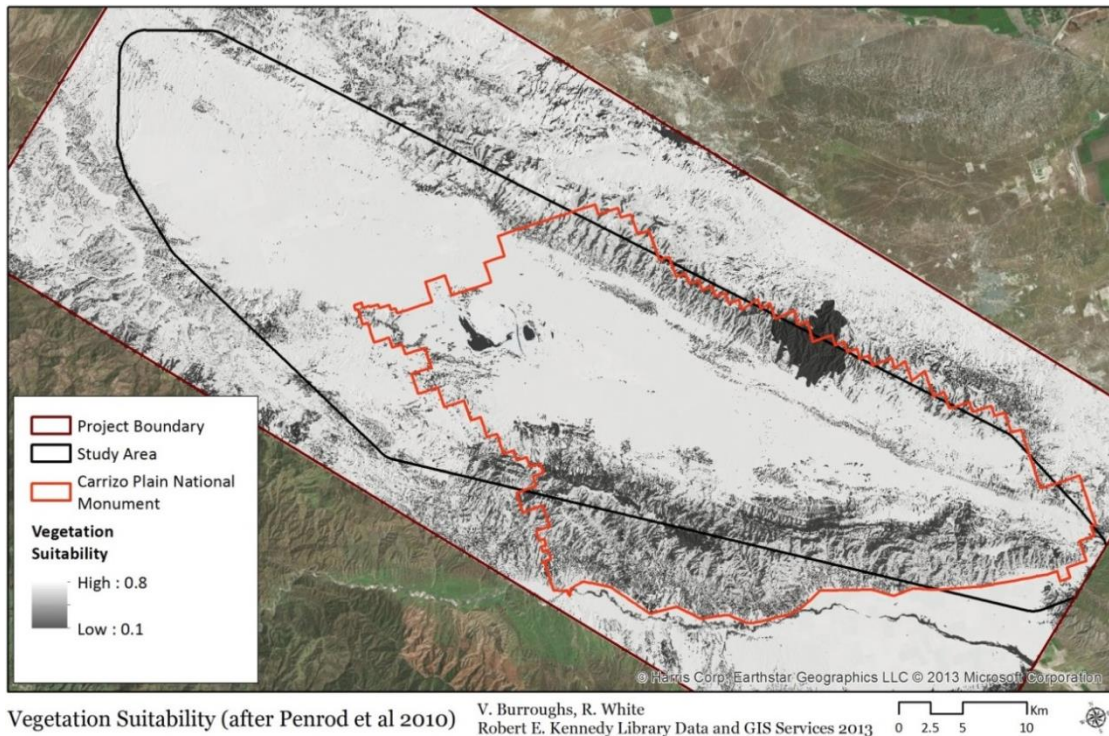


Figure 9: Vegetation suitability map for Carrizo based on parameters from Penrod et al. 2010.

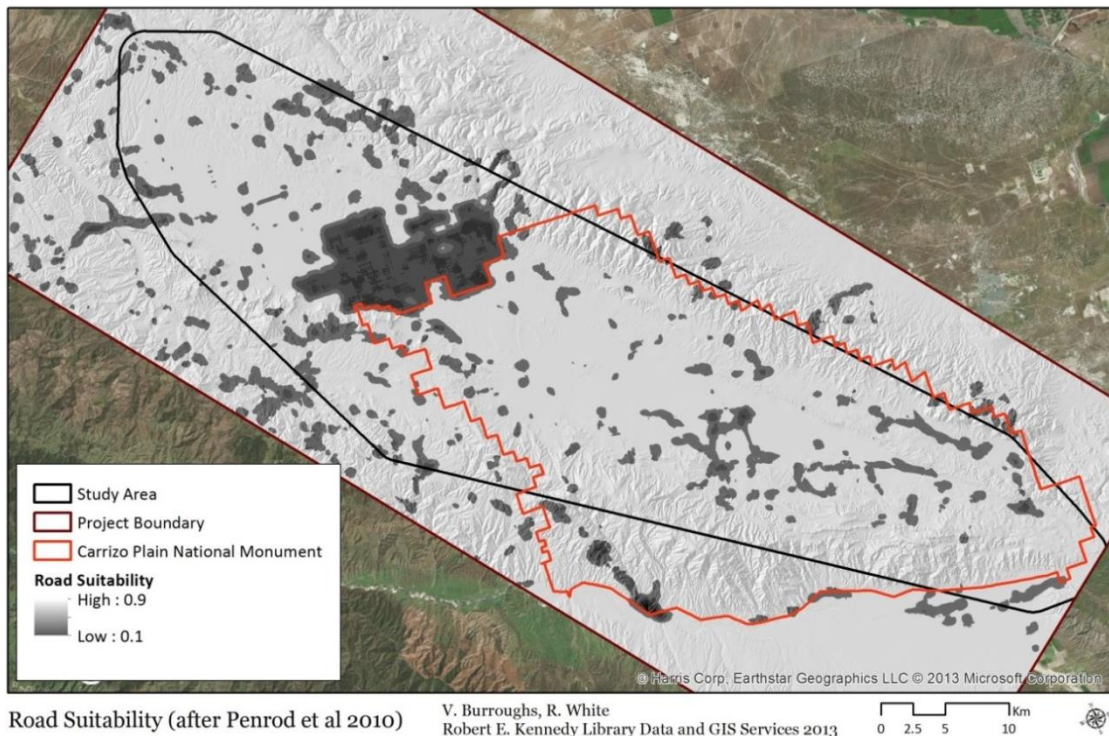


Figure 10: Road density map for Carrizo based on parameters from Penrod et al. 2010.

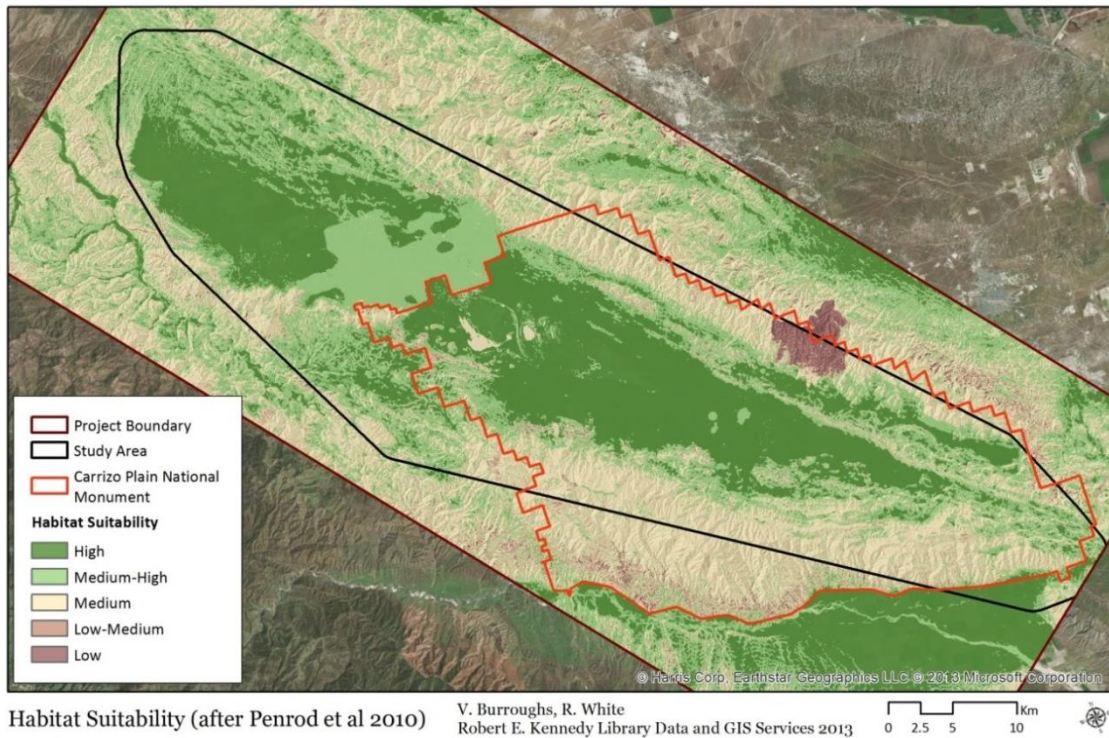


Figure 11: Depiction of Carrizo HSM using methods established by Penrod et al. 2010.

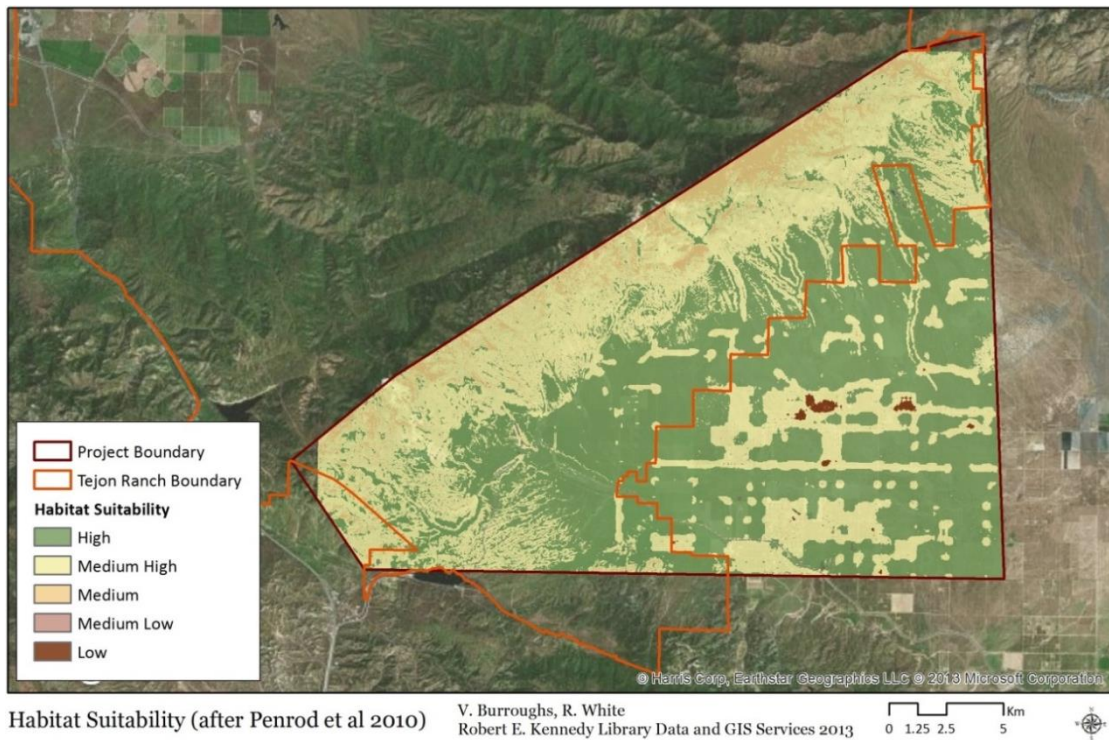


Figure 12: Application of Penrod et al. 2010 using HSM parameters to Tejon Ranch.

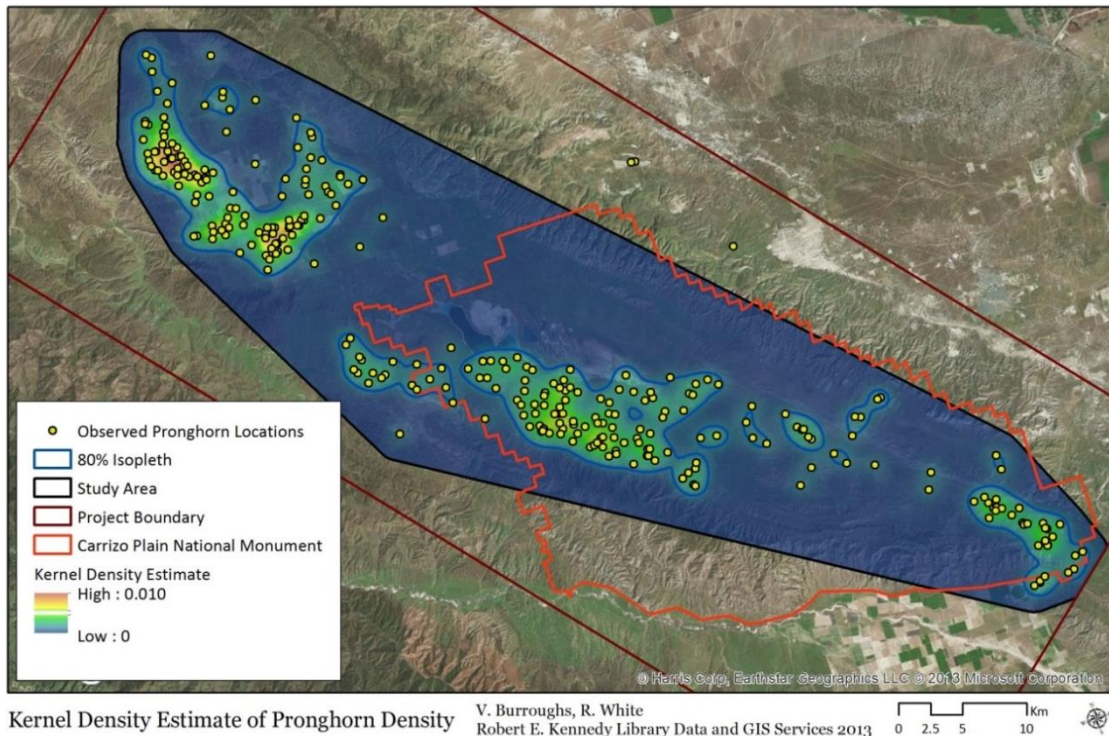


Figure 13: Kernel Density estimate of pronghorn density and 80% isopleth boundary.

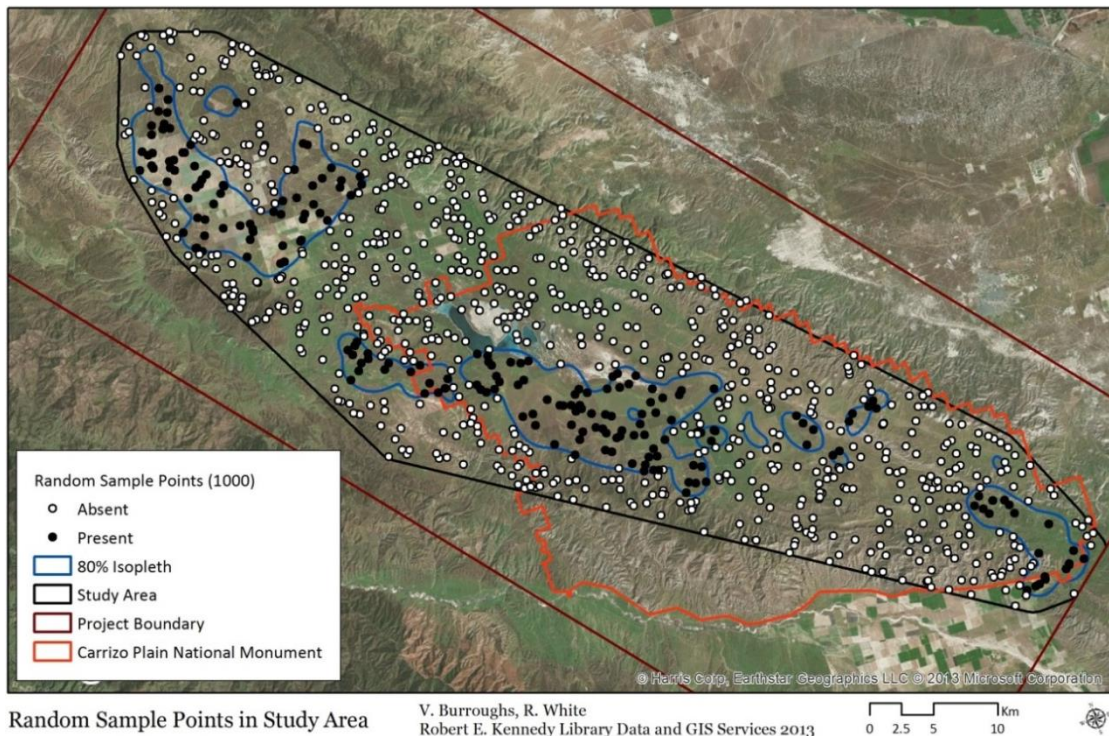


Figure 14: Random sample points (1000) generated to develop the Carrizo HSM.

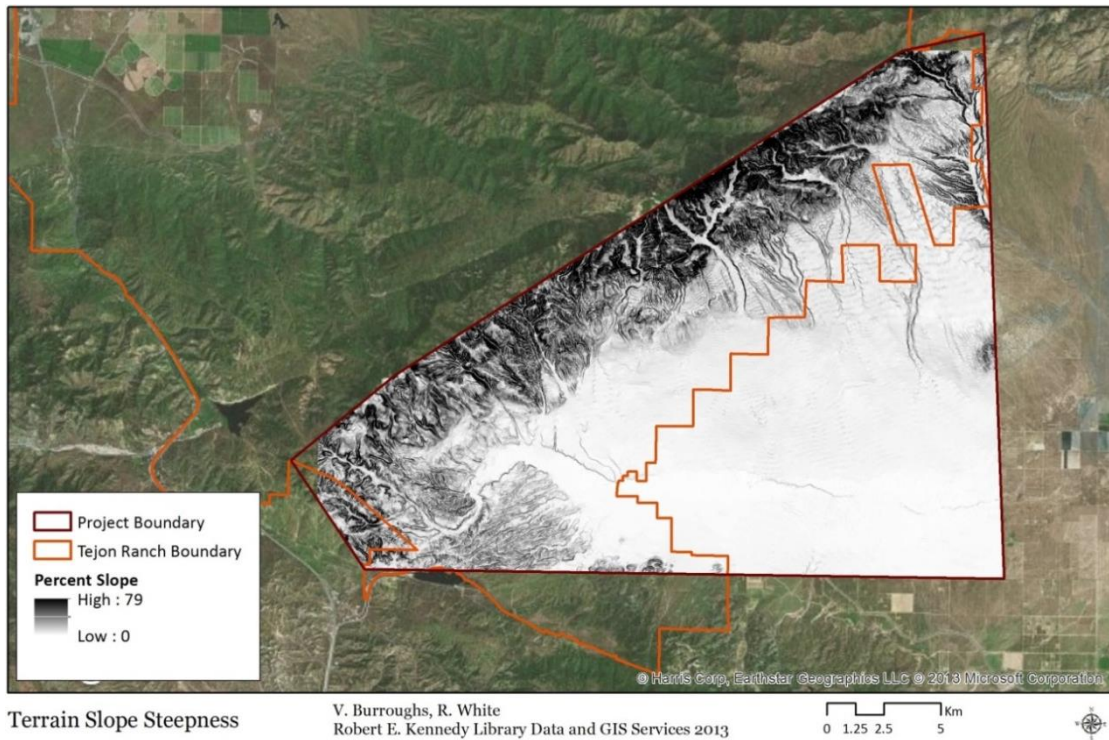


Figure 15: Terrain slope steepness for the Tejon Ranch HSM.

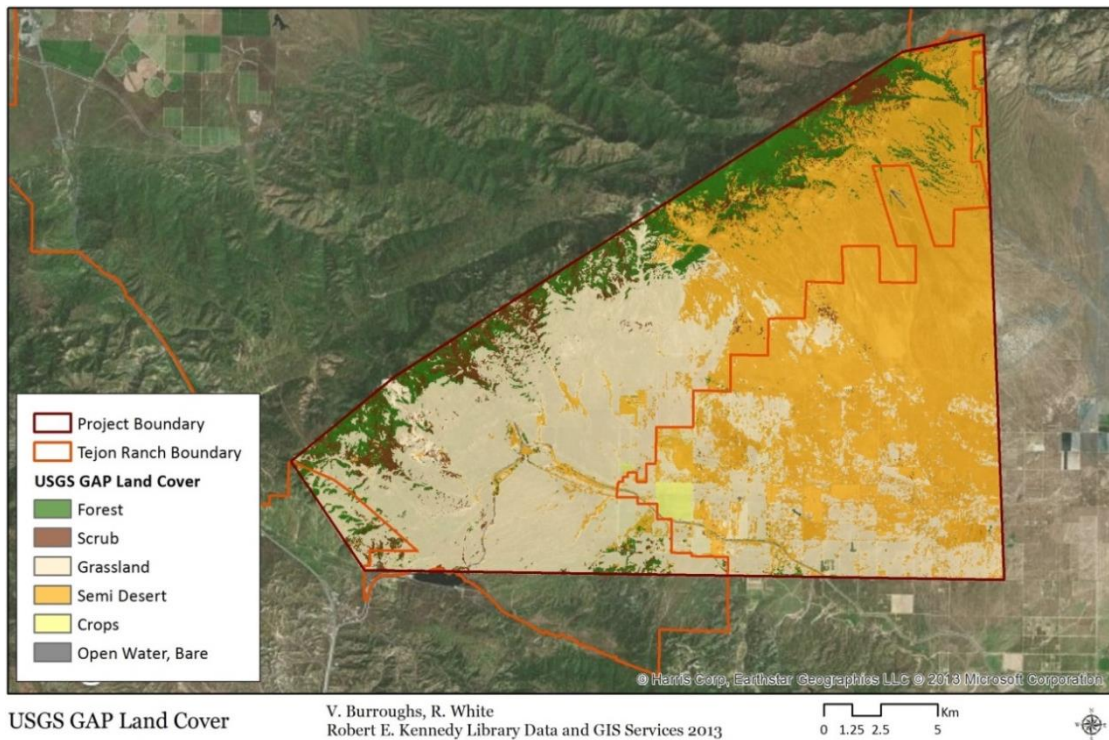


Figure 16: Vegetation composition categories for the Tejon Ranch HSM.

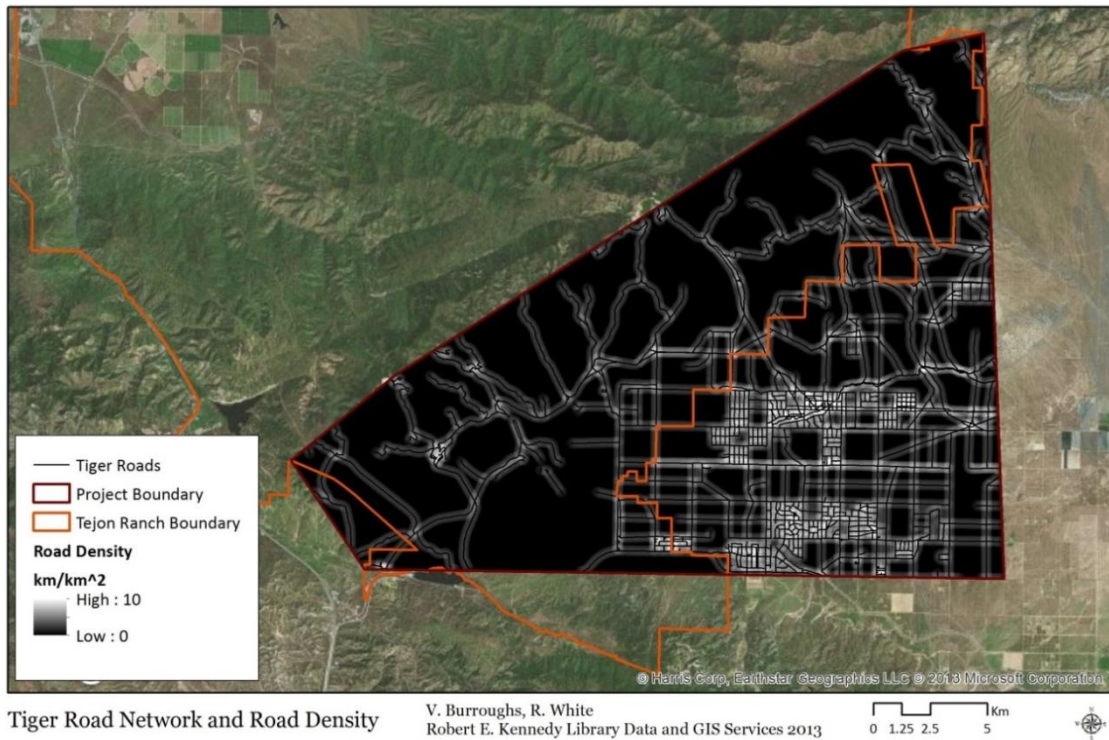


Figure 17: Roads and road density measurements for the Tejon Ranch HSM.

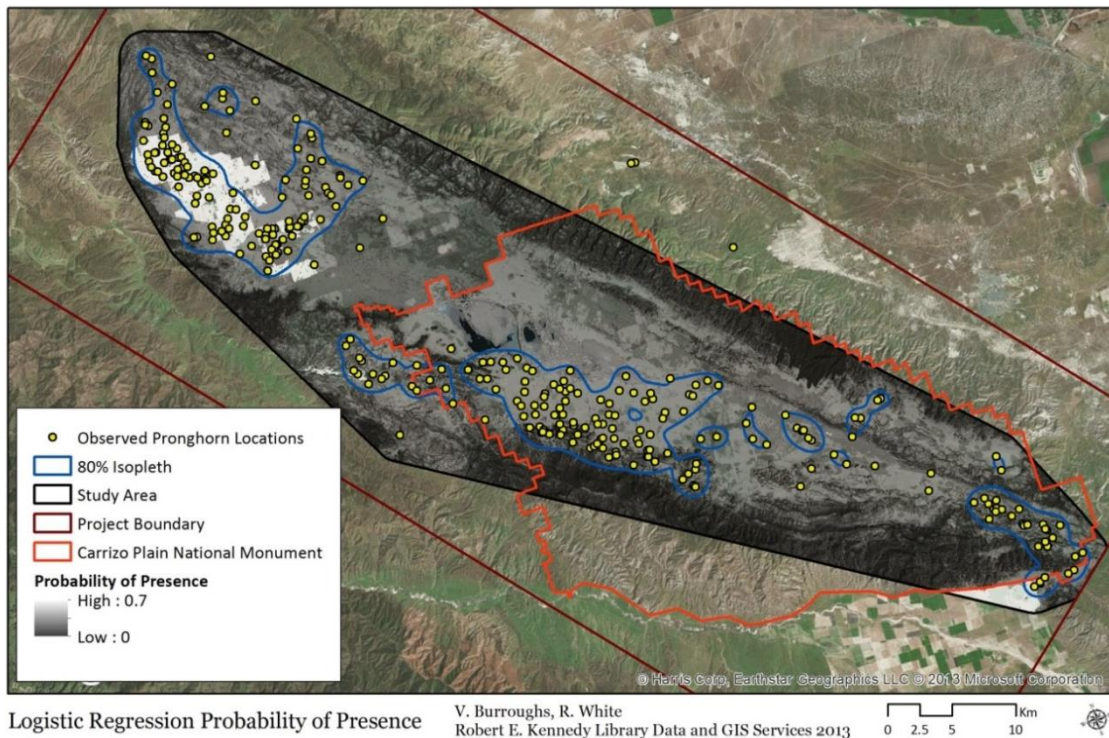


Figure 18: Continuous probability of pronghorn presence for the Carrizo Plain based on logistic regression analysis.

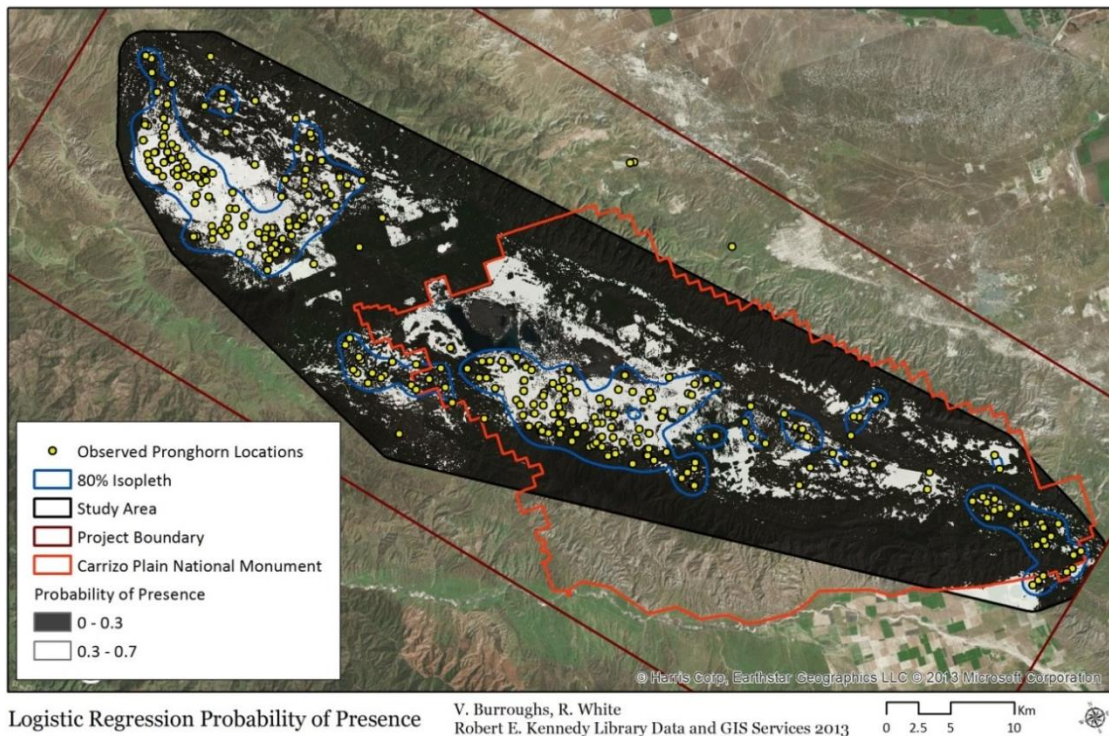


Figure 19: Binary probability of pronghorn presence for the Carrizo Plain derived from Kappa statistic cutoff of 0.3.

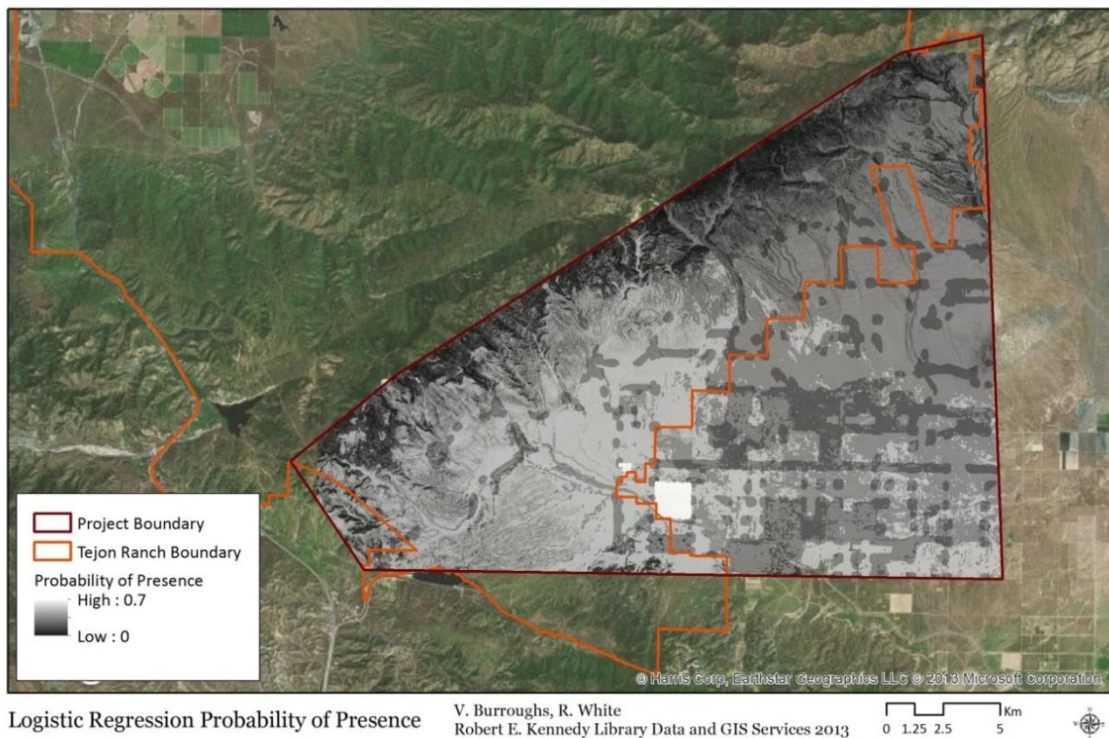


Figure 20: Continuous probability of pronghorn presence for Tejon Ranch based on the logistic regression analysis.

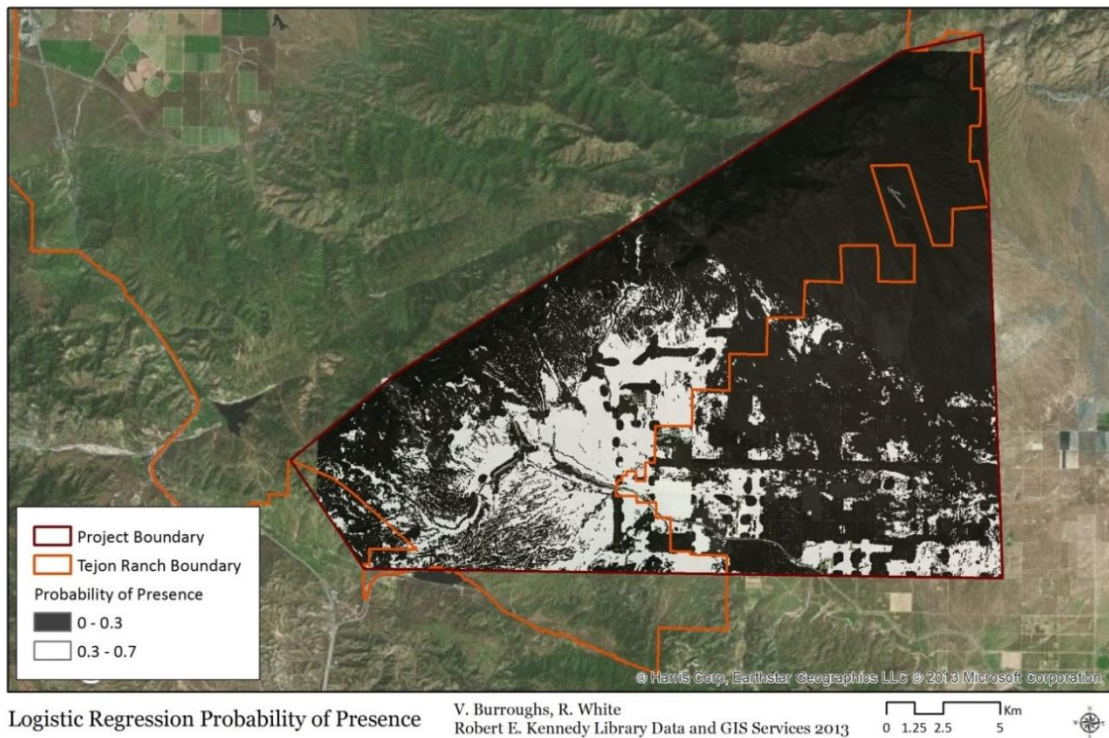


Figure 21: Binary probability of pronghorn presence for Tejon Ranch derived from Kappa statistic cutoff of 0.3.

## **APPENDIX II: MODEL BUILDING DOCUMENTATION**

### **Part I: Application of Penrod Model**

One initial objective of this project was to reproduce the HSM methods and results previously applied to the Carrizo Plain by Penrod et al. (2010). The methods described by Penrod were applied to the raster layers on both Carrizo and Tejon using the following general procedures.

- 1) Using the ArcGIS tool 'Reclassify', the percent slope layer was classified with the following break values  $<5\% = 1$ ,  $<5\% \times <20\% = 0.6$ ,  $>20\% = 0.3$
- 2) Similarly, the GapVeg layer was reclassified from vegetation categories, into the corresponding suitability values as presented in Appendix B of Penrod et al. (2010).
- 3) Finally, Road Density was reclassified using the following breaks (Penrod et al. 2010):

Density km/km <sup>2</sup>	Suitability Rating
0-0.5	0.9
0.5-1	0.8
1-2	0.8
2-4	0.4
4-6	0.3
6-8	0.2
8-10	0.1
>10	0

### **Part II: ArcGIS Documentation/Input Layer Development**

Three GIS input layers were used to develop the habitat suitability model: a slope layer, a vegetation layer, and a roads layer. The slope layer was derived as follows:

- 10m resolution Digital Elevation Model was downloaded from the USGS National Map. The 10m DEM was projected in ArcGIS 10 into the UTM coordinate system Zone 10, North American Datum 1983, to match the other layers in the analysis. Elevation of this

layer is recorded in meters. This layer was clipped to the project extent using the ArcGIS Extract by Mask tool.

<http://viewer.nationalmap.gov/viewer/>

- Slope raster (percent slope) was derived using Slope tool in ArcGIS, using the clipped 10m DEM as input.

### ***Vegetation Layer Development:***

Initially, to characterize vegetation for Carrizo and Tejon land cover data from the 2006 USGS National Land Cover Database was investigated. (<http://www.mrlc.gov/nlcd2006.php>)

An updated layer, containing better characterization of vegetation with the study area, particularly the croplands was also investigated from the 2011 National Agricultural Statistics Service layer:

<http://www.nass.usda.gov/research/Cropland/SARS1a.htm>

This layer was used in the initial modeling, but I found that other land cover layers were available, particularly suited for landscape-scale habitat analyses. The NASS layer provided good depiction of the croplands but did not further distinguish wildland vegetation types useful for characterizing the pronghorn locations. Other layers possible for vegetation are listed in Penrod et al. (2010).

Ultimately, the 30m resolution USGS GAP Land Cover layer was downloaded and clipped to represent vegetation and land cover for both the Carrizo and Tejon project areas:

<http://gapanalysis.usgs.gov/data/>

Vegetation categories from this layer follow the USGS National Vegetation Classification System (NVCS). For this study, vegetation classes were reclassified to the relevant cover types present on both the Carrizo and Tejon study sites as follows:

1. Forest
2. Scrub
3. Grass
4. Semi-Desert
5. Crops
6. Miscellaneous (including open water and burned areas).

See Table 2 for a breakdown of habitat types by model categories for the Carrizo Plain study area.

Table 3 provides similar information for Tejon Ranch.

Table 2: USGS Gap Land Cover class and merged VegClasses for Carrizo project area.

VegClass	USGS LandCover: ECOLSYS_LU	Area sq. km
Forest	Great Basin Pinyon-Juniper Woodland	264.60
Forest	California Central Valley Mixed Oak Savanna	71.54
Forest	California Lower Montane Blue Oak-Foothill Pine Woodland and Savanna	41.16
Forest	Mediterranean California Foothill and Lower Montane Riparian Woodland	8.40
Forest	California Coastal Live Oak Woodland and Savanna	7.35
Forest	California Montane Jeffrey Pine-(Ponderosa Pine) Woodland	1.20
Forest	North American Warm Desert Riparian Woodland and Shrubland	0.80
Forest	Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland	0.74
Forest	California Central Valley Riparian Woodland and Shrubland	0.22
Forest	Central and Southern California Mixed Evergreen Woodland	0.17
Forest	Southern California Oak Woodland and Savanna	0.17
Shrub	Southern California Coastal Scrub	578.50
Shrub	Mediterranean California Mesic Serpentine Woodland and Chaparral	165.64
Shrub	Southern California Dry-Mesic Chaparral	45.31
Shrub	Sonora-Mojave Semi-Desert Chaparral	2.70
Shrub	California Mesic Chaparral	1.68
Grassland	California Central Valley and Southern Coastal Grassland	1395.15
Grassland	California Mesic Serpentine Grassland	13.80
Semi-Desert	Sonora-Mojave Creosotebush-White Bursage Desert Scrub	71.88
Semi-Desert	Inter-Mountain Basins Mixed Salt Desert Scrub	57.92
Semi-Desert	Sonora-Mojave Mixed Salt Desert Scrub	35.25
Semi-Desert	Mojave Mid-Elevation Mixed Desert Scrub	33.66
Semi-Desert	Inter-Mountain Basins Big Sagebrush Shrubland	18.49

Crops	Cultivated Cropland	95.74
Crops	Pasture/Hay	48.27
Misc.	North American Warm Desert Bedrock Cliff and Outcrop	78.03
Misc.	Recently burned	22.60
Misc.	Open Water (Fresh)	4.13
Misc.	Southern California Coast Ranges Cliff and Canyon	0.88
Misc.	North American Warm Desert Pavement	0.70
Merged*	Developed, Open Space (roads)	145.59
Merged*	Developed, Low Intensity	6.15
Merged*	Developed, Medium Intensity	0.57
Merged*	Developed, High Intensity	0.02

\*Merged categories were blended with landcover categories in their immediate vicinity, further explained in the road layer development section.

Table 3: USGS GAP Land Cover class and merged VegClasses for Tejon Ranch.

VegClass	USGS Land Cover: ECOLSYS_LU	Area sq.km
Forest	Mediterranean California Mixed Oak Woodland	23.31
Forest	Great Basin Pinyon-Juniper Woodland	9.86
Forest	California Lower Montane Blue Oak-Foothill Pine Woodland and Savanna	1.93
Forest	Central and Southern California Mixed Evergreen Woodland	1.15
Forest	California Montane Jeffrey Pine-(Ponderosa Pine) Woodland	1.08
Forest	Mediterranean California Foothill and Lower Montane Riparian Woodland	0.65
Forest	Rocky Mountain Aspen Forest and Woodland	0.57
Forest	California Central Valley Mixed Oak Savanna	0.42
Forest	Inter-Mountain Basins Curl-leaf Mountain Mahogany Woodland and Shrubland	0.23
Forest	North American Warm Desert Riparian Woodland and Shrubland	0.14
Forest	Mediterranean California Red Fir Forest	0.05
Forest	California Coastal Live Oak Woodland and Savanna	0.03
Forest	Mediterranean California Dry-Mesic Mixed Conifer Forest and Woodland	0.02
Forest	Mediterranean California Mixed Evergreen Forest	0.02
Shrub	Southern California Dry-Mesic Chaparral	11.44
Shrub	Northern and Central California Dry-Mesic Chaparral	5.82
Shrub	Southern California Coastal Scrub	2.07
Shrub	California Montane Woodland and Chaparral	0.98
Shrub	Sonora-Mojave Semi-Desert Chaparral	0.80
Shrub	California Mesic Chaparral	0.14
Grassland	California Central Valley and Southern Coastal Grassland	184.40
Semi-Desert	Mojave Mid-Elevation Mixed Desert Scrub	145.72

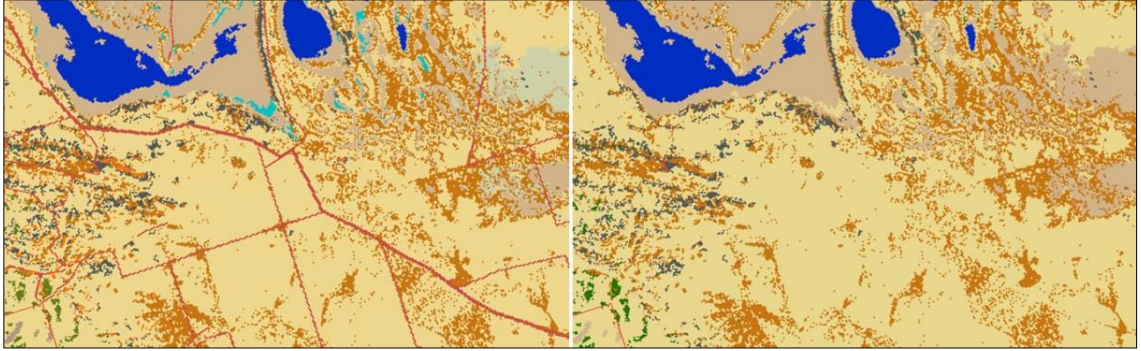
Semi-Desert	Sonora-Mojave Mixed Salt Desert Scrub	18.09
Semi-Desert	Inter-Mountain Basins Mixed Salt Desert Scrub	15.33
Semi-Desert	Inter-Mountain Basins Big Sagebrush Steppe	7.30
Semi-Desert	Sonora-Mojave Creosotebush-White Bursage Desert Scrub	5.22
Semi-Desert	Inter-Mountain Basins Big Sagebrush Shrubland	2.60
Semi-Desert	Inter-Mountain Basins Semi-Desert Grassland	0.09
Crops	Cultivated Cropland	2.59
Crops	Pasture/Hay	0.26
Misc.	Southern California Coast Ranges Cliff and Canyon	0.05
Misc.	North American Warm Desert Bedrock Cliff and Outcrop	0.02
Misc.	Open Water (Fresh)	1.13
Merged*	Developed, Open Space	23.79
Merged*	Developed, Low Intensity	0.13
Merged*	Developed, Medium Intensity	0.05

\*Merged categories were blended with landcover categories in their immediate vicinity, further explained in the road layer development section.

### ***Roads Layer Development:***

For analysis of road density roads were represented as a vector layer, the US Census TIGER Roads layer. Roads and urban areas were also included in the Land Cover layer. In order to simplify the representation of roads so just the vector roads were used for the density analysis, ‘roads’ represented as pixels in the Land Cover Layer, coded as ‘Developed Low Intensity’, were removed from the raster layer. To perform this operation, a majority filter with a 5x5 pixel moving window was applied to the USGS Gap Land Cover layer to replace ‘Developed Low Intensity’ pixels with the pixel value most common in its immediate surroundings. Within the study area there were very few locations other than roads represented as Developed Low Intensity.

The following image illustrates how the roads from the Land Cover layer were “blended” in using the above method:



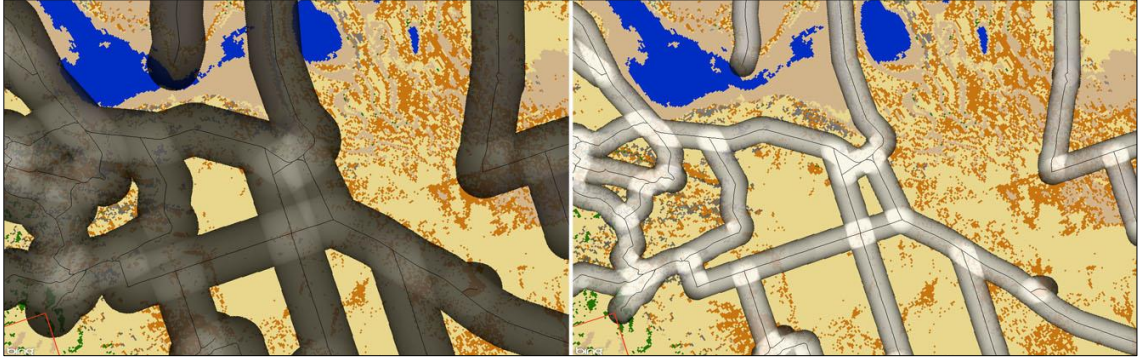
Roads for this analysis were represented by the US Census Tiger Roads database 2010 available from the US Census Bureau (<http://www.census.gov/cgi-bin/geo/shapefiles2010/layers.cgi>).

Roads were clipped to the extent of study area.

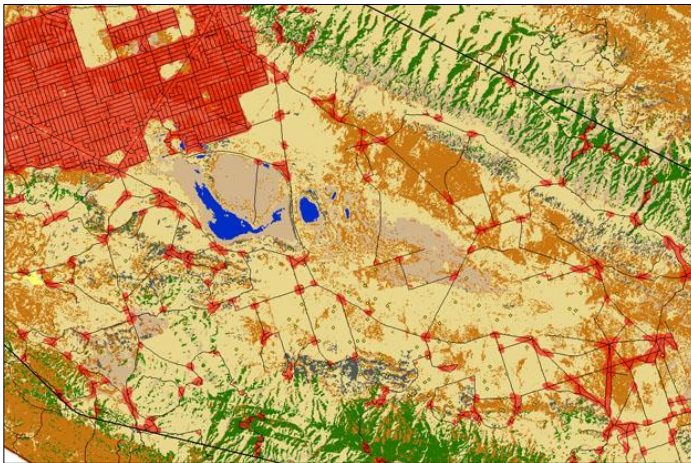
Road density was calculated using the following method. Note that Penrod et al. (2010) did not include a detailed description of how road density was calculated in their study. It is possible that road length within a 1km area was used, but this is not documented.

Initially, road density was calculated over a circular radius of 500m to allow for a generalized attribute of road density over large areas. The additional effect of a 500m radius is that of broadening the ‘high road density effect’ of a single road.

Upon revision, road density was calculated over a 250m radius, narrowing the effect of a single road, yet wide enough to allow multiple road segments to coalesce together to show areas of higher road density, as illustrated in the image below. The image below on the left illustrates the effect of the 500m radius and the image below on the right shows the effect of the reduced 250m radius:



The 250m radius was selected because restricting the radius too narrowly would show only strips of ‘high-road-density’ along the roads and immediate intersections, and would not indicate high road density found across broader areas. The application of this parameter is illustrated in the following image:



Road density was included in the model as a categorical variable, ‘High road density’ or not. A cutoff-value from the road density layer (above) was applied to create this categorical layer. Visually, a cutoff value of 3.3 was used as a means of indicating areas of high road density, while also eliminating the ‘presence’ effect of a single road.

In addition, the cutoff value of 3.3 represents the approximate midpoint in the road density ‘suitability’ ratings applied in Penrod et al. (2010), representing a possible binary division between ‘low’ road density and ‘high.’

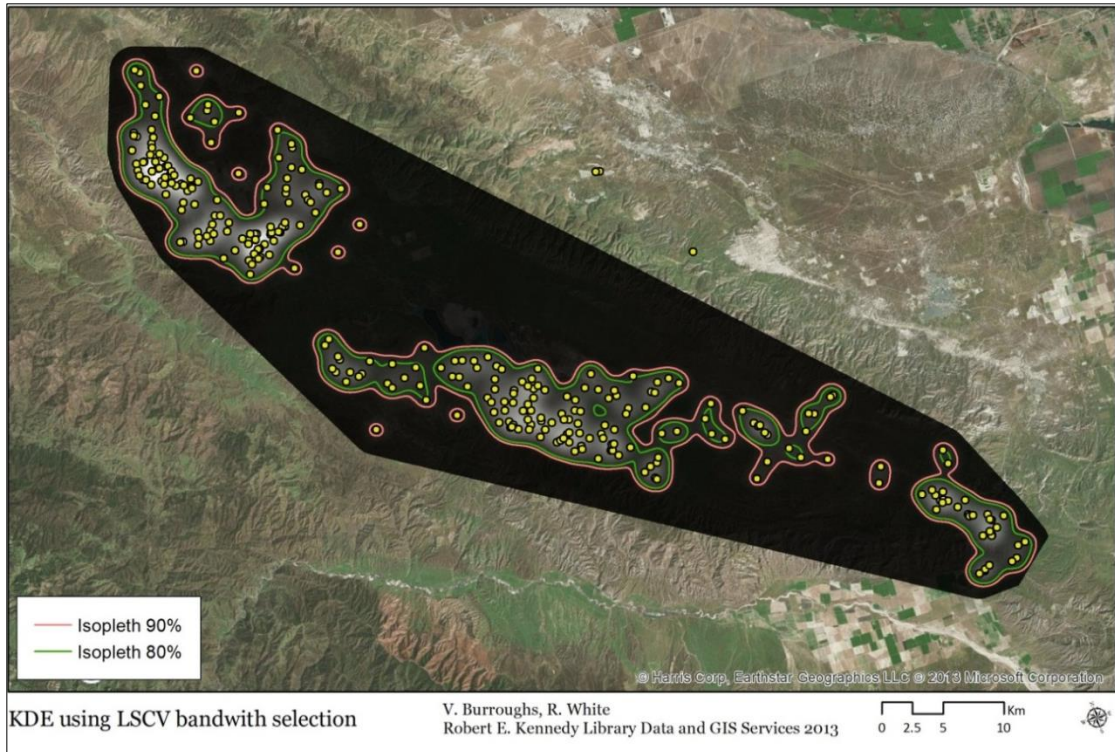
### **Part III: Application of Aerial Survey Data:**

Spreadsheets were obtained from the California Department of Fish and Wildlife containing aerial observations of pronghorn and geographic coordinates (latitude and longitude, referenced to WGS 84). The spreadsheet of observation data was added into ArcGIS as a csv file, and the ‘Display XY’ command was used to plot the recorded pronghorn locations. In order to compute further density metrics, pronghorn points were projected into the coordinate system NAD 83 UTM zone 10 with units in meters.

A rectangular project boundary extent was generated to encompass the Carrizo Plain and surrounding terrain.

A restricted modeling extent was generated using a 100% MCP of observation points, which was then extended by a 2km buffer to account for positional uncertainty of points.

Kernel Density Estimation with Least Squares Cross Validation was applied to identify occupied areas of the study area. Other validation methods were tested and visually examined, where the LSCV approach appeared to provide the most appropriate results. The other methods showed large, circular areas of high use, which did not match the distribution of points. It seemed possible that the long and narrow orientation of the points within the study area was important in how these methods compute bandwidth. Use of the 80% isopleth was based on the extension of ‘occupied’ area indicated by 90% isopleth extending into high-slope areas in the southern, central location of study area and at the sharp boundary of the crops areas in the northern position of the study area. Also, the 80% isopleth eliminated single-location points outside of the main areas of use, as depicted in the following image:



#### Part IV: Statistical Analyses Output

##### *Binary Logistic Regression Output:*

##### Logistic Regression Table

Predictor	Coef	SE Coef	Z	P	Odds Ratio	95% CI Lower Upper	
Constant	-1.68905	0.558224	-3.03	0.002			
GAP Veg							
Shrub	0.47787	0.575003	0.83	0.406	1.61	0.52	4.98
Grassland	1.09528	0.546735	2.00	0.045	2.99	1.02	8.73
Semi-desert	0.612586	0.620939	0.99	0.324	1.85	0.55	6.23
Crops	2.51127	0.642429	3.91	<0.0001	12.32	3.50	43.40
C_slope_10	-0.0544825	0.0092665	-5.88	<0.0001	0.95	0.93	0.96
RoadClass2c	-0.531518	0.260299	-2.04	0.041	0.59	0.35	0.98

Log-Likelihood = -436.063

Test that all slopes are zero: G = 136.941, DF = 6, P-Value = < 0.0001

#### Goodness-of-Fit Tests

Method	Chi-Square	DF	P
Pearson	1289.15	988	<0.001
Deviance	868.31	988	0.997
Hosmer-Lemeshow	13.97	8	0.082

#### Measures of Association:

(Between the Response Variable and Predicted Probabilities)

Pairs	Number	Percent	Summary Measures	
Concordant	123010	76.0	Somers' D	0.52
Discordant	38127	23.6	Goodman-Kruskal	0.53
Ties	654	0.4	Kendall's Tau-a	0.17
Total	161791	100.0		

#### *Univariate Logistic Regression Analysis for each input layer:*

*Binary Logistic Regression: Present versus GAP\_Veg*

Link Function: Logit

#### Response Information

Variable	Value	Count
Present	1	203 (Event)
	0	797
Total		1000

#### Logistic Regression Table

Predictor	Coef	SE Coef	Z	P	Odds Ratio	95% CI Lower Upper	
Constant	-3.33220	0.508850	-6.55	<0.0001			
GAP Veg							
Shrub	1.52940	0.545671	2.80	0.005	4.62	1.58	13.45
Grassland	2.16808	0.518826	4.18	<0.0001	8.74	3.16	24.17
Semi-desert	0.52771	0.590347	2.59	0.010	4.61	1.45	14.65
Crops	4.02535	0.605038	6.65	<0.0001	56.00	17.11	183.31

Log-Likelihood = -463.222

Test that all slopes are zero: G = 82.623, DF = 4, P-Value = <0.0001

\* NOTE \* No goodness of fit test performed.

\* NOTE \* The model uses all degrees of freedom.

Measures of Association:

(Between the Response Variable and Predicted Probabilities)

Pairs	Number	Percent	Summary Measures	
Concordant	74484	46.0	Somers' D	0.32
Discordant	22764	14.1	Goodman-Kruskal	0.53
Ties	64543	39.9	Kendall's Tau-a	0.10
Total	161791	100.0		

\* NOTE \* 3 time(s) the standardized Pearson residuals, delta chi-square, delta deviance, delta beta (standardized) and delta beta could not be computed because leverage (Hi) is equal to 1.

*Binary Logistic Regression: Present versus c\_slope\_10*

Link Function: Logit

Response Information

Variable	Value	Count
Present	1	203 (Event)
	0	797
Total		1000

Logistic Regression Table

Predictor	Coef	SE Coef	Z	P	Odds Ratio	95% CI	
						Lower	Upper
Constant	-0.708214	0.101201	-7.00	<0.0001			
C_slope_10	-0.0639301	0.0088543	-7.22	<0.0001	0.94	0.92	0.95

Log-Likelihood = -457.322

Test that all slopes are zero: G = 94.422, DF = 1, P-Value = <0.0001

Goodness-of-Fit Tests

Method	Chi-Square	DF	P
Pearson	1579.90	991	<0.0001
Deviance	905.65	991	0.975
Hosmer-Lemeshow	22.42	8	0.004

Table of Observed and Expected Frequencies:

(See Hosmer-Lemeshow Test for the Pearson Chi-Square Statistic)

Value	1	2	3	4	Group 5	6	7	8	9	10	Total
1											
Obs	4	5	6	14	16	18	25	34	36	45	203
Exp	1.4	4.4	9.3	16.5	22.4	26.3	28.8	30.2	31.3	32.5	
0											
Obs	96	95	94	86	84	82	75	66	64	55	797
Exp	98.6	95.6	90.7	83.5	77.6	73.7	71.2	69.8	68.7	67.5	
Total	100	100	100	100	100	100	100	100	100	100	1000

#### Measures of Association:

(Between the Response Variable and Predicted Probabilities)

Pairs	Number	Percent	Summary Measures	
Concordant	118318	73.1	Somers' D	0.47
Discordant	42858	26.5	Goodman-Kruskal	0.47
Ties	615	0.4	Kendall's Tau-a	0.15
Total	161791	100.0		

#### Binary Logistic Regression: Present versus RoadClass2c

Link Function: Logit

#### Response Information

Variable	Value	Count
Present	1	203 (Event)
	0	797
Total		1000

#### Logistic Regression Table

Predictor	Coef	SE Coef	Z	P	Odds Ratio	95% CI	
Constant	-1.34397	0.0834613	-16.10	<0.0001		Lower	Upper
RoadClass2c	-0.199712	0.249257	-0.80	0.423	0.82	0.50	1.33

Log-Likelihood = -504.202

Test that all slopes are zero: G = 0.663, DF = 1, P-Value = 0.415

\* NOTE \* No goodness of fit test performed.

\* NOTE \* The model uses all degrees of freedom.

Measures of Association:  
(Between the Response Variable and Predicted Probabilities)

Pairs	Number	Percent	Summary Measures	
Concordant	18643	11.5	Somers' D	0.02
Discordant	15268	9.4	Goodman-Kruskal	0.10
Ties	127880	79.0	Kendall's Tau-a	0.01
Total	161791	100.0		

\* NOTE \* 1 time(s) the standardized Pearson residuals, delta chi-square, delta deviance, delta beta (standardized) and delta beta could not be computed because leverage (Hi) is equal to 1.

**Correlation/pairwise correlation analyses:**

*General Regression Analysis: c\_slope\_10 versus GAP\_Veg, RoadClass2c*

Coefficients

Term	Coef	SE Coef	T	P
Constant	14.0598	0.98575	14.2631	<0.0001
GAP_Veg				
1	21.1158	1.41187	14.9559	<0.0001
2	-3.8079	1.16415	-3.2710	0.001
3	-5.1538	0.92838	-5.5515	<0.0001
4	3.8101	1.53356	2.4845	0.013
RoadClass2c	4.0822	0.80459	5.0737	<0.0001

Summary of Model

S = 16.6084    R-Sq = 25.08%    R-Sq(adj) = 24.70%  
PRESS = 277394    R-Sq(pred) = 24.20%

Analysis of Variance

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Regression	5	91763	91763	18352.6	66.5334	<0.0001
GAP_Veg	4	84662	78002	19500.6	70.6951	<0.0001
Roadclass2c	1	7101	7101	7100.8	25.7423	<0.0001
Error	994	274185	274185	275.8		
Lack-of-fit	4	547	547	136.7	0.4947	0.7396
Pure Error	990	273638	273638	276.4		
Total	999	365948				

*General Regression Analysis: c\_slope\_10 versus GAP\_Veg*

Coefficients

Term	Coef	SE Coef	T	P	VIF
Constant	17.3534	0.75099	23.1073	<0.0001	
GAP_Veg					
1	21.7636	1.42346	15.2893	<0.0001	1.09321
2	-4.5983	1.16794	-3.9371	<0.0001	1.08615
3	-5.4730	0.93769	-5.8366	<0.0001	1.03878
4	4.0663	1.55167	2.6206	0.009	1.11995

Summary of Model

S = 16.8137 R-Sq = 23.14% R-Sq(adj) = 22.83%  
 PRESS = 284370 R-Sq(pred) = 22.29%

Analysis of Variance

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Regression	4	84662	84662	21165.6	74.8695	<0.0001
GAP_Veg	4	84662	84662	21165.6	74.8695	<0.0001
Error	995	281286	281286	282.7		
Total	999	365948				

*General Regression Analysis: c\_slope\_10 versus RoadClass2c*

Coefficients

Term	Coef	SE Coef	T	P
Constant	11.4647	0.898117	12.7653	<0.0001
RoadClass2c	5.6083	0.898117	6.2445	<0.0001

Summary of Model

S = 18.7855 R-Sq = 3.76% R-Sq(adj) = 3.66%  
 PRESS = 353103 R-Sq(pred) = 3.51%

Analysis of Variance

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Regression	1	13761	13761	13760.8	38.9941	<0.0001
RoadClass2c	1	13761	13761	13760.8	38.9941	<0.0001
Error	998	352188	352188	352.9		
Total	999	365948				

*General Regression Analysis: RoadClass2c versus GAP\_Veg*

Coefficients

Term	Coef	SE Coef	T	P
Constant	0.0965943	0.0146147	6.60941	<0.0001
GAP_Veg				
1	-0.0793529	0.0277011	-2.86461	0.004
2	0.0968020	0.0227286	4.25903	<0.0001
3	0.0390935	0.0182479	2.14235	0.032
4	-0.0313769	0.0301962	-1.03910	0.299

Summary of Model

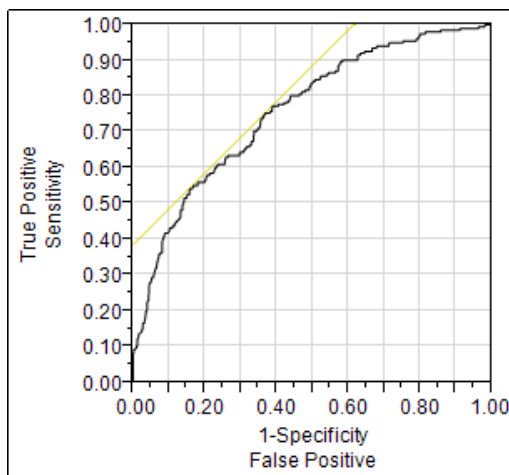
S = 0.327201    R-Sq = 2.61%    R-Sq(adj) = 2.21%  
 PRESS = 107.371    R-Sq(pred) = 1.83%

Analysis of Variance

Source	DF	Seq SS	Adj SS	Adj MS	F	P
Regression	4	2.850	2.850	0.712381	6.65398	<0.0001
GAP_Veg	4	2.850	2.850	0.712381	6.65398	<0.0001
Error	995	106.525	106.525	0.107061		
Total	999	109.375				

**Receiver Operating Characteristic/AUC**

*JMP Output - ROC Curve:*



<b>AUC</b>
0.76220