EFFICACY OF NATIVE GRASSLAND BARRIERS AT LIMITING PRAIRIE DOG DISPERSAL IN LOGAN COUNTY, KANSAS

BY

ZACHARY EDDY

B.A., Bethel College, 2000

A THESIS

submitted in partial fulfillment of the requirements for the degree

MASTER OF ARTS

Department of Geography
College of Arts and Sciences

KANSAS STATE UNIVERSITY
Manhattan, Kansas

2011

Approved by:

Major Professor
Dr. J.M. Shawn Hutchinson

Abstract

Prairie dogs (*Cynomys* spp.) are social, ground-dwelling rodents native to North American short- and mixed-grass prairie. They are also the main prey of the Federally-endangered black-footed ferret (*Mustela nigripes*). At the same time, prairie dog colonization is highly opposed by most agricultural landowners. Therefore nonlethal population management techniques must be investigated. This paper presents the results of research on the effectiveness of ungrazed vegetative barriers composed of native plants at limiting prairie dog dispersal away from a ferret reintroduction site in northwest Kansas. Data was collected on barrier quality and condition as well as estimates of population densities of immigrant prairie dogs, dispersing through the vegetative barrier to reoccupy previously extirpated colonies on properties surrounding the ferret reintroduction site. Using strip transects and aboveground visual counts to estimate population densities and visual obstruction ranking techniques to sample barrier condition, statistical analysis of the data indicated that while barrier condition increased over time, it was not effective at limiting prairie dog emigration from the black-footed ferret reintroduction site.

Table of Contents

Acknowledgements

My gratitude is owed to many people for the completion of this project. Foremost, I need to thank Mr. Charles Lee, K-State Research and Extension Wildlife Specialist, for his assistance both personally and professionally. For five years he's acted as a boss, teacher, and friend—for which I am grateful.

My advisor, Dr. J.M. Shawn Hutchinson, has also been a wonderful source of support during my time at Kansas State. Through him and my time working in the Geographic Information Systems Spatial Analysis Lab, I have learned that critical thinking and a desire to know more are the most important tools in the researcher's arsenal.

Additionally, thanks need to be given to the two other members of my Committee. Drs. Kendra McLachlan and Douglas Goodin gave their time and expertise to improving this document. As I transition from school to a career, I have already been able to apply principles learned in classes with them.

A debt of gratitude is also owed to the landowner's of all the sites on which this work was done. It is rare that a researcher is afforded the chance to conduct their work on over 25,000 acres of privately-owned land controlled by seven different parties. Their willingness to contribute to this work has been amazing. I need to give special thanks to Rob Manes, Randy Martin, Michelle Martin, Bill Beckman, and the rest of the Kansas Chapter of The Nature Conservancy. They have not only provided black-footed ferrets a home in northwest Kansas, but have also supplied me with one over the course of four summers—not to mention all their help with equipment breakdowns.

Finally, I need to thank my family. Mom, Dad, and Jennifer, I will never be able to repay you for the things you have given me. And I cannot forget Maura Cornell, who always reminds me to think about all the stars and all the fish. She also starts graduate school in August. Knowing that (and fearing she would get done before me) was motivation to finish writing this document.

INTRODUCTION

The prairie dog (*Cynomys* spp) is commonly regarded as a pest in agricultural communities (Jones 1999, Forrest and Luchsinger 2006, Lamb *et al*. 2006). The black-tailed prairie dog (*Cynomys ludovicianus*) is treated similarly by Kansas law (K. S. A. 80-1201-1203). Landowners are required to apply population management practices and reduce/eradicate prairie dog populations on their properties if an adjoining landowner files a complaint (U.S. Fish and Wildlife Service 2000). However, lethal control of prairie dogs through the use of toxicants is highly opposed by conservation organizations because of the prairie dog's status as a keystone species (Kotliar *et al.* 1999, Kotliar *et al.* 2006). One function of thriving colonies of prairie dogs is to provide food necessary for the success of the Federally-endangered black-footed ferret (*Mustela nigripes*) which has been reintroduced to the wild in 17 locations throughout the western U.S. by the U.S. Fish and Wildlife Service as well as one site each in Canada and Mexico (BFFRP 2009). Though, prairie dogs may not function as a keystone species throughout their range (Stapp 1998, Bartz *et al*. 2007), the importance of prairie dog colonies to black-footed ferret populations coupled with the ineffectiveness of lethal control measures has caused a great need for useful nonlethal methods for managing prairie dog populations (Knowles, C.J. 1986, Cable and Timm 1987, Franklin and Garrett 1989, Hoogland 2006a).

Nonlethal control of prairie dog populations is particularly important in Logan County, KS, where the U. S. Fish and Wildlife Service (USFWS) has reintroduced a number of black-footed ferrets on two privately-owned ranches. Prior to the reintroduction of the ferrets and continuing today, the Logan County commissioners and private landowners adjoining one of the ferret reintroduction sites were involved in litigation with the owners of the ferret reintroduction site (FRS) because of prairie dogs' tendency to emigrate from the FRS into the surrounding properties where they could potentially damage crops, compete with livestock for forage, and lower property values.

Attempts to force the FRS owners to eradicate prairie dogs have stalled in court because of the addition of black-footed ferrets. Since U.S. Federal law—specifically, the Endangered Species Act—preempts state law, the State of Kansas cannot force the owners of a FRS to kill prairie dogs, because it could adversely effect the reintroduced black-footed ferret populations (Barnhardt, Haverfield, and Haverfield vs. Board of Logan County Commissioners 2010). This highlights the need of the conservation community to know more about the effectiveness of nonlethal control measures, prairie dog population dynamics, and colony location choice in northwestern-Kansas.

The author conducted a study to investigate the effectiveness of a native grass vegetative dispersal barrier composed of ungrazed native pasture at limiting prairie dog emigration away from a black-footed ferret reintroduction site as well as monitor the reoccupation of extirpated colonies on properties adjacent to the FRS. Chapter 1 describes the purpose of the study and research questions investigated. In Chapter 2, a description of the study area is presented. Chapter 3 presents a review of primary and secondary literature pertinent to the study—such as prairie dog natural history, dispersal/colony expansion, methods of assessing population densities, and techniques for quantifying visual obstruction produced by vegetation within the barrier. The fourth chapter describes the methods used to study the effectiveness of the barrier. Results of the work are presented in Chapter 5. Chapter 6 is a discussion of the results, comparing them to previously reported results of other researchers and offering possible reasons for the results obtained in this project. The final chapter describes conclusions from the study, lists potential prairie dog management implications, and suggests further research needs.

CHAPTER 1 - PURPOSE

Background Information

In Logan County, Kansas, landowners of a large prairie dog colony complex, where the USFWS has reintroduced black-footed ferrets, have tried to ease tensions with their neighbors by creating native grassland barriers along their boundary fences. This was done in an attempt to limit prairie dog emigration from the release site to neighboring properties—where their presence is considered undesirable and lethal measures are employed to eliminate them. The barriers were created by constructing a 1-strand electric fence approximately 27 meters (30 yards) inside of the boundary fences. The electric fences prevented cattle from grazing within the 27-meter wide strip between the fences. This was done to allow vegetation (native grasses) to grow to its maximum height, which could prevent prairie dogs from colonizing new areas by creating a visual obstruction between properties. The barrier was constructed in 2007. Its design was based on several sets of research which suggested that visual obstructions and tall vegetation can limit prairie dog activity within an area (Snell and Hlavachick 1980, Cable and Timm 1988, Franklin and Garrett 1989).

Research Questions

There were several research questions addressed in this study regarding the population dynamics of prairie dogs dispersing through the native grassland barriers to reoccupy extirpated colonies.

1. Did sites bounded by a vegetative barrier (Treatment sites) exhibit less prairie dog activity than those not bounded by a barrier (Controls)?
   null hypothesis: There would be no difference in burrow activity between Treatments and Controls.
2. Was visual obstruction within the vegetative barrier greater than that along borders with Control sites?
   null hypothesis: There would be no difference in visual obstruction between the vegetative barrier and borders with Controls.
3. Did the condition of the barrier change over time (seasonally and/or from 2008 to 2009), particularly its height and spatial continuity?

null hypothesis: There was no change in barrier condition over time.

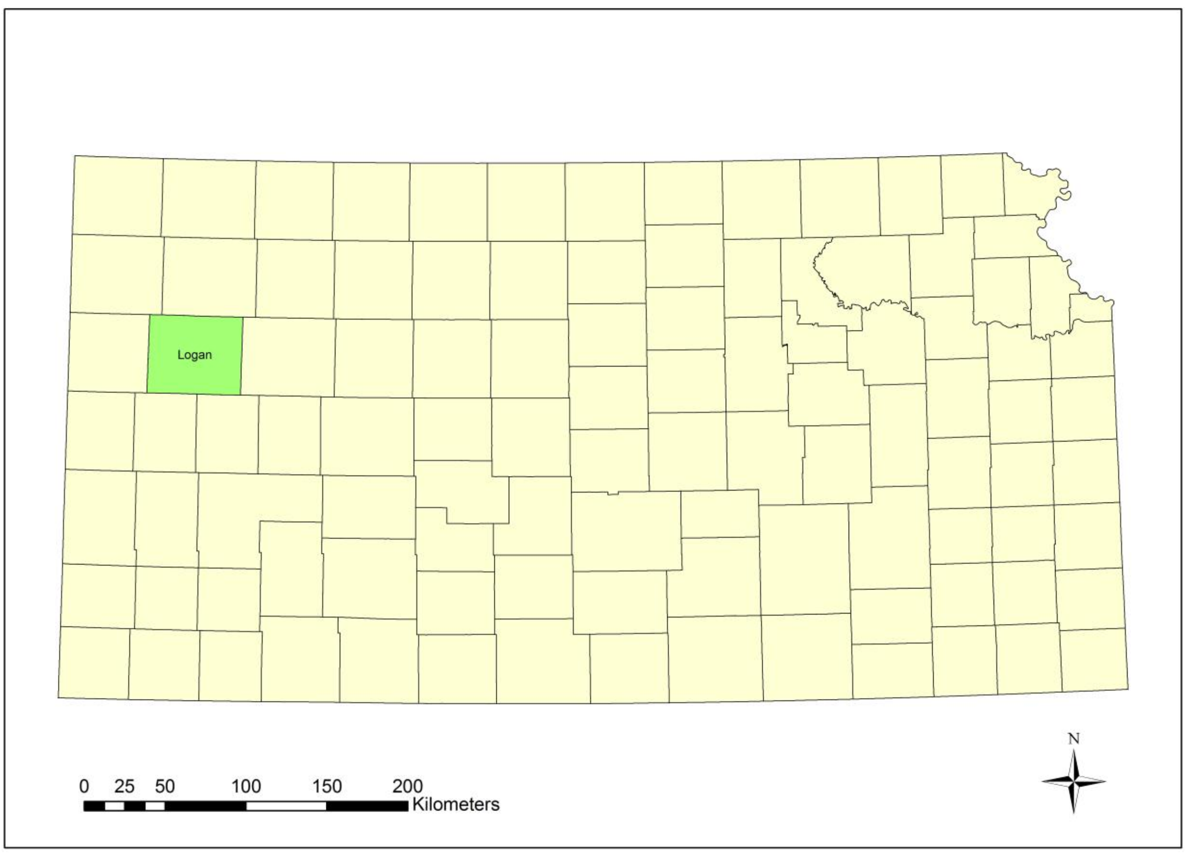
4. Were there any places along the barrier where vegetation was too short/sparse to create a visual obstruction?

hypothesis: there would be areas of poor visual obstruction within the barrier during each year of study.

CHAPTER 2 - STUDY AREA

All studies took place in Logan County in the northwestern part of Kansas (Fig. 2.1). The county seat is the town of Oakley (39° 7′ 41″ N, 100° 51′ 16″ W). Logan County is a rural, primarily agrarian area. Common land uses in the county include cattle grazing as well as dry-land and irrigated crop production. The nearest NOAA weather station is located at Goodland, KS (about 70 km northwest of FRS 1). Normal monthly temperature averages at that site range from 14.83° and 23.94°C during the months May-August. Average monthly precipitation ranges from 6.32- 8.99 cm from May to August. Mean annual precipitation is approximately 50.19 cm. The 2009 U.S. Census Bureau's population estimate for Logan County was 2,549 (U.S. Census Bureau 2011).

Figure 2.1 Logan County, KS.



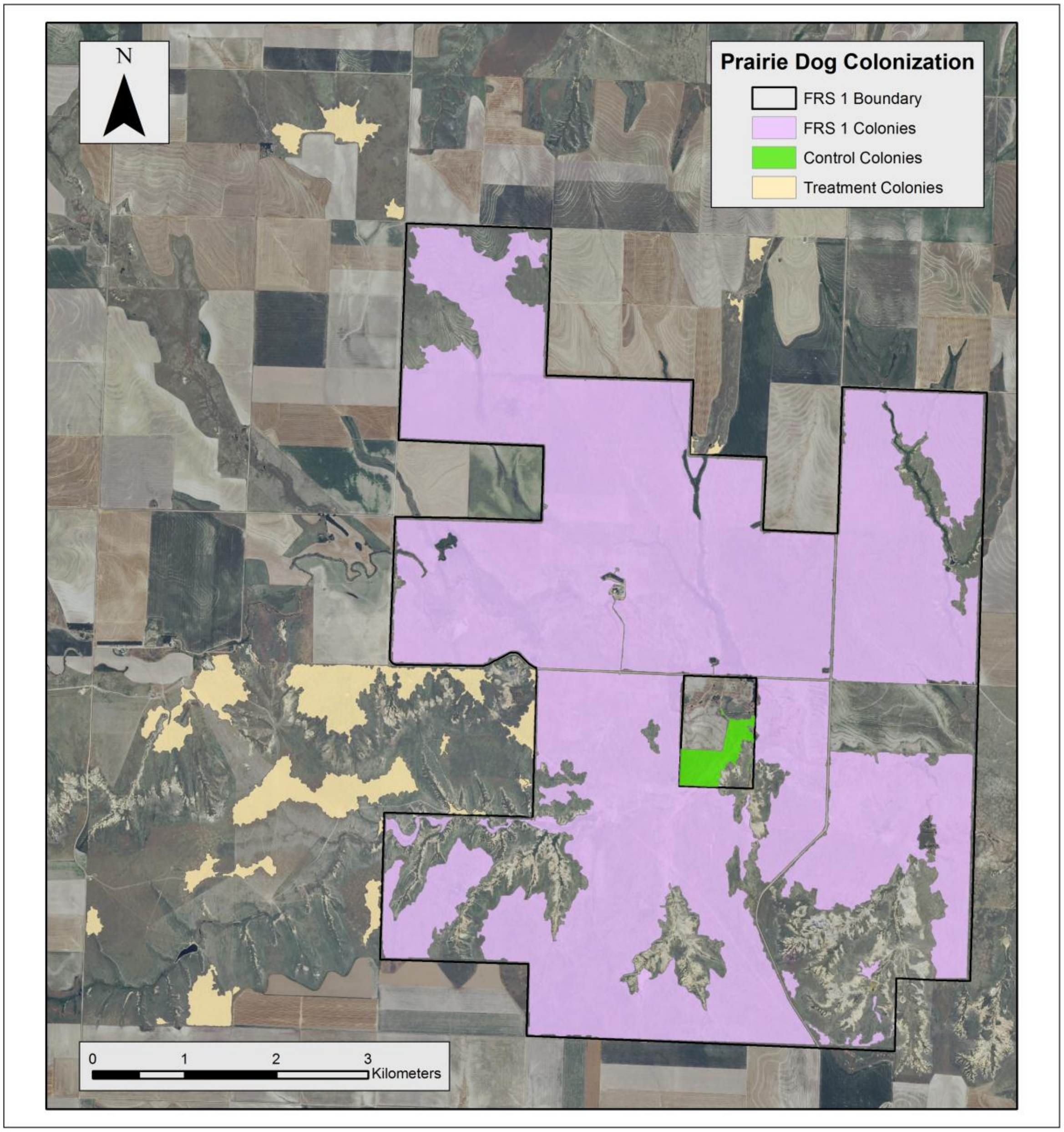
The study areas are on the border between the mixed- and shortgrass prairies. The EPA's level IV ecoregion map (Chapman *et al.* 2001) defines these areas as: Moderate Relief Rangeland (level IV code 25c) and Rolling Plains and Breaks (level IV code 27b). These ecoregions are short- and mixed-grass prairies, respectively. Approximately two-thirds of FRS 1 is within the Moderate Relief Rangeland class (with the boundary running from northwest to southeast through the property. The other one-third of FRS 1 and all of FRS 2—located approximately five miles east of FRS 1—are within the boundary of the Rolling Plains and Breaks class. However, the mapping of ecoregion boundaries is imprecise, and in nature, ecoregions generally do not change abruptly; instead transitioning along gradients (Loveland and Merchant 2004). Thus, there are no clear-cut differences in vegetation between study sites. While there is heterogeneity between vegetative community composition between areas under investigation, this is just representative of the mosaic of plant biodiversity in this transitional zone.

Areas under investigation were all representative examples of common prairie dog habitat, and the vegetative community in all pastures was characterized by mixed- and shortgrass prairie plant species. Common plant species within the area include: blue grama (*Bouteloua gracilis* [Willd. ex Kunth] Lag. ex Griffiths), buffalograss (*Buchloe dactyloides* [Nutt.] Engelm.), little bluestem (*Schizachyrium scoparium* [Michx.] Nash), sideoats grama (*Bouteloua curtipendula* [Michx.] Torr.) , purple threeawn (*Aristida purpurea* Nutt.), Western wheatgrass (*Pascopyrum smithii* [Rydb.] Á. Löve), broom snakeweed (*Gutierrezia sarothrae* [Pursh] Britton & Rusby), horseweed (*Conyza canadensis* [L.] Cronquist), Louisiana wormwood (*Artemisia ludoviciana* Nutt.), scarlet globe mallow (*Sphaeracea coccinea* [Pursh] Rydb.), Western ragweed (*Ambrosia psilostachya* DC.), and yucca (*Yucca glauca* Nutt.).

Research took place on seven privately-owned ranches. Two properties are sites where the USFWS has reintroduced black-footed ferrets. All other ranches bordered the two ferret reintroduction sites.

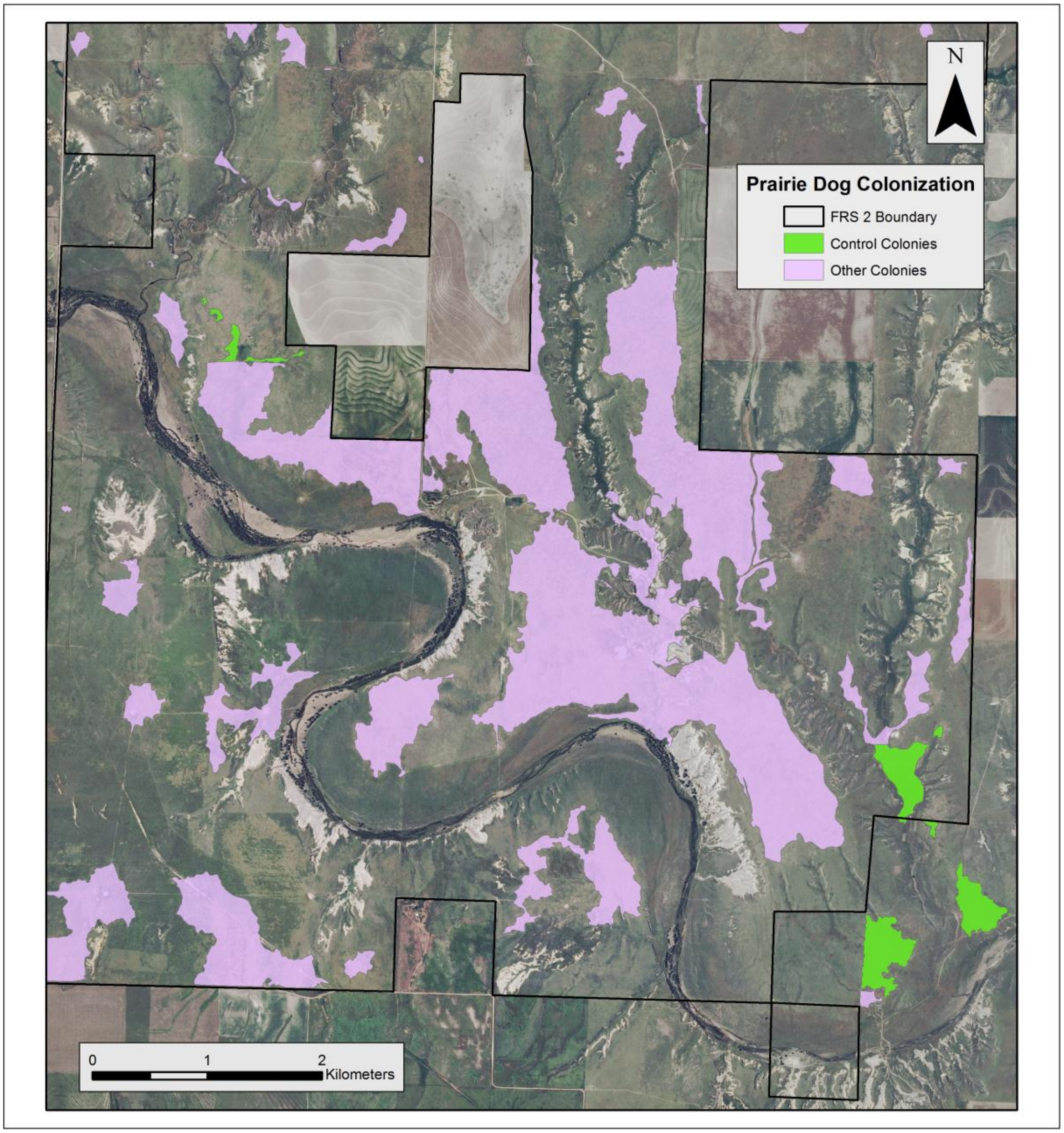
The two ferret reintroduction sites are large working cattle ranches. FRS 1 occupies approximately 4,014 hectares (Figure 2.2). In 2006, prior to the start of this study, the area colonized by prairie dogs was 2,603.61 hectares). The most In 2006, prior to the start of this study, the area colonized by prairie dogs was 2,603.61 hectares). The most recent measure of the area colonized was conducted during the summer of 2009. It indicated colonization had expanded to include 3,043.92 hectares. recent measure of the area colonized was conducted during the summer of 2009. It indicated colonization had expanded to include 3,043.92 hectares.

Figure 2.2 Overview of primary study colonies around FRS 1. The grassland barrier cannot be seen at this scale, but it runs the length of the exterior boundary fences (black lines outside of purple shaded areas). Prairie dog colonization shown on FRS 1 was mapped in 2006. The other colonies shown were mapped following the 2008 dispersal season.



Prairie dog population management at FRS 1 includes directed shooting and poisoning of prairie dogs within the vegetative barrier areas. During the years of this study, the colony complex itself received no rodenticide treatment. However, in the summer of 2010 after all data for this project had been collected, some areas within the colony complex were also baited with rodenticide to reduce population densities in hopes of lessening prairie dog emigration from the site.

Figure 2.3 Secondary study site at Ferret Reintroduction Site 2. Prairie dog colonization shown was mapped following the 2008 dispersal season.



The owners of FRS 2 have taken a more active role in prairie dog management at their property. FRS 2 is a 6,754.45 hectare ranch. In 2006, prairie dog colonization was measured at 981.24 hectares (Figure 2.3). That winter, the owners of FRS 2 identified a core area of prairie dogs in the center of their property where they wanted to maintain prairie dog populations. Areas away from this core were baited with rodenticide. This was done to provide buffers at the periphery of the ranch into which prairie dogs dispersing from the core area could immigrate to without impacting surrounding properties as much. Each winter since 2006, the owners reapply rodenticides to selected areas bordering other properties in their own effort to lessen dispersal away from their property. The extent of colonization on FRS 2 was last measured in 2010. This work showed that the area occupied by prairie dogs had expanded (primarily in the core areas where they want prairie dogs) to include 1,020.49 hectares. In addition to the annual application of rodenticides, the owners also allow directed shooting of prairie dogs in areas where colonization is unwanted (*i.e.* within colonies slated for rodenticide application).

The owners of the two ferret reintroduction sites also employ differing grazing management plans for their cattle. At FRS 1 the owners use a short duration cell grazing system similar to the Savory Grazing Method (Savory 1983). This method calls for high densities of cattle grazing small areas (“cells” within a pasture) for short durations (days-weeks) before being moved to the next grazing cell. Following grazing within one cell, the cell is then rested until sufficient regrowth has taken place that it can be grazed by livestock again.

The owner’s of FRS 1 never discussed their grazing management system with the author, so only estimates of the stocking rates can be made. There were an estimated 1,200-1,500 head of yearling cattle grazing within cells year round. Grazing cells were about 40 hectares each. Cattle were divided into several groups (approximately 4), and each group grazed in separate cells. Suggesting that cells were stocked at a rate of about 0.2 hectares of pasture per head. Groups were rotated to new cells in less than one week. Rotations continued year-round, and each cell was grazed by cattle twice per year.

Ferret Reintroduction Site 2 couples seasonal grazing (cattle are only allowed to graze pastures from approximately May 1-October 1) with a more traditional rotational grazing system and management plan produced by the Natural Resource Conservation Service (NRCS). The plan allows a lower density of livestock to graze a whole pasture for longer durations (weeks-months) before being moved into another pasture. The owner’s of FRS 2 have also further reduced stocking rates to accommodate more prairie dog grazing. Currently, they average between 7-9 hectares of pasture per cow-calf pair and rotate stock to new areas as pasture conditions warrant. Differences in grazing management systems could have effects on the productivity, composition, and physiognomy of plants in the pastures of each property (Heitschmidt and Walker 1983, Savory 1983, Briske *et al*. 2008).

The studies of barrier efficacy were conducted on three Treatment and three Control sites. Treatment sites were those in which a native grassland barrier had been established between those areas and FRS 1. Control sites were those not bounded by a grassland barrier. Each year of the study, two additional Control sites were added to the study by also investigating prairie dog dispersal on and around Ferret Reintroduction Site 2 (FRS 2)—located approximately 11 km east of FRS1. Because FRS 2 employed a different prairie dog management strategy than FRS 1, prairie dog movement back into study areas of FRS 2 could be monitored following population eradication with rodenticide (Fig. 2.3). These areas worked as Control sites, because native grassland vegetative barriers such as those surrounding FRS 1 had not been established at FRS 2.

All Treatment sites for the barrier efficacy study were located adjacent to FRS 1. These sites were called Treatment Sites 1, 2, and 3 (TS 1-3) (Figure 2.5). TS 1 is the largest Treatment site. TS 1 shares approximately 6.5 km of border with FRS 1. In 2007, prior to the start of the research, prairie dog colonization was measured as occupying 195.53 hectares of the property. Treatment site 2 (TS 2) was located at the northwest corner of FRS 1. It was the second largest Treatment site and only bordered the ferret reintroduction site at the SE corner of the Treatment site. The area colonized in 2007 was 28.68 hectares. TS 3 lies north of FRS 1. It was the smallest Treatment site, and shared approximately 0.5 kilometers of border with FRS 1. Prairie dog colonies occupied 8.58 hectares in 2007.

Control site 1 (CS 1) is located in the center of FRS 1 (Fig. 2.5). This property is not owned by the owners of FRS 1, and the prairie dogs are unwanted at CS 1. However, no native grassland vegetative barrier was ever established along the borders of the 2 properties. The length of those shared borders was about 4 kilometers. In 2007, total colonized area at CS 1 was 30.34 hectares.

The other Control sites were on/around FRS 2. However, because their prairie dog management plan did not include rodenticide application within the all of the same areas annually, Control sites 2 and 3 are different in years 1 and 2 of the study. Figure 2.6 shows the Control sites sampled in 2008 (the first year of the study).

Figure 2.5 Locations of the primary study properties.

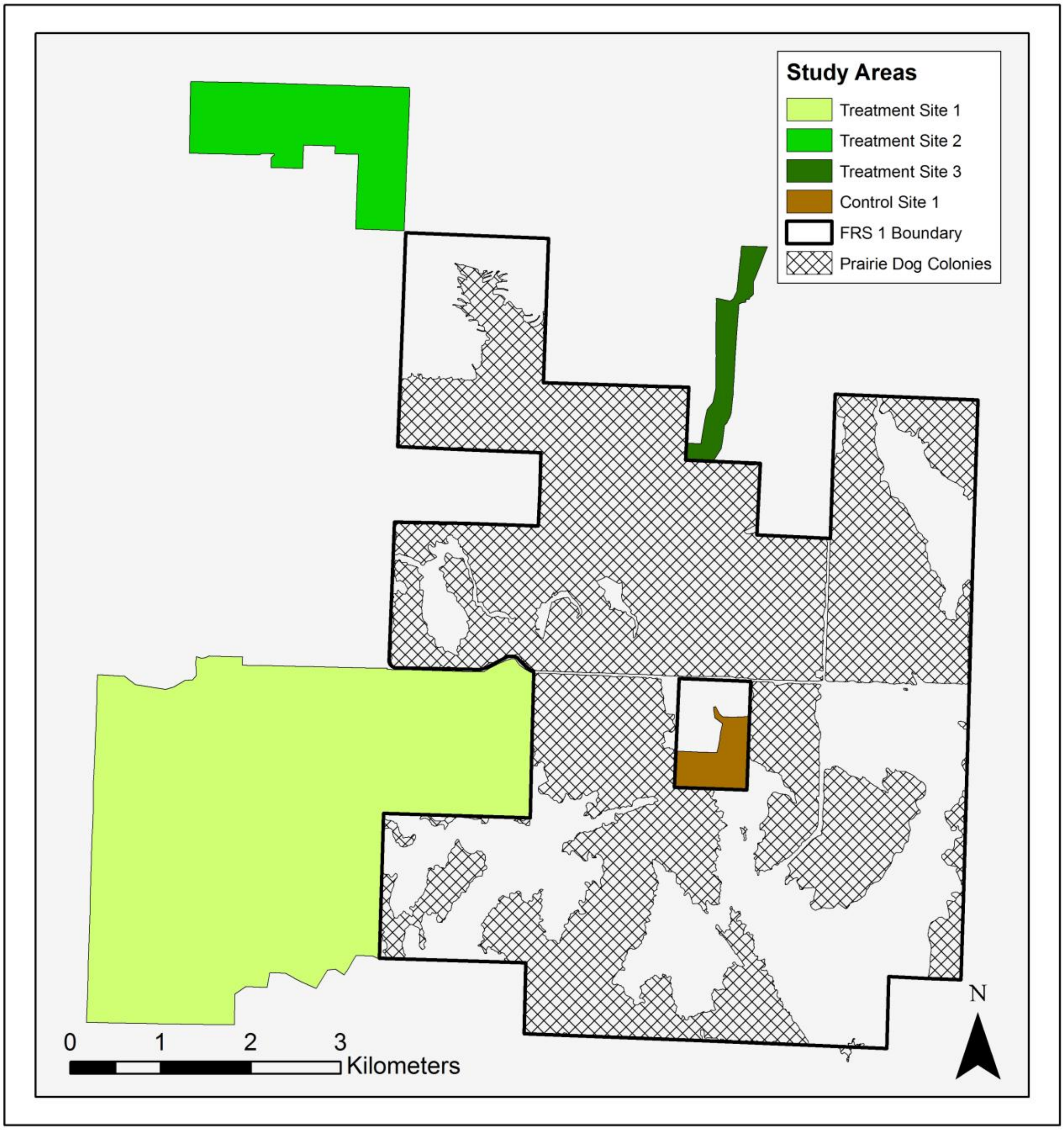
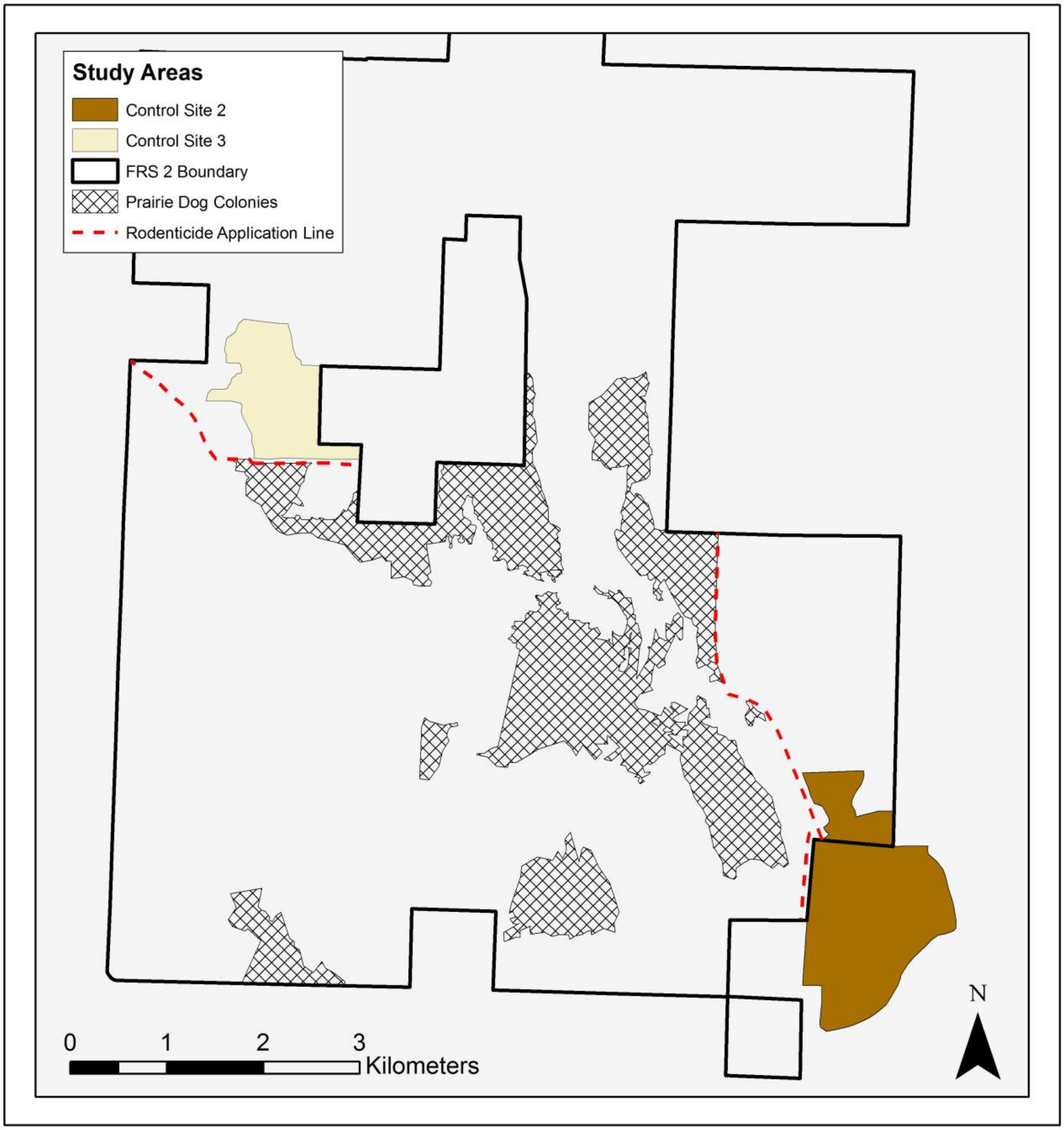
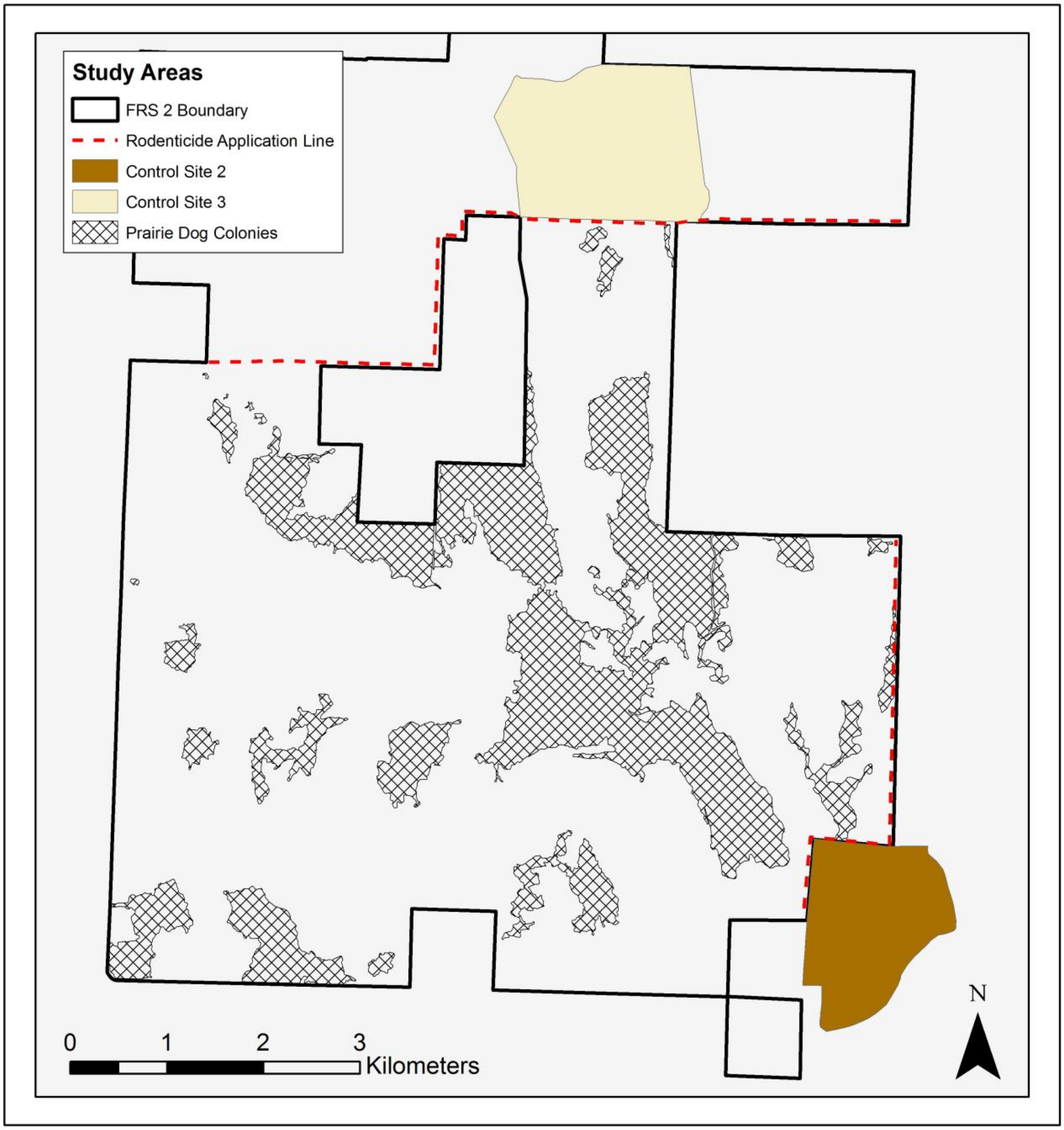


Figure 2.6 Secondary study sites in 2008 located at FRS 2 and one neighboring property. The rodenticide application lines show the boundary between the baited and unbaited areas of FRS 2. All colonies to the side of the line where each Control site was located were poisoned during the winter of 2007-2008.



In that year, samples were conducted on the north side of the fence that separates FRS 2 from the neighboring property to the south because it had been baited with rodenticide the previous winter. So during the summer of 2008, CS 2 was located on both FRS 2 and one neighboring property.

Figure 2.7 Secondary study sites in 2009 located at FRS 2 and one neighboring property. The rodenticide application lines show the boundary between the baited and unbaited areas of FRS 2. All colonies to the side of the line where each Control site was located were poisoned during the winter of 2008-2009.



That year, 57.09 hectares of prairie dog colonies were sampled. The owners of FRS 2 did not poison the bottom ground that composed the northern end of CS 2 in 2008, so during the second year of study that unextirpated area was excluded from sampling. Thus, in 2009, CS 2 was located wholly on the property of FRS 2's neighbor (Figure 2.7). The area sampled was 37.66 hectares.

Control site 3 was located in two different areas at FRS 2 during the study. In 2008, the area occupied by prairie dogs at CS 3 was 5.26 hectares (Figure 2.6). The next year, 4.12 hectares of new colonization in a different pasture was sampled (Figure 2.7).

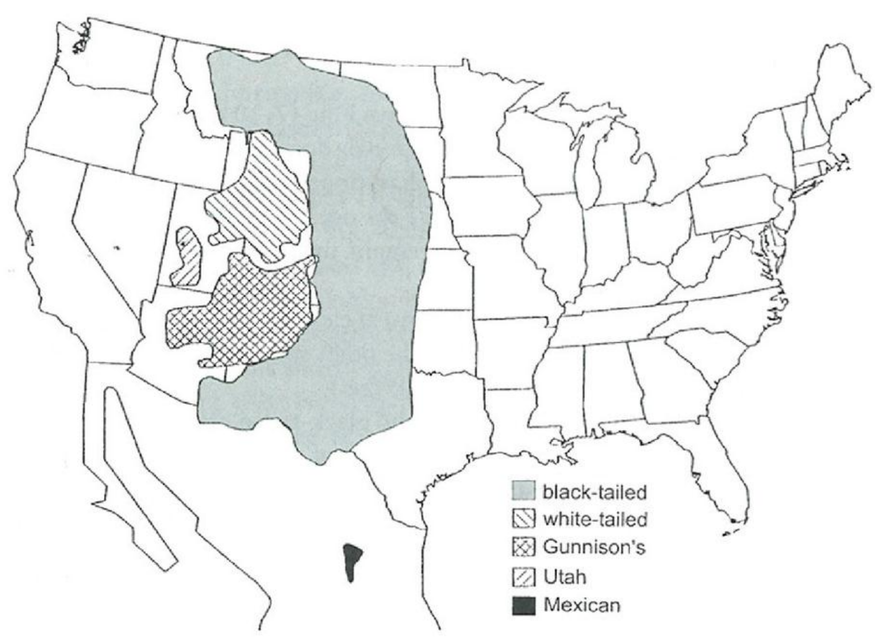
Prior to each study season, the prairie dog colonies at all sites (both Treatments and Controls) were baited with a chlorophacinone-based rodenticide. Chlorophacinone is an extremely effective toxicant (Fisher and Timm 1987) which resulted in the near-total eradication of prairie dogs from each study area. Application of rodenticide was necessary to ensure that prairie dog activity observed during the sampling periods was the result of prairie dogs dispersing away from source populations and immigrating to the study areas.

CHAPTER 3 - LITERATURE REVIEW

Prairie Dog Natural History

Prairie dogs are small, semi-fossorial rodents native to North America in the family Sciuridae. Five species of prairie dogs are currently recognized (Hoogland 1995, Hoogland 2006b). Of the five species, the black-tailed prairie dog (*Cynomys ludovicianus)* has the largest population and the widest geographic distribution (Hoogland 1995). Historically, their range covered much of the Great Plains—bounded by the Rocky Mountains in the west and the tallgrass prairie in the east. Colonies occurred throughout the mixed- and shortgrass prairies from northern Mexico to southern Canada (Figure 3.1). However, eradication campaigns, habitat loss, and outbreaks of sylvatic plague have reduced the populations to just 2% of historical estimates (Kotliar *et al* 1999, Lomolino and Smith 2001, Biggins *et al.* 2006, Facka *et al*. 2008). Current estimates of the size of the black-tailed prairie dog's range are between 500-800 thousand hectares (Biggins *et al.* 2006). This range is spread over a wide geographic area with local populations being found in areas of mixed- and shortgrass prairie from Mexico to southern Canada (Hoogland 1995).

Figure 3.1 Estimated historic ranges of the 5 prairie dog species. Adapted from Hoogland (2006b).



Black-tailed prairie dogs are social and live in colonies (King 1955, Koford 1958), with family groups, called coteries, occupying and defending their own territory within the colony (Hoogland 1995, Hoogland 2006b). Coteries are generally composed of 1 dominant male, 2-3 breeding females and 1-2 yearlings of either sex (Hoogland 2006b). These family groups live in underground burrows which can have "one to many" entrance holes on the soil surface (Hoogland 1995, Terrall 2006, 5, Verdolin *et al*. 2008).

Prairie dogs are born and reared in the burrow during the spring and do not emerge from it until May (Knowles 1985, Garrett and Franklin 1988, Hoogland 2006b). The time of juvenile first emergence is also the time when many prairie dogs choose to disperse away from their home burrows (Knowles 1985, Garrett and Franklin 1988). In most cases, dispersers are generally yearling males or older females dispersing from their natal territory (Garrett and Franklin 1988, Antolin *et al.* 2006, Hoogland 2006a).

Prairie Dog Dispersal and Colony Expansion

Prairie dog dispersal falls into two categories: intercolonial dispersal and intracolonial dispersal (which includes colony perimeter expansion). To avoid inbreeding, yearlings disperse from the natal coterie starting in the spring and lasting through the summer (Garrett and Franklin 1988). The period when intercolonial dispersal takes place lasts from May-August. These are also the months when intracolonial dispersal is at its peak—though this activity has been noted year-round (Garrett and Franklin 1988).

Dispersal and colony perimeter expansion is also the result of vegetation change—attributed to the grazing of prairie dogs and other herbivores within the colony boundaries as well as prairie dogs' tendency to forage in new grass growth on colony edges (Bonham and Lerwick 1979, Garrett and Franklin 1988, Winter *et al*. 2002). Overgrazing and vegetation clipping result in a shift in the species composition from grass-dominated to forb-dominated (O'Meilia 1976, Bonham and Lerwick 1979, Detling 2006), and the decrease in available forage for prairie dogs causes the population density of colonies to decline over time (Bevers *et al*. 1997). While intracolonial dispersal is most common, intercolonial dispersal is critical to gene flow (Antolin *et al.* 2006, Bevers *et al.* 1997, Dobson *et al*. 1997). The combination of intercolonial dispersal and colony expansion is the reason for what Antolin *et al*. call prairie dogs' "high rates of dispersal and colonization" (2006, 873).

Intercolonial dispersal is commonly done by utilizing roadways and trails or low-lying drainages (Knowles 1985, Garrett and Franklin 1988, Antolin *et al.* 2006). These landscape features must allow for faster or safer travel between colonies–increasing the dispersers' survivability. Prairie dog dispersal is nonrandom and directed towards already established colonies or areas of short vegetation and land where soils and slope will allow the successful establishment of a burrow (Cable and Timm 1987, Garrett and Franklin 1988, Antolin *et al.* 2006, Reading and Matchett 1997, Detling 2006, Hoogland 2006b, Assal and Lockwood 2007). Maximum recorded dispersal distances are approximately 6 km (Roach *et al*. 2001, Hoogland 2006a), and the average distance is 2-3 km (Garret and Franklin 1988, Roach *et al*. 2001).

The tendency of prairie dogs to disperse to areas of short vegetation is natural in that short vegetation is key for them to be able to visually identify predators, and is, therefore, key to a colony's survival (Cable and Timm 1987, Detling 2006, Hoogland 2006b). The grazing of prairie dogs and other animals within colonies and clipping of ungrazed vegetation by prairie dogs ensures short vegetation and allows clear views for predator detection. Snell and Hlavachick (1980) suggested that ungrazed vegetative buffer strips alongside a colony may limit colony expansion in the direction of the buffers.

This idea was the foundation of the deferred grazing method of prairie dog management (Cable and Timm 1987, Detling 2006). Cable and Timm (1987), working in western Nebraska on short- and mixed-grass rangelands, found that deferring cattle grazing (*i.e.* excluding cattle from grazing within prairie dog colonies) from May 1-September 1 reduced birthrates and population increases in prairie dog colonies during the second year of their study. In addition to population decreases, they also found that active colonized area decreased during the second year of the study. However, these results were not seen during the first year. They posited that the drought conditions of 1985 negated the effectiveness of deferred grazing that year. Thus, they suggest that in years with adequate precipitation, the practice of excluding cattle from pastures containing prairie dogs during the growing season may allow vegetation to grow high enough to affect the success of the colony (Snell and Hlavachick 1980, Uresk 1981, Cable and Timm 1987). Andelt (2006) agrees with this data, but suggests that the "benefits [of deferred grazing] are probably meager for many ranches on the short-grass prairies…" (Andelt 2006, 132). Tall vegetation not only causes increased predation of prairie dogs, but it can also impede dispersal by limiting visibility of the surrounding area (Cincotta *et al*. 1987, Antolin *et al.* 2006, Augustine *et al.* 2007).

Franklin and Garrett (1989) used burlap, strung between metal stakes to a height of one meter off the ground, to test whether man-made visual obstructions could reduce use of an area and limit colony expansion. In their study colony, they mapped all active burrows and estimated population densities within each cell of a 15x15 meter grid overlaid on top of the colony. They then installed the burlap barrier on the eastern side of the colony, and used a similar sized area on the western side as the Control. Their research showed that barriers creating a visual obstruction caused a marked decrease in the amount of dispersal to and use of areas that were blocked by a visual barrier of burlap. Colony expansion was also limited on the eastern margin of the colony while the western side showed an increase in colonized area. The results were similar when they replicated the study using cut and stacked Ponderosa pine trees to create visual obstructions at another site (Franklin and Garret 1989).

Since then, there has been little evidence supporting the success of visual barriers at limiting prairie dog activity (Hyngstrom 1995, Merriman *et al*. 2004, Foster-McDonald 2006, Terrall 2006). The study which most closely replicates the 27-meter wide (90 feet) ungrazed grassland barrier around FRS 1 was undertaken by David Terrall (2006) in his master's thesis at South Dakota State University. He studied the effectiveness of the vegetative buffer strips Snell and Hlavachick (1980) had earlier promoted. Terrall established vegetative buffer strips of 10, 25, and 40 meter widths alongside colonies by fencing those strips off with electric fences to exclude cattle grazing within the buffers. He collected information about the vegetation composition, height, and density within the buffer strips. He also quantified the "breakthrough" (number of newly active burrows) of prairie dogs through the buffer strips, expanding the perimeter of the colonies. Terrall used all this data to produce regression models for the effectiveness of each buffer. Like Cable and Timm (1987), Terrall's study was negatively impacted by drought during the first year of data collection, and no meaningful effect was produced by the buffer strips. However, in year 2 (with average to above average precipitation during the growing season), "breakthrough" was reduced in relation with vegetative height and density. Terrall used a modification of Robel's (1970) technique to quantify visual obstruction produced by rangeland vegetative communities. The technique will be discussed in more detail below, but the method is known as visual obstruction ranking (VOR). It is used to estimate plant biomass based on visual obstruction produced by the vegetative community. Terrall found that vegetation height should be at least 40 cm from ground level and VOR readings needed to be at least 10 cm to make a 40 meter wide barrier effective. With less vegetation height or visual obstruction, he suggested that barriers should be at least 100 meters wide.

While Terrall (2006) showed that "breakthrough" decreased in relation to increased VORs, only two sites exhibited no prairie dog "breakthrough". Those were 2 of the 40-meter width buffers. No other sites showed 100% effectiveness. Given the average VOR readings from the successful and unsuccessful sites, the author was then able to model the width of the other buffers that would be necessary to make all other study sites 100% effective at stopping "breakthrough". Terrall found that for the other buffer strips to completely stop colony expansion, those strips would need to be 85.1-103.1 meters wide.

The outgrowth of these nonlethal techniques has led to the creation of a 27 meter (90 feet) wide barrier strip of vegetation around the perimeter of FRS 1 which the study colonies surround. The barrier, which was in place at the end of summer in 2007, was made by constructing an electric fence 27 meters inside of the ranch's perimeter fences. This was done to exclude cattle from grazing in the barrier area, thereby creating a strip that was allowed to grow unimpeded. The hope was that this strip of vegetation would grow high and dense enough as to create a visual dispersal barrier and prevent re-infestation of neighboring properties by prairie dogs.

Assessing Prairie Dog Populations

There are three commonly used methods of estimating prairie dog population densities within colonized areas. Those are above-ground visual counts of prairie dogs within a defined area, the capture-mark-recapture (CMR) technique, and the use of burrow-entrance densities as an index for prairie dog abundance (Powell *et al*. 1994, Severson and Plumb 1998, Biggins *et al*. 2006). It has been shown that the CMR is the most accurate technique for estimating population densities. However, the labor required by this technique makes it impractical for many applications (Biggins *et al*. 2006).

Visual population estimations are obtained by counting the number of prairie dogs seen within a count plot of a known size over several days. Working in South Dakota in 1998, Severson and Plumb found that correlations between maximum visual counts (the most prairie dogs counted in one scan at each site) and mark-recapture estimates were the strongest. They compared the relationships between population density estimates derived from CMR, maximum aboveground visual counts, and active and total burrow density estimation techniques. They compared the results of their field studies with regression analysis using mean estimates for plot densities derived from CMR techniques as the independent variable X and maximum visual counts, mean visual counts, total active burrows with mounds, all active burrows (mounded or unmounded), and all burrows counted as the dependent variables. They found that maximum aboveground visual counts had the strongest correlation to average plot-level population density estimates obtained using CMR techniques. They used their data to create a mathematical model that can be used to refine estimates of prairie dog density. They found that the equation [X = (Y-3.04) / 0.4] can be used to match visual estimates to those obtained using the CMR technique, where Y = the maximum number of prairie dogs per hectare seen during the counts. However, this equation is only accurate when population densities from the observed maximum counts are between 5.2 and 40.8 prairie dogs per hectare prior to being incorporated in the model (Severson and Plumb 1998). Literature suggests that typical population densities are between 5-45 prairie dogs per hectare (Koford 1958, Reading *et al*. 1989, Biggins *et al*. 1993, Powell *et al*. 1994, Johnson and Collinge 2004, Biggins *et al*. 2006, USFWS 2009).

The least accurate method for assessing population densities is the use of burrow-entrances as an index for population density (Powell *et al*. 1994, Severson and Plumb 1998, Biggins *et al*. 2006). This is caused by the fact that each den may have more than one entrance on the ground surface (Hoogland 1995, Terrall 2006, Verdolin *et al*. 2008). However, this technique was chosen as one of the population density sampling methodologies employed in this study, because it is an efficient, replicable way to sample large areas. The belt transect technique for sampling burrow entrance density also allowed for expanded sampling into areas where colonization had not previously been identified by quickly digitizing additional transects with GIS software. This was important—especially prior to the first year of sampling—because it wasn't known if all previously occupied colonies would be recolonized, whether recolonized sites would cover the same extents as those measured in 2007, or whether new colonies would form in previously unoccupied territory.

When creating a technique for evaluating a location's ability to support the reintroduction of black-footed ferrets, Dean Biggins and his coauthors (1993) used a systematic sampling methodology employing strip transects to sample burrow densities within prairie dog colonies.

Their transects ran parallel through the colony and were spaced evenly at 60-meter intervals. The systematic sampling resulted in colonies of varying size to be sampled at the same proportion. Sixty-meter intervals between transects means that 5% of each colony is sampled. The researchers mounted a three-meter length piece of PVC pipe to a surveyor's wheel so that the pipe was perpendicular to the orientation of the transects. This created a three-meter wide swath as the researchers traversed each transect. They then collected the number of active and inactive burrows that fell within the swath of each transect. This allowed them to calculate the area sampled in each colony [total length of transects in meters * 3 meters (the width of the transects) = area sampled] to find the density of burrows per hectare.

They found that correlations between burrow densities and prairie dog densities were quite low, but could be strengthened by just using the density of active burrows (described as those with fresh scat on top of them) (Biggins *et al*. 1993). Their regression analyses showed that active burrow density was highly correlated to prairie dog densities in colonies of white-tailed prairie dogs (R^2= 0.95), but in colonies of black-tailed prairie dogs the data showed a lower correlation (R^2= 0.65) (Biggins *et al*. 1993).

Since this correlation is so low, many authors discount the use of burrow densities to assess prairie dog densities (King 1955, Powell *et al*. 1994, Severson *et al* 1998). However, data obtained from this technique does have some practical use (Biggins *et al*. 2006). Burrow-entrance density data derived from strip transects is the least labor-intensive of the three population density estimation techniques (Biggins *et al*. 2006). If research is being done solely to determine the presence of prairie dogs, this method can be employed with success. Also, when studying prairie dog colonies from a common geographic area (*i.e.* multiple colonies on the same property), differences in active burrow densities can indicate differences in population densities between those colonies.

Studies using belt transects to sample burrow densities have shown ranges of active burrow densities between 10 and 250 burrows per hectare (Biggins *et al*. 1993, Powell *et al*. 1994, Biggins *et al*. 2006). However, Johnson and Collinge (2004) found greater prairie dog densities and active burrow densities in their study of landscape context relationships to prairie dog colonies. Their data showed increased densities of populations and burrows was correlated to the "boundedness" of a colony by urbanization. Thus, when territory suitable for new colonization is not within the local habitat matrix of a colony (thereby restricting dispersal and colony expansion), prairie dog population densities and burrow densities will increase to levels beyond what are considered "normal".

In the end, it is up the researcher to decide which method is the most appropriate to use given the desired level of accuracy in the results as well as time and budgetary constraints. It is generally assumed that as the spatial extent of the study increases, the appropriate estimation technique also changes (Biggins *et al*. 2006).

Identifying Barrier Condition

Tall vegetation is an impediment to prairie dog dispersal (Cincotta *et al*. 1987, Antolin *et al.* 2006, Augustine *et al.* 2007). One limitation of vegetative barriers is their ineffectiveness if the vegetation does not get sufficiently tall and dense to create a visual obstruction. This will be an obvious problem in times of drought (Cable and Timm 1987, Terrall 2006). The grazing and clipping of vegetation by prairie dogs results in short vegetation within colonies (Cable and Timm 1987, Detling 2006, Hoogland 2006b). This means that if existing prairie dog colonies on the ferret reintroduction site are allowed to spread into the barrier, there will be lower vegetation height and possibly 'windows' in the barrier through which other prairie dogs may disperse. Similarly, areas where cattle can get into the barrier to graze or places that simply are not productive enough to produce tall vegetation can also be considered 'windows'.

Robel (1970) created a method to rank the degree to which a grassland community created visual obstructions and found this to be correlated to vegetative biomass within grassland communities. His technique is known as visual obstruction ranking (VOR). Working in Geary County, KS, Robel used a pole marked in one-decimeter increments. He stood this pole vertically in grassland vegetation and recorded the lowest mark that could be read on the pole from distances of 2, 3, and 4 meters from the VOR pole. At each distance, he recorded three readings off the pole. The three readings were taken from different heights: 0.5, 0.8, and 1.0 meters. Robel found that readings taken from a distance of four meters away from the pole and one meter off the ground showed the highest correlation (R^2=0.955) with biomass data obtained by clipping/weighing samples following the VOR data collection.

The VOR methodology has been adopted by many wildlife researchers as an efficient way of quantifying biomass and visual obstruction within grassland habitats (Higgins *et al*. 2005). Benkobi *et al*. (2000) modified Robel's (1970) technique by constructing a VOR pole with 2.54 cm increments to better sample visual obstruction in areas of shorter vegetation. Since they were interested in standing crop height and density, not just biomass, they also took four VOR readings (one facing each of the cardinal directions) at each sampling location. Like Robel in 1970, Benkobi and his coauthors took their readings at a distance of 4 meters from the VOR pole and with their eye line at one meter above the ground.

Benkobi *et al*.'s (2000) methodology was modified for the purposes of systematically sampling within the barrier and used to collect VOR data. With this data, a comparison can be made between the vegetation surrounding Control sites and Treatment sites—comparing visual obstruction created by vegetation at points within the barrier to determine whether barrier quality is dynamic over space/time. The VOR data can also be used to perform an inverse distance weighted (IDW) interpolation (Shepard 1968) to analyze variations within the vegetative community structure and extrapolate the VOR data throughout the barrier.

Inverse Distance Weighted Interpolation

IDW has been chosen as the interpolation technique used in this study because it adheres to Tobler's Law (Tobler 1970, Longley *et al*. 2005). He said that distance matters in spatial relationships. The closer two events are, the more they are related. The IDW function works on this premise. It allows the researcher the option to control the weight given to each sampling point—meaning that when predicting the value of an output cell, the formula assumes that local characteristics have a higher degree of influence over the value than points further away. The degree of influence (k power) can be set by the user. A k power set at 1.0 assumes that there is a constant rate of change in the influence of control points as distance from the output cell increases. A power ≥ 2.0 assumes more influence locally and a more swift loss of influence as distance increases (Chang 2006).

IDW is also an exact interpolator which means that the minimum/maximum values on the interpolated surface come from the minimum/maximum known values at each control point. The interpolated surface must pass through all control points, so the values given at each control point are the values sampled in the field (Chang 2006) and predicted values tend toward the sampled mean (Schloeder 2001).

Interpolation has been shown to be a powerful tool for natural scientists to use to make predictions of unknown values from sampled data (Robertson 1987, Liebhold *et al*. 1993)—perhaps even more accurate and useful than least squares regression (Karl 2010). Inverse distance weighted interpolation has been widely used in an array of ecological studies: modeling landscapes capable of supporting grizzly bear reintroductions (Merrill *et al*. 1999), quantifying/predicting land cover change (Wickham *et al*. 2000), mapping soil properties (Schloeder *et al*. 2001), and predicting the biomass of underwater vegetation (Valley *et al*. 2005).

In their study of aquatic vegetation, Valley *et al*. (2005) assessed the accuracy of spline, kriging, and IDW interpolation at predicting the biomass of underwater aquatic vegetation. They used a combination of sonar and field sampling to determine the biomass and height of vegetation at control points in three lakes. They next used the three interpolation techniques to predict the biomass of vegetation at un-sampled points in the lakes. Regression modeling was employed to determine the accuracy of the three interpolations—focusing on the error between predicted and actual values of biomass. The authors found that kriging was the most accurate method for lake-wide interpolations. However, at smaller scales, IDW may be the better technique to use. They ultimately suggest that "both kriging and IDW can be used to monitor both local and lake-wide effects of stressors on aquatic vegetation abundance….[while] the performance of spline interpolation was inferior" (Valley *et al*. 2005, 23).

In this case, IDW regression was chosen for two reasons. First, the primary reason for conducting the interpolation was to produce a graphic showing the VOR readings at each point sampled. This goes along with the reasoning for a systematic sampling of visual obstruction within the barrier. It was important to ensure that all parts of the barrier had been included in the samples to more accurately characterize obstruction throughout the barrier and produce graphics that were completely accurate at each sampling location. Thus, a technique that was an exact interpolator was needed to ensure that the output raster is true to the field sampled data at each sampling location. Secondly, when predicting VOR data throughout the barrier, very little emphasis on the data sampled >100 meters from a particular point needed to be placed, because at this scale, variations in microclimate and elevation may be more important factors influencing vegetation height and density than distant vegetation composition, and IDW allows the researcher to define how much weight to give distant points when determining local characteristics.

CHAPTER 4 - METHODS

To determine the effectiveness of native grassland vegetative barriers at limiting prairie dog emigration from FRS 1, three datasets were collected: a sampling of newly active burrows on FRS 1's neighboring properties, visual population density estimates at Control and Treatment properties, and samples showing the amount of visual obstruction created by the vegetation within the barrier.

During the winters prior to both sampling seasons, all active prairie dog colonies within approximately 4.8 kilometers (3 miles) of the ferret reintroduction sites were baited with chlorophacinone-based rodenticides to eradicate prairie dogs and ensure that any activity observed within the study colonies was the result of prairie dogs dispersing to the study sites during each dispersal season. The most likely source of prairie dogs immigrating to the extirpated areas was from the FRS since research has indicated it is unusual for prairie dogs to disperse more than 6 kilometers (Hoogland 2006a), with 2.4 kilometers being the average distance (Garrett and Franklin 1988). Rodenticide application was done by the Logan County (KS) Weed Department or USDA-APHIS employees to ensure that the application was conducted according to label directions and was effective. The first round of rodenticide application was done between October 2007 and March 2008. Sampling followed from April-August 2008. Following the first year of data collection, colonies to be studied the second year were then re-baited with rodenticide between October 2008 and March 2009. The second period of burrow and population density data collection occurred from April-August 2009.

Active Burrow Sampling

The study took place on six privately-owned ranches in Logan County, Kansas. The pastures all bordered large prairie dog colony complexes where the U.S. Fish and Wildlife Service has reintroduced the black-footed ferret (Figures 2.2 and 2.3). The primary use for all study sites is the seasonal grazing of cattle, and all pastures had been colonized for several years by prairie dogs until the winter of 2007-2008. Each site is dominated by shortgrass prairie plant species, and the topography makes these sites ideal prairie dog habitat.

A total of 20 colonies on three Treatment sites (those bordered by the vegetative barrier) and eight colonies on three Control sites were studied. The Control sites were locations where prairie dogs had been eradicated with rodenticide next to ferret release sites, but there was no vegetative barrier in place on either side of the fence.

To determine the effectiveness of native grassland vegetative barriers at managing prairie dog dispersal, a modified version of the protocols employed by Biggins *et al*. (1993) and Powell *et al*. (1994) to sample for the amount of burrow entrance (hereafter, "burrows") activity within a colony was employed. However, in this study a 20-meter spacing between transects was used (instead of the 60-meter spacing used in the two cited studies) resulting in sampling more of each colony. A larger sample taken in each colony should make estimates of active burrow densities more accurate. The methodology calls for the researcher to sample burrow activity along a system of transects that span the width of a prairie dog colony. This allows a person to estimate the number of active burrows per acre within a colony. It is assumed the greater the number of active burrows per acre, the greater the number of prairie dogs inhabiting the colony (Biggins *et al*. 1993). Colonies under investigation were sampled four times, from late-April until early-August, in 2008 and four times again in 2009. Each year, colonies were first sampled in late-April or early-May and repeated samples were taken at approximately four-week intervals until the four samples had been collected over the spring and summer. These sample periods correspond to the times of peak dispersal as identified by other researchers (Garrett and Franklin 1988).

Transects used to sample burrow activity were parallel, spaced evenly, 20 meters apart running the width of the colony. This design resulted in sampling approximately 15% of the total colonized area in each study colony. Transects were oriented either north-south or east-west to match the general orientation of the prairie dog colony they passed through (Figure 4.1).

Figure 4.1 Example of burrow activity transect system. This map is of CS 1 and shows transects oriented in both directions to match colony orientation.

A geographic information system (ESRI's ArcMap 9.2 [http://www.esri.com/]) was used to design transects in the office, and then they were transferred to a global positioning system (GPS) receiver. In the field, the author would start the GPS receiver and load the transect file so that the transect lines could be seen on the screen. This allowed him to traverse transects on an all-terrain vehicle (ATV) and collect the burrow data in an efficient manner. To conduct the samples, a belt transect methodology (Powell *et al*. 1994) was employed, by fixing a three-meter length of PVC pipe to the front of an ATV and using the GPS receiver to record the position of every active prairie dog burrow which fell under the width of the pipe as the transects were traversed (Figure 4.2).

Figure 4.2 This photo shows how the ATV was configured to sample for burrow activity along the belt transects. The three-meter length piece of pipe attached to the front of the ATV. The GPS is contained in a protective box strapped between the handlebars. The burrow in the foreground fell under the swath of the pipe and was marked active as indicated by the dead, clipped vegetation beside the burrow entrance.

A burrow was considered to be within the swath of the transect if at least 50% of the burrow entrance was under the PVC pipe as it passed over the burrow (Severson and Plumb 1998). When sampling burrow densities, burrows exhibiting any of the following characteristics were considered active: visually sighting a prairie dog at the burrow entrance, clipped vegetation surrounding the burrow, freshly manicured burrow mounds, or fresh scat at the burrow entrance (Powell *et al*. 1994). When sampling colonies, only the locations of newly active burrows (burrows which had not previously been indentified as active) were recorded.

To determine active burrow densities within each colony, ESRI's ArcMap 9.2 (http://www.esri.com/) GIS software was used to clip transect shapefiles by the actual dimensions of the colony polygons (which were collected following the dispersal seasons and will be discussed in greater detail below) and the sum of the lengths of all transects in each colony were calculated. The sum was then multiplied by the width of each transect (3 meters), which resulted in the number of square meters sampled. This was converted to hectares, and the number of burrows sampled in the colony during each sampling interval was divided by the sampled area. This product showed the density (# of burrows per hectare) of newly activated burrows in each colony during each round of sampling. By determining active burrow density, the data from each colony was standardized for the area sampled—allowing comparisons to be made between colonies.

The data was compared using a 2-sample *t*-test. This test compares numbers obtained from two independent samples and determines whether the datasets are significantly different from each other by comparing their means. It can be argued that active burrow entrances within a prairie dog colony are not completely independent of one another, but the test was chosen after consulting with statisticians from Kansas State University's Statistical Consulting Lab, because it was a simple, yet robust test for differences between two populations.

In cases where sampled data displays non-normality or where sample sizes are small ($n < 15$ for either population), nonparametric statistical analysis will also be conducted. The test chosen for assessing differences between two samples nonparametrically was the Wilcoxon rank-sum test (equivalent to the Mann-Whitney *U* test) or the Wilcoxon signed-rank test for matched pairs.

Four comparisons were made: Burrow densities in Treatment colonies ("Treatments") vs. burrow densities in Control colonies ("Controls") during the summer of 2008, Treatments vs. Controls during the summer of 2009, Treatments 2008 vs. Treatments 2009, and Controls 2008 vs. Controls from 2009.

The area occupied by each of the study colonies was also monitored. This began by mapping all known prairie dog colonies on four study properties (TS 1-3 and CS 1) during the winter of 2007-2008 (Figure 2.2). Study colonies were again mapped after the prairie dog dispersal seasons in 2008 and 2009 as well. Two-sample *t*-tests were used to compare the data from year to year: 2007 vs. 2008, 2008 vs. 2009, and 2007 vs. 2009. However, in this analysis, only matched samples were compared. Thus, direct comparisons of colony sizes from one year to the next were made. For example, study colony 14 at TS 1 was not mapped in 2007, but it was mapped in 2008 and 2009. Therefore, when comparing colony sizes measured in 2007 and 2008, colony 14 was excluded from testing. However, colony 14 was included in the analysis of years 2008 and 2009, because it had been mapped following those dispersal seasons.

Colonies were mapped by identifying the outermost active burrows on the colony perimeter. Once the perimeter burrows had been identified, a GPS receiver (mounted aboard an ATV) was used to collect a shapefile of each colony's spatial extent. These files were then transferred to a GIS where the area occupied by prairie dogs each year was calculated.

Visual Population Density Estimates

Visual prairie dog surveys were conducted within eight colonies (five Treatments and three Controls) to determine minimum population density estimates at each site. Two-sample *t*-tests were used to compare the population densities on Treatment and Control sites. Visual counts were conducted during the months of June, July, and August of 2008 and 2009 immediately following the 2^{nd}, 3^{rd}, and 4^{th} rounds of burrow activity transects. Treatment colonies sampled for this study were called T 1, T 2, T 3, T 4, and T 5. T 1-3 were in colonies 2, 8, and 9 at Treatment Site 1 (TS 1). T 3 was in the northwestern colony at TS 2. And T 5 was located in the southern colony of TS 3. Control colonies sampled were called C 1-3. C 1 was located in the main colony at CS 1. C 2 and C 3 were both located at FRS 2 in 2008 and 2009, however, different colonies were sampled in those years, because of the differing prairie dog management plan employed by that property's owners in each year (as explained in Chapter 2).

The methodology to visually sample the selected colonies was a modification of the methods employed by Powell *et al*. (1994) and Severson and Plumb (1998). To collect the data, sampling areas needed to be clearly defined, so that prairie dog population estimates could be standardized per area sampled, as well as allowing the study to be repeated in the same areas over two years. Counts were conducted on areas of the study colonies identified by the author using natural (yucca plants, hills and draws, etc.) and manmade (fence posts, water tanks, etc.) landmarks to delineate the perimeter of the observation area. The sizes of the study areas were calculated in a GIS environment following the creation of polygon shapefiles for each sampling area. The polygons were collected by using a GPS receiver to record the perimeters of the areas by traveling from one perimeter landmark to the next. The set-up locations that the observer would view the visual count study areas from were selected as the sites from which the study area could be seen most completely. Generally, the set-up locations were on hills looking down on the adjacent study areas.

Sampling started after a 15 minute wait following the observer's arrival at the set-up location. The 15 minute wait allowed prairie dogs time to acclimate to the observer's presence. The counting procedure was done in three intervals. Each counting interval lasted at least five minutes to ensure that colonies with low densities and those with high densities were viewed a similar length of time. During each interval, the observer would scan the observation area 3 times using 10x50 binoculars. Thus, in the 3-five minute scan intervals, a total of nine counts were recorded. Following each scan of the observation area, the researcher recorded the number of prairie dogs seen. This procedure was done at approximately the same time on three consecutive days with similar weather conditions (or as close to consecutive as weather conditions allowed). The maximum count obtained during the three days was considered to be the best estimate of population size (Severson *et al*. 1998) for that portion of the colony during that sampling period and should have represented about two-thirds of that area's total population (Biggins *et al*. 1993).

When completing the statistical tests on the visual count data, it had to first be standardized by the area sampled. Thus, the maximum count observed at each sample site during each of the 3-day rounds of counts, was divided by the number of hectares in the observation area to obtain a product that showed the number of prairie dogs per hectare. For example, following the first round of counts at Visual Site 1, a maximum of 28 prairie dogs was observed. This was divided by the area sampled (13.95 hectares), resulting in a minimum population density estimate of 2.01 prairie dogs per hectare.

Once all data had been standardized, the data obtained from Treatment colonies in 2008 and 2009 was pooled and compared to that from the Control colonies. Data was pooled to increase the sample size used in the statistical tests. Again, statistical analysis was conducted using 2-sample *t* tests and Wilcoxon rank-sum tests (where nonparametric statistics were warranted).

Also, the average number of burrows each prairie dog used in study colonies can be calculated by extrapolating burrow sampling data from the colonies where visual prairie dog counts were conducted and comparing it to data reported by Biggins *et al*. (1993). They found that in black-tailed prairie dog colonies, each prairie dog used an average of 3.9 burrow entrances.

Assuming that sampled estimates of population density were consistent outside of sampled areas in colonies, the estimated number of prairie dogs inhabiting a colony can be calculated by multiplying the density estimate with the colony's area. The number of active burrows can be similarly extrapolated by multiplying density estimates by the colonized area. Then by dividing the active burrow estimate by the population estimate, the number of burrows used by each prairie dog can be estimated.

Barrier Condition Sampling

VOR data was collected in the barrier twice during each study year. In 2008, VOR data was collected in May and November. In 2009, visual obstruction in the barriers was sampled during the months of May and October. These months were chosen to characterize barrier condition before and after the summer growing seasons. Data was collected at 200-meter intervals along borders with Treatment sites and at 100-meter intervals along borders with Controls. The higher sampling rate around Controls was done to increase the number of samples around the smaller Control sites. VOR data was collected along all shared borders with FRS 1 and up to 1.5 km past shared borders (Figure 4.3), because known prairie dog dispersal distances indicate that prairie dogs do not have to disperse through a shared border to emigrate to neighboring properties (but could instead, take a more circuitous route) (Garret and Franklin 1988, Hoogland 2006a).

To gather visual obstruction data, a modification of Benkobi *et al*.'s (2000) technique was used. They used a pole with 2.54 cm increments to more accurately rank short vegetation than the methodology employed by Robel *et al*. (1970). Like Benkobi *et al*. (2000), the pole was modified to 2.54 cm intervals and the VOR pole was viewed from a height of 1 meter and distance of four meters in the cardinal directions from the VOR pole (Figure 4.4).

Figure 4.3 This map shows the VOR sampling locations around FRS 1.

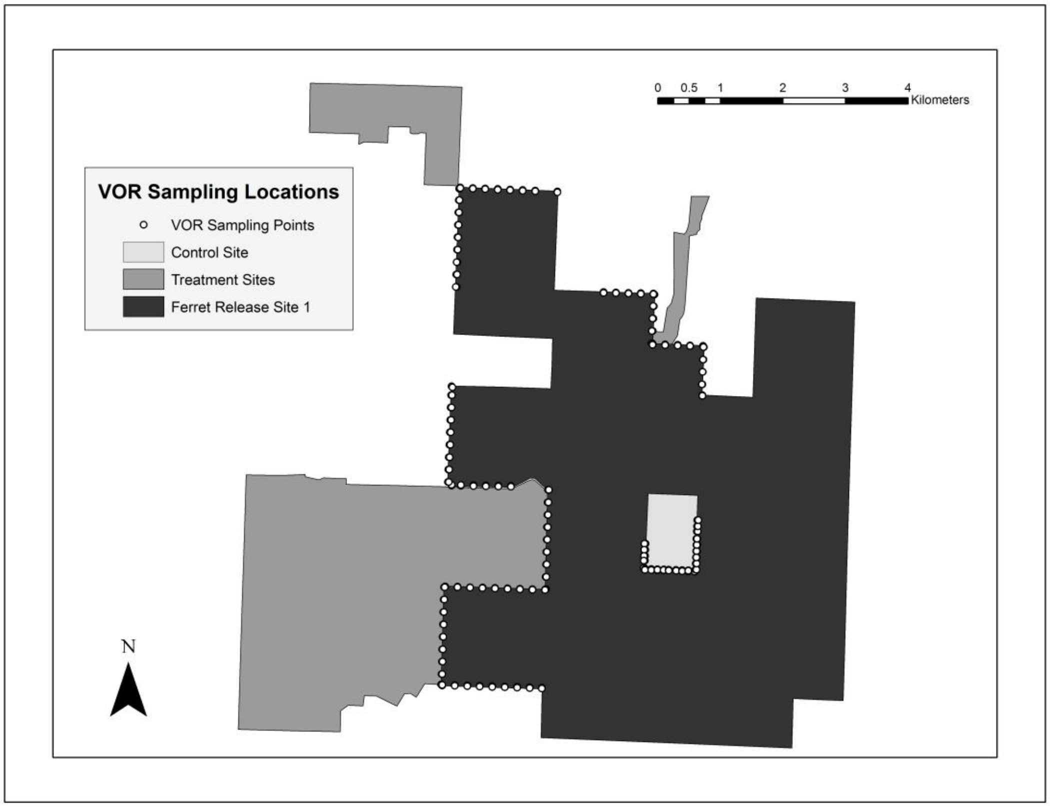
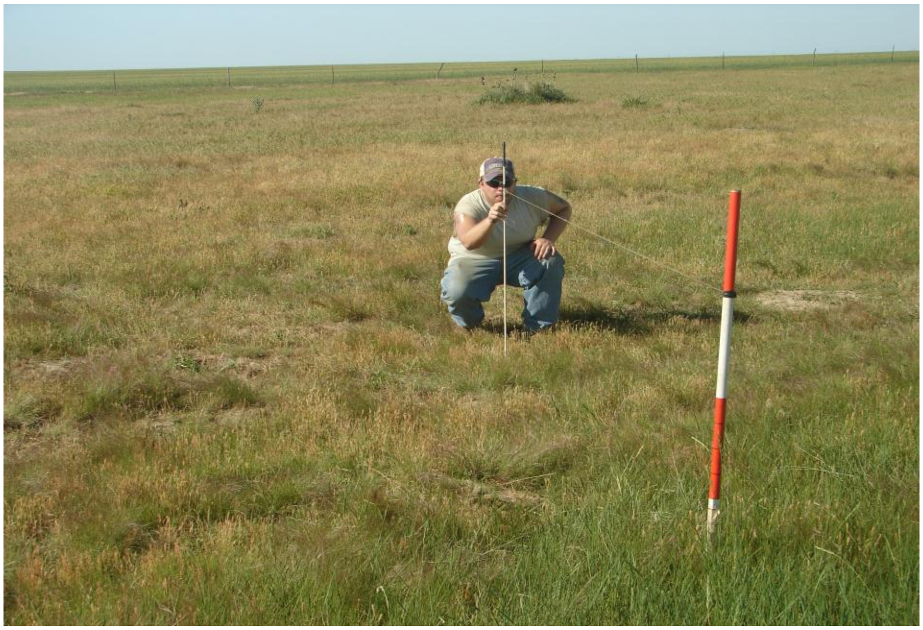


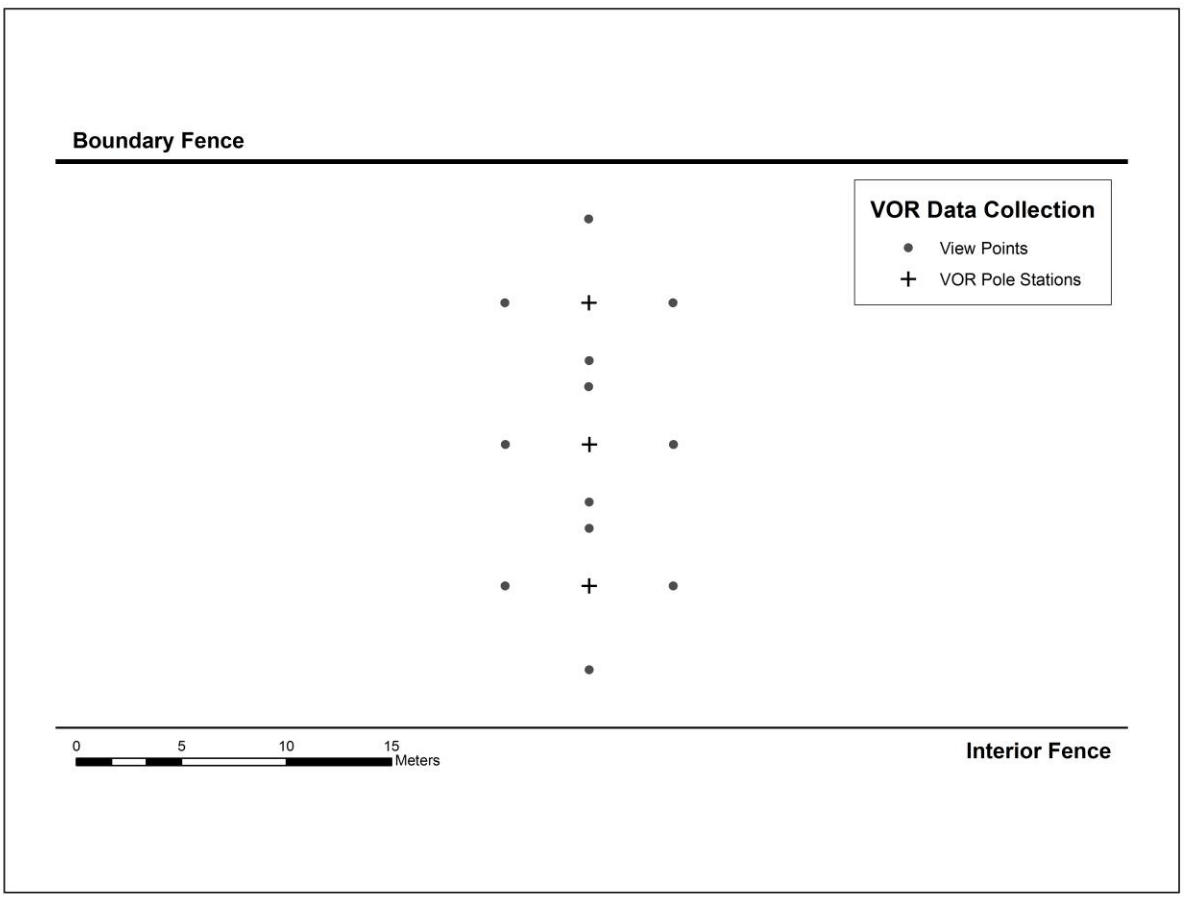
Figure 4.4 This photo shows how one person collected the VOR data. The spike attached to the VOR pole was pushed into the ground. A four-meter string was attached to the VOR pole and was tied to a plastic rod at the other end that had been marked at one meter, so the height and distance from which the VOR pole was viewed was exactly replicated at each location.



A spike was attached to the bottom of the pole allowing it to stand independently, so data could be collected by one person. Each interval on the pole was numbered. The lowest mark on the pole was one—marked 2.54 cm from the bottom of the pole. Rankings increased by one with each 2.54 cm interval up to the top of the pole. The highest rank was 36 (equal to 91.44 cm). The lowest number not obscured by vegetation was recorded as the VOR. In locations where the bottom of the pole could be clearly seen, the VOR was recorded as 0.

As stated above, a systematic sampling methodology was used. Data was collected at 100-meter intervals at borders with Controls and 200-meter intervals in the barrier along Treatments. At each sampling location, observations were made at three stations within the barrier: approximately 6.25 meters from the boundary fence, the barrier midpoint (13.5 meters from the boundary fence), and 6.25 meters from the interior fence (Figure 4.5).

Figure 4.5 Diagram of the 3 VOR pole stations at each sampling location within the barrier.



At each station, four VOR readings were taken—one facing each cardinal direction—then the height of the tallest vegetation touching the pole was recorded. The four VOR readings at each station were then averaged to determine the average visual obstruction at each station.

Once all four datasets had been collected, statistical comparisons of average visual obstruction could be compared. Using 2-sample *t*-tests, mean VOR data collected within the barrier around Treatment sites was compared to the data collected along borders with Control sites. Next, matched pairs *t*-tests were conducted on data obtained within the vegetative barrier prior to and after each growing season (May 2008 vs. November 2008 and May 2009 vs. October 2009)—indicating whether barrier condition improved during the spring and summer. Mean VOR's prior to the growth season and after the growth season were tested against each other (May 2008 vs. May 2009 and November 2008 vs. October 2009). These last two comparisons indicate whether barrier condition was similar from year to year.

Following the statistical comparison tests, a way to display the results from around FRS 1 graphically and extrapolate conditions throughout the rest of the barrier was needed. For this application, inverse distance weighted (IDW) interpolation (Shepard 1968) was chosen. ArcMap's Spatial Analyst toolbox was used to complete the interpolation.

The first step in the process was to produce a point shapefile showing the locations of all the VOR stations throughout the barriers. This shapefile was then joined to a data table containing the VOR data from one of the sampling periods. Once these steps were complete, data exploration was conducted using the Geostatistical Analyst extension in ArcMap 9.3. This step was undertaken to calculate the root mean square prediction error (RMSPE). The RMSPE is calculated during cross-validation of the IDW model in the Geostatistical Analyst. As ESRI says within their online help section, "In cross-validation, each measured point is removed and compared to the predicted value for that location" (ESRI 2008). RMSPE is a measure of the error between the actual and predicted values for each point. And it is necessary to know when conducting an IDW interpolation, because RMSPE is equal to the optimal power value (p) in the IDW algorithm. The RMSPE calculated for each sampling period were 1.677, 2.336, 2.54, and 3.567 (respectively for May '08, November '08, May '09, and October '09). The parameters input in the Geostatistical Wizard (within the Geostatistical Analyst extension) used to obtain the stated RMSPE values were a search radius was a 100 m by 100 m simple ellipse. The model incorporated a range of 1-3 neighbors within the calculation.

After obtaining optimal power values for each sampling period, the interpolation process was run. The power of the interpolation was set as the RMSPE calculated for each sampling period in the previous step, and the z-value calculated was the average VOR taken from the four readings at each VOR station. A fixed search radius was set at 100 meters (half the distance between each VOR sampling location in the barrier). The minimum number of points to use in the interpolation was 0.

Following the interpolations, output rasters were reclassified to display five classes of visual obstruction rankings. The classes were numbered 1-5. Class 1 includes mean visual obstruction of 0-4 cm from ground level. Class 2 shows mean obstruction from 4.01-8 cm. Class 3 represents 8.01-12 cm of visual obstruction. Class 4 shows 12.01-16 cm. Class 5 represents all areas where mean VORs were greater than 16.01 cm.

Aside from statistical analysis and interpolation of the VOR data, it was also used to quantify the number of VOR pole stations which exhibit a station average visual obstruction ranking of less than 3.94 (10 cm). Terrall (2006) calculated that a 40 meter wide vegetative barrier with visual obstruction of at least 10 centimeters was necessary to effectively stop prairie dog breakthrough. Since the barrier studied was approximately 27 meters wide (90 feet), it was assumed that for the barrier to be effective, visual obstruction throughout the width of the barrier must be at least the 10 cm recommended by Terrall. Thus, any VOR pole stations where the average of the four VOR readings taken was less than 10 cm were considered as areas where the barrier could not be effective at stopping dispersal.

The analysis of active burrow distances to barrier windows was conducted within a GIS environment. First the VOR pole station shapefile was joined to a data table from each of the VOR sampling periods. The tables were then queried for any records where the station average VOR was less than or equal to 3.94 (10 cm). Since the vegetative barrier studied was only about 27 meters wide, one VOR pole station (of the three at each sampling location) identified as having a station average VOR of less than 3.94 (10 cm) was assumed to make the whole barrier ineffective at that sampling location. These areas were called “windows”. The selected records and point data were then exported to individual shapefiles showing the locations of each pole station where visual obstruction was less than Terrall’s (2006) recommended 10 cm during each sampling period.

Using those "window" shapefiles, a "Near" analysis in ArcGIS of active burrow (those sampled in using burrow activity transects) distances to the closest "window" in the barrier could be conducted. It was assumed that distance to each active burrow would increase as barrier quality increased (and the number of "windows" decreased).

Burrow activity samples in each year were pooled together. Thus, all four shapefiles containing data from each sampling period in 2008 were merged to produce a single output shapefile showing every burrow sampled on Treatment sites in 2008. The process was repeated for 2009 data. Distances from burrows to windows were then calculated comparing all 2008 sampled burrows to barrier windows indentified in May 2008. The merged 2008 burrow data were then analyzed for distances to windows in November 2008. This process was repeated with the 2009 burrow and barrier window datasets.

When inputting data into the Near tool, the active burrow shapefiles were entered into the "Near Features" field and the window shapefiles were entered into the "Input Features" field. This resulted in a field being added to the attribute table of each active burrow shapefile. The field was called "Near_Distance", and it showed the distance in meters to the closest VOR pole station identified as being a window. The updated attribute tables were then exported to .dbf files, so they could be opened in a spreadsheet and further analysed. The distance data was compared using 2-sample *t*-tests to determine whether distances from burrows to windows differed during sampling periods. Statistical comparisons were made comparing distances to windows in May 2008 vs. distances in November 2008, distances in May 2009 vs. distances in October 2009, distances in May 2008 vs. those in May 2009, and distances from burrows to window locations in November 2008 vs. distances in October 2009.

CHAPTER 5 - RESULTS

Active Burrow Sampling

The density of newly active burrows was sampled over four sampling periods (at approximately 30-day intervals) during the months of late-April to early-August in 2008 and 2009. A total of 20 Treatment colonies and seven Control colonies were sampled in 2008. However, colonies 11, 13, 14, and 15 at Treatment Site 1 (TS 1) were unknown to the researcher during the first 2-3 sampling periods of 2008. Thus, during sampling periods 1 and 2 in 2008, samples were collected in 16 Treatment colonies. Three more colonies were identified prior to the third sample. In the fourth sample period a 20th colony was identified. In 2009, all Treatment colonies studied the previous year were recolonized and again included in the study. Thus, there were 20 Treatment colonies sampled and eight Controls. The increase in Control colonies was the result of changing CS 3 as described in Chapter 2.

The average number of newly activate burrows per hectare sampled in Treatment colonies during 2008 ranged from 3.14-72.73(Table 5.1). Within individual colonies, data ranged from 0-200 newly active burrows per hectare. In Control colonies the average number of newly active burrows exhibited a range between 1.6 and 134.68 during the four sampling periods. Actual densities of newly active burrows within those colonies were between 0.0 and 516.67 newly active burrows per hectare. In 2009, average new burrow density within ranged from 1.3 to 94.5 newly active burrows per hectare during sampling periods 1-4. Over the four samples, data ranges were between 0.0-300.burrows per hectare in the colonies. Control colonies averaged 7.22-56.84newly active burrows per hectare during sampling periods 1-4, with a data range of 0-135.71 newly active burrows per hectare.

Table 5.1 Univariate statistics of burrow activity data. Range and mean units are newly active burrows per hectare. N represents the number of colonies sampled.

	Sample 1			Sample 2			Sample 3			Sample 4		
Group	N	Range	Mean	N	Range	Mean	N	Range	Mean	N	Range	Mean
Controls 2008	7	0.0-5.05	1.28	7	0.0-100.0	36.25	7	7.51-400.0	140.9	7	7.98-516.67	131.7
Treatments 2008	16	0.0-30.39	3.14	16	0.0-91.3	27.21	19	0.0-200.0	69.57	20	22.84-166.67	57.68
Controls 2009	5	0.0-28.57	7.22	8	0.0-88.89	34.46	8	22.22-100.0	56.84	8	12.89-135.71	54.67
Treatments 2009	20	0.0-12.73	1.3	20	0.0-100.0	23.78	20	23.08-233.33	66.99	20	30.5-300.0	94.5

The maximum density of newly active burrows during any period sampled in 2008 was 516.67 newly active burrows per hectare. This data was obtained during sampling period 4 in colony 1 at Control Site 2 (CS 2). The colony was 1.18 hectares (of which approximately 0.18 hectares was sampled). During sample period 4 in 2008, 93 newly active burrows were recorded in the 0.18 hectares sampled. In 2009, the maximum density of newly active burrows obtained during any sampling period was 300 burrows per hectare. This data was obtained at colony 6 of TS 1. Colony 6 comprised 0.2 hectares in 2009 (of which 0.03 hectares was sampled). During sample period 4, when the maximum was observed, a total of nine newly active burrows were sampled in colony 6.

Statistical comparisons of the means derived from burrow activity sampling were made in the following groupings: newly active burrows/ha in Treatment colonies vs. newly active burrows/ha in Control colonies during each of the sampling periods (four samples in each year), pooled data from each of the sampling periods in Treatment colonies vs. that from Controls during each year (this was done to increase sample sizes of Treatments and Controls), and all data collected in 2008 was pooled and compared all the data collected in 2009 (to determine whether burrow activity different between years).

In 2008, comparisons of Treatments and Controls during each sampling period showed no statistically significant difference in the number of active burrows sampled as indicated by the resulting P-values (Table 5.2).

Table 5.2 Results of burrow activity comparisons for each sampling period in 2008 and 2009.

Year	Sample	Control Mean	Treatment Mean	*t* Value	P-value
2008	1	1.27857	3.13938	0.886663	0.3861
	2	36.2486	27.2106	-0.52069	0.6169
	3	140.919	69.575	-1.19882	0.2733
	4	131.669	57.681	-1.072	0.3241
2009	1	7.222	1.302	-1.07974	0.3392
	2	34.4638	23.777	-0.84158	0.4169
	3	56.84	66.987	0.72305	0.4768
	4	54.67	94.5	1.964061	0.0623

The values from each of the four comparisons were greater than 0.05 which means the null hypothesis of no difference between Controls and Treatments is not rejected. P-values from the same comparisons for the data obtained in 2009 were similar in that none of the values were less than the critical value of 0.05. Thus, there was no statistically significant difference in burrow activity at Treatment and Control colonies in 2009 (Table 5.2). During each of the eight burrow activity sampling periods there was no difference in the number of newly activated burrows per hectare in Control and Treatment colonies.

Since there were fewer than 15 Control colonies in any of those comparisons, and because the datasets were not normally distributed, nonparametric statistics were also used to validate the conclusions obtained from the two-sample *t*-tests. P-values resulting from Wilcoxon rank-sum tests between burrow activity within Controls and Treatments during each sampling period in 2008 ranged between 0.6857 and 1.0. For the 2009 comparisons, P-values were between 0.063-0.647. All P-values agree with the results of the parametric analyses and suggest no significant difference in burrow activity between Treatment sites and Control sites.

Data collected in Controls and Treatments were then pooled to compare the overall burrow activity results for each of the two years of study. Again, there was no significant difference between burrow activity in either of the comparisons (Table 5.3).

Table 5.3 Results of pooled burrow activity comparisons.

Year	Control Mean	Treatment Mean	*t* Value	P-value
2008	77.5286	41.7062	-1.43408	0.1622
2009	41.5138	46.6415	0.576519	0.5659

These two comparisons were more robust than those comparing Controls and Treatments during each sample period, because the pooled comparisons had larger samples for each of the populations (Controls and Treatments). Sample sizes for pooled comparisons in 2008 were 71 and 28 samples, respectively for Treatments and Controls. In 2009, pooled comparisons had 80 samples in the Treatment class and 29 samples from Control colonies. However, the datasets were not normally distributed (Figures 5.1 and 5.2), so nonparametric tests were also conducted. The P-values resulting from the Wilcoxon rank-sum analyses were 0.795 and 0.689, respectively for 2008 and 2009. Thus, the results obtained from these tests backup and reaffirm the previous *t*-test results as well as the analyses conducted on the data from each sampling period.

Figure 5.1 Box plots and histograms of pooled burrow activity data from 2008.

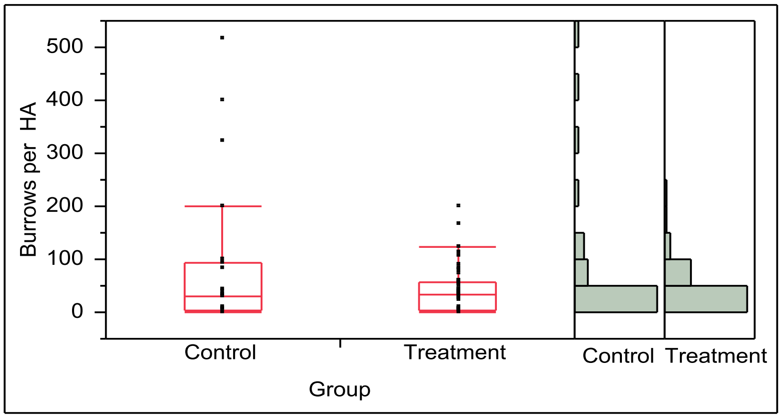
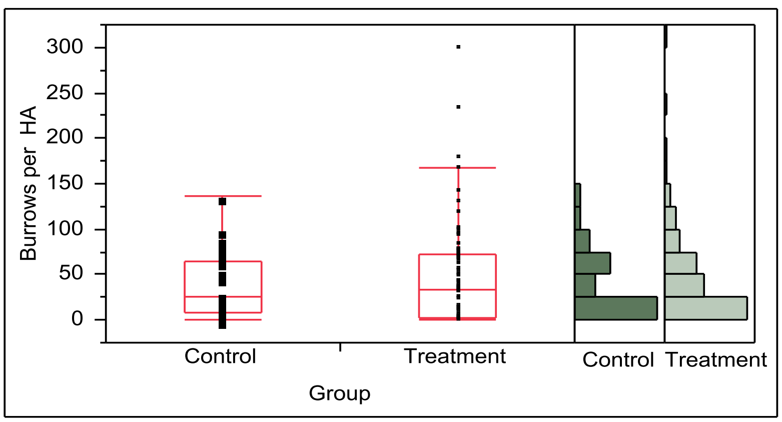


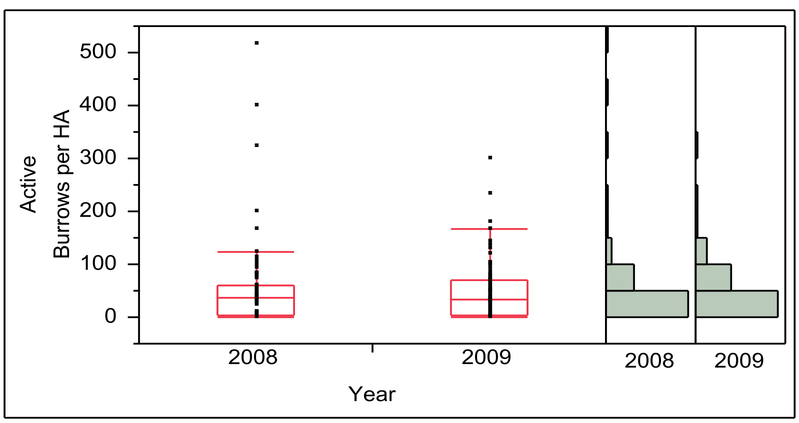
Figure 5.2 Box plots and histograms of pooled burrow activity data from 2009.



The statistical tests comparing Controls to Treatments indicated that there was no difference in newly activated burrow densities between either of the groups during 2008 and 2009. These results provide evidence that the native grassland vegetative barrier surrounding FRS 1 was not effective at limiting prairie dog emigration from the property. At the 95% confidence interval, only one comparison of 10 approached significance. The *t*-test of Controls vs. Treatments from the 4th sampling period in 2009 results in a P-value of 0.0623 which approaches the critical value 0.05. Nonparametric analysis of the data from the same period also resulted in a P-value of just over 0.06. However, in that sample, the difference in means between the 2 populations is just under 40 burrows per hectare, and Treatment colonies exhibit the higher mean density. Thus, in the only comparison approaching a statistically significant difference, it is a higher density of active burrows on Treatment colonies which causes the difference—the opposite of the intended effect of vegetative barriers.

A 2-sample *t*-test was also done to compare all data collected in 2008 to that of 2009. The analysis was done to determine whether the density of burrows sampled was similar in both years of study. The mean density of newly active burrows sampled in 2008 was approximately 51.84 burrows per hectare sampled. The 2009 mean was 45.28 burrows per hectare sampled (Figure 5.3). The *t*-test resulted in a *t*-value of -0.713 and a P-value of 0.4768. Nonparametric analysis yielded a P-value of 0.9594. This indicates there was no statistically significant difference between newly active burrow densities in 2008 and 2009. Since active burrow density was used as a proxy for prairie dog population density, it can be assumed that if colonized acreage was also stable in both years, similar numbers of prairie dogs dispersed to and reoccupied the study colonies in 2008 and 2009.

Figure 5.3 Box plots and histograms of all data collected in 2008 compared to data from 2009.



To determine whether the area recolonized by immigrant prairie dogs was similar from year to year, matched pairs *t*-tests were carried out. Three comparisons were made: 2007 area vs. 2008 area, 2008 area vs. 2009 area, and 2007 area vs. 2009 area. Again, a 95% confidence interval was used, so P-values less than 0.05 denote statistically significant differences between samples.

The tests indicated one significant difference in colony sizes between years (Table 5.4). The general increase in colony sizes between 2007 and 2008 is statistically significant (P = 0.0061). Analysis did not suggest a significant difference between area colonized in 2008 and 2009 or between colony sizes when testing 2007 area vs. 2009 area.

Table 5.4 Results of matched pair comparisons of colony size. (Sample sizes are listed under "N". Since these were matched pairs, the sample size from each population is half of the number given under "N". "Mean 1" represents the 1st variable in the pairing, while "Mean 2" gives the average of the 2nd variable. * denotes significance).

Pairing	N	Mean 1	Mean 2	P-value
2007 area vs. 2008 area	38	15.5161	22.7889	0.0061*
2008 area vs. 2009 area	46	20.717	20.3943	0.9080
2007 area vs. 2009 area	34	16.4288	23.7312	0.1756

None of the datasets were normally distributed, and Wilcoxon signed-rank analysis was chosen to perform nonparametric analysis. The results of those 3 tests were P-values of < 0.0001, 0.7572, and 0.0267 respectively for the pairings 2007-2008, 2008-2009, and 2007-2009. The results differ from the parametric analysis results for the final pairing. While *t*-testing did not indicate a significant difference (at the 95% confidence interval) between colony sizes in 2007 or 2009, Wilcoxon signed-rank analysis did suggest that those samples were significantly different.

Statistical analysis did not indicate a significant difference in dispersal and reoccupation of extirpated colonies between 2008 and 2009. It is assumed that each year a similar number of prairie dogs dispersed from the FRS to each of the study areas (as indicated by no significant differences in newly active burrow densities or area colonized). However, while colony sizes made a significant increase in 2008 and possibly in 2009, there is no way to determine whether colony expansion (following the construction of the barrier fences in 2007) was caused by an increased number of immigrant prairie dogs, because there had been no burrow activity data collected prior to the summer of 2008. In this case, colony size is only an important indicator of prairie dog activity in relation to the number of active burrows within the colony.

Visual Population Density Estimates

Visual surveys to estimate population density (prairie dogs per hectare) at a sub-sample of study colonies was undertaken to further investigate the effects of native grassland barriers at limiting prairie dog dispersal away from FRS 1. Surveys were conducted during three sampling periods in the 2008 dispersal period and again in 2009. The maximum number of prairie dogs counted at each sample colony during each sampling period was used to determine the colony's minimum population density estimate for each period. Once density estimates had been calculated, two-sample *t*-tests were used to investigate differences in population densities at Treatment and Control colonies. Four comparisons were made. First, all density estimates from Control colonies in 2008 were pooled and compared to all density estimates from 2008 Treatment colonies. The same comparison was tested used data from 2009. Control data from 2008 and 2009 was pooled and compared to all Treatment data from 2008 and 2009. Finally, all 2008 data was pooled and compared to all 2009 data to test for differences between overall population densities between years.

Overall, visual surveys yielded a range of minimum population density estimates between 0.34-18.84 prairie dogs per colonized hectare (Table 5.5). Average population density in colonies sampled was 5.21 prairie dogs per hectare.

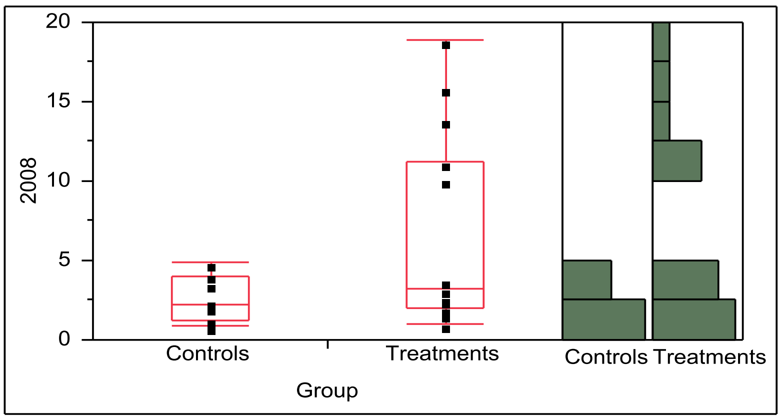
Table 5.5 Results of visual population density surveys.

	2008			2009		
Plot	1st Sample	2nd Sample	3rd Sample	1st Sample	2nd Sample	3rd Sample
T 1	2.01	1.65	2.58	3.23	3.23	1.72
T 2	1.78	0.97	2.01	0.4	0.75	0.34
T 3	15.94	10.14	18.84	12.32	10.87	7.25
T 4	3.27	2.72	3.81	2.59	3.81	2.59
T 5	10.11	11.24	13.86	10.49	11.24	13.48
C 1	3.61	4.08	4.86	4.86	5.02	5.33
C 2	N/A	2.11	2.39	1.08	1.62	2.16
C 3	0.94	1.18	1.3	3.77	7.55	7.55

In 2008, Treatment densities ranged from 0.97-18.84 dogs per hectare (mean = 6.73). Control colony densities ranged between 0.94-4.86 prairie dogs per hectare. The mean density sampled in 2008 Control colonies was 2.56 dogs/ha. The minimum population density observed was recorded during the first sampling period at C 3 (located at FRS 2). The maximum density observed was recorded during sample period 3 in T 3. Statistical analysis of these data resulted in a *t* value of 2.544—giving a P-value of 0.021. Those results indicate a statistically significant difference between population densities at Control and Treatment colonies. However, mean density is higher within Treatment colonies—suggesting vegetative barriers were ineffective in 2008.

The data was also analyzed with nonparametric statistics because sampled population densities were not normally distributed (Figure 5.4). Wilcoxon sum-rank testing did not indicate a significant difference between each of the samples (P = 0.1651).

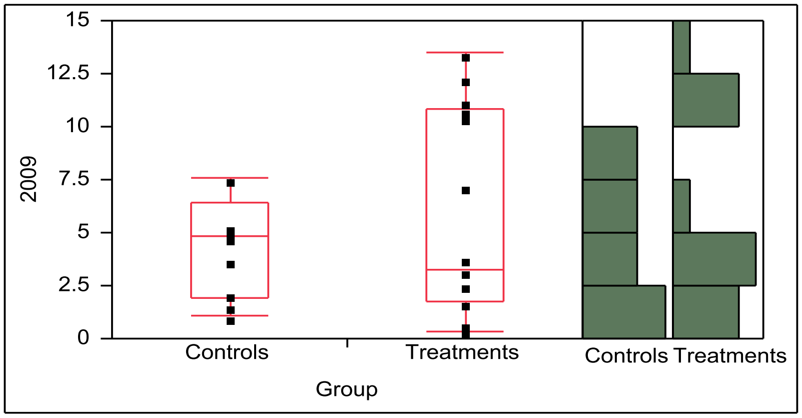
Figure 5.4 Box plots and histograms of the distribution of population density data from 2008.



In 2009, Treatment densities ranged from 0.34-13.48 prairie dogs per hectare. The mean population density was 5.62 dogs/ha. Within Control colonies, the range of densities observed was between 1.08 and 7.55 prairie dogs per hectare (mean = 4.33). During the 2nd year of study, both the minimum and maximum population densities were recorded in the same sampling period. In the final sample period of 2009, 6 prairie dogs were counted at T 2 (colony 8 at TS 1). The count resulted in a density estimate of 0.34 dogs per hectare. During the same period, 36 prairie dogs were observed in plot T 5 (southern colony at TS 3)—a density of 13.48 dogs per hectare. Two-sample *t*-testing yielded a *t* value of 0.882 and a P-value of 0.3874. There was no statistically significant difference between population densities in Controls and Treatments.

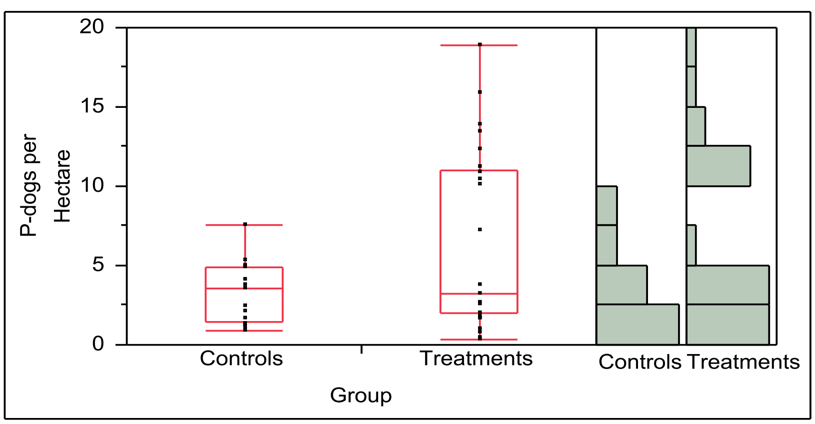
However, the datasets were not normally distributed (Figure 5.5), so Wilcoxon sum-rank analysis was used to further explore the data. The P-value from the test was 0.9050, agreeing with the *t*-test result suggesting no significant difference between population densities on Control and Treatment sites in 2009.

Figure 5.5 Box plots and histograms of population density data from 2009



Control and Treatment data from both years of study were then pooled, and an overall Treatment vs. Control density comparison was made. This third test was conducted to increase each group's sample size, thereby increasing the robustness of the statistical test. The pooled Treatment mean density was 6.17 prairie dogs per hectare. The group mean from Control plots was 3.49 prairie dogs per hectare (Figure 5.6). The analysis gave a *t* value of 2.414 and a P-value of 0.0202. However, nonparametric analysis did not indicate a statistically significant difference between Control and Treatment densities (P = 0.2828).

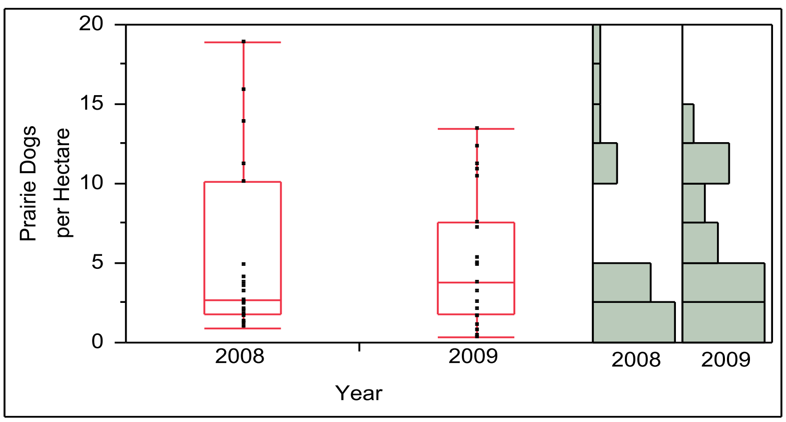
Figure 5.6 Box plots and histograms of the pooled population densities in Control and Treatment colonies.



The significant P-value in one of the overall population density comparisons suggests that population densities in Control colonies was significantly different than those in Treatment colonies. However, here again, the Treatment group exhibits the higher mean population density—indicating that recolonization of those areas was unimpeded by vegetative barriers. Furthermore, nonparametric analysis did not support the results of *t*-test calling into question whether the difference in population density was significant.

The final statistical analysis, comparing all 2008 data to all 2009 data, showed that the mean population density derived from 2008 estimates was 5.28 prairie dogs per hectare, while the 2009 mean was 5.14 dogs/ha (Figure 5.7).

Figure 5.7 Box plots and histograms showing the distribution of all population density data collected in 2008 and 2009.



The test resulted in a *t* value of -0.104 and a P-value of 0.9177—a highly insignificant difference in means. The result of the *t*-test was corroborated by Wilcoxon rank-sum analysis (P = 0.7095) These results indicate that population densities in study colonies were similar between years of study.

The average number of burrows each prairie dog used in study colonies was estimated by extrapolating burrow sampling data from the colonies where visual prairie dog counts were conducted and compared to data reported by Biggins *et al*. (1993). They found that in black-tailed prairie dog colonies, each prairie dog used an average of 3.9 burrow entrances. In 2008, calculations of burrow use (# active burrows per prairie dog) ranged between 1.46-216.18 active burrows per prairie dog (Table 5.6). The mean was 28.99 active burrows per prairie dog. The following summer (2009), active burrows per prairie dog ranged from 0-93.24 with an average of 15.38 burrows per individual.

Table 5.6 Estimated number of active burrows per known prairie dog following visual population estimates.

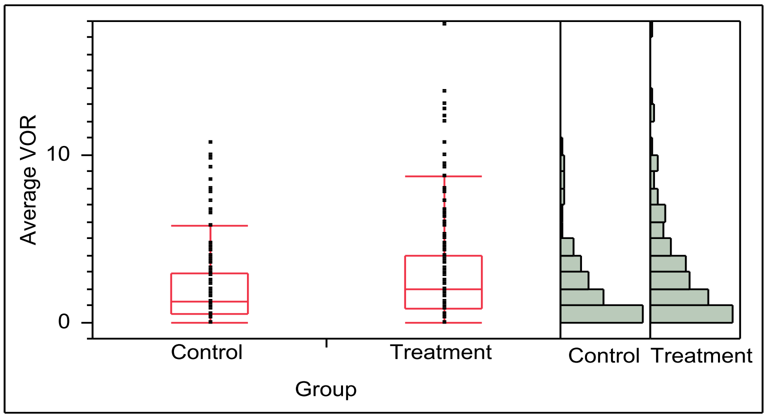
	2008			2009		
Site	Sample 1	Sample 2	Sample 3	Sample 1	Sample 2	Sample 3
TS 1	2.77	33.81	15.18	4.04	10.99	31.80
TS 2	3.74	38.48	19.07	26.13	34.37	93.24
TS 3	1.46	10.55	3.09	2.94	7.24	12.91
TS 4	3.31	15.05	8.83	0.63	9.25	18.99
TS 5	4.10	5.02	5.68	1.56	4.85	10.52
CS 1	11.02	10.40	7.10	2.83	4.56	12.32
CS 2	N/A	152.71	216.18	21.01	33.20	5.97
CS 3	6.91	69.34	22.98	0.00	13.25	6.62

Barrier Condition Sampling

Visual obstruction rank measurements were conducted within the vegetative barriers before and after the summer growing seasons in 2008 and 2009. There were 88 sampling locations within the vegetative barrier surrounding Treatment sites. At each sampling location there were three VOR pole stations; totaling 264 VOR samples collected during each sampling period. In 2008, there were 59 sampling locations along borders with Control sites—for a total of 177 visual obstruction ranking samples. When Control sites had to be changed at FRS 2 in 2009, it caused a change to one of the Control VOR sampling sites as well. Thus, in the second year of study, 63 sampling locations (189 VOR pole stations) were established along borders with Controls.

During the first sampling period in May 2008, Treatment mean VOR's calculated at each VOR pole station ranged from 0-17.75 (0-45.09 cm). The mean ranking from all Treatment VOR station averages was approximately 2.87 (7.29 cm), and the median was 2 (5.08 cm). Along borders with Control sites, the data ranged from 0-10.75 (0-27.31 cm). The mean of the station averages was approximately 2.03 (5.16 cm), while the median ranking along Control borders was 1.25 (3.18 cm). Two-sample *t*-testing yielded at *t* value of 3.368 and a P-value of 0.0008. The hypothesis test is highly significant. However, the distribution of the data is again non-normal (Figure 5.8). Nonparametric analysis of visual obstruction sampled along the borders of Control and Treatment sites resulted in a P-value of 0.0022, suggesting that as a whole, visual obstruction was higher within the vegetative barrier than along Control boundaries.

Figure 5.8 Boxplots and histograms of the visual obstruction samples collected in May 2008.



In the second visual obstruction sampling period (November 2008), pole station VOR averages ranged from 0-18.25 (0-46.36 cm) within the vegetative barrier. The mean and median rankings were approximately 3.54 (8.99 cm) and 2.75 (6.99 cm), respectively. Obstruction caused by vegetation along borders with Control sites ranged from 0-10.5 (0-26.67 cm). The mean of all pole station averages at Control site boundaries was 1.8 (4.57 cm) and the median was 1 (2.54 cm) (Figure 5.9). Both parametric and nonparametric statistical analysis of these data resulted in P-values of < 0.0001. This clearly indicates that following the growing season of 2008 visual obstruction created by vegetation height and density was significantly higher in the native grassland barrier than along boundaries with Control sites.

In May 2009 the spring barrier condition samples were collected. Within the barrier along Treatment borders VOR pole station ranking averages ranged between 0 and 16.5 (0-41.91 cm). The mean was approximately 4.52 (11.48 cm) and the median VOR was 3.75 (9.53 cm). At Control borders, VOR averages ranged from 0 to 11.75 (0-29.85 cm). The mean of was approximately 2.65 (6.73 cm) and the median was 2.25 (5.72 cm) (Figure 5.10). When comparing VOR's within the barrier and along Control sites for this sample, the t value was 8.014, and the P-value obtained was less than 0.0001. Wilcoxon sum-rank analysis also resulted in a P-value of < 0.0001. Therefore, at the beginning of the second year of study, visual obstruction was again higher within the barrier than along Control boundaries.

Figure 5.9 Boxplots and histograms of the visual obstruction samples collected in November 2008.

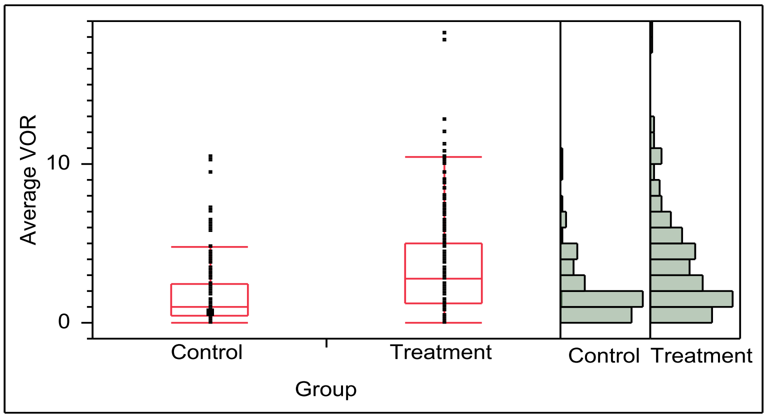
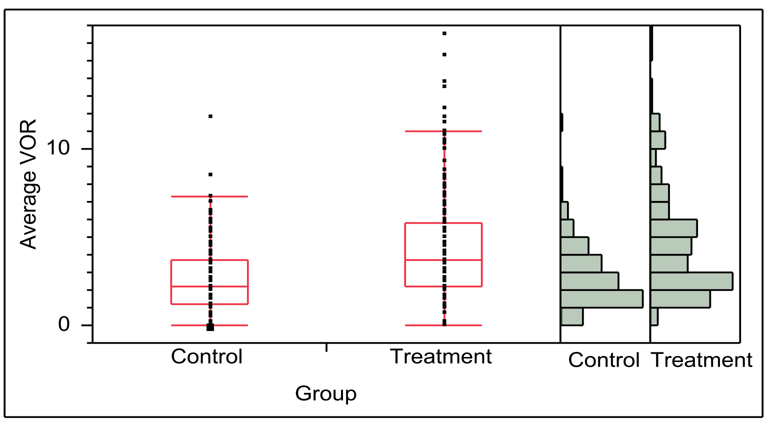
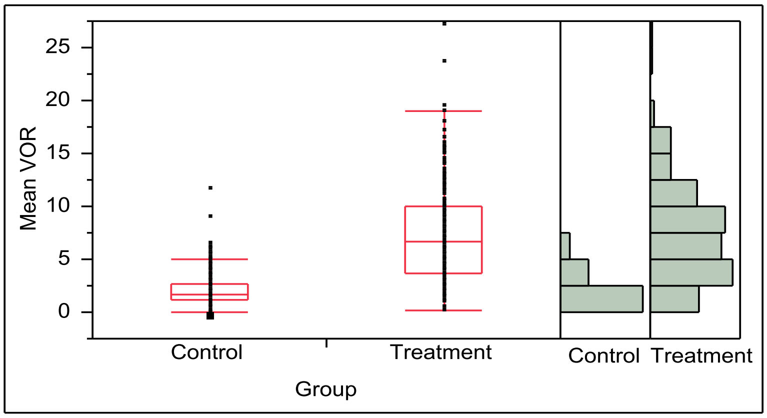


Figure 5.10 Boxplots and histograms of the visual obstruction samples collected in May 2009.



The final sampling of visual obstruction along borders with study sites occurred in October 2009. Rankings from pole stations within the barrier adjacent to Treatment sites ranged from 0.25-27.25 (0.64-69.22 cm). The mean and median calculated from pole station ranking averages were 7.26 (18.44 cm) and 6.75 (17.15 cm), respectively. Along Control boundaries the station averages ranged between 0 and 11.75 (0-29.85 cm). The mean ranking average was approximately 2.23 (5.66 cm) and the median was 1.75 (4.45 cm) (Figure 5.11).

Figure 5.11 Boxplots and histograms of the visual obstruction samples collected in October 2009.



Statistical analysis of these two groups yielded a *t* value of 16.561 and a P-value of less than 0.0001. While the Treatment data from this period approached normality, Control data was skewed, so Wilcoxon sum-rank analysis was included for this comparison as well. The P-value obtained from that analysis was also less than 0.0001. As with the previous three comparisons, during the final round of data collection visual obstruction was significantly higher within the barrier than along borders with Control sites.

Overall, the design of the vegetative barrier did result in taller, more dense vegetation than what surrounded Control sites (Table 5.7).

Table 5.7 Univariate statistics of Visual Obstruction Rank (VOR) data. Values are rankings of visual obstruction calculated from the average of data collected at each VOR pole station.

	Treatment			Control		
Sample	Range	Mean	Median	Range	Mean	Median
May 2008	0-17.75	2.87	2.00	0-10.75	2.03	1.25
November 2008	0-18.25	3.54	2.75	0-10.50	1.80	1.00
May 2009	0-16.50	4.52	3.75	0-11.75	2.65	2.25
October 2009	0.25-27.25	7.26	6.75	0-11.75	2.23	1.75

Morever, the univariate data within the table shows that during each growing season, visual obstruction increased through the summer within the native grassland barriers, while along borders with Control sites, visual obstruction rankings decreased from spring to fall. This suggests that if vegetative barriers were effective at limiting dispersal, they would be more effective toward the end of the dispersal season—as visual obstruction increases.

Further analysis indicated whether the increase in vegetation height and density was statistically significant during each year of study—by comparing the spring VOR data to that collected in the fall. Matched pairs *t*-tests of VOR data obtained within the vegetative barrier was done comparing VOR data from the spring of 2008 to that from fall 2008 and spring 2009 VOR data to fall 2009 data. Wilcoxon signed-rank analysis was used to make nonparametric matched pairs comparisons, because none of the datasets were normally distributed.

The matched pairs *t*-test of 2008 data yielded a *t* value of 3.455 and a P-value of 0.0006. Wilcoxon testing gave a P-value of 0.0008. The tests comparing data from 2009 resulted in a *t* value of 11.17 and a P-value of less than 0.0001, while the nonparametric analysis yielded a P-value of less than 0.0001. All tests were highly significant and indicating barrier condition improved during each growing season.

Knowing that barrier condition improved from spring to fall in 2008, means that it stands to reason that barrier condition was better at the start of the 2009 growing season than it was during the beginning of the 2008 season. A matched pairs *t*-test was used to determine the veracity of that assumption. Again working from a standard null hypothesis, the analysis was conducted, and resulted in a *t* value of 8.756. Hypothesis testing yielded a P-value of less than 0.0001—well below the critical value of 0.05. Thus, the samples are significantly different. However, because the data was not normally distributed, the parametric results were corroborated with a Wilcoxon signed-rank test which also gave a P-value of < 0.0001.

The previous test also suggests that improvement in barrier condition during the 2009 growing season would result in greater visual obstruction created by vegetation within the barrier in fall 2009 than in fall 2008. Parametric and nonparametric tests comparing visual obstruction at the end of each growing season both resulted in P-values of less than 0.0001 So again, there was a significant difference in barrier condition between years.

Statistical analysis indicated that visual obstruction was greater within the barrier than along borders with Control sites during each sample period. Moreover, barrier condition improved during each growing season, as well as improving overall from 2008 to 2009. However, it seems as if the vegetative barrier had no effect at limiting prairie dog dispersal from

FRS 1 to surrounding properties--indicating there were places within the barrier in which vegetation did not provide adequate cover to create a visual deterrent to emigration.

Inverse distance weighted interpolation was used to analyze VOR data from each sampling period and produce graphics to help visualize the condition of the barrier around FRS 1 (Figures 5.1-5.4). Data was categorized to aid in the visualization. Five data classes were created based on visual obstruction heights measured in centimeters. In each map, Classes 4 and 5 represent the best areas within the barrier. Class 3 is a marginal class representing visual obstruction between 8 and 12 centimeters high. Classes 1 and 2 are areas of poor barrier condition and were defined as “windows” through the barrier.

Figure 5.1 Results of IDW interpolation on May 2008 VOR data. (Note on Figures 5.1-5.4: Interpolation results are only valid for the 27-meter wide barrier as those are the only portions ungrazed by cattle. The interpolation results are shown stretching outside of the barrier only to aid visualization. Also, no barrier was in place around CS 1 in the center of FRS 1. However, interpolation results around that property are also shown to highlight the differences between visual obstruction within the vegetative barrier and around Control sites where there was no vegetative barrier.)

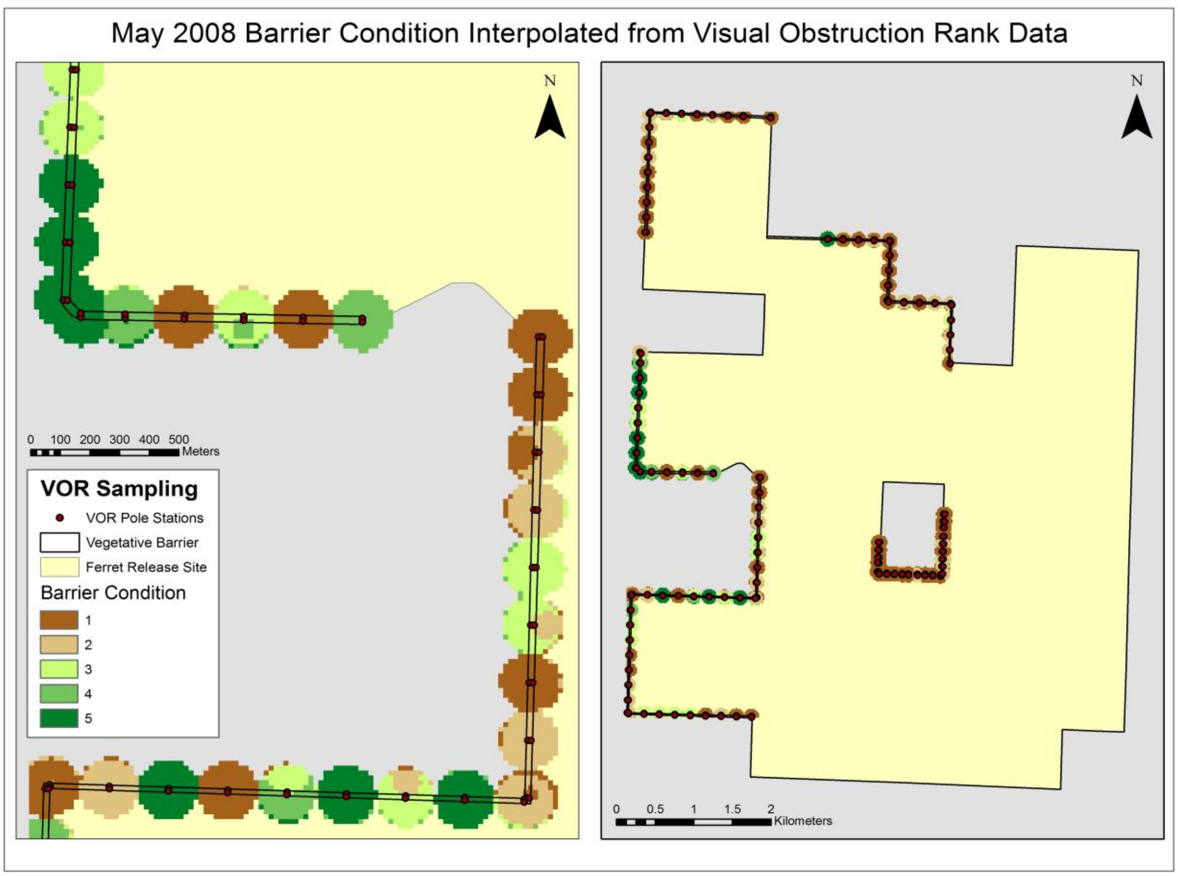


Figure 5.2 Results of IDW interpolation on November 2008 VOR data.

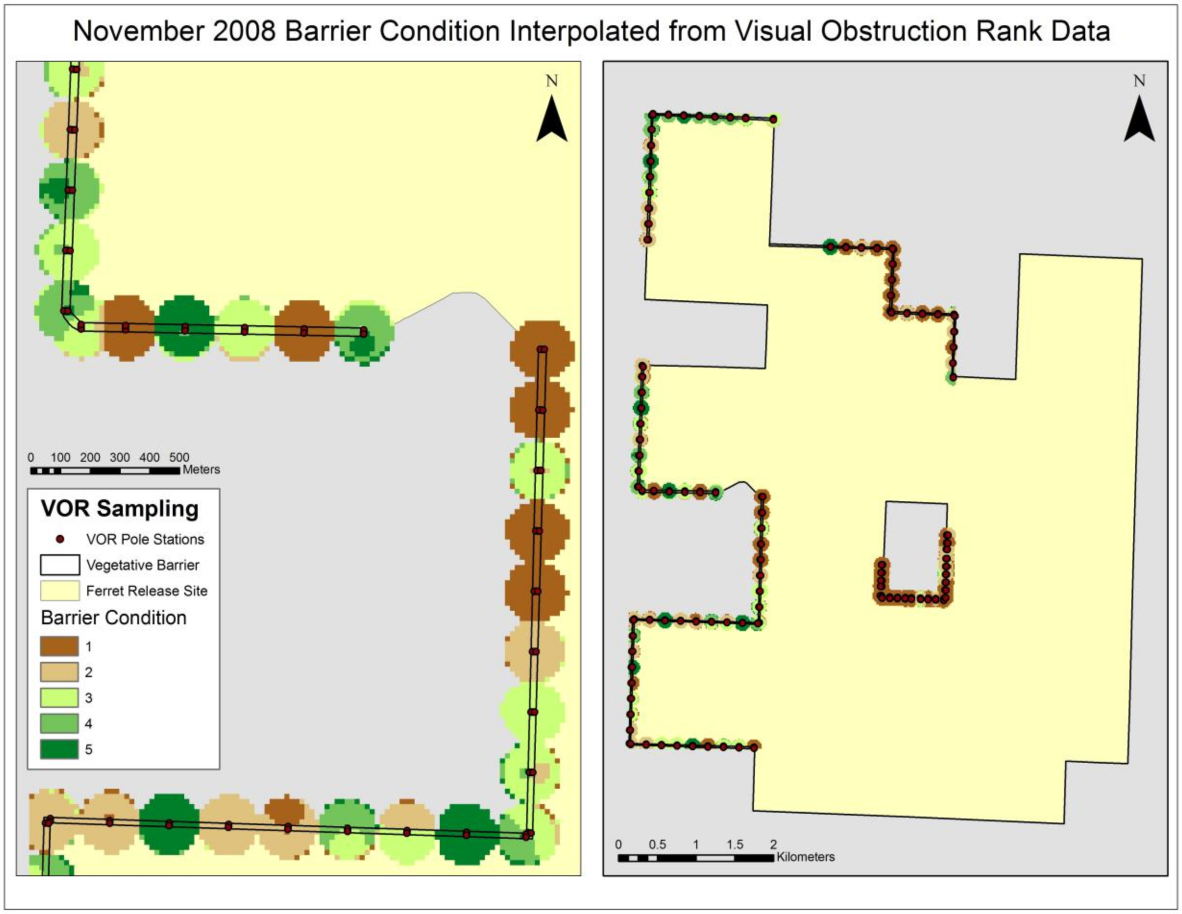


Figure 5.3 Results of IDW interpolation on May 2009 VOR data.

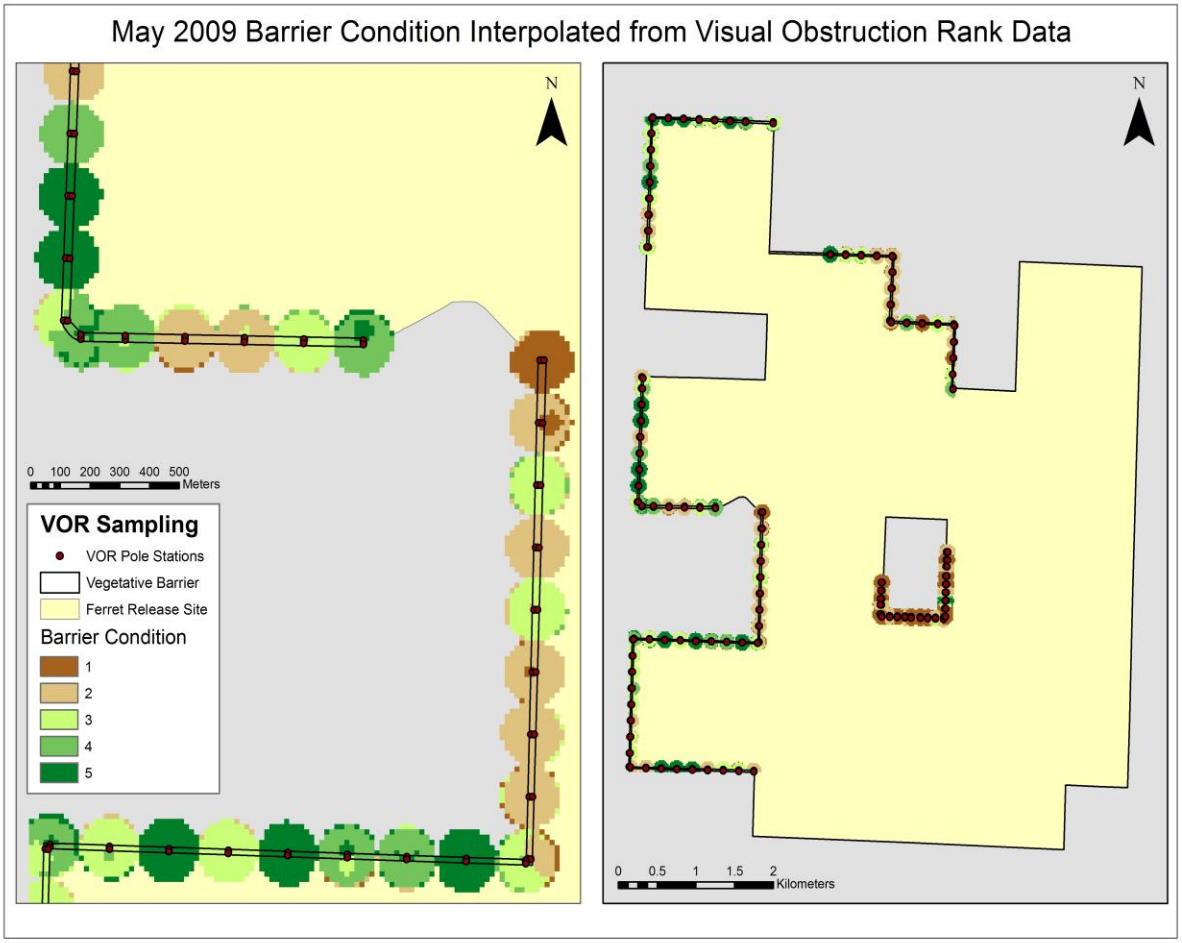
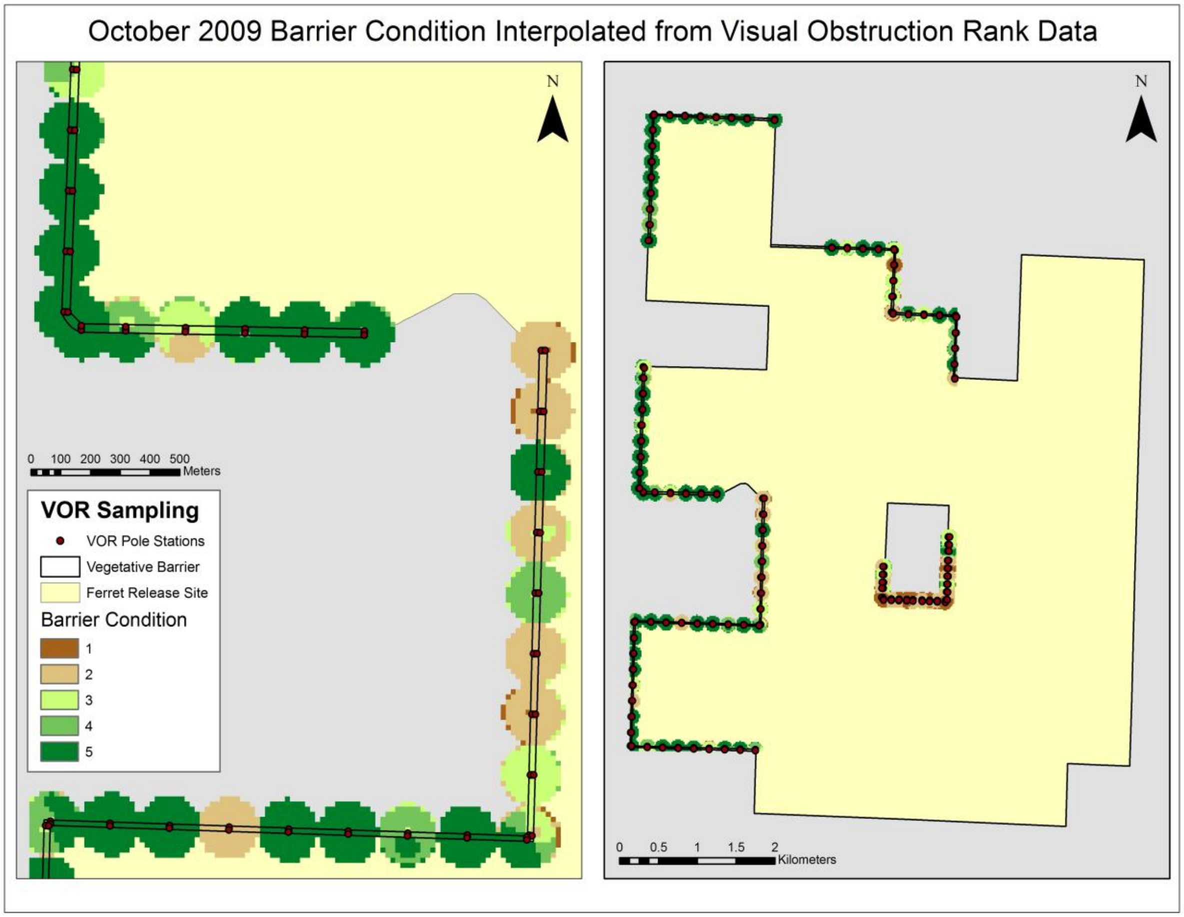


Figure 5.4 Results of IDW interpolation on October 2009 VOR data.



When sampling barrier condition in May 2008, 72 of 88 sampling locations had at least one VOR pole station that averaged a visual obstruction ranking of less than 3.94 (10 cm) and were defined as "windows" in the barrier (Table 5.8). Two hundred forty-four of the 264 individual VOR pole stations averaged less than 10 cm of visual obstruction.

Table 5.8 The number of VOR sampling locations identified as "windows" and averaged distance from active burrows to barrier windows.

Sample	# VOR stations < 10 cm	Total # of Windows	Avg. Distance of Sampled Burrows to Windows (meters)
May 2008	244	72	1092.58
November 2008	207	64	1068.1
May 2009	179	61	1137.38
October 2009	111	36	1242.61

November 2008 VOR sampling exhibited a reduction in the number of locations identified as barrier windows. Sixty-four sample locations were labeled as windows, and 207 pole stations averaged less than 10 cm of visual obstruction. In the spring of 2009, barrier condition had improved a bit more. In May 2009, 61 sampling locations were identified as windows and 179 pole stations averaged less than 10 cm of obstruction. The final barrier condition sampling period came after the 2009 growing season in October. During that period, 111 pole stations averaged less than 10 cm of visual obstruction and barrier windows occurred at 36 of the 88 total sampling locations.

It was unclear if increases in barrier condition (as indicated by *t*-tests) and decreases in the number of barrier windows were associated with increased distances from windows to active burrows sampled on Treatment sites. Distances of active burrows sampled in 2008 from barrier windows in May 2008 averaged 1092.58 meters with a range of distances between 6.95-3285.18 meters. In November 2008, the average distance had decreased to 1068.1 meters, and the range was 6.75-3275.08 meters. When the burrow data from 2009 was merged, distances from those burrows to the windows identified in the May 2009 barrier sampling ranged from 11.48-3314.26 meters. The average distance was 1137.38 meters. The mean distance from burrow windows identified during the final barrier sampling period in October 2009 was 1242.61 meters with values ranging from 11.48-3496.7 meters.

Four statistical comparisons of the distance data were conducted. Using matched-pair *t*-tests, distances to windows identified in spring samples were compared to distances to fall barrier windows for each year of data collection. The third and fourth tests compared distances from burrows to windows identified in May 2008 to distances from May 2009 windows and distance from windows in autumn of 2008 and 2009.

All *t*-tests yielded significant P-values indicating that each pairing was statistically dissimilar (Table 5.9).

Table 5.9 Statistical comparisons of distances from active burrows to barrier windows. (Values under "Mean 1" represent the first variable in each pairing. Values under "Mean 2" represent the second variable.)

Pairing	Mean 1	Mean 2	*t* Value	P-value
Distance May 2008 vs. November 2008	1092.58	1068.1	-13.0159	< 0.0001
Distance May 2009 vs. October 2009	1137.38	1242.61	58.42551	< 0.0001
Distance May 2008 vs. May 2009	1092.58	1137.38	2.934056	0.0034
Distance November 2008 vs. October 2009	1068.27	1242.61	11.25475	< 0.0001

Only the first comparison yielded a negative *t*-value. Thus, the change in window locations from May 2008 to November 2008 resulted in a shorter average distance from burrows to windows in November 2008 than in May 2008. In all other comparisons, the distances increased with each VOR sampling. This suggests that generally, as barrier condition improves, distances from each burrow to a window within the barrier increase.

However, nonparametric Wilcoxon sum-rank analysis of the four comparisons resulted in significant P-values for three of the pairings. The difference between one pairing was insignificant at the 0.05 confidence interval. The comparison of distances to windows measured in May 2008 and May 2009 yielded a P-value of 0.0523—very nearly significant at the desired alpha level.

CHAPTER 6 – DISCUSSION

Burrow Activity and Population Density Estimates

The goal of this project was to determine the effectiveness of a native grassland barrier at controlling prairie dog emigration away from a black-footed ferret reintroduction site. The barrier was created around the perimeter of the FRS in an attempt to prevent prairie dog emigration as a way to ease tensions between owners of the reintroduction site and owners of neighboring properties. Neighboring landowners perceive immigrant prairie dogs as nuisance animals that degrade pastures and compete with livestock for forage. To counter potential economic impacts on their livestock's profitability, they resort to lethal population control. These actions can be expensive, ineffective at long-term population control, and cause death to non-target animal species (Knowles 1986, Uresk 1987, Miller *et al*. 1990, Andelt 2006, Forrest and Luchsinger 2006).

Ideally the barrier would have stopped all prairie dog dispersal, but the project's results did not indicate that outcome. Each year, all study colonies were recolonized by what are assumed to be immigrant prairie dogs from ferret reintroduction sites. Although rodenticide application is often not 100% effective at completely eliminating all individuals, chlorophacinone-based rodenticides are typically more effective prairie dog toxicants than zinc phosphide-based products—which are in more common usage (Fisher and Timm 1987, Hygnstrom *et al*. 1998, Andelt 2006, Forrest and Luchsinger 2006). Thus, it is unlikely that individual prairie dogs in these colonies survived the rodenticide application.

Results of burrow activity samples during the 1^{st} sampling period each year also indicate near total eradication of local populations following rodenticide application. During the 1^{st} sampling period in 2008, a total of 117 active burrows were recorded. That computes to an active burrow density of 1.62 active burrows per hectare. Moreover, all of those burrows were in study colonies directly adjacent to existing colonization at the ferret reintroduction sites (either across a property line or rodenticide application line). Strictly speaking, these study colonies were not separate entities from the ferret reintroduction site colonies—most especially at Control sites and locations where the condition of the vegetative barrier between FRS 1 and Treatment sites was poor. Therefore, immigration from the FRS to these study colonies is better characterized as intracolonial dispersal. As Garrett and Franklin (1988) noted, intracolonial dispersal can happen throughout the year. It is possible that some of the active burrows sampled were not from prairie dogs that survived poisoning, but were activated by residents of the FRS expanding their territory into study areas prior to the first burrow activity sampling period. This argument is further strengthened when recalling that these burrow activity samples were conducted before the first barrier condition data collection—when barrier condition was at the poorest level seen.

Sample period 1 in 2009 yielded 44 active burrows. Twenty-seven of these burrows were directly adjacent to existing FRS colonies. Though this suggests that there may have been a higher percentage of survivors in colonies, overall active burrow density was at the lowest level sampled (0.62 burrows per hectare).

In 2008, 6,040 burrows were sampled from 482.84 hectares of colonization. The next year, 6,371 burrows were sampled in 476.1 colonized hectares. The winter rodenticide applications were successful enough that it is likely the bulk of prairie dog activity at study sites was the result of prairie dog immigration from the ferret reintroduction sites.

Each year, the highest average densities of newly activated burrows were recorded in the early summer periods for both Control and Treatment colonies. In general the highest densities were recorded in the 3rd sample—except for Treatments in 2009, which exhibited the highest density during the 4th sample. This suggests that dispersal was peaking around the time the third samples were conducted—late-June to early-July. Statistical analyses conducted on burrow activity data showed no differences between densities at Controls and Treatments.

However, the average density of newly active burrows sampled each period were on the low end of what other authors have reported for active burrow densities within colonies—10-250 burrows per hectare (Table 5.1) (Biggins *et al*. 1993, Powell *et al*. 1994, Biggins *et al*. 2006). This can be explained by the fact that when collecting samples only burrows which were newly active and had not previously been identified as such were recorded. Therefore, the only samples where all active burrows along transects got recorded were during the first period each year. During those periods, it is likely that little intercolonial dispersal had taken place and the density of active burrows was at its lowest point.

Another way to make a comparison to other reported burrow densities would be to calculate the sum of all burrows recorded as active over all four sample periods each year and use that sum to calculate the number of active burrows per hectare within each colony. Using that method to calculate yearly mean burrow densities within Control and Treatment colonies yields higher active burrow densities (Table 6.1) that are more in line with previously reported densities of 10-250 burrows per hectare (Biggins *et al*. 1993, Powell *et al*. 1994, Biggins *et al*. 2006). Though, with all averages above 150 active burrows per hectare, the new figures seem to be on the high end of the spectrum.

Table 6.1 Yearly burrow density averages (active burrows per hectare).

	2008	2009
Control Colonies	288.34	150.49
Treatment Colonies	160.87	186.59

This is surprising in that visual population density estimates suggested that study colonies had lower than normal prairie dog population densities (as stated in Chapter 5). Visual surveys resulted in a range of minimum population density estimates between 0.34-18.84 prairie dogs per hectare (Table 5.4). Average population density in colonies sampled was 5.21 prairie dogs per hectare. This average is barely within the most commonly reported population densities of 5-45 prairie dogs per hectare (Koford 1958, Reading *et al*. 1989, Biggins *et al*. 1993, Powell *et al*. 1994, Johnson and Collinge 2004, Biggins *et al*. 2006, USFWS 2009). Low densities may be explained by the fact that sampled colonies were all newly recolonized—prior to the formation of coteries, which generally consist of 1 breeding male, 2-3 breeding females, and 1-2 yearlings of each sex (Hoogland 1995).

Statistical tests of visual density estimates showed either no difference between populations or significantly higher densities at Treatment colonies. Again, these data all support the conclusion that grassland barriers surrounding FRS 1 did not limit prairie dog emigration. Furthermore, similar means between Control and Treatment data collected on population and burrow densities suggests that prairie dog activity was essentially the same for each management regime. In other words, similar densities of prairie dogs were making similar use of their space at both Control and Treatment sites.

Given these results, how is it possible that low densities of prairie dogs activated an above average number of burrows per hectare? It could be argued that the revised burrow activity densities stated previously are artificially inflated. The sampling methodology employed in this study (sampling only newly active burrows) was well suited to determining when recolonization of study colonies was at its highest and lowest. However, since all active burrows were not sampled each month, it makes direct comparisons of this data to reported active burrow densities problematic. Anecdotally, it can be stated that burrows did not always stay active throughout the summer. For instance, a burrow recorded as active during the 1st or 2nd sampling period would not always show signs of activity during the 3rd or 4th sampling periods. Without samples of all active burrows, not just those newly activated, the number of burrows which went dormant following activation can not be quantified, though personal experience in the colonies lead to the belief that the percentage is low (< 10% of sampled burrows). Taking this into account, active burrow densities are probably well in line with what other researchers have reported.

Therefore, data shows that study colonies generally had relatively low population densities compared to other researchers' results and active burrow densities that were average compared to other reported values. This suggests that when recolonizing extirpated areas, prairie dogs maximize their territory by making use of more burrows than they would in more densely-populated colonies. That result agrees with Hoogland (1995). He studied a single colony for 15 years. Over that time the population ranged from 92-250 prairie dogs, but the number of burrow entrances (which, for simplicity, are called "burrows" in this document) was always about 1,600.

Estimates of burrow use by prairie dogs at the study sites indicate that, given the opportunity, prairie dogs will use more than the 3.9 burrow entrances per individual that Biggins *et al*. (1993) calculated for black-tailed prairie dogs (Table 5.5). Although estimates from one study site (CS 2 in 2008) are inconceivably high (over 150 active burrows per prairie dog), the rest seem more reasonable—especially in light of Biggins *et al*.'s (1993) calculations of 11.5 burrows per white-tailed prairie dog. Excluding the two highest values from calculations for 2008 the mean number of active burrows per individual was 20.48. In 2009, the average was 15.38 active burrows per individual. This provides further evidence that prairie dogs will maximize their territory and the use of burrows within it.

Farmers and ranchers overwhelmingly have a negative view of prairie dog colonization (Lamb *et al*. 2006). For many, the absence of prairie dog colonies is a sign of good land stewardship. It can be suggested that when they consider levels of prairie dog colonization on their lands, the primary concern is the size of the area colonized—not population densities.

Indeed, it has even been discussed that way by researchers. Uresk and Paulson (1988) investigated the effect of prairie dogs on a pasture's carrying capacity for cattle in South Dakota—suggesting that for every 20 hectares of colonization, a pasture could support about 2 less animal units per month. However, their study was conducted on existing colonization, not newly reoccupied colonies. Without established reproducing coteries, densities are not high enough to make those estimates of prairie dog impact reliable. Though prairie dogs are known to quickly rebound from population reductions (Knowles 1986), during the first year of recolonization potential competition should be lessened. Therefore, in this area, population densities are the important consideration when thinking of prairie dog management plans.

It is common to overestimate prairie dog population densities. When coupling that with high levels of burrow activity and recolonization of the same areas following consecutive years of rodenticide application, the owners of the study sites perceived a more acute impact from prairie dogs than is realistic. The financial costs associated with annual treatment with rodenticides are high enough to strengthen that perception and allow any level prairie dog colonization to be intolerable.

For example, at Treatment Site 1 in 2008, there were approximately 352.64 hectares colonized by prairie dogs. Of that, about 52.9 hectares were sampled for burrow activity. A total of 5,067 active burrows were sampled there throughout the summer. If each burrow stayed active, the estimate of average burrow density would be 95.78 active prairie burrows per hectare. Extrapolating the results to the total colonized area suggests that there would have been about 33,776 active burrows on the property following the summer of 2008.

Label directions state that rodenticide must be applied to all active burrows within a colony. Local rodenticide applicators charge approximately $0.29 per active burrow to bait with chlorophacinone-based poisons. The less effective zinc phosphide baits have been shown to cost between $11.05-$19.16 dollars per hectare (Andelt 2006). Estimating $0.29 * 33,776 burrows equals $9,795.04 in poison application costs. If the owners had used zinc phosphide and the midpoint in the range of costs given by Andelt—$15.11 dollars per hectare— is used to calculate expenditures, the owners of TS 1 would have spent $5,328.39 in 2008. The cost of baiting with chlorophacinone in 2009 would have been $10,778.14. At $15.11 per hectare, it would have cost $5,445.34 for the less effective rodenticide.

Barrier Condition

Analysis of barrier condition data revealed that while visual obstruction increased within the vegetative barrier from spring to fall each year it decreased along borders with Control sites (Table 5.6). Statistical analysis showed sample populations in each comparison were significantly different from each other—corroborating this assertion.

Within the vegetative barrier, the number of sampling locations identified as windows decreased each sampling period. The distances from active burrows to the nearest window also increased significantly in the autumn of 2009. In 2008, even though the number of windows decreased from May to November, the average distance from burrows to windows decreased as well. Reviewing the results of the inverse distance weighted interpolations shows that the area of the barrier where conditions improved most significantly between May and November 2008 was the northwest corner of FRS 1 (Figures 5.1 and 5.2). This area borders Treatment Site 2, but the properties only corner against each other, and the small amount of border is actually separated by a road. Thus, improvements in this part of the barrier affect distances to few burrows, because most of the burrows recorded on TS 2 would have been closest to the same window (located in the northwest corner of FRS 1). Furthermore, though the differences in distance were deemed significant by statistical analysis, the difference in means was only about 30 meters, which is a relatively small difference.

Even though distances from windows to active burrows generally increased, they are well within the range of distances given as average intercolonial dispersal distances (Table 6.2).

Table 6. 2 Distance from newly activate burrows sampled in 2008 and 2009 to barrier windows.

Sampling Period	Mean	Range
May 2008	1092.58	6.95-3285.18
November 2008	1068.1	6.75-3275.08
May 2009	1137.38	11.48-3314.26
October 2009	1242.61	11.48-3496.7

The mean distance from burrows to windows ranged from 1,068.1-1,242.61 meters. Other researchers have reported that dispersing prairie dogs commonly travel 2-3 kilometers (Garrett and Franklin 1988, Roach *et al*. 2001). Moreover, the maximum distance of any burrow to the nearest window was 3.49 km from a window—just over half as far as the reported maximum distance of 6 km (Roach *et al*. 2001, Hoogland 2006a). This again indicates that prairie dog activity at study sites is most likely the result of prairie dog immigration from one of the closest ferret reintroduction sites.

CHAPTER 7 – CONCLUSIONS

Prairie dogs are a well-studied, highly-recognizable species. The issue of prairie dogs on the landscape is highly complex with valid arguments to be made from all sides. The ineffectiveness, expense, and potential for danger to non-target species along with the need for prairie dogs as prey for black-footed ferrets must be measured against their potential to cause conflict within stakeholders' communities. Associated with long-term prairie dog colonization is a shift in local vegetative communities and potential economic impact. These are perceptions that persist even though vegetation within colonies has been shown to be higher in nutritive content and more easily digested than off-colony forage (Coppock *et al*. 1983, Krueger 1986).

Conflicting evidence about prairie dogs' impact on cattle as well as their importance to associated species highlights the need for further research into nonlethal population management techniques. The research presented in this document focused on the effectiveness of native grassland vegetative barriers at limiting emigration away from a black-footed ferret reintroduction site. The barrier was approximately 27 meters wide. It was designed to exclude cattle from grazing up to boundary fences—allowing vegetation to reach its maximum height.

All research suggested the barrier, as constructed, was not effective at slowing or stopping dispersal away from the ferret reintroduction site. While barrier condition improved over time, areas of poor quality were always present. Prairie dogs reestablished extirpated colonies in both years of study, and active burrows were well within average dispersal distances for prairie dogs. Though visual counts indicated that population densities of immigrant prairie dogs were low, burrow activity levels were high. This leads to repeated high-cost rodenticide application by landowners whom do not want prairie dogs on their properties and continued tension between stakeholders.

Efforts were made through directed shooting and rodenticide application (by the landowners and USDA-APHIS employees) to keep prairie dogs from establishing burrows within the barrier. However, colonies did move into the barrier area in some locations. From time to time cattle also made it through the interior electric fence and grazed within the barrier. Both of these unintended events could have had negative impacts on the barrier's ability to reach its maximum height. Periods of drought could also impede native grassland vegetation from reaching its maximum potential. However, when reviewing the yearly climate summaries from the closest National Weather Service weather station in Goodland, KS, it is clear that was not the case in 2008 or 2009 (NWS 2009, NWS 2010). In 2008, total precipitation was slightly above (1.68 cm) the 30-year average from 1971-2000. The following year was even wetter—7.42 cm above normal. During both those years, precipitation that fell as snowfall was below average, suggesting that most of the precipitation occurred in months too warm for snow—probably meaning that the bulk of the moisture fell during the growing seasons in 2008 and 2009.

Aside from biotic and climatic sources of error in this study, other potential sources came from the experimental design. Small sample sizes (especially for Control colonies hurt the statistical validity of the inferences made. Moreover, the fact that two-thirds of the Control sites were located on/around FRS 2 (under different prairie dog and cattle grazing management plans) further compounds potential problems. However, access to study sites can be a common problem for field scientists. This is particularly true when working on private lands. That is a long way of saying that study sites had to be located on the properties whose owners would grant permission to complete the work.

That said, all statistical comparisons were corroborated with nonparametric tests in an attempt to overcome problems with sample sizes/distributions. Both the parametric and nonparametric analyses showed overwhelmingly that the native grassland barrier around FRS 1 was unsuccessful at limiting emigration.

Ultimately, this work suggests that, alone, native grassland vegetative barriers of this width will not be an effective management technique in northwest Kansas. However, the research does lead to a number of questions which, if answered, could shed more light onto prairie dog population dynamics and dispersal in the area. Foremost, additional research on immigrant population densities should be conducted. In this study, there was not enough time to visually sample densities intensively. A larger sample size of study colonies would benefit conservationists by more accurately estimating the number of prairie dogs emigrating from the source populations each year. Furthermore, anecdotal evidence suggested that the number of active burrows per prairie dog was dynamic through the summer. Sampling all active burrows during each sampling period could lead to a better understanding of active burrow dynamics and prairie dog use of space. Barrier width effects on prairie dog emigration should also be investigated. Terrall (2006) suggested that when vegetative biomass could not cause average VORs of at least 10 cm and vegetation height of 40 cm barrier width should be increased to 100 meters. Finally, in 2010, the native grassland vegetative barriers were modified to enhance their effectiveness. Poultry wire and an additional electrified wire were installed on the boundary fences at ground level to further impede dispersal. This modification needs to be studied in the same depth.

References

Antolin, M. F., Savage, L. T. and R. J. Eisen. 2006. Landscape features influence genetic structure of black-tailed prairie dogs (*Cynomys ludovicianus*). *Landscape Ecology* 21:867-875.

Assal, T. J. and J. A. Lockwood. 2007. Utilizing remote sensing and GIS to detect prairie dog colonies. *Rangeland Ecology and Management* 60(1):45-53.

Augustine, D. J., Cully, J. F. and T. L. Johnson. 2007. Influence of fire on black-tailed prairie dog colony expansion in shortgrass steppe. *Rangeland Ecology and Management* 60(5):538-542.

Bartz, S. E., Drickamer, L. C. and M. J. C. Kearsley. 2007. Response of plant and rodent communities to removal of prairie dogs (*Cynomys gunnisoni*) in Arizona. *Journal of Arid Environments* 68:422-437.

Benkobi, L., Uresk, D. W., Schenbeck, G. and R. M. King. 2005. Protocol for monitoring standing crop in grasslands using visual obstruction. *Journal of Range Management* 53(6):627-633.

Bevers, M., Hof, J., Uresk, D. W. and G. L. Schenbeck. 1997. Spatial optimization of prairie dog colonies for black-footed ferret recovery. *Operations Research* 45(4):495-507.

Biggins, D. E., Miller, B. J., Hanebury, L. R., Oakleak, B., Farmer, A. H., Crete, R. and A. Dodd. 1993. A technique for evaluating black-footed ferret habitat. *In*: J. L. Oldemeyer, D. E. Biggins and B. J. Miller [EDS.]. Proceedings of the symposium on the management of prairie dog complexes for the reintroduction of the black-footed ferret. Washington, DC, USA: U. S. Fish and Wildlife Service, Biological Report 13, pp. 73-88.

Biggins, D. E., Sidle, J. G., Seery, D. B. and A. E. Ernst. 2006. Estimating the abundance of prairie dogs. *In*: J. L. Hoogland [ED.] Conservation of the Black-Tailed Prairie Dog: Saving North America’s Western Grasslands. Island:Washington, DC, USA. pp. 94-107.

Black-Footed Ferret Recovery Program (BFFRP). 2009. “Reintroduction.” Available at: http://www.blackfootedferret.org/reintroduction.htm. Accessed 9 June 2011.

Bonham, C. D. and A. Lerwick. 1979. Vegetation changes induced by prairie dogs on shortgrass range. *Journal of Range Management* 29(3):221-225.

Briske, D. D., Derner, J. D., Brown, J. R., Fuhlendorf, S. D., Teague, W. R., Havstad, K. M., Gillen, R. L., Ash, A. J. and W. D. Willms. 2008. Rotational grazing on rangelands: reconciliation of perception and experimental evidence. *Rangeland Ecology and Management* 61(1):3-17.

Cable, K. A. and R. M. Timm. 1987. Efficacy of deferred grazing in reducing prairie dog reinfestation rates. *In*: D. W. Uresk, G. L. Schenbeck, R. Cefkin [TECH COORDS]. Eigth Great Plains Wildlife Damage Control Workshop Proceedings; 28-30 April 1987; Rapid City, SD, USA. Rapid City, SD, USA: Great Plains Damage Control Workshop. p. 46-49. Available at: http://digitalcommons.unl.edu/gpwdcwp/53. Accessed 20 September 2010.

Chang, K. 2006. Spatial interpolation. Introduction to geographic information systems. 3rd ed. McGraw-Hill: New York, USA. pp. 318-347.

Chapman, S. S., Omernik, J. M., Freeouf, J. A., Huggins, D. G., McCauley, J. R., Freeman, C. C., Steinauer, G., Angelo, R. T. and Schlepp, R. L. 2001. Ecoregions of Nebraska and Kansas (color poster with map, descriptive text, summary tables, and photographs). Reston, Virginia, USA. U. S. Geological Survey (map scale 1:1,950,000).

Cincotta, R. P., Uresk, D. W. and R. M. Hansen. 1987. A statistical model of expansion in a colony of black-tailed prairie dogs. *In*: D. W. Uresk, G. L. Schenbeck and R. Cefkin [TECH COORDS.]. Eighth Great Plains Wildlife Damage Control Workshop Proceedings. USDA Forest Service General Technical Report RM-154. Rapid City, SD, USA. pp. 30-32.

Coppock, D. L., Detling, J. K., Ellis, J. E. and M. I. Dyer. 1983. Plant herbivore interactions in a North American mixed-grass prairie. 1: effects of black-tailed prairie dogs on intra-seasonal aboveground biomass, nutrient dynamics, and plant species diversity. *Oecologia* 56:1-9.

Detling, J. K. 2006. Do prairie dogs compete with livestock? *In*: J. L. Hoogland [ED.] Conservation of the Black-Tailed Prairie Dog: Saving North America's Western Grasslands. Island:Washington, DC, USA. pp. 65-88.

Dobson, F. S., Chesser, R. K., Hoogland, J. L., Sugg, D. W. and D. W. Foltz. 1997. Do black-tailed prairie dogs minimize inbreeding? *Evolution* 51(3):970-978.

Environmental Systems Research Institute (ESRI). 2008. “How inverse distance weighted (IDW) interpolation works”. *ArcGIS Desktop 9.3 Help*. Available at: http://webhelp.esri.com/arcgiSDEsktop/9.3/index.cfm?TopicName=How_Inverse_Distance_Weighted_%28IDW%29_interpolation_works. Accessed 28 March 2011.

Facka, A. N., Ford, P. L. and G. W. Roemer. 2008. A novel approach for assessing density and range-wide abundance of prairie dogs. *Journal of Mammalogy* 89(2):356-364.

Fisher, D. D. and R. M. Timm. 1987. Laboratory trial of chlorophacinone as a prairie dog toxicant. *In*: D. W. Uresk, G. L. Schenbeck, and R. Cefkin [TECH COORDS.]. <u>Eighth Great Plains Wildlife Damage Control Workshop Proceedings</u>. USDA Forest Service General Technical Report RM-154. Rapid City, SD, USA. Available at: http://digitalcommons.unl.edu/cgi/viewcontent.cgi?article=1064&context=gpwdcwp. Accessed 07 March 2011.

Forrest, S. C. and J. C. Luchsinger. 2006. Past and current chemical control of prairie dogs. *In*: J. L. Hoogland [ED.] <u>Conservation of the Black-Tailed Prairie Dog: Saving North America’s Western Grasslands</u>. Island:Washington, DC, USA. pp. 115-128.

Foster-McDonald, N. S., Hyngstrom, S. E. and S. P. Korte. 2006. Effects of a visual barrier fence on behavior and movements of black-tailed prairie dogs. *Wildlife Society Bulletin* 34(4):1169-1174.

Franklin, W. L. and M. G. Garrett. 1989. Nonlethal control of prairie dog colony expansion with visual barriers. *Wildlife Society Bulletin* 17:426-430.

Garrett, M. G. and W. L. Franklin. 1988. Behavioral ecology of dispersal in the black-tailed prairie dog. *Journal of Mammalogy* 69(2):236-250.

Gordon Barnhardt, Larry Haverfield, and Elizabeth Haverfield vs. Board of County Commissioners of Logan County, KS. 2010. Case No. 2009 CV 5. (15th Judicial District, Logan County, KS).

Heitschmidt, R. and J. Walker. 1983. Short duration grazing and the Savory grazing method in perspective. *Rangelands* 5(4):147-150.

Higgins, K. F., Jenkins, K. J., Clambey, G. K., Uresk, D. W., Naugle, D. E., Norland, J. E. and W. T. Barker. 2005. Vegetation sampling and measurement. *In*: C. E. Braun [ED.].

Techniques for wildlife investigations and management. 6th edition. Wildlife Society: Bethesda, Maryland, USA. pp. 524-553.

Hoogland, J. L. 1995. The Black-Tailed Prairie Dog: Social Life of a Burrowing Mammal. University of Chicago Press: Chicago, Illinois, USA. 557pp.

Hoogland, J. L. 2006a. Demography and population dynamics of prairie dogs. *In*: J. L. Hoogland [ED.] Conservation of the Black-Tailed Prairie Dog: Saving North America's Western Grasslands. Island:Washington, DC, USA. pp. 27-52.

Hoogland, J. L. 2006b. Social behavior of prairie dogs. *In*: J. L. Hoogland [ED.] Conservation of the Black-Tailed Prairie Dog: Saving North America's Western Grasslands. Island:Washington, DC, USA. pp. 7-27.

Hyngstrom, S. E. 1995. Plastic visual barriers were ineffective at reducing recolonization rates of prairie dogs. *In*: R. E. Masters and J. G. Huggins [EDS.]. Tulsa, Oklahoma, USA. National Wildlife Research Center, Fort Collins, Colorado, USA. pp. 74-76.

Hyngstrom, S. E., McDonald, P. M. and D. R. Virchow. 1998. Efficacy of three formulations of zinc phosphide for managing black-tailed prairie dogs. *International Biodeterioration and Biodegradation* 42:147-152.

Johnson, W. C. and S. K. Collinge. 2004. Landscape effects on black-tailed prairie dog colonies. *Biological Conservation* 115:487-497.

Jones, S. 1999. Becoming a pest: prairie dog ecology and the human economy in the Euroamerican west. *Environmental History* 4(4):531-552.

Karl, J. W. 2010. Spatial predictions of cover attributes of rangeland ecosystems using regression kriging and remote sensing. *Rangeland Ecology and Management* 63(3):335-349.

King, J. A. 1955. Social behavior, social organization, and population dynamics in a black-tailed prairie dog town in the Black Hills of South Dakota. *Contributions from the Laboratory of Vertebrate Biology* 67:1-123.

Knowles, C. J. 1985. Observations on prairie dog dispersal in Montana. *Prairie Naturalist* 17(1):33-40.

Knowles, C. J. 1986. Population recovery of black-tailed prairie dogs following control with zinc phosphide. *Journal of Range Management* 39(3):249-251.

Koford, C. B. 1958. Prairie dogs, white-faces and blue gramma. *Wildlife Monographs* 3:1-78.

Kotliar, N. B., Baker, B. W., Whicker, A. D. and G. E. Plumb. 1999. A critical review of assumptions about the prairie dog as a keystone species. *Environmental Management* 24(2):177-192.

Kotliar, N. B., Miller, B. J., Reading, R. P. and T. W. Clark. 2006. The prairie dog as a keystone species. *In*: J. L. Hoogland [ED.] Conservation of the Black-Tailed Prairie Dog: Saving North America's Western Grasslands. Island:Washington, DC, USA. pp. 53-64.

K. S. A. § 80-1201-1203. Available at: http://kansasstatutes.lesterama.org/Chapter_80/Article_12/#80-1202. Accessed 08 March 2011.

Krueger, K. 1986. Feeding relationships among bison, pronghorn, and prairie dogs: an experimental analysis. *Ecology* 67:760-770.

Lamb, B. L., Reading, R. P. and W. F. Andelt. 2006. Attitudes and perceptions about prairie dogs. *In*: J. L. Hoogland [ED.] Conservation of the Black-Tailed Prairie Dog: Saving North America's Western Grasslands. Island:Washington, DC, USA. pp. 108-114.

Liebhold, A. M., Rossi, R. E. and W. P. Kemp. 1993. Geostatistics and geographic information systems in applied insect ecology. *Annual Review of Entomology* 38:303-327.

Lomolino, M. V. and G. A. Smith. 2001. Dynamic biogeography of prairie dog (*Cynomys ludovicianus*) towns near the edge of their range. *Journal of Mammalogy* 82(4):937-945.

Longley, P., Goodchild, M. F., Maguire, D. J. and D. W. Rhind. 2001. Inverse distance weighting. Geographical Information Systems and Science. Wiley: West Sussex, England. pp. 333-337.

Loveland, T. R. and J. M. Merchant. 2004. Ecoregions and Ecoregionalization: Geographical and Ecological Perspectives. *Environmental Management* 34(Suppl. 1):S1-S13.

Merrill, T., Mattson, D. J., Wright, R. G. and H. B. Quigley. 1999. Defining landscapes suitable for restoration of grizzly bears *Ursus arctos* in Idaho. *Biological Conservation* 87:231-248.

Merriman J. W., Zwank, P. J., Boal, C. W. and T. L. Bashore. 2004. Efficacy of visual barriers in reducing black-tailed prairie dog colony expansion. *Wildlife Society Bulletin* 32:1316-1320.

Miller, B., Wemmer, C., Biggins, D. and R. Reading. 1990. A proposal to conserve black-footed ferrets and the prairie dog ecosystem. *Environmental Management* 14(6):763-769.

National Weather Service (NWS). 2009. Climate report: Goodland, KS. 04 January 2009. Available at: http://www.crh.noaa.gov/gld/scripts/localdata.php?loc=apptxt&data=TOPCLAGLD.20090108.1759. Accessed 04 August 2011.

National Weather Service (NWS). 2010. Climate report: Goodland, KS. 01 January 2010. Available at: http://www.crh.noaa.gov/gld/scripts/localdata.php?loc=apptxt&data=TOPCLAGLD.20100106.1634. Accessed 04 August 2011.

O'Meilia, M. E. 1980. Competition between prairie dogs and beef cattle for range forage. Oklahoma State University [master's thesis].

Powell, K. L., Robel, R. J., Kemp, K. E. and M. D. Nellis. 1994. Aboveground counts of black-tailed prairie dogs: temporal nature and relationship to burrow entrance density. *Journal of Wildlife Management* 58(2):361-366.

Reading, R. P. and R. Matchett. 1997. Attributes of black-tailed prairie dog colonies in northcentral Montana. *Journal of Wildlife Management* 61(3):664-673.

Roach, J. L., Stapp, P., van Horne, B. and M. F. Antolin. 2001. Genetic structure of a metapopulation of black-tailed prairie dogs. *Journal of Mammalogy* 82:946-959.

Robertson, G. P. 1987. Geostatistics in ecology: interpolating with known variance. *Ecology* 68(3):744-748.

Savory, A. 1983. The Savory grazing method or holistic resource management. *Rangelands* 5(4):155-159.

Schloeder, C. A., Zimmerman, N. E. and M. J. Jacobs. 2001. Comparison of methods for interpolating soil properties using limited data. *Soil Science Society of America Journal* 65:470-479.

Severson, K. E. and G. E. Plumb. 1998. Comparison of methods to estimate population densities of black-tailed prairie dogs. *Wildlife Society Bulletin* 26(4):859-866.

Shepard, D. 1968. A two-dimensional interpolation function for irregularly-spaced data. Proceedings of the 1968 23rd ACM National Conference. The Association for Computing Machinery. pp. 517-524.

Snell, G. R. and B. D. Hlavachick. 1980. Control of prairie dogs—the easy way. *Rangelands* 2:239-240.

Stapp, P. 1998. A reevaluation of the role of prairie dogs in Great Plains grasslands. *Conservation Biology* 12(6):1253-1259.

Terrall, D. F. 2006. Use of natural vegetative barriers to limit black-tailed prairie dog town expansion in western South Dakota. South Dakota State University [master's thesis].

Tobler W. 1970. A computer movie simulating urban growth in the Detroit region. *Economic Geography* 46(2):234-240.

Uresk, D. W., J. G. McCracken, and A. J. Bjugstad. 1981. Prairie dog density and cattle grazing relationships. *In*: R. J. Johnson and R. M. Timm [TECH COORDS.]. Fifth Great Plains Wildlife Damage Control Workshop Proceedings; 13-15 April 1981; Lincoln, NE, USA. Lincoln, NE, USA: Great Plains Damage Control Workshop. p. 199-201. Available at: http://digitalcommons.unl.edu/gpwdcwp/145/. Accessed 21 September 2010.

Uresk, D. 1987. Relation of black-tailed prairie dogs and control programs to vegetation, livestock, and wildlife. *In*: J. L. Capinera [ED.]. Integrated pest management on rangeland: a shortgrass prairie perspective. Boulder, CO, USA: Westview Press. pp. 312-323. Available at: http://www.fs.fed.us/rmrs/docs/pubs/forest-grasslands-archive/pdrelat2.pdf. Accessed 9 September 2010.

Uresk, D. and D. D. Paulson. 1988. Estimated carrying capacity for cattle competing with prairie dogs and forage utilization in western South Dakota. *In*: Management of amphibians, reptiles, and small mammals in North America: proceedings of the symposium. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, General Technical Report RM-166. Fort Collins, CO, USA. pp. 387-390. Available at: http://www.fs.fed.us/rmrs/docs/pubs/forest-grasslands-archive/carryingcap.pdf. Accessed 13 May 2011.

U.S. Census Bureau. 2010. State and county quickfacts: Logan County, KS. Available at: http://quickfacts.census.gov/qfd/states/20/20109.html. Accessed 01 February 2011.

U.S. Fish and Wildlife Service. 2000. Endangered and threatened wildlife and plants: 12-month finding for a petition to list the black-tailed prairie dog as threatened. *Federal Register* 65(24): 5476-5488.

U.S. Fish and Wildlife Service. 2009. Endangered and threatened wildlife and plants: 12-month finding on a petition to list the black-tailed prairie dog as threatened or endangered. *Federal Register* 74:63343-63366. Available at: http://www.fws.gov/mountain-prairie/species/mammals/btprairiedog/74FR63343.pdf. Accessed 9 September 2010.

Valley, R. D., Drake, M. T. and C. S. Anderson. 2005. Evaluation of alternative interpolation techniques for the mapping of remotely-sensed submersed vegetation abundance. *Aquatic Botany* 81:13-25.

Verdolin, J. L., Lewis, K. and C. N. Slobodchikoff. 2008. Morphology of burrow systems: a comparison of Gunnison's (*Cynomys gunnisoni*), white-tailed (*C. leucurus*), black-tailed (*C. ludovicianus*), and Utah (*C. parvidens*) prairie dogs. *The Southwestern Naturalist* 53:201-207.

Wickham, J. D., O'Neill, R. V. and K. B. Jones. 2000. A geography of ecosystem vulnerability. *Landscape Ecology* 15:495-504.

Winter, S. L., Cully, J. F. Jr. and J. S. Pontius. 2002. Vegetation of prairie dog colonies and non-colonized shortgrass prairie. *Journal of Range Management* 55(5):502-508.