Black-Tailed Prairie Dog (*Cynomys ludovicianus*) Response to Seasonality and Frequency of Fire

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BLACK-TAILED PRAIRIE DOG RESPONSE TO SEASONALITY AND FREQUENCY OF FIRE

ABSTRACT

Fragmentation of the landscape, habitat loss, and fire suppression, all a result of European settlement and activities, have precipitated both the decline of Black-tailed prairie dog (*Cynomys ludovicianus*) populations and the occurrence of fire throughout the Great Plains, including the Shortgrass steppe of northeastern New Mexico. The presence of Black-tailed prairie dogs, a keystone species, and the occurrence of fire both play a vital role in maintaining the integrity and diversity of grassland ecosystems. In addition to serving as prey for the endangered Black-footed ferret (*Mustela nigripes*), Black-tailed prairie dogs conspicuously alter grassland landscapes and provide foraging, shelter, and nesting habitat for a diverse array of grassland species.

While agricultural politics have interfered with efforts to list the species as threatened under the Endangered Species Act, a lack of knowledge regarding the mechanisms for colony expansion have hindered the development of recovery plans that effectively manage for the persistence of prairie dog colonies. Previous studies have shown that manipulations of vegetation, through prescribed fire and brush removal, encourage and facilitate colony expansion. Here, with the use of Geographic Information Systems (GIS) and Remote Sensing technology, we determine how Black-tailed prairie dogs have responded to growing season and dormant season burns at 3, 6, and 9 year intervals. Knowledge of prairie dog response to seasonality and frequency of fire may aid in the development and implementation of prescribed fire regimes that can be used to effectively manage Shortgrass ecosystems.

INTRODUCTION

Since its introduction in 1969 by Robert T. Paine, the term keystone species has evolved to describe a species whose presence is crucial in maintaining the organization and diversity of their ecological community and is exceptional to the rest of the community in its importance (Mills *et al*., 1993; Paine, 1969). Another keystone trait is that the activities of the species are not wholly duplicated by other members of the community (Kotliar *et al*., 1999). The Black-tailed prairie dog (*Cynomys ludovicianus*) significantly alters grassland ecosystems, and it is estimated that more than 150 different species associate and depend on prairie dogs as a food source and upon their colony sites for shelter from weather and predation (Hoogland, 2006). For these reasons, the Black-tailed prairie dog is considered by ecologists to be a keystone species in grassland ecosystems (Miller and Cully, 2001). In an effort to maximize biodiversity it has been suggested that strongly interactive species, such as the Black-tailed prairie dog, be a top priority for protection and management programs; a necessity in re-establishing and sustaining ecosystem structure and stability (Miller *et al.*, 1994, Soule et al., 2005).

Of the estimated 150 different species that benefit at some level from prairie dog activities, one of the most notable is the Black footed ferret (*Mustela nigripes*)—an endangered species whose primary food source is the Black-tailed prairie dog (Anderson *et al*., 1986). The Kiowa and Rita Blanca National Grassland Ranger District, located in Shortgrass steppe in the southern Great Plains, is receiving consideration as a proposed release site for the Black-footed ferret. However, the prairie dog population in this area is not extensive enough to support a release, and management for the persistence of prairie

dog colonies has been hindered by lack of knowledge regarding the mechanisms for colony expansion.

Previous studies (Milne-Laux and Sweitzer, 2006; Northcott *et al.*, 2008) have shown that manipulations of vegetation, through fire and brush removal, facilitate Black-tailed prairie dog colony expansion. Numerous other studies (King, 1955; Koford, 1958; Butler, 1995; Garrett and Franklin, 1998) have shown that the reestablishment of periodic fire is fundamental to the ecological restoration of grassland ecosystems. However, prior to reintroducing large-scale fire as a management tool, the appropriate fire season, frequency, and fire effects on ecosystem components need to be determined. The purpose of this study is to examine how the seasonality and frequency of prescribed fire affects Black-tailed prairie dog colony expansion in the Shortgrass steppe of northeastern New Mexico.

LITERATURE REVIEW

European Settlement of the Grasslands

The North American grasslands once formed one of the largest prairies in the world, stretching 1,000 miles from east to west and 2,000 miles from north to south (USDA Forest Service). Prior to European settlement, approximately 162 million ha of extensive and unbroken grasslands existed throughout the Great Plains (Samson and Knopf, 1994). At the time of early exploration by Meriwether Lewis and William Clark, the Great Plains were lightly settled by native people and the grasslands supported a diverse array of plant and animal species. In May of 1804, Lewis and Clark traversed the Great Plains and, over the course of 14 months, documented and described the new landscapes, native people, plants, and wildlife that they had encountered (Johnsgard, 2003). In the years after the Lewis and Clark expedition, settlers streamed across the Mississippi River and settled the Great Plains (Johnsgard, 2003). The pristine and extensive grasslands were divided, domestic cattle populations rose, and the prairie ecosystem was converted into farmland and rangeland (Cunfer, 2005).

The Homestead Act of 1862, a landmark law that was responsible for helping settle much of the American West, is perhaps the single most influential event associated with shaping the character and biological diversity of the Great Plains today. Under this act, which was signed into law on May 20, 1862, up to 160 acre parcels of land were granted to citizens and newly arrived immigrants who wished to settle in the western United States. Between 1862 and 1976, 600,000 families took advantage of the Homestead Act, and 270 million acres of the western United States were turned over to private citizens who converted the prairies to farmland and rangeland (Kennedy and

Cohen, 2013). This increase in settlement in the western US resulted in a significant loss of native grasslands and soils as the homesteaders set out to make a living through agriculture (Bradsher, 2012).

The homesteading rush began to slow around 1917 when America entered World War I; however, by this time most of the productive agricultural land was already claimed (Bradsher, 2012). Sod-busting incentives and a demand for wheat after the war accelerated the cultivation of the Great Plains, and new machinery, such as disc plows, and a series of wet years in the late 1920s led to the plowing of more than 33 million acres (Hazlett *et al*., 2009). Very little value and effort was placed into keeping the native grassland intact, and by the 1930s almost all of the prairie and forest had been broken to the plow and converted into an agricultural cash crop. The environmental consequences of these activities would not be realized until the rains stopped and the drought and the winds came. Between 1931 and 1938, the worst drought in US history spread across 75 percent of the country and affected 27 states—crops died and the dust from over-plowed and over-grazed land began to blow (Cunfer, 2005).

An estimated 23.5 million acres of plowed farmland lost between 2.5 -5 inches of top soil. Furthermore, the organic matter and nitrogen content was carried away with the wind-blown soils, leaving behind very poor soil. When all was said and done many farms were bankrupt and deserted, leaving approximately 2.5 million farmers as refugees (Bradsher, 2012). As part of the Roosevelt Administration's New Deal program, the U.S. government put an end to the practice of giving away public lands and bought back 11.3 million acres of overgrazed and mismanaged farmland and rangeland. On June 20, 1960,

3.8 million acres of the purchased land was turned over to the USDA Forest Service and later designated National Grasslands (Perry, 1999).

Grassland system loss across the Great Plains is currently estimated at 70%. The direct loss of prairie to urbanization, and exploitation by overgrazing and farming practices has stressed and destroyed much of what once was a pristine and continuous grassland ecosystem (Samson and Knopf, 1994). Furthermore, grassland species are in constant competition with humans for available habitat and resources. Habitat loss and fragmentation of the grasslands restricts and alters the spatial distribution of grassland species more than any other factor (Johnson, 1995). The potential for grassland species extinction, as a result of human activities, habitat loss, and fragmentation of the landscape is best exemplified by the 100 previously unknown species of vertebrate animals that were first described by Lewis and Clark. Of these 100 species, approximately 40 percent have a state or federal level designation indicating that active protection or conservation concern is warranted. Of these, 13 species are now classified as nationally endangered (Johnsgard, 2003).

Prehistoric Black-Tailed Prairie Dog Range

Prehistoric fossil evidence collected throughout North America indicates that black-tailed prairie dogs, *Cynomys ludovicanus,*resided on the Great Plains. Although two records from the Senecan period (2.5 -1.8x10^6 years ago) show the genus *Cynomys* to be present in the Central Great Plains, the subgenera was indeterminate (Goodwin, 1995). Further fossil evidence suggests that black-tailed prairie dogs were present in the southern and central Great Plains in the Sappan (early Irvingtonian, 1.8 -0.75x10^6 years ago), and in Nebraska and Colorado during the Sangamon (125,000-75,000 years ago).

The geographically most complete and accurate fossil records for prairie dogs is from the Wisconsin (75,000-10,000 years ago), which indicate that the range of black-tailed prairie dogs extended as far north as central Nebraska and east into western Iowa and Kansas, well east of its current range, and as far south as southeastern Chihuahua, Mexico and as far west as southwestern New Mexico (Goodwin, 1995).

Throughout the Pleistocene, the black-tailed prairie dog retained its core range in the central and southern Great Plains, despite major climatic shifts that were associated with five glacial-interglacial cycles (Goodwin, 1995; Lomolino and Smith, 2003). The climatic and environmental oscillations during the late Pleistocene altered the vegetation of the Great Plains from conifer woodland to expanded grassland—neither of which caused a change or reduction in prairie dog ranges (Goodwin, 1995).

Historical Black-Tailed Prairie Dog Range

Despite the broad and relatively stable distribution of the prairie dog colonies throughout the Pleistocene (Goodwin, 1995), historical evidence suggests that black-tailed prairie dog population numbers and occupied acreage have declined dramatically over the last 100 years (Northcott *et al., 2*008). It is estimated that prairie dog colonies once covered between 40 and 160 million ha of mixed and short grass prairies of North America (Marsh, 1984; Hoogland, 2006). This historic range spanned thousands of square kilometers from southern Canada to northern Mexico, with portions in 12 states – Arizona, New Mexico, Colorado, Oklahoma, Kansas, Montana, Nebraska, North Dakota, South Dakota, Oklahoma, Texas, and Wyoming (Hall and Kelson, 1959; Mulhern and Knowles, 1997).

Anglo-European settlement of the grasslands, with widespread cultivation and grazing during the late 19th century, has had a long lasting effect on the ecological structure of the Great Plains (Brockway *et al*., 2002). Prairie dog population numbers and area of occupied habitat began to decline almost immediately after European settlement because of human persecution and habitat destruction (McCain *etal*., 2002). Prior to changes in land use by the settlers, a systematic inventory of the distribution of prairie dogs was never performed; therefore, it is impossible to accurately estimate the average size or density of prairie dog colonies in pre-settlement times. Rough estimates from the Bureau of Biological Survey indicate that all five species of prairie dog occupied approximately 40 million ha in the early 1900s (Merriam, 1902, Knowles *et al.*, 2002) Other data from the same time period, such as regional maps that were created from land surveys or from poisoning and eradication efforts, illustrate that prairie dogs once occupied between 2%- 15% of the Great Plains; with numerous colonies measuring as large as 20,000 ha in size. Some exceptionally large colonies occupied more than 2 million ha (Hoogland, 2006).

Recent Black-Tailed Prairie Dog Range and Status

Today, the lands currently occupied by prairie dog colonies are thought to represent less than 2% of their historical range (Anderson *et al*., 1986; Stapp 1998). By the 1960s, the large and numerous black tailed prairie dog colonies of yesteryear had been reduced to 600,000 ha (Lomolino *et al*., 2003), and this trend continues today. In 1995, it was estimated that black tailed prairie dogs occupied approximately 540,000 ha in the United States, by 1998 this number had been reduced to 280,000-320,000 ha (McCain *et al.*, 2002). More recent estimates from the United States Fish and Wildlife

Service suggest that the cumulative area occupied by prairie dogs is between 500,000 – 800,000 ha of 160 million ha of potential habitat in their former geographic range (Miller and Cully Jr, 2001). Although their former range and numbers have decreased dramatically, black-tailed prairie dogs can still be found in Canada, Mexico, and all of the states that were part of its historic range—with the exception of Arizona, from which the species has been extirpated (Mulhern and Knowles, 1997).

Habitat availability for prairie dogs and other grassland species continues to be threatened and reduced by municipal, agricultural, and industrial development (Mulhern and Knowles, 1997). Furthermore, approximately two thirds of prairie dog colonies are restricted to small, isolated colonies—making them more susceptible to local extinction(Hoogland, 2006). In 1998, the National Wildlife Federation petitioned the U.S. Fish and Wildlife Service to list the black-tailed prairie dogs as a threatened species under the Endangered Species Act. This petition led to the species being listed as a federal candidate for listing as threatened until 2004. Although it was warranted for listing but precluded, conservation efforts increased drastically and yearly reviews were conducted to determine to the status of the listing. Under these conservation efforts, recreational shooting and poisoning efforts declined or were eliminated on public lands (Johnson and Cully, 2004).

The "warranted, but precluded" designation by the USFWS also flamed an already heated and complex political debate over the relationship among black-tailed prairie dogs, agricultural interests, and wildlife. Advocates of the agricultural industry promote the management of publicly owned lands in ways that benefit the livestock industry by citing arguments that historic estimates of prairie dog numbers are greatly

inflated and that prairie dogs reduce the carrying capacity for livestock and other wildlife. (Miller *et al.*,2007). In 2007, another petition to list the species was filed, but the USFWS determined in 2009 that listing the species was not warranted. Although a species must meet only one of the five criteria used by the USFWS to list a species under the Endangered Species Act, agricultural politics have hindered efforts to list the black-tailed prairie dog, despite the fact that the species meets all five of the criteria (see Miller *et al* 2007 for a discussion of all five criteria).

Prairie Dog Range in New Mexico

It is estimated that prairie dogs occupied approximately 4,838,460 ha in New Mexico in 1919 (Mulhern and Knowles, 1997). By 1980, this area had been reduced by 96%, to an estimated 200,000 ha (Henderson, 1979; Mulhern and Knowles, 1997). In New Mexico, the black-tailed prairie dog has been assigned an S2 ranking by the New Mexico Natural Heritage program –meaning they are rare. Colonies were once extensive and abundant east of the Rio Grande and in the grasslands, open woodlands, and semi-desert habitats of the southwestern section of the state, but they have since been extirpated (Johnson and DeLay, 2000).

Today, current black-tailed prairie dog colonies in New Mexico can be found in small numbers in the eastern plains (Johnson and Delay, 2000). The BLM and White Sands Missile Range have also reported that prairie dogs have been extirpated from several sites, with only 140 ha of occupied habitat remaining on BLM land, and 300 ha on the Missile Range (Mulhern and Knowles, 1997).

Prairie Dogs within Kiowa and Rita Blanca National Grasslands

As part of their research involving spatial patterns of plague, Tammi Johnson and Jack Cully Jr. (2004) mapped prairie dog colonies on five national grasslands that had experienced epizootic die-offs, including the Kiowa and Rita Blanca National Grasslands. Cully and Johnson visited all locations in which prairie dogs had previously occupied, and mapped only the active areas of each colony to show year-to-year changes. The results of Cully and Johnsons mapping efforts indicated that active colony areas on the Kiowa and Rita Blanca National Grasslands in 1999, 2001, 2002, and 2003 were 696; 1,663; 2,186; and 2,750 ha, respectively (Johnson and Cully, 2004).

Human-Prairie Dog Conflicts

In part, the considerable reduction of prairie dog populations can be attributed to eradication efforts carried out on a very large scale (Anderson *et al.*, 1986). In some years, prairie dogs were intentionally poisoned on more than 19,760,000 acres in the United States (Clark, 1989). Throughout history, prairie dogs and grazing mammals of the Great Plains coexisted and flourished within their native habitats (Miller *et al*., 1994). However, since the early 1900s, much attention has been focused on the economic impact that prairie dogs have on rangeland productivity. Soon after the Great Plains were settled, ranchers began to view prairie dogs as competitors with cattle for rangeland resources, and they became concerned that prairie dogs were both consuming and destroying grasses and other forage plants (Hollister, 1916; Antolin *et al*., 2002).

In 1902, the US Department of Agriculture published a paper titled “The prairie dog of the Great Plains,” in which it was estimated that range productivity decreased as much as 50-75% when prairie dogs were present on the landscape (Merriam, 1902).

Other studies, such as the one Koford (1958) published, suggested that prairie dogs negatively impact forage crop and pasturage directly by removing it, and indirectly through the long-term influence their burrowing activities have on certain plant species (Koford, 1958). The result of these studies was government funded eradication programs and poisoning campaigns that are responsible for reducing prairie dog populations to less than two percent of their historical populations as of several decades ago(Whicker and Detling, 1988; Anderson *et al*., 1986; Marsh, 1984).

Merriam's (1902) estimates were not based on scientific evidence, and research later showed they were a ten-fold exaggeration with levels of competition between 4% and 7% (Uresk and Paulson, 1989). Both Hansen and Gold (1977) and O'Meilia et al. (1982) reported that there was not a significant difference in the market weight of livestock that were living on the same landscape as prairie dogs and livestock that did not coexist with prairie dogs. Derner et al. (2006) found a 5.5% reduction in cattle weight when 20% of the pasture was prairie dog colony and 14% reduction in weight when 60% of the pasture was occupied by prairie dogs. For a review of competition between prairie dogs and wildlife see Miller et al. (2007). Numerous other studies showed that grazing mammals, including domestic cattle, preferentially graze in areas in which prairie dogs towns are present—where the forage is often more succulent and nutritious (Krueger, 1986; Knowles, 1986; Whicker and Detling, 1988).

Although these studies sparked a change in the scientific and conservation communities' attitudes toward prairie dogs, political complexity regarding the relationship among black-tailed prairie dogs, agricultural interests, and wildlife species has increased in recent years (Miller *et al.,* 2007). Despite all the modern research that

indicates prairie dogs have little effect on range productivity, the agricultural industry actively promotes management of public lands in ways that benefit the livestock industry. These management programs include eradication and poisoning campaigns that have been carried out on a very large scale, affecting millions of ha of land (Mulhern and Knowles, 1997). In the1980s, the US government spent over six million dollars to eradicate 185,600 ha of prairie dogs in South Dakota, including a 110,000 ha complex — which was the largest remaining at the time. In the early 1990s, it was estimated that 80,000 ha of prairie dogs were eliminated annually (Miller *et al.,* 1994), with almost every federal land management agency having being involved in these efforts (Mulhern and Knowles, 1997). Much of this effort still continues today, with statewide legislation in Wyoming, Colorado, Kansas and South Dakota either mandating, or allowing for, control of prairie dogs. New Mexico, Nebraska, Montana, North Dakota, Oklahoma, and Texas do not mandate control; however, assistance may be available to landowners who believe they have a prairie dog population problem (Mulhern and Knowles, 1997).

Sylvatic Plague

The most persistent threat facing black-tailed prairie dogs today is Sylvatic plague—which is caused by *Yersinia pestis*, a bacterium that spreads through contact between flea vectors and mammalian hosts (Barnes, 1982; Perry and Featherson, 1997; Coolinge *et al*., 2005). Y. *pestis*, was first introduced to North America from Asia and was first recorded in the United States in 1899 on ships in port in numerous US cities (Antolin *et al.,* 2002). The bacterium was first reported to affect prairie dogs around 1945 when plague-positive fleas were discovered in prairie dog burrows in Kansas; two epizootic die-offs occurred around the same time in Texas and Colorado (Cully *et al.,*

2010). Prairie dogs have neither a natural immunity nor adaptive protection against the plague, which often results in 100% mortality rates when plague enters a colony (Coolinge *et al*., 2005).

The mortality rate is not the only impact that plague has on a colony, even after the death of many individuals; plague persists in the colony and results in a longer population recovery time (Mulhern and Knowles, 1997). Between 1999 and 2005, Jack Cully and his colleagues mapped the perimeter of four prairie dog complexes that had a history of plague, and compared this data to two prairie dog complexes that had never been affected by the disease. The researchers found that in areas in which plague had been present, colony sizes were smaller and the distance between colony sites was greater when compared to the sites in which the disease had never been reported. The researchers also found that the proportion of potential habitat that was occupied was less in plague-positive areas when compared to plague-negative areas (Cully *et al*., 2010). Furthermore, Knowles and Knowles (1994) suggested that prairie dogs have continued to exist, despite plague, simply because of their large, highly dispersed populations. Further reductions in population numbers could make prairie dogs much more susceptible to local and regional extinctions (Knowles and Knowles, 1994).

Keystone Role of Prairie Dogs

Since its introduction in 1969 by Robert T. Paine, the term *keystone species* has evolved to describe a species that is exceptional, relative to the rest of the community, in its importance and whose presence is crucial to maintaining the organization and diversity of their respective ecological community (Paine, 1969). Power et al. (1996) further defined this broadly applied term to describe any species whose impact on its

community or ecosystem is disproportionately large relative to its abundance. Kotliar et al. (1999) added that the functional role is not wholly duplicated by other members of the community. Increasingly, many scientists agree that prairie dogs meet the criterion of a keystone species because they are a strongly interactive species that have large effects on their community structure and function, and these effects are disproportionately large relative to their abundance (Kotliar, 1999). Prairie dogs are crucial to the structure and function of native prairie ecosystems, and it is estimated that more than 150 different species of vertebrates, arachnids, protozoans, and invertebrates associate and benefit from prairie dogs and their colony sites (Hoogland, 2006).

Through their burrowing activities, prairie dogs influence grassland ecosystem structure, composition, and function by conspicuously altering landscapes and providing foraging, shelter, and nesting habitat for a diverse array of species (Whicker and Detling, 1988; Hoogland, 2006). Within colonies, black-tailed prairie dogs excavate an elaborate system of deep burrows that cover several acres to hundreds of acres of land. The burrow entrances are characterized by large rim craters—some measuring up to 12 centimeters high and two meters in diameter—that serve to prevent flooding, improve underground ventilation, and provide a vantage point when scanning for predators (Hoogland, 1995). Burrows provide nesting and shelter sites for many grassland species, such as burrowing owls (*Athene cunicularia*) and tiger salamanders (*Ambystoma tigrinum*). Prairie dogs also serve as prey for a number of predators, including American badgers (*Taxidea taxus*), coyotes (*Canis latrans*), ferruginous hawks (*Buteo regalis*) and the Black-footed ferret (*Mustela nigripes*), which is an endangered species (Whicker and Detling, 1988; Hoogland, 2006).

In addition to their burrowing activities, prairie dogs facilitate predator detection by voiding the area surrounding their burrow entrances of any vegetation that is higher than 30 centimeters. These activities create an open habitat that is preferred by species such as Mountain plovers (*Charadrius montanus*), and alters species composition and structure of plant communities (Kotliar, 1999). Through grazing and clipping activities, prairie dogs create a distinct and recognizable disturbance patches that are characterized by minimal woody vegetation and short, clipped grasses. It has been suggested that the decline and removal of prairie dogs may have played a role in grassland conversion to shrublands and woodland vegetation states (Johnson and Delay, 2000). Everett (2002) found shrub cover was 7.5 times higher outside of prairie dog colonies than inside of colonies on Thunder Basin National Grassland in Wyoming. Similarly, mesquite (*Prosopis* spp.) increased from 27% to 61% of the cover in the first 23 years after prairie dogs were removed from an area in Texas (Weltzin *et al*., 1997). In Chihuahua, List (1997) documented that there was a 14% increase in mesquite in the first eight years after a prairie dog colony was poisoned.

Historic Range and Variability

Little question exists that the loss of native prairie has had a profound effect on grassland communities; however, conservation efforts have been hampered by limited knowledge regarding the historical conditions in which the grasslands evolved. The structure, composition, and function of today's prairies bear little resemblance to those of the past. A comparison of the historical context of the prairies provides a baseline for information regarding the frequency and extent of major ecological drivers, as well as an indication of the presence and impact of disturbances. This information can serve as a

foundation for the development of conservation strategies that maintain and restore grassland ecosystem communities and the ecological processes that maintain them (Samson *et al*., 2004).

Historic range and variability (HRV) is the full variation and range of conditions, of historical ecosystem characteristics and processes occurring across multiple scales of time and space (Kean *et al.,* 2009). HRV is a relatively new concept that is based on the idea that historical variation and range of conditions provides a representative time series of reference conditions to guide land management (Aplet and Keeton, 1999; Kean *et al*., 2009). The ultimate goal of ecosystem management is said to be a healthy, sustainable ecosystem that can maintain its structure and organization through time (Whitford and DeSoyza, 1999; Kean *et al*., 2009). To achieve this goal, land must be managed as a whole—all organisms, patterns of abundance, and connectivity of their habitats, and the ecological processes that influence these organisms on the landscape must be taken into consideration (Kean *et al*., 2009). Furthermore, the historical conditions and variations that represent the broad envelope of conditions that support landscape resilience and self-organization, must also be considered (Swetnam *et al*., 1999; Kean *et al.*, 2009). Analysis of HRV can help identify and describe dynamic changes in ecosystems, such as vegetation types that were important historically, but are less abundant today.

Ecological drivers, such as drought, fire, and grazing, significantly influence the composition and distribution of grassland communities (Knopf *et al.*, 2004). Two assumptions of HRV are: (1) ecosystems are dynamic, and their responses to changing processes are represented by past variability, and (2) ecosystems are complex and have a range of conditions within which they are self-sustaining and beyond this range they

transition to disequilibrium (Kean *et al*, 2009). Based on these assumptions, we should expect that when ecological drivers act with characteristic behavior, ecosystems will exhibit characteristic composition and behavior (Samson *et al.,* 2004). Often times, a degraded system can be restored along successional pathways once the historical physical environment is reestablished. In other cases; however, changes in landscape connectivity and organization, loss of native species pools, shifts in species dominance, trophic interactions and/or invasion by exotics, and concomitant effects on biogeochemical processes can push a degraded system to a persistent, alternative state that is resilient to traditional restoration techniques (Suding *et al.,* 2004).

The ecological phenomena that drive ecosystem structure, composition, and function normally vary within bounded ranges; however, if threshold values are exceeded, rapid, nonlinear changes to an alternative state may occur. Once an ecological threshold is crossed, an ecosystem loses its ability to return to the original steady-state after disturbance or change by external forces, and is deflected toward a new state (Burkett *et al.,* 2005). Analysis of HRV may be useful in guiding land managers to restore and maintain the ecosystem conditions that sustained biological diversity prior to the dramatic changes that occurred after European settlement of the Great Plains, especially if ecological thresholds have not been crossed.

Historic Grazing and Fire Regimes within the Grasslands

Drought, grazing, fire, and the grasslands of the Great Plains have a long relationship that evolved over the course of several million years. Increased aridity during the Miocene-Pliocene transition (7-5 million years ago) gave rise to the North American Grasslands (Anderson, 1990). During this dry period, grasses, which are better

adapted to drought, spread at the expense of forest vegetation and the number of grazing ungulates rose (Axelrod, 1985; Anderson, 1990). Disturbance from grazing and fire impact both the spatial and temporal patterns of animal and plant communities in grassland ecosystems. Historically, native herbivores moved nomadically in response to changes in vegetation associated with fire and precipitation patterns (Samson *et al*., 2004). According to historic archaeological records, the time lag for return to previously grazed areas was estimated to be 1-8 years, which allowed for natural rest and re-growth of the vegetation (Irby *et al.*, 2002; Samson *et al*., 2004). The interaction of fire and grazing led to a natural habitat mosaic of short, mid and tall seral stages (Samson *et al.*, 2004).

After European settlement of the Great Plains, destruction of bison, an increase in domestic livestock, fragmentation of the landscape, and fire suppression resulted in dramatic changes in the number and distribution of native herbivores (Hartnett *et al*., 1997; Samson *et al*., 2004). Furthermore, current fire regimes bear little resemblance to those of the past, with fire size and frequency decreasing significantly since the 1800s (Bahre, 1991; Ford and McPherson, 1996). Although it is known that Native Americans used grassland fires as a management tool to modify habitat and to aid in driving and attracting wild game during their hunts, (Bahre, 1985; Pyne, 1982; Ford and McPherson, 1996) historic fire frequency and extents cannot be accurately estimated due to the absence of direct evidence from fire-scarred trees. Our best estimates of historic fire frequencies in the Great Plains come from studies in which charcoal fragments were taken from lake sediment cores to construct historical records of fire frequency in the northern Great Plains. This research indicated that post-settlement charcoal deposition

was much lower than pre-settlement, suggesting that fire frequency decreased after European settlement of the Great Plains (Umbanhower, 1996; Ford and McPherson, 1996).

Post-settlement decline in fire frequency and size can most likely be attributed to active fire suppression by the European settlers and the removal of fine fuel by overgrazing. Fragmentation and habitat loss from agriculture and urban development have also affected the natural fire process in which grasslands evolved (Gottfried *et al*., 1995; Hartnett *et al*., 1997; Frank *et al*., 1998; Brockway *et al.*, 2002). Fragmentation, cultivation, and cattle grazing substantially reduced the amounts of aboveground biomass that normally serves as fine fuel for fire (McGinnies *et al*., 1991; Frank *et al*., 1998; Hart and Hart, 1997; Brockway *et al.*, 2002). Along with fire suppression programs that were implemented in the 1950s, the decrease in standing biomass and fragmented prairie landscape decreased the probability of ignition and spread of grassland fires, virtually eliminating fire in modern prairie ecosystems (Brockway *et al*., 2002). The interaction of fire with the grasslands is vital to sustaining ecosystem integrity; however, uncertainty exists concerning the season, frequency, and methods that would be most beneficial in restoring fire in Shortgrass steppe (Mutch, 1994; Brockway *et al.*, 2002).

Importance of Fire in Grassland Ecosystems

Fire, a commonly recognized component of the disturbance regime in grassland ecosystems, disrupts community structure and changes the physical environment, resources, or availability of space. Fire in the grasslands is heavily influenced by climatic variability; precipitation, temperature, wind, and lighting all play a role in determining

fuel dynamics and ignition rates (Swetnam and Betancourt, 1997). Grassland fire provides numerous benefits by increasing the rate of nutrient turnover, regulating plant communities, reducing woody species, suppressing the growth of fire-intolerant plants, and discouraging invasion of non-native species (Pyne, 1982; McPherson, 1997; DeBano *et al*., 1998; Brockway *et al*., 2002). Numerous studies (Anderson, 1982; Dyer *et al*., 1982; Knapp and Seastedt, 1986) suggest that productivity increases in grassland ecosystems when plant litter is removed by fire, light grazing, or mechanical removal. Grassland plants in the Great Plains have evolved alongside periodic drought, fire, and grazing and have adapted the ability to die down to underground organs, leaving only dead tops exposed aboveground. The advantage to having their growing points beneath the surface of the soil allows grasses to avoid desiccation during drought, insulation from heat during a fire, and from below ground tissue removal during grazing (Gleason 1922, Anderson 2006). Furthermore, research by Golley and Golley (1972) suggests that an adaptation to grazing resulted in grasses producing more biomass than can be decomposed of; therefore, periodic fire and light grazing are necessary to maintaining a healthy grassland system.

Relationship between Prairie Dogs and Fire

Prairie dog burrowing and grazing activities significantly influence grassland ecosystem function and composition; therefore, factors that influence their distribution and abundance are of great importance to rangeland managers (Augustine *et al*., 2007). Disturbance from fire is known to influence the spatial distribution of many plant and animal species in shortgrass steppe ecosytems; however, the degree to which fire

influences prairie dog colonies in not well known (Brockway *et al*., 2002; Ford and Johnson, 2006, Augustine *et al*., 2007).

Prairie dog colonization occurs mainly during the spring in open, flat grasslands with minimal woody vegetation (King, 1955; Koford, 1958; Butler, 1995). Previous studies, by Milne-Laux and Sweitzer (2006) and Northcott and his colleagues (2008) have shown that manipulations of vegetation, by fire and brush removal, are conducive to prairie dog colony expansion, which occurred and was directed towards experimental plots that had been treated by fire. In 2007, Augustine et al., examined the expansion rates of prairie dog colonies into two areas that had been affected by a prescribed burn, and one that had been affected by a wildfire, in the shortgrass steppe of the Comanche National Grassland in Baca County, CO. The results of this study showed that the expansion of black-tailed prairie dog colonies into burned areas of the shortgrass steppe was twice the expansion rate into unburned areas. The authors also noted that the study was conducted during a period of colony expansion associated with drought conditions, which further allowed them to conclude that burning in the shortgrass steppe can encourage colony expansion, even under dry conditions (Augustine *et al.,* 2007).

These previous studies have shown that interaction of fire with shortgrass steppe is important for prairie dog colony expansion because it discourages the growth of dense and tall vegetation—areas in which prairie dogs have a difficult time colonizing (Garrett and Franklin, 1988; Wolff, 1999). Augustine et al. and others (Knapp *et al.,* 1999) also suggest that mammalian herbivores are attracted to post-fire growth that is of enhanced forage quality, and this may be another factor that enhances prairie dog expansion into burned areas.

Kiowa and Rita Blanca National Grasslands Site Description

In 1960, Congress designated 136,562 acres of New Mexico land in Mora, Harding, and Union Counties as the Kiowa National Grassland, and 77,183 acres of land in Dallam County Texas and 15,639 acres of land in Cimarron County Oklahoma as The Rita Blanca National Grassland (USDA Forest Service). These Grasslands, one of twenty publicly owned National Grasslands administered by the USDA Forest Service, encompass approximately 230,000 acres of numerous small government parcels that are intermingled with privately owned tracts of land. Although they are managed as one, the Kiowa and Rita Blanca Grasslands are separated into three geographic units—eastern and western parts of the Kiowa National Grassland in New Mexico, and the Rita Blanca National Grassland in Oklahoma and Texas (USDA Forest Service).

The Kiowa and Rita Blanca are semiarid, mostly treeless, grasslands dominated by grasses and forbs, and are characterized by large grazing mammals and burrowing animals. These grasslands are also home to a variety of arthropods, birds, and mammals that characterize the southern Great Plains and play important roles in ecosystem functioning of the shortgrass prairie (Ford and McPherson, 1996). Our study site, K46, lies within the shortgrass steppe of the eastern section of the Kiowa National Grasslands, which falls entirely within Union County, New Mexico. The steppe habitat is the most extensive habitat type in these grasslands, and is characterized by relatively level plains with undulating hills that are covered by mid-grass prairie and shortgrass steppe vegetation (Ford and McPherson, 1996). For northeastern New Mexico, annual precipitation is approximated at 15 inches with a bimodal distribution in precipitation occurring with an increase in rainfall in April and a second pulse, associated with a

monsoon effect, occurring in July and August. Temperatures average 70 °F between June and September in the summer months and 35°F during the winter months (Hazlett *et al.*, 2009; Ford and McPherson, 1996).

Previous Research on Study Site K46

Research on a long-term experimental fire research site, K46, on the Kiowa National Grasslands has shown that there are significantly different responses by plant and animal species to the season and frequency of fire. The 18-year study was initiated in 1995 by Brockway to examine the effects of prescribed fire in the Shortgrass steppe in northeastern New Mexico (36°31' 20" N, 103° 3' 30"W), during the growing season and the dormant season at three, six, and nine year intervals (Ford, 2007). Although this region has been altered by grazing, fragmentation, and fire suppression, it is still dominated by mixed grass or short grass communities (Brown, 1994; Ford, 2007). Thus far, the study has shown differential responses (positive, negative or unaffected), by small mammals to fire, based largely on life history requirements. Mammals that responded negatively to fire were those that live and forage in dense vegetation and use plant debris for their nests. In this situation, fire removes both their food source and habitat—which elicits a negative response (Kaufman *et al.*, 1990; Ford, 2007). In contrast, those mammals that respond positively to fire are those that inhabit open areas, feed on insects that are attracted to new vegetation growth, and live in burrows, which offer protection from the fire (Ford, 2007).

Ford and Johnson (1997), researchers from the USDA Forest Service Rocky Mountain Research Station, examined the effects of the dormant and growing-season burns on biological soil crusts and perennial grasses on K46, the long-term experimental

fire research site on the Kiowa National Grasslands. Through their analysis, the authors found that growing-season burns reduced fire severity and had less of an impact on soil crusts; however, burning during the growing-season negatively impacted grass cover for up to two years after the fire. The dormant season fire had a larger impact on biological soil crusts, but the grass cover recovered in as little as two months after the burn. Based on these results, the researchers were able to conclude that the Shortgrass steppe is able to recover from fire within three to 30 months, depending on the season. The authors also note that the Shortgrass steppe was in a drought during the course of this study, and warn that weather patterns must be taken into consideration when deciding on an appropriate return interval for prescribed burns, which may affect the recovery time of the vegetation and soils after a burn. These findings provide a strong basis for predictions regarding which fire treatment will most facilitate prairie dog colony expansion.

Direct Observation and Aerial Survey Methods

Methods for estimating the area inhabited by prairie dogs vary. Two common methods involve the use of aerial or satellite imagery and mapping software to trace polygons around the colony. The first method involves connecting the outermost burrow entrances to create a polygon around the perimeter of the colony. This method can be both time-consuming and difficult because the outermost burrow entrances tend to have small mounds, and the vegetation is often higher in comparison to the central portion of the colony (King, 1955; Hoogland, 1995). A second method for estimating the area involves tracing a polygon around the *clip-zone,* the central area surrounding a colony site where vegetation is clipped low, either for consumption or to enhance predator detection. Tracing the clip-zone can be difficult during periods of drought because

grazing is especially heavy, and it is hard to discern between bare ground that is created by drought and bare ground that is created by heavy grazing and clipping (King, 1955; Hoogland, 1995).

Of the two methods, connecting the outermost burrow entrances often yields more accurate estimates; however, tracing the clip-zone is often quicker and more visible in aerial and satellite imagery (Hoogland, 1995). For smaller areas, such as national parks, national wildlife refuges, and Native American reservations, the boundaries of prairie dog colonies can be delineated, on foot, by marking the outermost burrow entrances with a Global Positioning System (GPS) receiver. Although this method is the most accurate, it is also the most time consuming and not possible in areas where access is not permitted (Hoogland, 2006).

By viewing ASCS black and white aerial photos, Cheatheam (1973), was the first to utilize aerial photographs to census prairie dog colony abundance. Cheatheam assessed the effectiveness of using aerial photos to survey prairie dog colonies in the Texas Panhandle by ground verifying, contacting landowners, and by examining inhabited areas by flying low to the ground. The results of this study showed that 88-94% of the colonies that were detectable in the aerial photos were actually present on the ground, although some limitations to this method were found. First, small colonies between 0.5-4 ha, have little vegetation removed, and are harder to detect in aerial photos. Second, some of the colonies detected on the photos were found to be inactive, due to disease or other factors, when they were ground verified. Despite these limitations, Cheatheam found aerial photograph analysis to be a cost effective and time saving method for estimating large prairie dog populations. In contrast, Sidle et al. (2012) found that aerial surveys without

ground-truthing overestimated prairie dog numbers by 94% on Pawnee and Comanche National Grasslands.

Schenbeck and Myhre (1986) analyzed aerial photographs to assess pre- and post-treatment effects of zinc phosphide poison on a prairie dog town on the Buffalo Gap National Grassland in South Dakota. The pre-treatment photos were pre-existing black and white photos and post-treatment photos were color infrared (CIR) aerial photos. The researchers delineated the colonies by viewing the mounds and outlining them; transferred colonies to topographic maps using a zoom transfer scope; and measured colony area using a planimeter. This technique was found to be a cost-effective method for viewing overall population changes in black-tailed prairie dogs.

In 2010, researchers from the University of New Mexico Biology Department, utilized a digital orthophoto quarter quadrangle (DOQQ) survey method to survey and monitor the distribution of prairie dog colonies across eastern New Mexico. DOQQ panchromatic air photos collected in 1996 and 1997 were surveyed, and 24,400 ha of Black-tailed prairie dog disturbance was revealed—with 89% of towns still identifiable on the ground in a field check survey in 2003. To determine changes between the 1996-97 and 2005, gains and losses of prairie dog colonization were identified and major shifts in the spatial distribution were determined. The results of their comparison between the 1996-97 and 2005 DOQQs allowed the authors to determine that the DOQQ survey method is most beneficial for the detection, depiction, and the potential for spatial analysis of the distribution of towns over an area (Johnson *et al.*, 2010).

METHODS AND MATERIALS

Study Site Location

The Kiowa and Rita Blanca National Grasslands encompass approximately 93,100 hectares (230,000 acres) of Shortgrass prairie in the southern Great Plains (Fig. 1) In six counties within New Mexico, Texas, and Oklahoma, this grassland ranger district consists of numerous, small Government parcels that are intermingled with privately owned tracts of land (USDA Forest Service, 2012). Unit K46 (Fig. 2), the Rocky Mountain Research Station Kiowa Long-term Experimental Fire Research Site (36°31' 20" N, 103° 3' 30" W), occupies a total of 160 hectares (395 acres) of the Cibola National Forest in Union County, New Mexico (Ford, 2007). The northern boundary of K46 borders a tract of privately owned land that is used for agriculture, while the western boundary is bordered by a fence that separates the site from another grassland parcel. The eastern and southern boundaries of the study site are bordered by Country Road A077 and New Mexico State Road 411, respectively (Fig. 3).

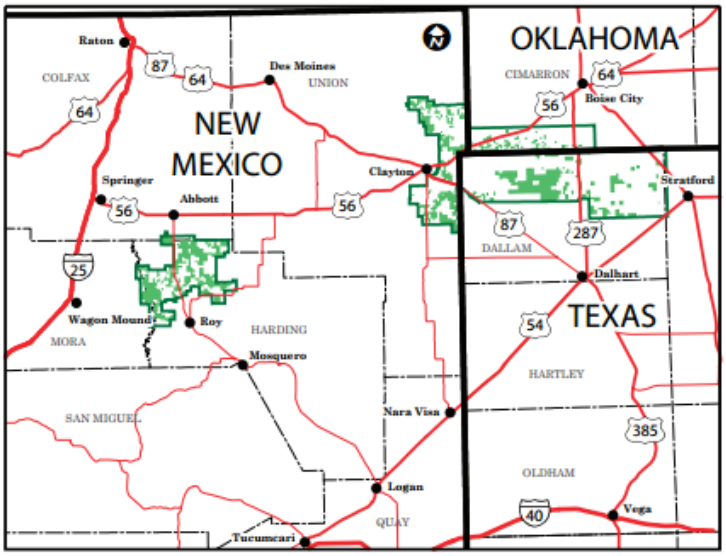

Figure 1 - USFS Map of the Kiowa and Rita Blanca National Grasslands

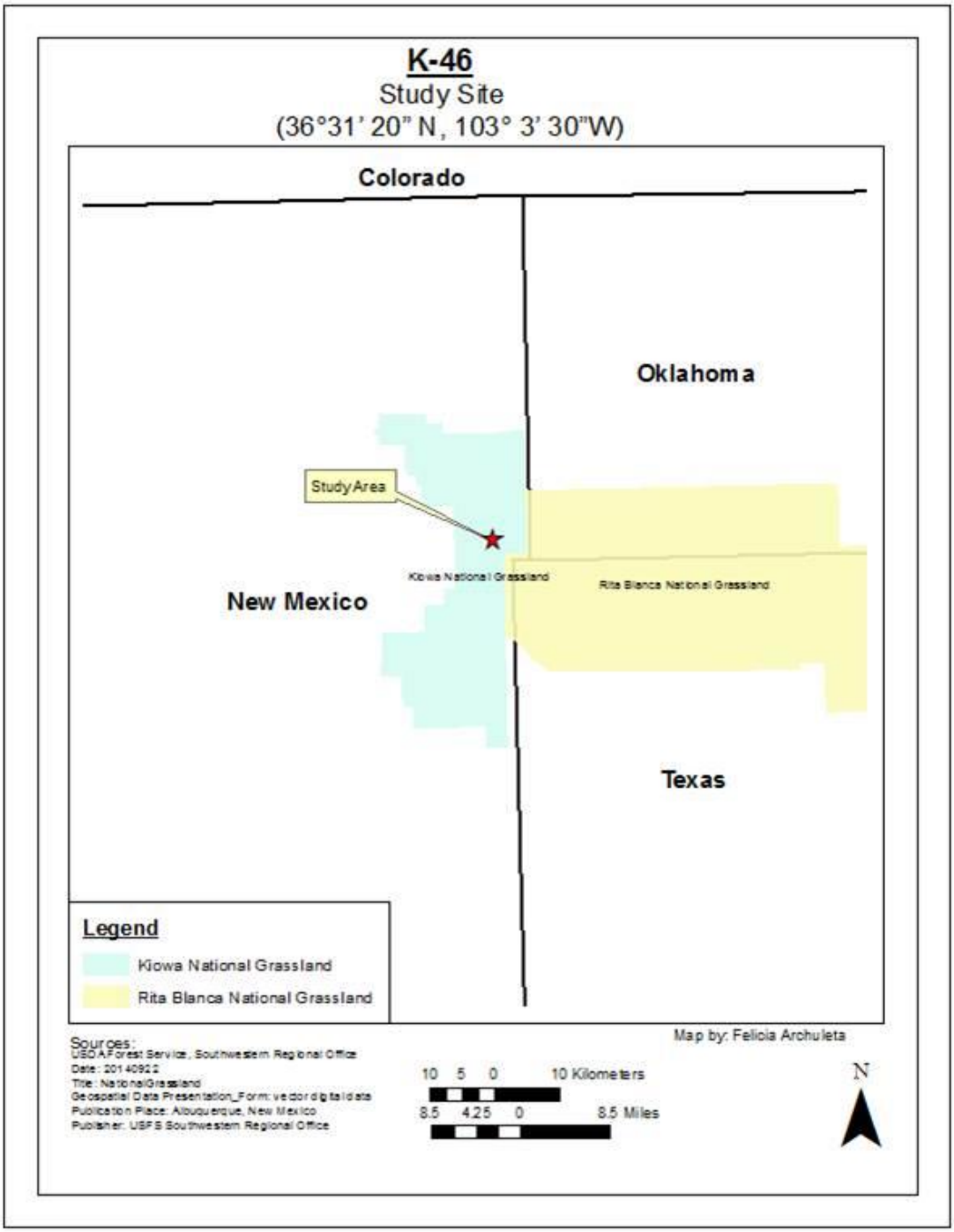

Figure 2 - Map of the Study Site Location. Map showing the study site location within the eastern Kiowa National Grassland (shaded area within New Mexico territory).

Figure 3 - Location Map of K46 and Boundaries. The northern boundary of K-46 borders a tract of privately owned land used for agriculture, while the western boundary is bordered by a fence that separates the site from another grassland parcel. The eastern and southern boundaries of the study site are bordered by Country Road A077 and New Mexico State Road 411, respectively.

Site Description

The most extensive grassland vegetation type on K-46 is Shortgrass steppe, which accounts for approximately 50 percent of New Mexico's grassland vegetation (Ford, 2006; Dick-Peddie, 1993). Although the study site was grazed by cattle until 1990, it has never been plowed and is relatively homogenous and flat with an elevation of 1472 m at the northwest corner to 1455 m at the southwest corner (Ford and Johnson, 2006). Biannual average temperatures range from 11 to 26^{o}C April through September, and from

3 to 12^{o}C October through March. The majority of the precipitation received occurs from May through September, with peak rainfall occurring in July. Mean annual precipitation (MAP), Fig. 8 in Appendix 4, was approximately 356 mm from 1931-1960; however, the total precipitation throughout the course of the study has been measured as low as 50 percent below MAP. Drought conditions, ranging from moderate to exceptional, existed in 2003 and again from 2011-2013 (Ford and Johnson, 2006).

Experimental Design

The experimental design is a completely randomized application of seven treatments and five replicates. Beginning in 1997, dormant and growing-season burns were planned to be performed at three, six and nine year intervals (Fig. 4). Growing season burns were applied in July, dormant season burns were applied in April, and control plots were left unburned (Brockway, 1995; Ford, 2006). The treatments were each replicated 4-5 times and were randomly assigned to 35-140m x 140m plots with 60m of unburned area separating the plots. Metal fence posts were set in the corners of each plot and labeled with the assigned plot number to designate the boundaries. ‘‘Black lines’’ were burned around the inside perimeter of each burn plot using drip torches, and the interior of each plot was burned using a strip headfire. (Ford, 2006).

Overall, five rounds of experimental fire treatments were completed, beginning in 1997 when the first treatments were applied. After the initial burn treatments, some burns had to be rescheduled due to drought and fire restrictions or due to too much rain. Further fire treatments were suspended indefinitely in 2013 due to drought. Throughout the course of the study, the 3 year, dormant season plots (3D) were burned 5 times, the 6 year dormant season plots (6D) were burned 3 times, and the 9 year dormant season plots

(9D) were burned twice. The 3 year growing season plots (3G) were burned 4 times, and both the 6 year growing season plots (6G) and the 9 year growing season plots (9G) were burned twice.

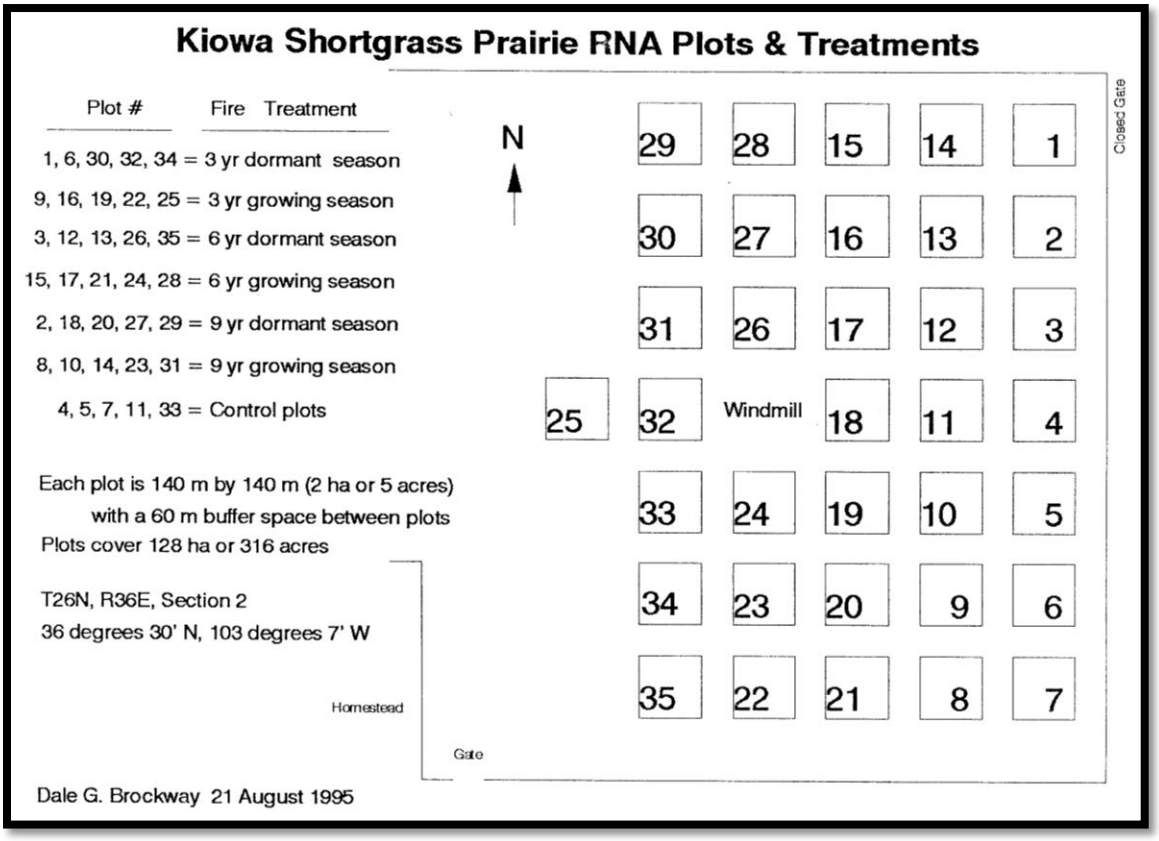

Figure 4 - Map of Designated Plot Numbers and Treatment Assignments

Survey Methods

To determine how the prescribed fire treatments have affected Black-tailed prairie dog colony expansion, we examined the location, size, and pattern of the colony within the study area over the course of eight years, 2005, 2006, 2009, 2010, 2011, 2012, 2013, and 2014. In order to make use of all available data, two well established survey methods were used to map the spatial location and size of the colony boundary on K46. The first

method, direct observation, involves the use of a GPS receiver and direct observation to record the coordinates of the outermost burrow entrances of the colony. Although the direct observation method is the most accurate, data was not available for 2012 and 2013; therefore, an additional method involving visual interpretation of high resolution satellite imagery was employed to estimate the location and size of the colony boundary during these years. These survey methods are very well established, and were used to track the location, range, and movement of the colony on the research site, to better understand which, if any, treatment plots were preferred for colonization.

Direct Observation

Polygon layers (ArcView shapefiles) of the colony boundary on the study site were obtained from the USFS Kiowa and Rita Blanca National Grasslands. The USFS developed these files based on data that was collected by Jack Cully of the US Geological Survey Biological Survey at Kansas State University. To map prairie dog colonies, which were accessed and traversed with an ATV, Cully used a hand-held Trimble GeoExplorer3 GPS unit set to obtain positional readings every second. In order to show year-to-year changes in abundance of prairie dogs, only the active areas of each colony were mapped. Burrow entrances that were actively in use by the prairie dogs were distinguished from inactive burrow entrances by examining them for the presence of fresh digging, tracks, scat, and visual identification. The GPS coordinates of each of the active burrow entrances were recorded and uploaded to Environmental Systems Research Institute, Inc. (ESRI) ArcMap^{TM} software to create a shapefile. The shapfiles that were created from the GPS locations were collected in 2005, 2006, 2009, 2010, and 2011. In addition, with the use of a hand-held Garmin GPSMAP 60Cx GPS unit, we delineated the 2014 colony

boundary by walking the study area and recording the GPS locations of the outermost, active burrow entrances.

Image Analysis

In aerial and satellite imagery, Black-tailed prairie dog mounds can be seen as bright, roughly circular spots that typically are clumped spatially and surrounded by a light halo resulting from vegetation around the mounds being clipped by the prairie dogs. Images from 2012 and 2013, which were purchased from DigitalGlobe Inc., were examined in ESRI ArcMap™ and ERDAS Imagine software, one screen at a time, moving left to right, then down and right to left, to identify the boundary of the prairie dog colony on the research site. The colony boundary was designated by well-defined mounds and a definite contrast between the site and the surrounding landscape. A polygon was drawn around the boundary in each image by either tracing the clipped-vegetation halo surrounding the mounds, or by tracing the outermost burrow entrances. (Johnson *et al.*, 2010).

The colony boundary in the 2012 image, which was taken on September 10, 2012 by the Pleiades satellite, was identified by tracing a polygon around the *clip-zone*, the central area surrounding a colony site where vegetation is clipped low. Due to the exceptional drought conditions, the colony boundary in the 2013 image, which was taken on February 10, 2013 by the GeoEye satellite, was identified by connecting the outermost burrow entrances to create a polygon around the perimeter of the colony. This method of connecting the outermost burrow entrances visible in the image was used in lieu of creating a polygon around the clip-zone because it was difficult to identify bare ground created by the drought and bare ground created by grazing and clipping.

Mapping Analysis

A digital orthophoto quarter quadrangle (DOQQ) aerial image of the area, which was taken in 2009, was obtained from the New Mexico Resource Geographic Information System (RGIS) online repository and uploaded to ESRI ArcMap™. The boundaries of the treatment plots in this image were visible and could be distinguished from the surrounding vegetation. A polygon was traced around the boundary of each of the 35 treatment plots, plot number labels were applied, and the area of each of plot was calculated. An Arcview shapefile (Fig. 5) with each of the delineated treatments was created to serve as the "input feature" in our clip analysis.

Figure 5 - DOQQ Image Processing. This image shows the DOQQ image that that was processed to create a .shp file that represented the boundaries of each of the 35 treatments and served as the "clip feature" in our analysis

For each year's representative colony boundary (2005, 2006, 2009, 2010, 2011, 2012, 2013, and 2014), we used the clip analysis function in ESRI ArcMap™, to create a new feature class containing a geographic subset of the colony boundary layer and the delineated treatment layer. By assigning the colony boundary polygon as the clip feature

and the delineated treatment plot layer as the input feature, we were able to create a new feature class that contains only those portions of the prairie dog colony that fall within the boundaries of the treatment plots. If a treatment plot was overlapped by the colony boundary polygon, the area of overlap was clipped out and calculated in the new feature class (Fig. 6). We took the area of overlap in each treatment plot and we divided it by the respective treatment plot's total area and multiplied by 100 to get the percentage of coverage in each plot.

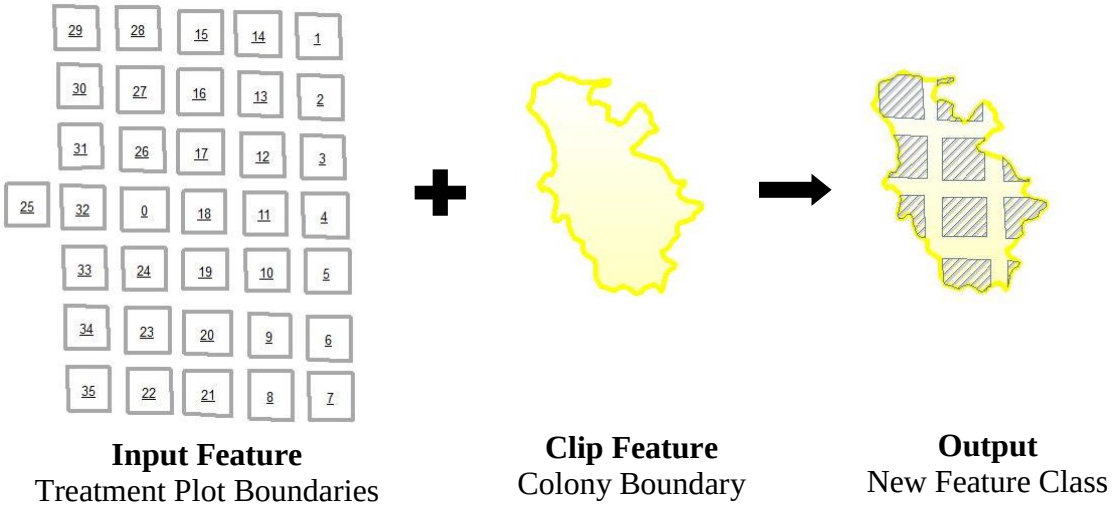

Figure 6 - Illustration of ESRI ArcMap™ Clip Function. This image illustrates the clip function that was used to calculate the total area of overlap/coverage of the prairie dog colony on the treatment plots.

Statistics

Our experiment involves the manipulation of two independent variables with multiple groups. The independent variables are burn season (growing and dormant) and burn frequency (3, 6, and 9 years). Our statistical analysis was performed in order to determine how these two factors affected prairie dog colony expansion, which was our

dependent variable. Our categorical data was coded and entered into IBM SPSS Statistics Software (Appendix 4). Upon running the descriptive statistics of the data, we determined that the data was not normally distributed. Attempts to transform the data and normalize it were not successful; therefore, we determined whether the assumptions of the Mann-Whitney U Test and the Kruskal-Wallis Test were met.

The Mann-Whitney U test is used to compare differences between two independent groups when the dependent variable is either ordinal or continuous, but not normally distributed. Since our independent variable, season, has two independent groups we used the Mann-Whitney U test to determine if the mean area covered by the colony on the experimental research site differs between the growing season and the dormant season burn treatments. The Kruskal-Wallis test is also a nonparametric test that can be used to determine if there are statistically significant differences between two or more groups of an independent variable. The other independent variable in our study, frequency, has three independent groups, so the Kruskal-Wallis test was used to determine if the mean area of coverage by the colony on the experimental research site differs between the three, six, and nine year burn frequency treatments.

RESULTS AND CONCLUSION

Results

Direct Observation and Image Analysis

The final maps and results from our mapping analysis of the prairie dog colony boundaries, which were acquired through direct observation and image analysis are in Appendix 1. The total area of the colony increased every year, with the exception of 2006—which decreased in size from the previous year (Table 1). The results of the colony boundaries that were delineated by direct observation from Jack Cully indicated that in 2005, 2006, 2009, 2010, and 2011 the colony covered approximately 183,016 m^2, 46,505 m^2, 220,497 m^2, 313,152 m^2, and 316,110 m^2, respectively. We also utilized the direct observation method to delineate the colony boundary in April, 2014—which was found to cover approximately 680,183 m^2. Moreover, through image analysis, we were able to delineate the colony boundary in the 2012 and 2013 satellite imagery. The results of this analysis indicated that the colony covered approximately 596,852 m^2 in 2012 and 621,528 m^2 in 2013.

Table 1 - Total Area (m^2) of the Prairie Dog Colony by Year

Year	**Total Area of Prairie Dog Colony (m^2)**
2005	183,016
2006	46,505
2009	220,497
2010	313,152
2011	316,110
2012	596,852
2013	621,528
2014	680,183

Descriptive Statistics and Normality Tests

Analysis that was performed using the clip-function in ArcMap™, gave us the total amount of overlap of the prairie dog colony on the individual treatment plots, results in Appendix 4. When the percentage of overlap was analyzed with the combined seasonal and frequency treatments, it was found that the mean percentage of coverage/overlap of the prairie dog colony was highest on the 9 year burn frequency plots that were burned during the dormant season. This analysis also indicated that the lowest mean percentage of coverage occurred on the 3 year burn frequency plots that were burned during the dormant season, Figure 15 in Appendix 2.

When the treatments were split by season, results shown in Table 2, there were 80 instances in which the prairie dog colony overlapped the growing season treatment plots, and 66 instances in which the colony overlapped the dormant season plots. The percent of coverage/overlap by the prairie dog colony was higher in the dormant season plots, with an average of 64.53 percent (SD = 37.40). The average coverage of the prairie dog colony on the growing season plots was 60.59 percent, (SD=36.76). The assumption of normality was tested via examination of the unstandardized residuals. Review of the Kolmogorov-Smirnov tests for normality, Table 3, indicate that the scores recorded for the percent of coverage/overlap of the prairie dog colony on the growing season plots, $D(80)=0.166$, $p<.001$, deviated significantly from a normal distribution. Scores for the dormant season, $D(66)=0.235$, $p<.001$, also deviated significantly from a normal distribution. The Shapiro-Wilk test results also indicated that both the growing and dormant season scores deviated significantly from normal with $p<.001$ for both.

When the treatments were split by burn frequency, results in Table 4, there were 42 instances in which the prairie dog colony overlapped the control plots, 27 in which the colony overlapped the three year burn plots, 39 for the six year burns, and 38 for the nine year burn treatments. Furthermore, the average percent of coverage/overlap by the prairie dog colony was 66.91 in the control plots, (*SD* = 33.03), which was only slightly higher than the average percent of coverage in the 9 year plots at 63.05, (*SD*= 40.14). The average percent of coverage was lowest in the 3 year treatments at 56.3 percent, (*SD*=41.29), and was slightly higher in the 6 year treatments at 60.97 percent, (*SD*=35.46). This analysis also indicated that the data for all four of the frequency treatments, was negatively skewed—indicating a build-up of high scores. The data also exhibited negative kurtosis for all of the frequency treatments. Furthermore, Kolmogorov-Smirnov tests for normality, in Table 5, indicate that the scores recorded for the percent of coverage/overlap of the prairie dog colony on the treatment plots deviated from a normal distribution for each of the frequency treatments. $D(42)=0.262$, $p<.001$, $D(27)=.244$, $p<.001$, $D(39)=.146$, $p=.036$, and $D(38)=.227$, $p<.001$ for the control, 3 year frequency, 6 year frequency, and 9 year frequency treatments, respectively. The Shapiro-Wilk test results also indicated that all of the burn frequency scores deviated significantly from normal.

Further review of the histograms (Appendix 2) and normal Q-Q plots (Appendix 3), also suggest a non-normal distribution of the data for all four frequency treatment types and both seasonal treatment types. The histograms all show a deviation from a normal bell-shaped distribution, and Q-Q plots all indicate that the scores deviate from the normal.

Table 2 - Descriptive Statistics with percentage of cover/overlap as the dependent variable and season as the independent variable

			Growing Season N=80		Dormant Season N=66	
			Statistic	Std. Error	Statistic	Std. Error
Percent Coverage	Mean		60.5988	4.11286	64.5273	4.60386
	95% Confidence Interval for Mean	Lower Bound	52.4123		55.3327	
		Upper Bound	68.7852		73.7218	
	5% Trimmed Mean		61.7639		66.1348	
	Median		67.5000		73.0500	
	Variance		1353.247		1398.907	
	Std. Deviation		36.78650		37.40197	
	Minimum		.00		.10	
	Maximum		100.00		100.00	
	Range		100.00		99.90	
	Interquartile Range		78.58		69.90	
	Skewness		-.352	.269	-.502	.295
	Kurtosis		-1.442	.532	-1.344	.582

Table 3 - Normality Tests with percentage of cover/overlap as the dependent variable and season as the independent variable

		Kolmogorov-Smirnov[a]			Shapiro-Wilk		
	Season	Statistic	df	Sig.	Statistic	Df	Sig.
Percent Coverage	Growing	.166	80	.000	.857	80	.000
	Dormant	.235	66	.000	.821	66	.000

Table 4 - Descriptive Statistics with percentage of cover/overlap as the dependent variable and frequency as the independent variable

			Control		3 year		6 year		9 year	
			N=42		N=27		N=39		N=38	
			Stat.	Std. Error	Stat.	Std. Error	Stat.	Std. Error	Stat.	Std. Error
% Cover	Mean		66.91	5.097	56.37	7.94	60.98	5.678	63.05	6.512
	95% C.I. for Mean	Upper Bound	56.62		40.04		49.48		49.86	
		Lower Bound	77.21		72.71		72.47		76.25	
	5% Trimmed Mean		68.19		57.01		62.20		64.50	
	Median		67.20		61.50		69.20		83.55	
	Variance		1091		1704		1258		1611	
	Std. Deviation		33.03		41.29		35.46		40.14	
	Minimum		10.80		.20		.00		.10	
	Maximum		100.0		100.0		100.0		100.0	
	Range		89.20		99.80		100.0		99.90	
	Interquartile Range		59.63		86.80		67.10		78.52	
	Skewness		-.398	.365	-.103	.448	-.469	.378	-.544	.383
	Kurtosis		-1.29	.717	-1.88	.872	-1.16	.741	-1.51	.750

Table 5 - Tests of Normality with percentage of cover/overlap as the dependent variable and frequency as the independent variable

		Kolmogorov-Smirnov			**Shapiro-Wilk**		
	Frequency	**Statistic**	**df**	**Sig.**	**Statistic**	**df**	**Sig.**
Percent Cover	Control	.262	42	.000	.834	42	.000
	3 year	.244	27	.000	.800	27	.000
	6 year	.146	39	.036	.882	39	.001
	9 year	.227	38	.000	.787	38	.000

Data Transformation and Nonparametric Tests

Attempts to transform the data were not successful in normalizing the burn frequency and seasonality data; therefore, we chose to run non-parametric tests. The results of the nonparametric tests are displayed in Table 6, part a. For the independent variable season, an independent samples Mann-Whitney U Test was used to test whether the distribution of percent coverage of the prairie dog colonies was the same across the dormant and growing season treatments. The results indicated that there was not a significant difference in the distribution of percent coverage across categories of season, p=.516. Furthermore, an independent samples Kruskal-Wallis test was used to test whether the distribution of percent coverage of the prairie dog colonies was the same across frequency treatments. The results, part b. in Table 6, indicated that there was not a significant difference in the distribution of percent coverage across categories of frequency, p=.536.

Table 6 - Non-parametric Test Results. Non-parametric test results for the seasonal and frequency groups.

Independent Variable Group	**Null Hypothesis**	**Test**	**Significance**
a. Season	The distribution of % Cover is the same across categories of season	Independent Samples Mann-Whitney U Test	0.524
b. Frequency	The distribution of % Cover is the same across categories of frequency	Independent Samples Kruskal-Wallis Test	0.556

Conclusion

Based on the results of our mapping and clip analysis, as well as our nonparametric statistical tests, we conclude that there was not a significant difference in the percentage of coverage/overlap of the prairie dog colony amongst any of the burn treatments. When the two treatments groups, season and frequency, were combined, the 9 year burn frequency plots that were burned in the dormant season had the highest mean coverage. The lowest mean percentage of coverage/overlap of the prairie dog colony was on the 3 year burn frequency plots that were burned during the dormant season. When the two treatment types, season and frequency, were split, the results of the Mann-Whitney U test indicated that the percentage of coverage/overlap of the prairie dog colony was not significantly different amongst the seasonal treatments. Therefore, we retained on our null hypothesis: the distribution of percentage of coverage/overlap of the colony was the same across the dormant season and growing season plots. The Kruskal-Wallis test results for the frequency treatments were similar. There was not a significant difference in the percentage of coverage/overlap of the prairie dog colony amongst any of the frequency treatments. Once again, these results led us to retain our null hypothesis: the distribution of percentage coverage/overlap of the colony was the same across the frequency treatments—3 year burns, 6 year burns, 9 year burns, and control/unburned plots.

DISCUSSION

Discussion

Historically, the Black-tailed prairie dog was one of the most conspicuous and characteristic features of the Great Plains. It is estimated that prairie dog colonies once covered between 40 and 60 million ha of shortgrass prairies within their historic range—from southern Canada to northern Mexico, with portions in 12 states (Marsh, 1984). Today, the lands currently occupied by prairie dog colonies are thought to represent less than 2% of their historical range (Anderson *et al*., 1986). Furthermore, recent estimates from the United States Fish and Wildlife Service suggest that the cumulative area occupied by prairie dogs is between 500,000 and 800,000 ha of 160 million ha of potential habitat in their former geographic range (USFWS, 2013). This considerable reduction of prairie dog populations can be directly attributed to human activities. Despite their ability to maintain a broad and relatively stable distribution throughout major climatic shifts during the Pleistocene, prairie dog population numbers and area of occupied habitat began to decline almost immediately after European settlement. Human persecution, habitat destruction, and sylvatic plague continue to pose a threat to the persistence of the black-tailed prairie dog.

The large-scale decline of the prairie dog populations has raised concerns, not only about the long-term survival and population viability of the prairie dogs, but also for the long-term survival and viability of the numerous grassland species that associate and depend on prairie dogs and their colony sites for survival. Agricultural politics and a lack of knowledge regarding the mechanisms for colony expansion have hindered the development of recovery plans that effectively manage for the persistence of prairie dog

colonies. Previous studies have shown that manipulations of vegetation, through prescribed fire and brush removal, encourage and facilitate colony expansion. Numerous other studies suggest that the reestablishment of periodic fire is fundamental to the ecological restoration of grassland ecosystems. A lack of knowledge regarding the historic range of variability of the Southern Great Plains has hindered the development and implementation of prescribed fire regimes that can be used to effectively manage Shortgrass ecosystems.

Prior to reintroducing large-scale fire as a management tool, the appropriate fire season, frequency, and fire effects on ecosystem components need to be determined. Knowledge of prairie dog response to seasonality and frequency of fire may aid in the development and implementation of prescribed fire regimes that can be used to effectively manage Shortgrass ecosystems. Based on results from previous studies, we expected that prairie dog colony expansion would be directed towards plots that have been treated by prescribed fire during the dormant-season. In these studies, dormant-season burns resulted in a quicker recovery of grasses, when compared to growing-season burns. Although the results of our study suggested that there was not a significant difference in prairie dog colonization amongst our season and frequency treatment groups, we did see a higher average in the 9 year, dormant season plots.

Throughout the duration of the study (1995-Today) the research site, along with the entire state of New Mexico, has experienced periods of drought. Between 2011 and 2012, when the drought was classified as exceptional, there was a large exodus of the prairie dog population on to a neighboring site that had been recently burned and had a slight regeneration of new, green forage. Upon returning to the site in April 2014, I

observed a healthy population on the site; however, I also observed that the colony boundary had expanded significantly since the last time it had been measured. Further inspection of satellite imagery confirmed that the colony perimeter now encompasses almost all of K46. When compared to the growth of the colony perimeter in previous years, this new colony perimeter grew larger and faster than what had previously been observed.

Changes in climate can alter fuel loads, available forage, and fire return intervals. These patterns must be taken into consideration when deciding on an appropriate return interval for prescribed burns, which may affect the recovery time of the vegetation and soils after a burn. Swetnam and Betancourt (1997) cited a phenomenon in which increased grass and fine fuel production during wet years are usually followed by large fire events and fewer fuels during dry years result in small fire events. This interaction of climatic variability and fire may explain why colonization amongst the fire treatments was not significantly different. Periodic drought, along with frequent burning, altered the fuel dynamics and available forage of the study site, which may have forced the prairie dogs to expand their colony in a way that allowed them to exploit any available forage. The need to survive and find available forage may be forcing the prairie dogs to expand into areas that they would not typically or preferentially inhabit.

Suggestions for Future Investigators

For future investigators, I suggest splitting the data between periods of drought occurrence and periods of normal precipitation, so that we may see if the effects of drought were muting the effects of the burn frequency and burn season on colonization patterns. Further research on the effects of season and frequency of prescribed fire on

other grassland plant and animal species will also allow for the development of strategies in the use of prescribed fire as a tool to maintain ecosystem health and integrity. Additionally, a long term, large landscape scale investigation of effects of season and frequency of prescribed burns is needed to better understand how fire interacts with, maintains, and sustains shortgrass ecosystems.

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APPENDIX 1

MAPPING ANALYSIS- FINAL MAPS

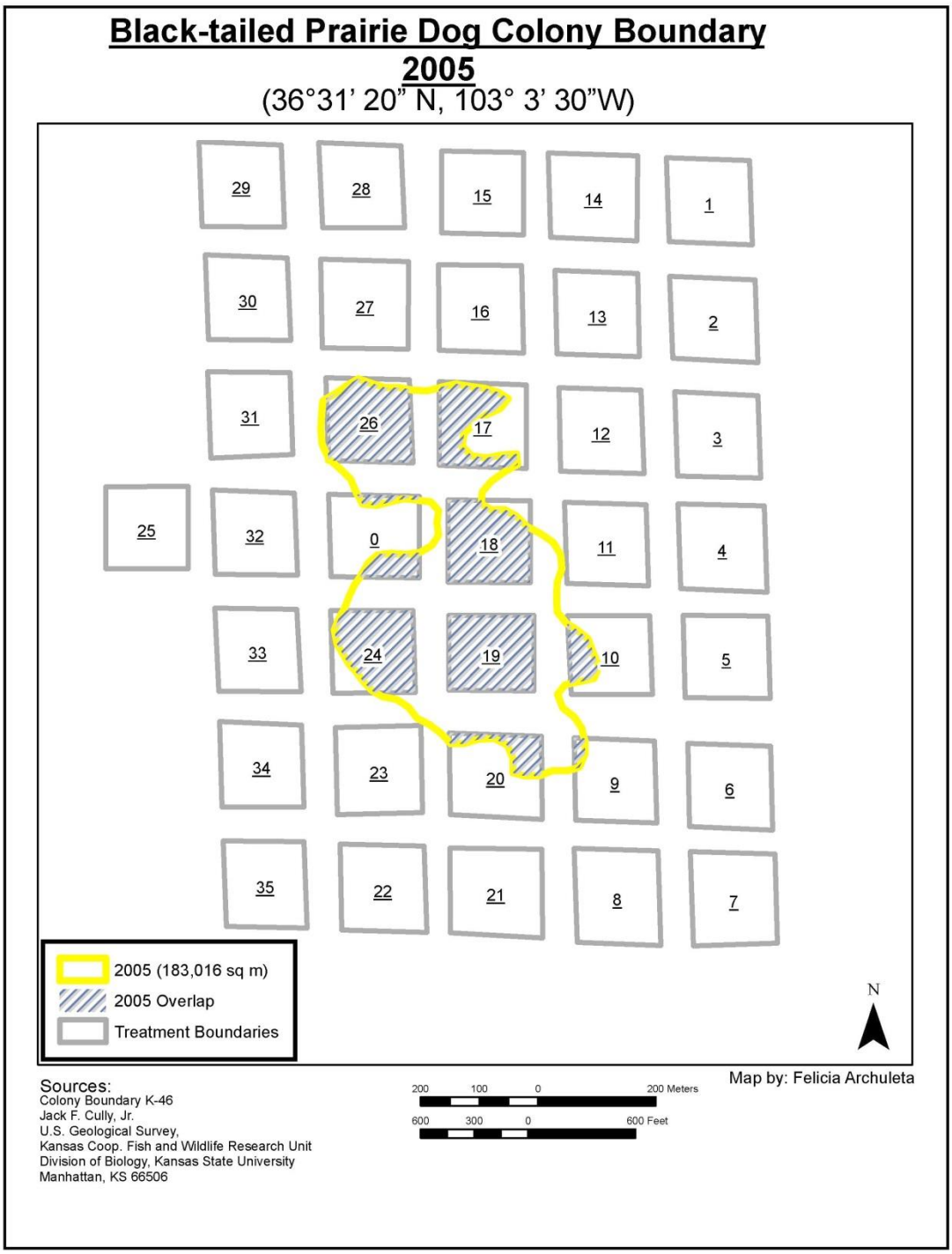

Figure 7 - Map of the 2005 Prairie Dog Colony Boundary. 2005 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray.

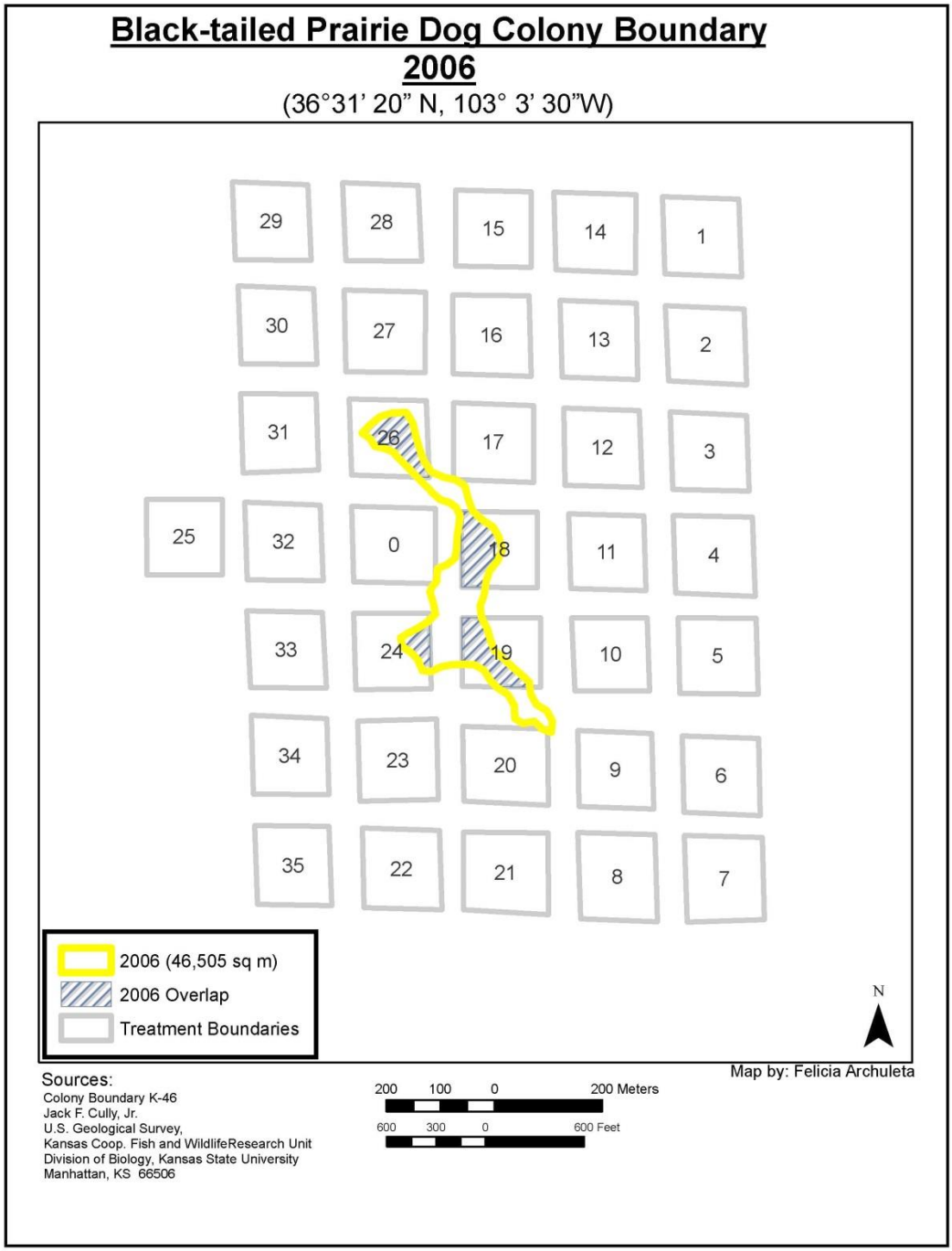

Figure 8 - Map of the 2006 Prairie Dog Colony Boundary. 2006 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray

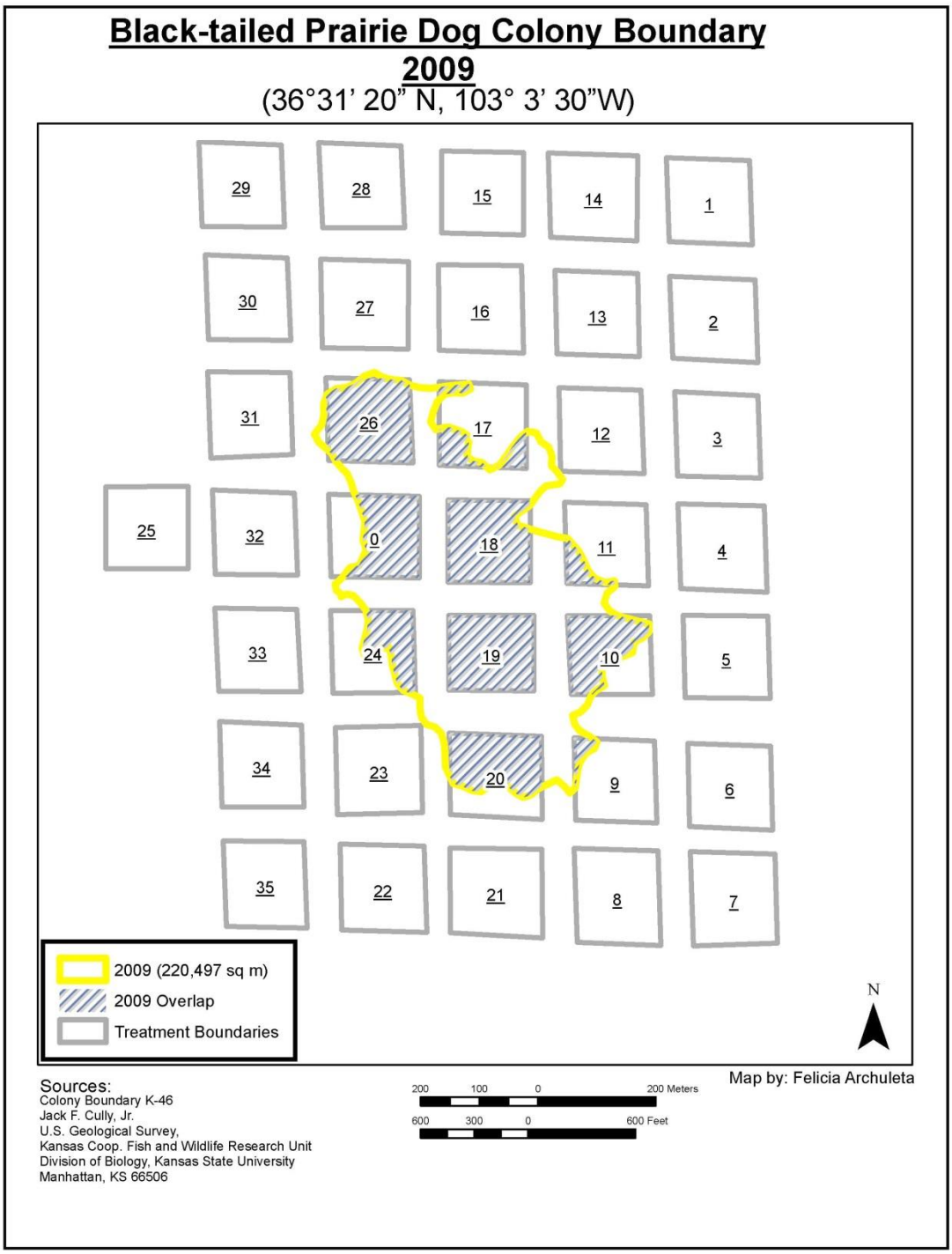

Figure 9 - Map of the 2009 Prairie Dog Colony Boundary. 2009 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray

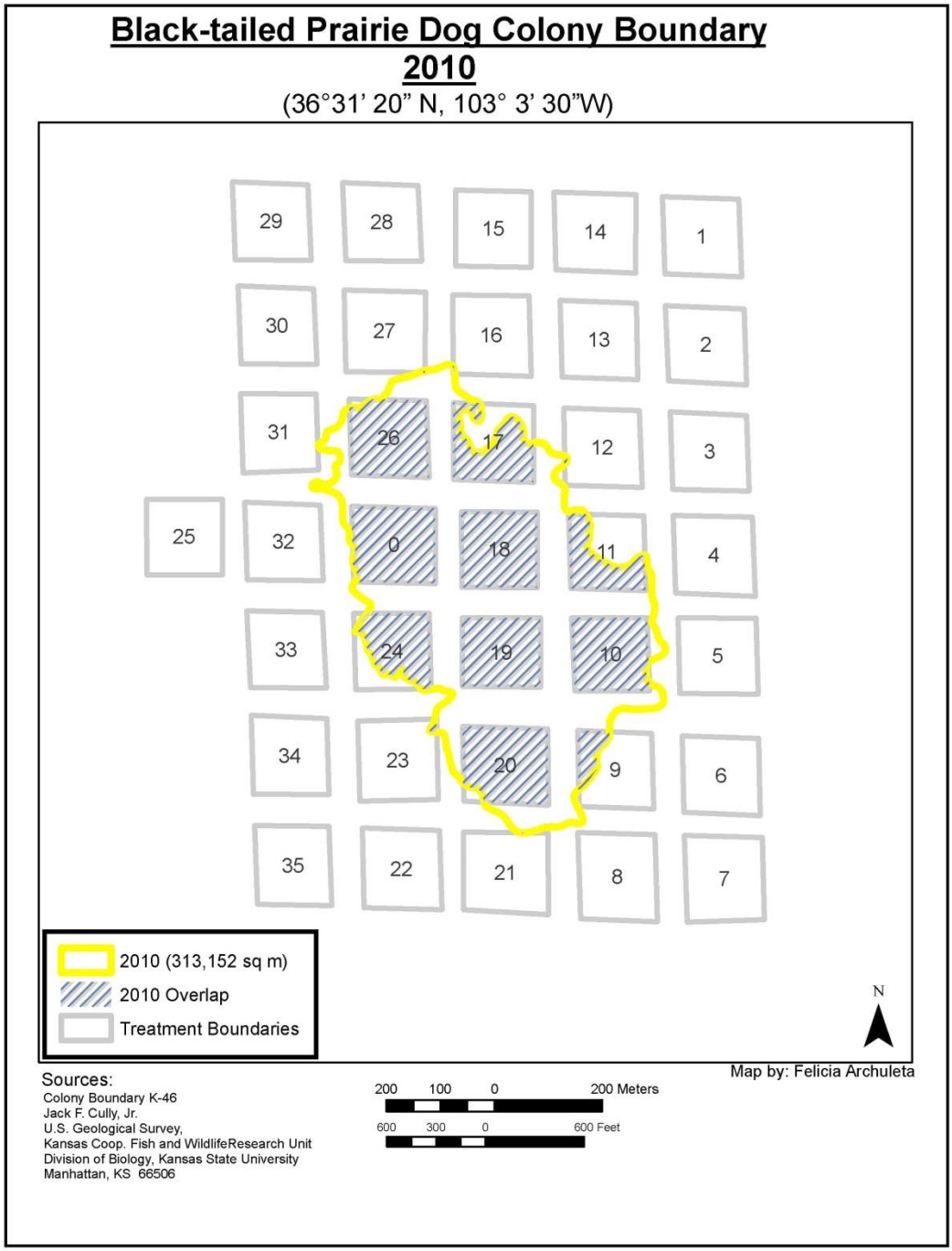

Figure 10 - Map of the 2010 Prairie Dog Colony Boundary. 2010 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray

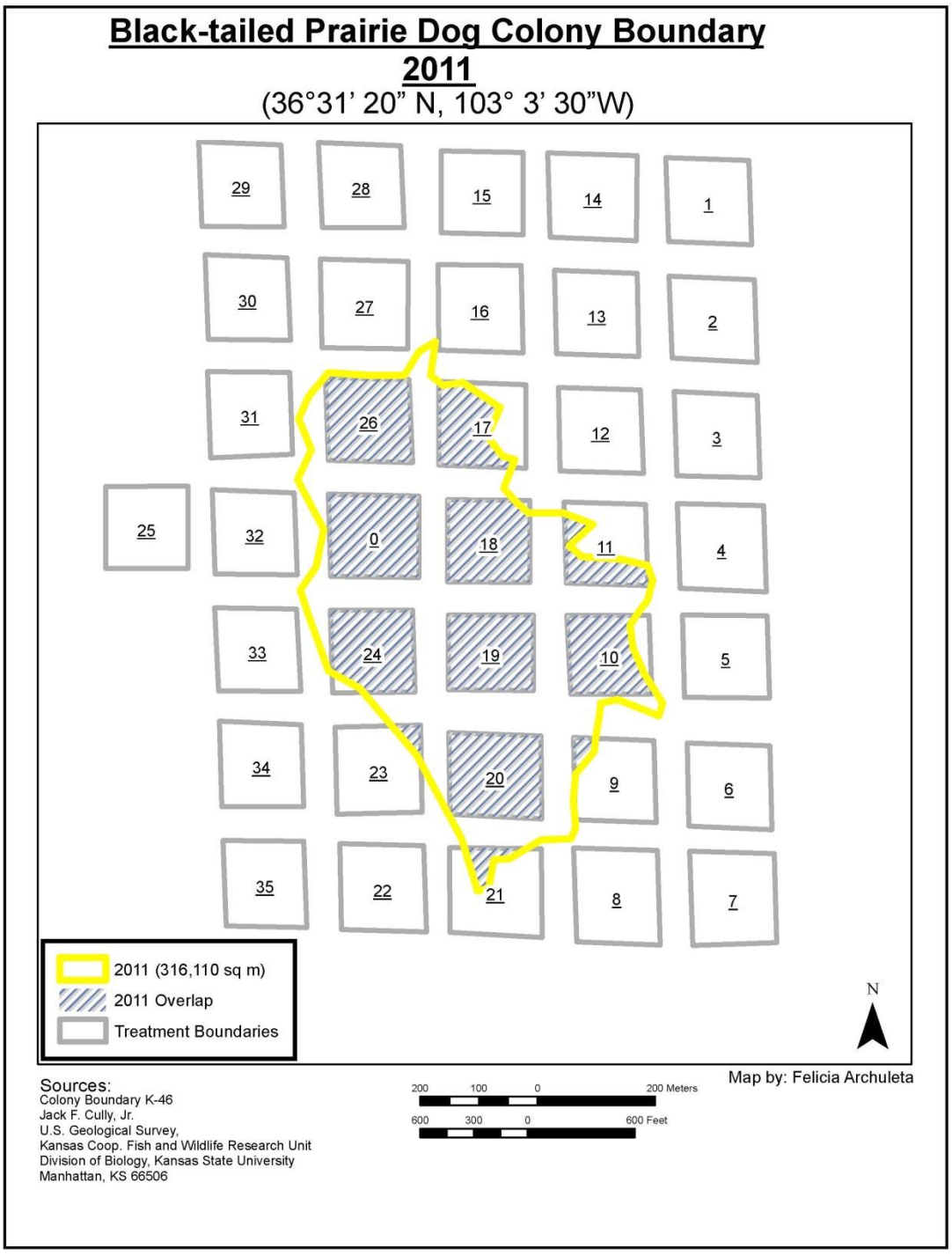

Figure 11 - Map of the 2011 Prairie Dog Colony Boundary. 2011 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray

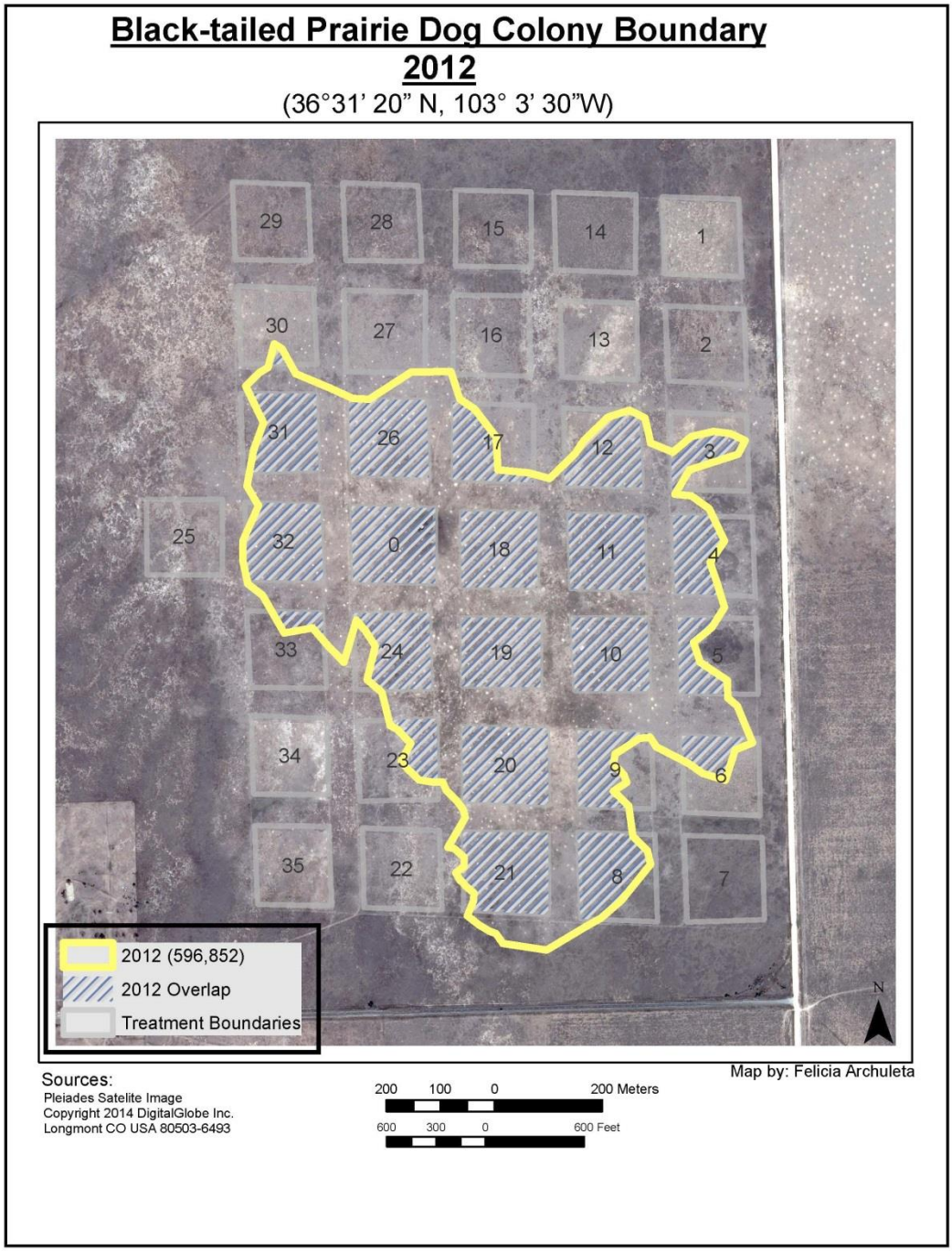

Figure 12 - Map of the 2012 Prairie Dog Colony Boundary. 2012 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray

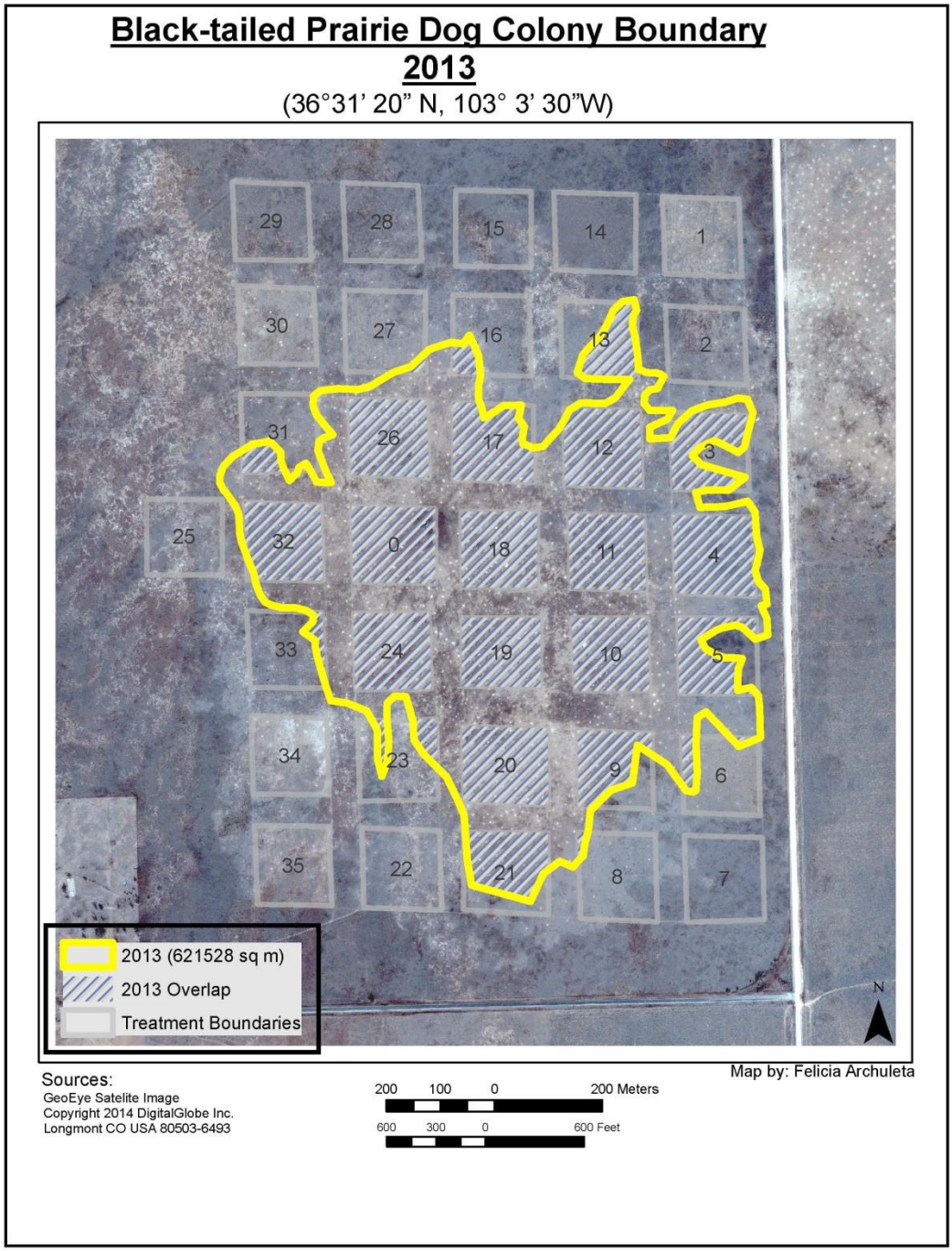

Figure 13 - Map of the 2013 Prairie Dog Colony Boundary. 2013 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray

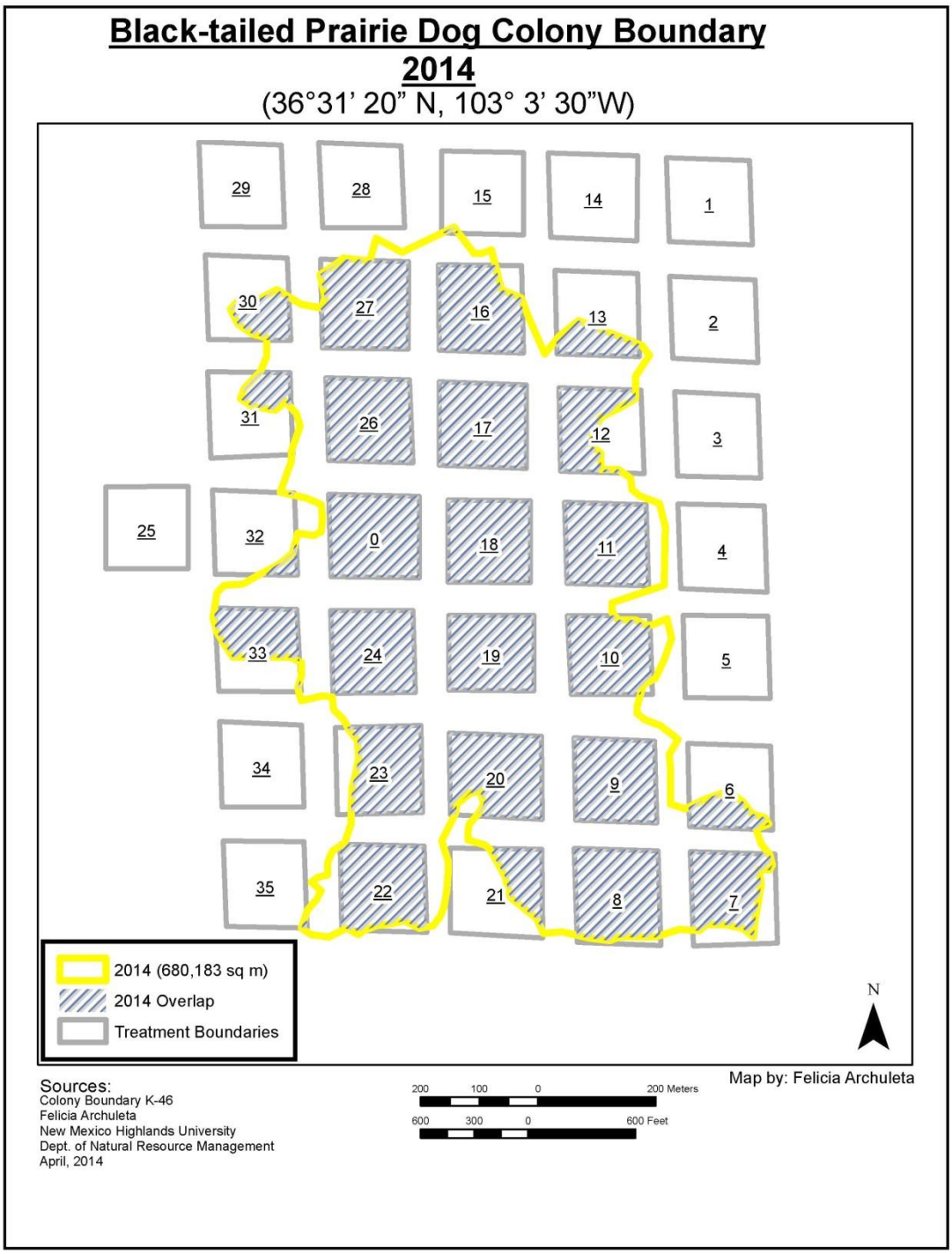

Figure 14 - Map of the 2014 Prairie Dog Colony Boundary. 2014 prairie dog colony boundary (yellow) overlaid on K-46 treatment plots with overlapping areas shaded in gray

APPENDIX 2

HISTOGRAMS

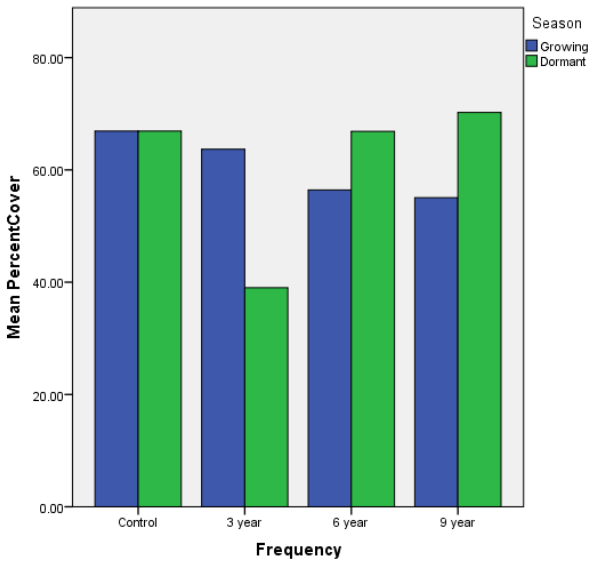

Figure 15 - Histogram of the mean percentage of coverage/overlap by frequency and season. Mean percentage of coverage/overlap of the prairie dog colony on the control, 3 year, 6 year, and 9 year burn frequency plots by season.

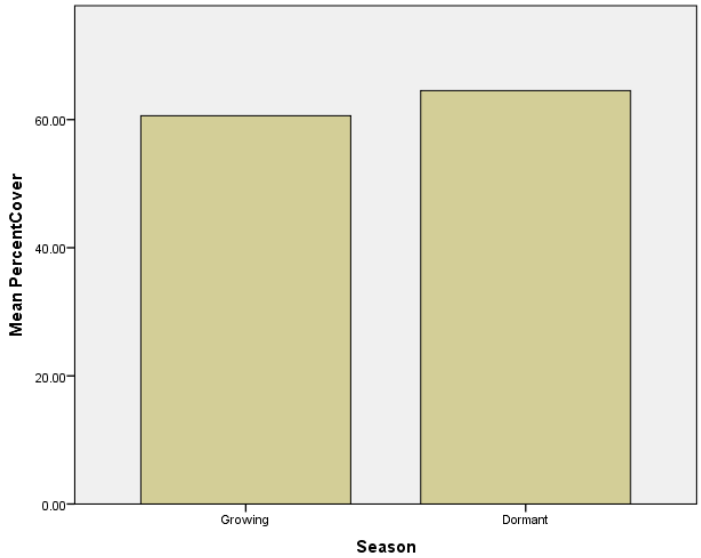

Figure 16 - Histogram of the mean percentage of coverage/overlap by season. Mean percentage of coverage/overlap of the prairie dog colony on the dormant and growing season.

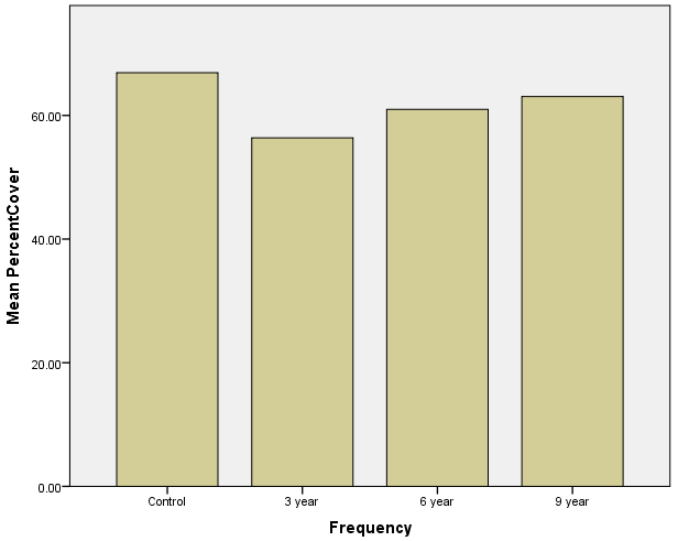

Figure 17 - Histogram of the mean percentage of coverage/overlap by burn frequency. Mean percentage of coverage/overlap of the prairie dog colony on the control, 3 year, 6 year, and 9 year burn frequency plots.

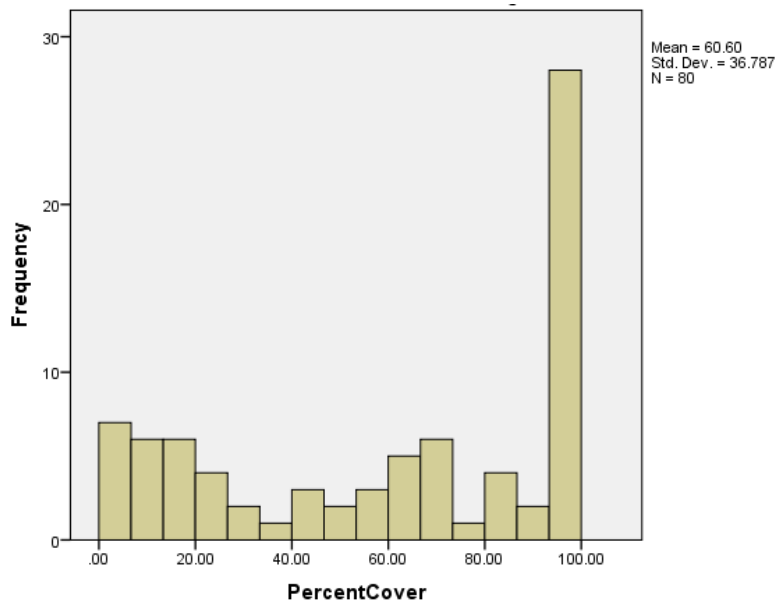

Figure 18 - Histogram of Growing Season Data. This graph illustrates the non-normal distribution of the mean percentage of coverage of the prairie dog colony on the growing season plots.

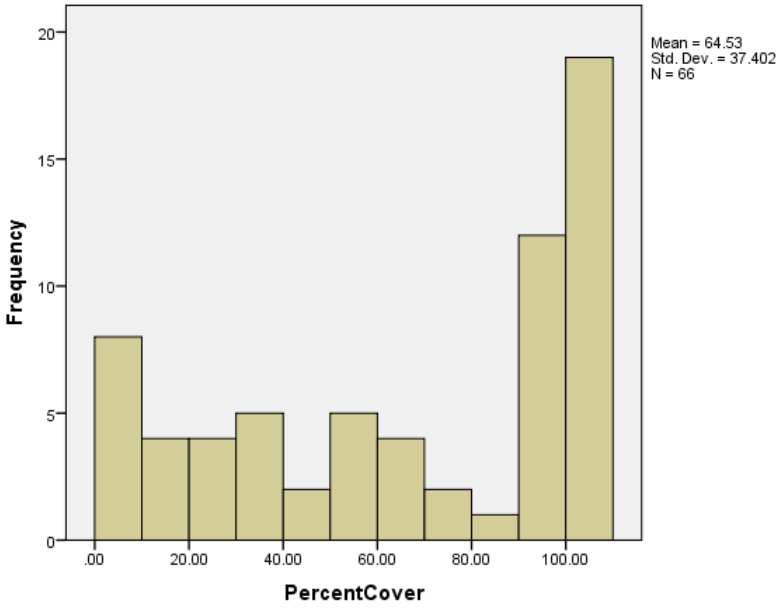

Figure 19 - Histogram of Dormant Season Data. This graph illustrates the non-normal distribution of the mean percentage of coverage of the prairie dog colony on the dormant season plots.

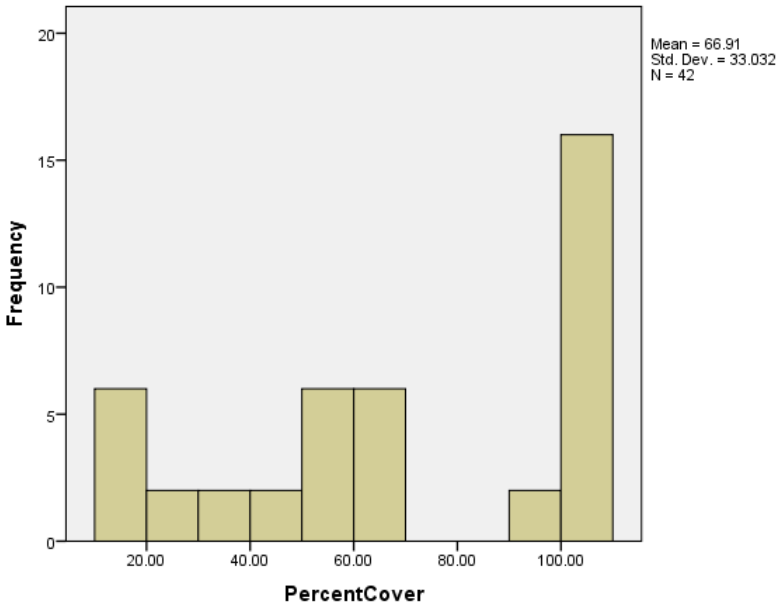

Figure 20 - Histogram of Control Plot Data. This graph illustrates the non-normal distribution of the mean percentage of coverage of the prairie dog colony on the control plots.

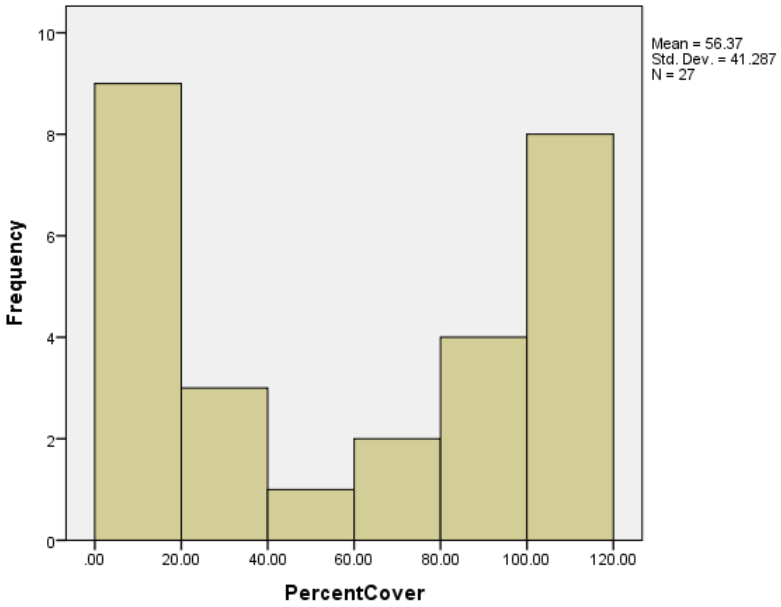

Figure 21 - Histogram of 3 Year Burn Frequency Data. This graph illustrates the non-normal distribution of the mean percentage of coverage of the prairie dog colony on the 3 year burn frequency plots.

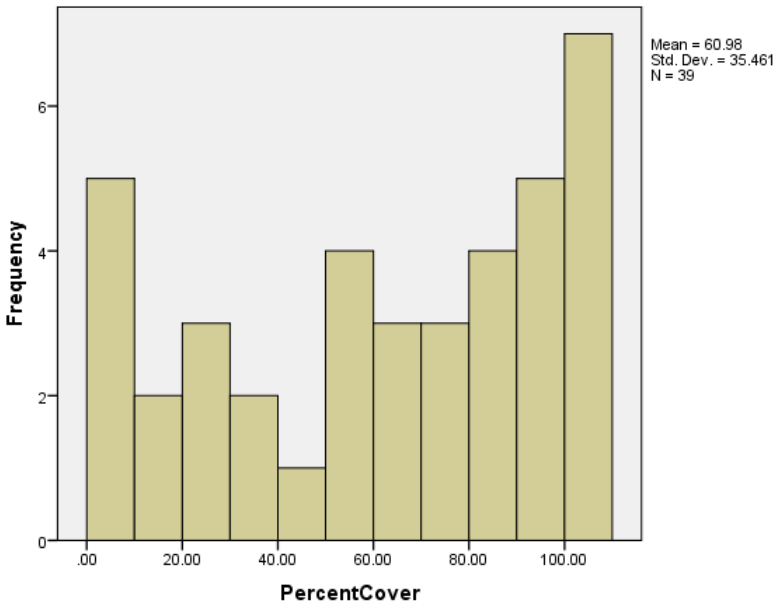

Figure 22- Histogram of 6 Year Burn Frequency Data. This graph illustrates the non-normal distribution of the mean percentage of coverage of the prairie dog colony on the 6 year burn frequency plots.

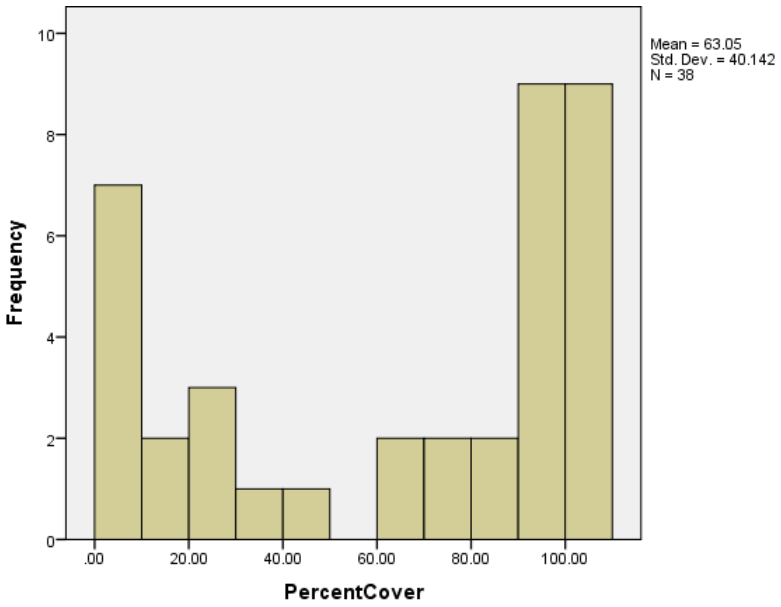

Figure 23 - Histogram of 9 Year Burn Frequency Data. This graph illustrates the non-normal distribution of the mean percentage of coverage of the prairie dog colony on the 9 year burn frequency plots.

APPENDIX 3

SCATTER PLOTS

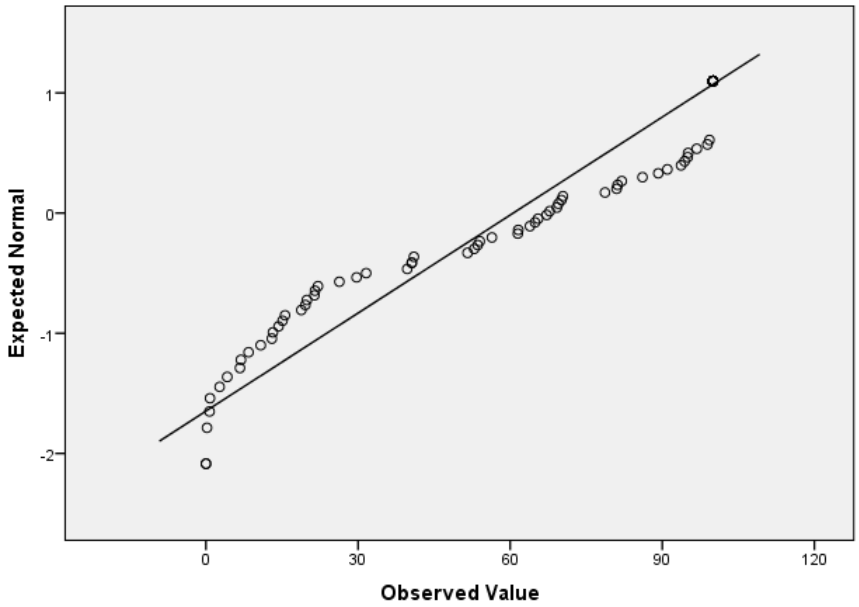

Figure 24 - Normal Q-Q Plot of Growing Season Data. Normal Q-Q plot of the mean percentage of coverage of the prairie dog colony on the growing season plots.

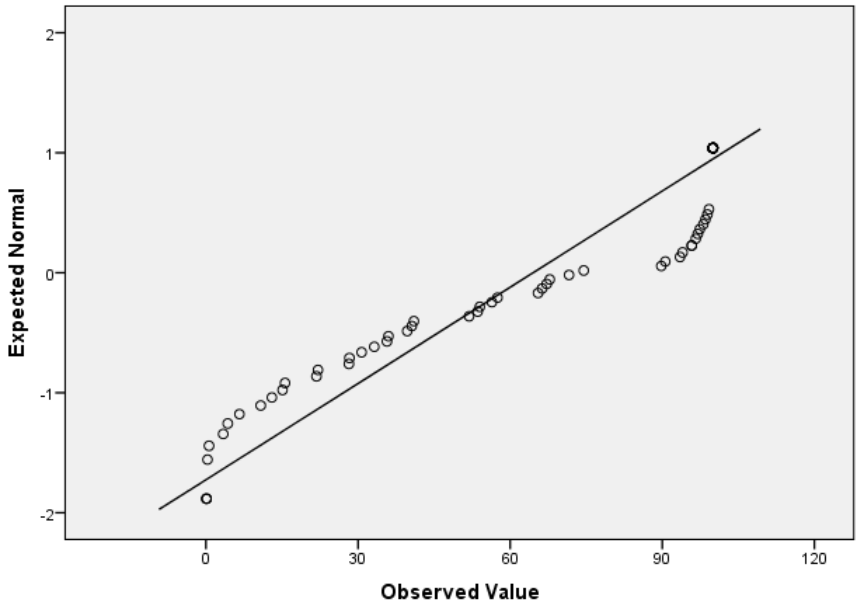

Figure 25 - Normal Q-Q Plot of Dormant Season Data. Normal Q-Q plot of the mean percentage of coverage of the prairie dog colony on the dormant season plots.

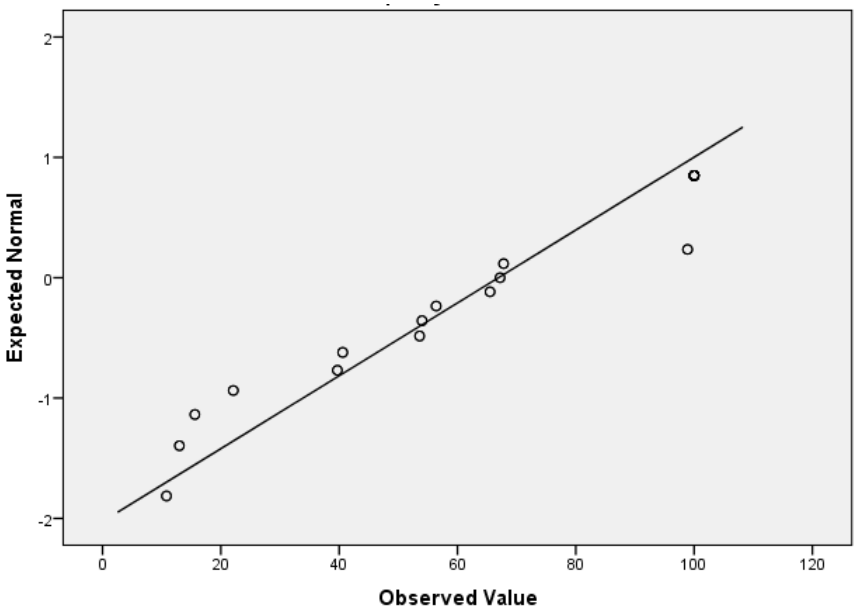

Figure 26- Normal Q-Q Plot of Control Data. Normal Q-Q plot of the mean percentage of coverage of the prairie dog colony on the control plots.

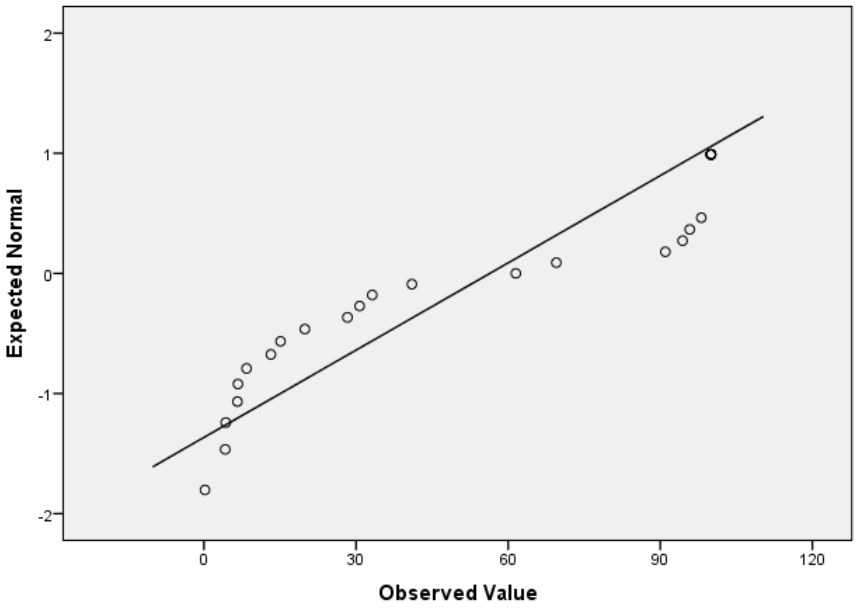

Figure 27 - Normal Q-Q Plot of 3 Year Burn Frequency Data. Normal Q-Q plot of the mean percentage of coverage of the prairie dog colony on the 3 year burn frequency plots.

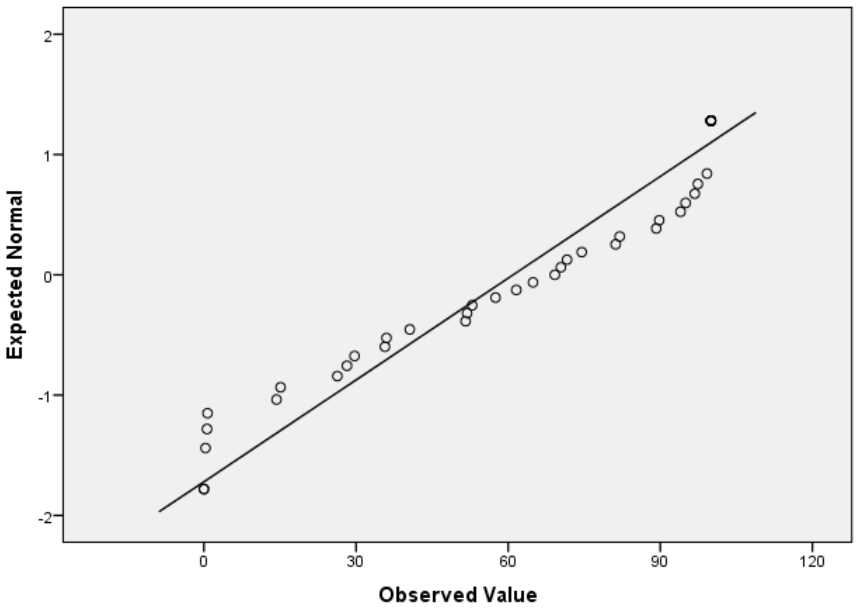

Figure 28 - Normal Q-Q Plot of 6 Year Burn Frequency Data. Normal Q-Q plot of the mean percentage of coverage of the prairie dog colony on the 6 year burn frequency plots.

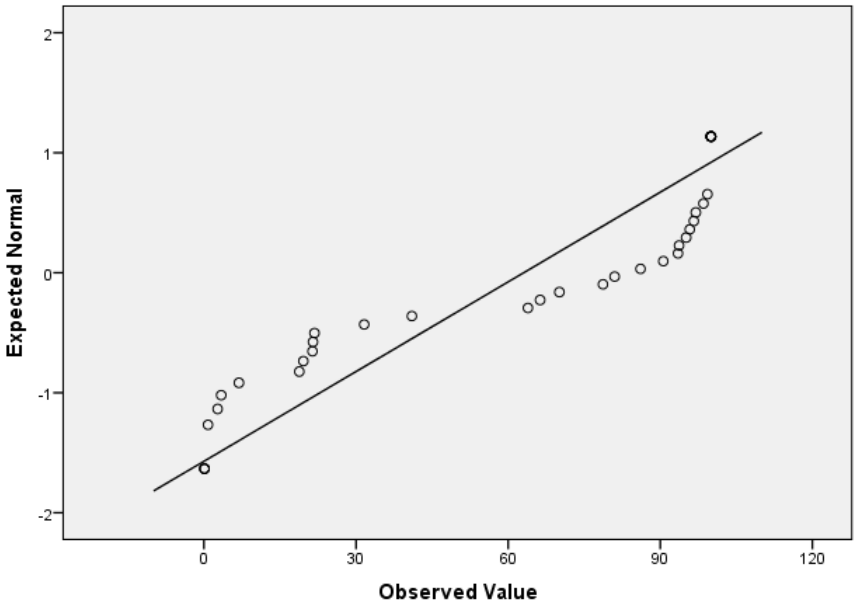

Figure 29 - Normal Q-Q Plot of 9 Year Burn Frequency Data. Normal Q-Q plot of the mean percentage of coverage of the prairie dog colony on the 9 year burn frequency plots.

APPENDIX 4

ANNUAL TEMPERATURE AND PRECIPITATION DATA

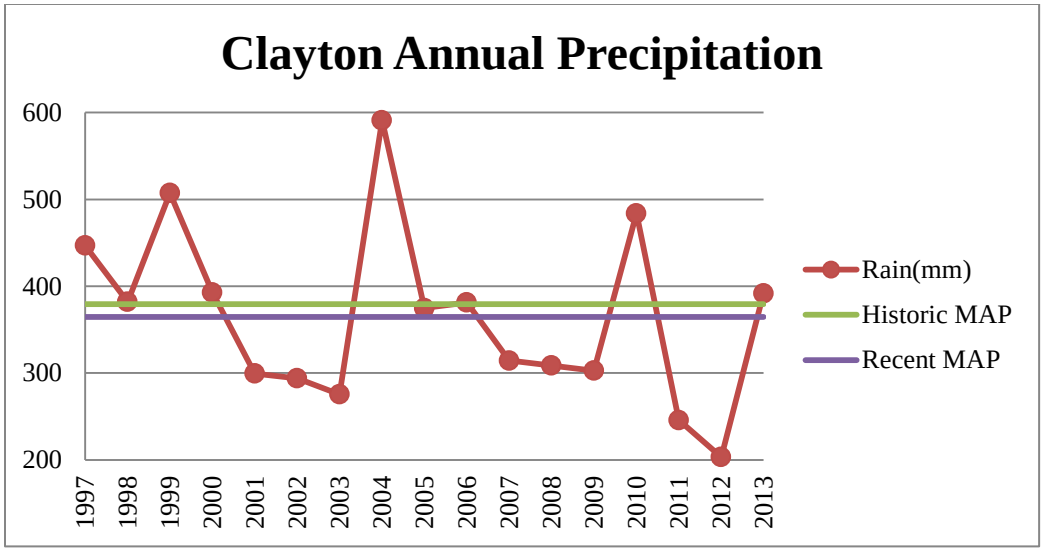

Figure 30 – Mean Annual Precipitation for Clayton, NM. This graph illustrates the mean annual precipitation recorded from 1997-2013 in Clayton, NM

Table 7 - Mean Annual Precipitation for Clayton, NM. This table lists the historic mean annual precipitation (MAP), recent MAP, and the MAP recorded from 1997-2013 in Clayton, NM.

Year	Rain(mm)	Historic MAP	Recent MAP	Rain(in.)
1997	446.786	379.31	364.43	17.59
1998	382.016	379.31	364.43	15.04
1999	507.238	379.31	364.43	19.97
2000	392.684	379.31	364.43	15.46
2001	299.466	379.31	364.43	11.79
2002	293.878	379.31	364.43	11.57
2003	275.59	379.31	364.43	10.85
2004	591.058	379.31	364.43	23.27
2005	374.904	379.31	364.43	14.76
2006	381.254	379.31	364.43	15.01
2007	314.452	379.31	364.43	12.38
2008	308.61	379.31	364.43	12.15
2009	302.768	379.31	364.43	11.92
2010	483.87	379.31	364.43	19.05
2011	245.872	379.31	364.43	9.68
2012	203.2	379.31	364.43	8
2013	391.6962	379.31	364.43	15.42111

APPENDIX 5

CODED DATA

DATA COLLECTED

Year	Plot #	Overlap Area (m2)	Plot Area (m2)	% Cover	Season Code	Season	Freq. Code	Frequency
2005	26	18165.84	20221	89.8	1	D	2	6yr
2005	17	10852.03	21017	51.6	0	G	2	6yr
2005	0	4710.98	21319	22.1	1	D	0	Control
2005	0	4710.98	21319	22.1	0	G	0	Control
2005	18	17718.43	19559	90.6	1	D	3	9yr
2005	24	15988.26	19690	81.2	0	G	2	6yr
2005	19	18471.00	18471	100.0	0	G	1	3yr
2005	10	3525.73	18720	18.8	0	G	3	9yr
2005	20	4773.47	21924	21.8	1	D	3	9yr
2005	9	810.21	19231	4.2	0	G	1	3yr
2006	26	7287.55	20221	36.0	1	D	2	6yr
2006	17	0.23	21017	0.0	0	G	2	6yr
2006	18	8012.56	19559	41.0	1	D	3	9yr
2006	24	2966.02	19690	15.1	0	G	2	6yr
2006	19	7580.98	18471	41.0	0	G	1	3yr
2006	20	19.96	21924	0.1	1	D	3	9yr
2009	26	19007.40	20221	94.0	1	D	2	6yr
2009	17	5527.05	21017	26.3	0	G	2	6yr
2009	0	13966.55	21319	65.5	1	D	0	Control
2009	0	13966.55	21319	65.5	0	G	0	Control
2009	18	18887.85	19559	96.6	1	D	3	9yr

Year	Plot #	Overlap Area (m2)	Plot Area (m2)	% Cover	Season Code	Season	Freq. Code	Frequency
2009	11	2490.11	19196	13.0	1	D	0	Control
2009	11	2490.11	19196	13.0	0	G	0	Control
2009	24	7986.76	19690	40.6	0	G	2	6yr
2009	19	18471.25	18471	100.0	0	G	1	3yr
2009	10	11970.71	18720	63.9	0	G	3	9yr
2009	20	14527.12	21924	66.3	1	D	3	9yr
2009	9	1283.29	19231	6.7	0	G	1	3yr
2010	27	19.47	22324	0.1	1	D	3	9yr
2010	16	31.89	21014	0.2	0	G	1	3yr
2010	26	19704.99	20221	97.4	1	D	2	6yr
2010	17	13649.05	21017	64.9	0	G	2	6yr
2010	12	52.33	20021	0.3	1	D	2	6yr
2010	0	21079.54	21319	98.9	1	D	0	Control
2010	0	21079.54	21319	98.9	0	G	0	Control
2010	18	19559.11	19559	100.0	1	D	3	9yr
2010	11	10368.43	19196	54.0	1	D	0	Control
2010	11	10368.43	19196	54.0	0	G	0	Control
2010	24	13869.90	19690	70.4	0	G	2	6yr
2010	19	18471.00	18471	100.0	0	G	1	3yr
2010	10	18592.32	18720	99.3	0	G	3	9yr
2010	23	177.15	21561	0.8	0	G	3	9yr
2010	20	21001.46	21924	95.8	1	D	3	9yr

Year	Plot #	Overlap Area (m2)	Plot Area (m2)	% Cover	Season Code	Season	Freq. Code	Frequency
2010	9	3817.66	19231	19.9	0	G	1	3yr
2010	21	5.12	23109	0.0	0	G	2	6yr
2011	26	20222.50	20221	100.0	1	D	2	6yr
2011	17	12955.60	21017	61.6	0	G	2	6yr
2011	0	21319.00	21319	100.0	1	D	0	Control
2011	0	21319.00	21319	100.0	0	G	0	Control
2011	18	19263.30	19559	98.5	1	D	3	9yr
2011	11	7801.14	19196	40.6	1	D	0	Control
2011	11	7801.14	19196	40.6	0	G	0	Control
2011	24	18703.69	19690	95.0	0	G	2	6yr
2011	19	18471.00	18471	100.0	0	G	1	3yr
2011	10	16114.28	18720	86.1	0	G	3	9yr
2011	23	1483.30	21561	6.9	0	G	3	9yr
2011	20	21923.50	21924	100.0	1	D	3	9yr
2011	9	1618.11	19231	8.4	0	G	1	3yr
2011	21	3315.60	23109	14.3	0	G	2	6yr
2012	30	862.00	19865	4.3	1	D	1	3yr
2012	32	18995.00	19820	95.8	1	D	1	3yr
2012	33	3070.00	19686	15.6	0	G	0	Control
2012	33	3070.00	19686	15.6	1	D	0	Control
2012	31	16254.00	20065	81.0	0	G	3	9yr
2012	24	16141.00	19690	82.0	0	G	2	6yr

Year	Plot #	Overlap Area (m2)	Plot Area (m2)	% Cover	Season Code	Season	Freq. Code	Frequency
2012	23	6818.00	21561	31.6	0	G	3	9yr
2012	6	6659.00	20068	33.2	1	D	1	3yr
2012	3	7166.00	20061	35.7	1	D	2	6yr
2012	12	14921.00	20021	74.5	1	D	2	6yr
2012	27	18.00	22324	0.1	1	D	3	9yr
2012	26	20220.00	20221	100.0	1	D	2	6yr
2012	0	21319.00	21319	100.0	0	G	0	Control
2012	0	21319.00	21319	100.0	1	D	0	Control
2012	4	11417.00	21307	53.6	0	G	0	Control
2012	4	11417.00	21307	53.6	1	D	0	Control
2012	5	8052.00	20290	39.7	0	G	0	Control
2012	5	8052.00	20290	39.7	1	D	0	Control
2012	8	15995.00	22820	70.1	0	G	3	9yr
2012	9	11826.00	19231	61.5	0	G	1	3yr
2012	10	18720.00	18720	100.0	0	G	3	9yr
2012	11	19196.00	19196	100.0	0	G	0	Control
2012	11	19196.00	19196	100.0	1	D	0	Control
2012	17	11121.00	21017	52.9	0	G	2	6yr
2012	18	19559.00	19559	100.0	1	D	3	9yr
2012	19	18471.00	18471	100.0	0	G	1	3yr
2012	20	21924.00	21924	100.0	1	D	3	9yr
2012	21	22379.00	23109	96.8	0	G	2	6yr

Year	Plot #	Overlap Area (m2)	Plot Area (m2)	% Cover	Season Code	Season	Freq. Code	Frequency
2013	32	19445.00	19820	98.1	1	D	1	3yr
2013	33	2127.00	19686	10.8	0	G	0	Control
2013	33	2127.00	19686	10.8	1	D	0	Control
2013	31	3924.00	20065	19.6	0	G	3	9yr
2013	24	19690.00	19690	100.0	0	G	2	6yr
2013	23	4645.00	21561	21.5	0	G	3	9yr
2013	6	3035.00	20068	15.1	1	D	1	3yr
2013	3	14369.00	20061	71.6	1	D	2	6yr
2013	12	19867.00	20021	99.2	1	D	2	6yr
2013	13	10337.00	19928	51.9	1	D	2	6yr
2013	27	759.00	22324	3.4	1	D	3	9yr
2013	26	20221.00	20221	100.0	1	D	2	6yr
2013	0	21319.00	21319	100.0	0	G	0	Control
2013	0	21319.00	21319	100.0	1	D	0	Control
2013	4	21307.00	21307	100.0	0	G	0	Control
2013	4	21307.00	21307	100.0	1	D	0	Control
2013	5	13645.00	20290	67.2	0	G	0	Control
2013	5	13645.00	20290	67.2	1	D	0	Control
2013	8	625.00	22820	2.7	0	G	3	9yr
2013	9	13374.00	19231	69.5	0	G	1	3yr
2013	10	18720.00	18720	100.0	0	G	3	9yr
2013	11	19196.00	19196	100.0	0	G	0	Control

Year	Plot #	Overlap Area (m2)	Plot Area (m2)	% Cover	Season Code	Season	Freq. Code	Frequency
2013	11	19196.00	19196	100.0	1	D	0	Control
2013	16	2776.00	21014	13.2	0	G	1	3yr
2013	17	18753.00	21017	89.2	0	G	2	6yr
2013	18	19559.00	19559	100.0	1	D	3	9yr
2013	19	18471.00	18471	100.0	0	G	1	3yr
2013	20	21924.00	21924	100.0	1	D	3	9yr
2013	21	15992.00	23109	69.2	0	G	2	6yr
2014	0	21319.00	21319	100.0	0	G	0	Control
2014	0	21319.00	21319	100.0	1	D	0	Control
2014	6	5675.00	20068	28.3	1	D	1	3yr
2014	7	15190.00	22410	67.8	0	G	0	Control
2014	7	15190.00	22410	67.8	1	D	0	Control
2014	8	21373.00	22820	93.7	0	G	3	9yr
2014	9	19231.00	19231	100.0	0	G	1	3yr
2014	10	17808.00	18720	95.1	0	G	3	9yr
2014	11	19196.00	19196	100.0	0	G	0	Control
2014	11	19196.00	19196	100.0	1	D	0	Control
2014	12	11512.00	20021	57.5	1	D	2	6yr
2014	13	5623.00	19928	28.2	1	D	2	6yr
2014	15	140.00	19578	0.7	0	G	2	6yr
2014	16	19133.00	21014	91.0	0	G	1	3yr
2014	17	21017.00	21017	100.0	0	G	2	6yr

Year	Plot #	Overlap Area (m2)	Plot Area (m2)	% Cover	Season Code	Season	Freq. Code	Frequency
2014	18	19559.00	19559	100.0	1	D	3	9yr
2014	19	18471.00	18471	100.0	0	G	1	3yr
2014	20	20498.00	21924	93.5	1	D	3	9yr
2014	21	6871.00	23109	29.7	0	G	2	6yr
2014	22	19902.00	21081	94.4	0	G	1	3yr
2014	23	16977.00	21561	78.7	0	G	3	9yr
2014	24	19690.00	19690	100.0	0	G	2	6yr
2014	26	20221.00	20221	100.0	1	D	2	6yr
2014	27	21646.00	22324	97.0	1	D	3	9yr
2014	30	6108.00	19865	30.7	1	D	1	3yr
2014	31	4287.00	20065	21.4	0	G	3	9yr
2014	32	1314.00	19820	6.6	1	D	1	3yr
2014	33	11103.00	19686	56.4	0	G	0	Control
2014	33	11103.00	19686	56.4	1	D	0	Control
2014	35	115.00	20102	0.6	1	D	2	6yr