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Relationships Between Water Developments and Select Mammals on the U.S. Army Dugway Proving Ground, Utah

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RELATIONSHIPS BETWEEN WATER DEVELOPMENTS AND SELECT MAMMALS ON THE U.S. ARMY DUGWAY PROVING GROUND, UTAH

by

Bryan M. Kluever

A dissertation submitted in partial fulfillment
of the requirements for the degree

of

DOCTOR OF PHILOSOPHY

in

Wildlife Biology

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2015

ABSTRACT

Relationships Between Water Developments and Select Mammals on the U.S. Army Dugway Proving Ground, Utah

by

Bryan M. Kluever, Doctor of Philosophy

Utah State University, 2015

Major Professor: Dr. Eric M. Gese
Department: Wildland Resources

The adding of water developments to arid environments is commonplace. Water developments in arid regions have several aims including promoting urban development, improving habitat for domesticated ungulates, and to benefit game species. A growing body of literature has suggested that the impacts of water developments on certain species can be negative or not in accordance with management objectives. A negative effect has been posited as a driving factor behind a change in the canid community in the Great Basin Desert; kit foxes (*Vulpes macrotis*) populations have declined and coyote (*Canis latrans*) populations have increased since the mid-twentieth century. From 2010 to 2013, I conducted a series of investigations on the U.S Army Dugway Proving Ground (DPG). My overall objective was to determine if effects of water developments occur on DPG for a host of mammalian species. I found no evidence that water developments affected rodent abundance. I also found no evidence that water developments influenced black-tailed jackrabbit (*Lepus californicus*) relative abundance, but observed a 10% decrease in coyote relative abundance in areas associated with removal of free water availability. I documented no visits to water sites by coyotes in 16% of seasonal home ranges, and <5 visits within 39% of home ranges. In addition, 25% of coyote home ranges did not contain a water site, territory sizes were not influenced by number of water sites contained with home ranges, and coyotes with home ranges associated with water developments did not exhibit territorial shifts when water was removed. I also found that kit fox survival, relative abundance, and coyote induced mortality were not influenced by removal of water availability. Furthermore, kit fox home ranges varied from areas associated with water developments for several environmental characteristics known to be important kit fox habitat components. My findings suggested that water developments had no influence on the rodent community, jackrabbits, or kit foxes, and did not represent a requisite habitat component for coyotes. I suspect the observed changes in the Great Basin canid community may be attributed to a combination of factors that were largely unrelated to significant increases of free water availability.

(201 pages)

PUBLIC ABSTRACT

Relationships Between Water Developments and Select Mammals on the U.S. Army Dugway Proving Ground, Utah

Bryan M. Kluever

Water is essential to life. Three general forms of water exist: pre-formed water that is available in food, metabolic water that is created as a byproduct of life processes (e.g., metabolism of fat or breakdown of carbohydrates), and free water (i.e., water available for drinking). As humans settle arid environments, the addition of man-made free water sources (e.g., sewage ponds, catchment ponds) often occurs. In addition, a tool commonly used to increase the abundance or distribution of wildlife species in desert environments is the addition of water sources, usually specifically designed to benefit game species like bighorn sheep (*Ovis canadensis*), mule deer (*Odocoileus hemionus*), and chukar partridge (*Alectoris chukar*). In recent decades, some scientists have argued that adding water sources to deserts may have little to no effect on desert species because they are adapted to living in desert conditions, and have thus evolved to obtain their water needs in preformed and/or metabolic form. Scientists have also suggested that adding water sources to desert environments may actually harm some individual species and alter the arraignments of groups of similarly related species, known as communities. I conducted four studies at the U.S. Army Dugway Proving Ground to determine if man-made water sources have an influence on the rodent community, jackrabbits, and the canid community at the U.S. Army Dugway Proving Ground, Utah. I found that turning off water sources had no effect on abundance of rodent communities or jackrabbits. I found that a portion of coyotes used water sources and coyotes were only slightly less common near water sources once they were turned off. In addition, a portion of coyotes rarely or never drink from water sources and that coyotes did not leave their territories if water sources accessible to them were turned off. My final study revealed that turning off water sources did not influence kit fox survival or abundance, and that kit fox territories differed from areas associated with water sources in several key environmental characterizes, which may suggest that areas associated with water sources were not historically used by kit foxes. In summary, these findings suggest that water developments have little impact on the species that I studied.

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CHAPTER 1

INTRODUCTION

Discerning the impacts of the human footprint on wildlife species and communities has become a central tenet of ecology and related sciences (Leu et al. 2008). The pervasiveness of man-made infrastructure on the majority of terrestrial landscapes has even spurred several subfields of scientific investigation, such as road ecology (Forman et al. 2002, van der Ree et al. 2015) and urban wildlife management (Adams and Lindsey 2009, McCleery et al. 2014). Anthropogenic activities can have variable impacts on a myriad of species, and the impacts of such activities are not always all-encompassing for the same species across space and/or time (Larsen et al. 2010, Morrison and Mathewson 2015), making it difficult to distinguish broad-scale effects. Furthermore, impacts of anthropogenic activities on wildlife can be indirect, and as a result more difficult to detect and quantify (Krausman and Cain 2013, Morrison and Mathewson 2015). Though extensive research has investigated the impacts of select anthropogenic activities on wildlife habitat relationships and population state variables/vital rates [e.g., energy development (Wilson et al. 2013, Brittingham et al. 2014, Jones et al. 2015, Ramirez and Mosley 2015), many others remain largely unexplored.

Adding anthropogenic water sites (hereafter water developments) to arid environments in western United States is commonplace. These water developments can have several aims, including promoting urban development (i.e., sewage and catchment ponds) (Kristan and Boarman 2003, improving grazing habitat for livestock (Brooks et al. 2006, Holecheck et al. 2010, LaBaume 2013), and benefiting target wildlife species

(Larsen et al. 2012, Krausman and Cain 2013). Free water has been reported as a key limiting factor on individual species and communities (Leopold 1933), especially in arid systems (Rosenstock et al. 1999, Krausman et al. 2006). As a result, the construction and maintenance of water developments in arid regions is a common practice in the United States. The intent of such developments are to enhance populations, influence animal movements, effect distributions of wildlife species, and facilitate livestock grazing (Simpson et al. 2011, Larsen et al. 2012). At the end of the last century, 10 of 11 western USA state wildlife agencies reported ongoing water development programs with combined annual expenditures >$1,000,000 US dollars (Rosenstock et al. 1999), and as of 2012, nearly 7000 water developments had been constructed in western United States (Larsen et al. 2012). Furthermore, water developments have been utilized as a mitigation technique to offset the impact of military activities (Broyles 1995). Larsen et al. (2012) forecasted that the importance of water developments to support wildlife will increase as a conservation and management tool in the western United States.

Despite their prevalence, the utility of both general and wildlife-designated water developments has been questioned. Researchers speculate whether increased availability of free water benefits or harms species that are adapted to desert or arid conditions (Burkett and Thompson 1994, Cain et al. 2008). Essentially, the notion that direct uptake of free water by wildlife translates to a biological benefit (Leopold 1933) or that water use always equates to water need, has been challenged under certain conditions. Furthermore, some have argued that water developments may be deleterious, either by spreading disease, encouraging exotic species, hindering wilderness values, or negatively influencing populations of non-water dependent wildlife by increasing predation, predation risk, or competition (Broyles 1995, DeStefano et al. 2000, Bleich 2005, Simpson et al. 2011, Hall et al. 2013).

Investigations lending empirical insight to the impacts of water developments on wildlife are rare, because the majority of studies have only indexed the uptake of free water at the species or population level (Cambell and Remington 1979, Rosenstock et al. 2004, Morgart et al. 2005, Lynn et al. 2006, Whiting et al. 2010). Though such studies have merit, they are unable to determine frequencies of free water use at the individual level and whether the use of free water translates to a biological or ecological effect. Furthermore, the effects of water developments on wildlife can be either direct or indirect. Larsen et al. (2012) defined the direct effects of water developments as those associated with the intake of free water [e.g., an increase in chukar (*Alectoris chukar*) survival due to chukar water intake]. In contrast, indirect effects included, but were not limited to, exploitative or interference competition with other species or conspecifics, or altered vulnerability to predation [e.g., a decrease in black-tailed jackrabbit (*Lepus californicus*) survival due to an increase in coyote (*Canis latrans*) survival, engendered by coyote water intake at water developments].

Investigations on direct effects of water developments on wildlife are sparse. Larsen et al. (2010) found chukars utilized water developments in several mountain ranges but space use and diet were influenced by water development use only in certain ranges. A seminal study including a before-after/control-impact (BACI) design (Morrison et al. 2001, Cain et al. 2008) showed that bighorn sheep (*Ovis canadensis*) used water developments, but reduction of water availability at developments did not influence their population parameters or space use. Broyles and Cutler (1999) also reported water developments did not impact bighorn sheep population parameters, and Krausman and Etchberger (1995) found no spatial affinity for water developments. Hall et al. (2013) observed coyote relative abundance was similar in areas with and without water developments and springs despite coyote use at water sites, suggesting no direct effect of water consumption for this species.

Investigations of indirect effects of water developments on wildlife are also lacking. Cutler and Morrison (1998) found that species richness and relative abundance for small mammals and reptiles, species believed not to consume free water, did not differ in relation to areas adjacent to dry or wet water developments, suggesting predation rates at sites did not differ, or that increased predation was compensatory. DeStefano et al. (2000) observed a negative relationship between leporid [(black-tailed jackrabbit and desert cottontail (*Sylvilagus audubonii*)] and coyote relative abundance in relation to proximity to water developments in the Sonoran Desert. This finding suggested an indirect effect of coyotes on the leporid community, either by way of predation or a lagomorph behavioral response to increased predation risk (Apfelbach et al. 2005). Conversely, Hayden (1966) observed that black-tailed jackrabbit abundance was higher near water sources in the Mojave Desert, and believed this finding could be attributed to leaky or overflowing water tanks supporting patches of vegetation with high levels of preformed water, which were preferred forage for jackrabbits.

A specific indirect effect of water hypothesis (Larsen et al. 2012) has been theorized for the canid community in portions of the Great Basin Desert. Specifically, it has been argued that water developments constructed during the 1970s-1990s on and near the US Army Dugway Proving Ground (DPG) removed the arid system limitations of coyotes, which compete with kit foxes (*Vulpes macrotis*) for habitat, space, and food (Arjo et al. 2007, Nelson et al. 2007, Kozlowski et al. 2008). The justification for this theory is largely premised on the differential physiological demand of free water by coyotes and kit foxes. Golighlty and Omhart (1984) reported that the amount of wet prey biomass required per day to meet energy and water requirements (i.e., preformed water) is 504 and 1780 g for coyotes, and 101 and 175 g for kit foxes. Put in simpler terms, in the absence of available free water, coyotes must consume over 3x the wet prey biomass to meet water versus energy requirements, while kit foxes need consume less than 2x the amount. This disparity in kit foxes' and coyotes' abilities to utilize preformed water, coupled with behavioral differences for the two species (Golightly and Omhart 1983), and the addition of anthropogenic water developments has reportedly led to an expansion of coyote habitat on DPG, which has in turn attributed to increases in coyote populations and reductions of kit foxes by increasing interference competition and intraguild predation (TRIES 1997, AGEISS 2001, Arjo et al. 2007, Kozlowski et al. 2012). However, a recent investigation did not lend support to water developments being an attributing factor to canid community changes at DPG. Hall et al. (2013) observed no difference in coyote or kit fox activity in areas with and without free water at DPG, but observed that kit foxes regularly visited water sources in the Mojave Desert. The latter finding suggests that kit foxes may utilize water sources if they are located in areas containing environmental characteristics that constitute kit fox habitat. To date, no studies have chronicled the frequency of water development visitations by individual kit foxes or coyotes. Clearly, there is a need to gain a better general understanding of the impacts of water developments on wildlife. In addition, there is a need to determine if direct and indirect effects of water developments occur for a host of species that have received little attention. Finally, there is a need to test the hypothesis that water developments have attributed to a reported change in the canid community of the Great Basis Desert.

In this dissertation, I evaluated the impact of water developments for a host of mammalian species, both at the species and community level, at the U.S. Army Dugway Proving Ground (DPG). In chapter 2, I used both an observational and before-after-control-impact (BACI) design in concert with a capture-mark-recapture framework to determine if water developments have an indirect effect on rodent total abundance, a metric often utilized in community ecology investigations. The results of this study will allow wildlife managers and conservation strategists to predict the outcome of adding wildlife water developments to the rodent community, which could be of marked importance in areas where rare or imperiled rodent species occur.

In chapter 3, I examined the role of water developments on black-tailed jackrabbit and coyote populations at DPG. I used a BACI design in concert with scat deposition surveys, spotlight surveys, and a generalized linear mixed models to test whether water availability directly impacted coyote and/or indirectly impacted black-tailed jackrabbit abundance. Previous investigations on these species and water developments have been purely observational and revealed contrary findings (Hayden 1966, DeStefano et al. 2000, Hall et al. 2013). My results will allow wildlife managers, mitigation strategists, and rangeland managers to forecast the impact of water developments for both species.

In chapter 4, I expanded my investigations of the impacts of water developments on coyote space use. Specifically, I examined individual water development use by coyotes. In addition I examined the spatial affinity of coyote home ranges to water developments, a third order selection process (Johnson 1980), while incorporating a water availability manipulation design component. Previous researchers on DPG suggested that water developments were a requisite habitat component for coyotes, as all coyote home ranges were centered around water sources (AGEISS 2001). Furthermore, no previous investigations have reported on individual-based use of water developments by coyotes. My results will shed light on the overall influence of water developments on coyotes in the Great Basin Desert, which could have implications for both coyote management and kit fox conservation.

In chapter 5, I report on my final field investigation, which examined the impact of water developments on both kit fox demography and space use. We used a before-after and BACI design to determine if water developments influence kit fox survival and abundance. In addition, we determined individual based visitation frequencies to water developments by kit foxes and compared kit fox home ranges to areas associated with water developments for several environmental variables reported to be important kit fox habitat components. The results of this study will determine whether water developments have contributed to a decline in the kit fox population at DPG, which will have implications to kit fox conservation efforts in Great Basin Desert and throughout their entire range.

The results of these four studies will contribute significantly to our current knowledge of the relationship between water developments and wildlife. Wildlife officials, rangeland managers, and urban development planners in arid regions alike can use my results to assess and predict the impacts of existing or proposed water developments on the species I have investigated. In addition, my empirical based investigations on water developments will help bridge the gap between the reported disparity between conceptual and data driven investigations on the nebulous topic of the ecological impacts of water developments (Simpson et al. 2011, Larsen et al. 2012, Krausman and Cain 2013).

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CHAPTER 2

INFLUENCE OF WILDIFE WATER DEVELOMENTS ON THE RODENT COMMUNITTY IN THE GREAT BASIN DESERT[1]

ABSTRACT

Rodent communities comprise a majority of the mammalian diversity within a host of ecosystems and provide a significant portion of the available biomass consumed by higher trophic organisms. Rodents also influence the structure and composition of vegetation communities and contribute to the supply of several ecosystem services. The importance of rodent communities, coupled with increases in anthropogenic modifications to landscapes, creates a need for determining if such modifications affect rodent communities. The construction of artificial water developments for wildlife in the deserts of the western United States is commonplace, but developments have been proposed to have unintentional negative impacts to desert adapted species and communities. To test these negative impact ideas, we employed a BACI and observational design over four summers to determine if water developments influenced rodent abundance. We found no evidence that water developments negatively impacted rodent abundance. Estimates of total rodent abundance for trapping grids that were distant and proximate to water developments under the observational design framework were similar [i.e., overlap of 95% confidence intervals (CIs)] during 87.5% (7 of 8) of trapping sessions. The 95% CIs for rodent abundance at distant and proximate grids

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[1] Co-authors are Eric M. Gese and Steven J. Dempsey; chapter is formatted for Journal of Mammalogy

under the BACI design framework overlapped during 75% (3 of 4) of trapping sessions prior to the water manipulation; in all three cases abundance was lower at proximate grids. Following the water manipulation, non-overlap of 95% CIs occurred during 100% (4 of 4) of trapping sessions; rodent abundance estimates were lower at proximate grids in all cases. Rodent abundance appeared to be driven by a combination of spatial (e.g., vegetation structure heterogeneity) and temporal (e.g., variations in seasonal precipitation) factors. Our findings suggested that water developments on our study area did not significantly impact rodent communities.

INTRODUCTION

Rodent communities comprise a majority of the mammalian diversity within a host of ecosystems (Feldhamer et al. 2007; Jones and Safi 2011; Merritt 2010), provide a significant portion of the available biomass consumed by predators (Andersson and Erlinge 1977; Sieg 1987), and can influence the structure and composition of vegetation communities (Sieg 1987; McMurray et al. 1997). Recent investigations have found that rodent assemblages contribute to the provision of ecosystem services such as groundwater recharge, regulation of soil erosion, regulation of soil productivity potential, soil carbon storage, and forage availability (Martinez-Estevez et al. 2013; Longland and Ostaja 2013). Factors reported to influence such communities included predation (Stapp 1997; Henke and Bryant 1999; Shenbrot 2014), competition (Heske et al. 1994; Stokes et al. 2009; Shenbrot 2014), precipitation and primary productivity (Ernest et al. 2000; Thibault et al. 2010; Shenbrot 2014), vegetation characteristics (Whitford and Steinberger 1989; Hernandez et al. 2005; Thompson and Gese 2013), and both natural and anthropogenic disturbances (Ramirez and Hornocker 1981; Medin and Clary 1989; Kutiel et al. 2000). Of these, anthropogenic disturbances are receiving increasing attention due to their growing pervasiveness on many natural landscapes.

Investigations have revealed negative impacts on rodent communities for a host of anthropogenic disturbances, including urbanization (Umetsu and Pardini 2007; Shenko et al. 2012), livestock grazing (Medin et al. 1989), predator removal (Henke and Bryant 1999), and military training activities (Shenko et al. 2012). Conversely, disturbances such as roads and road traffic (Bissonette and Rosa 2009; Rotholz and Mandelik 2013), wind farm construction (de Lucas et al. 2005), and military training activities (Thompson and Gese 2013) have been shown to have benign or positive impacts. The variability of the effects of anthropogenic disturbance on rodent communities suggests a myriad of ecological processes are likely at play, that disturbances may have direct or indirect impacts on communities, and that similar types of disturbances may not impact communities occurring in different areas in the same fashion. Hence, there may be a need to limit inference outside the scope of site-specific investigations and address the role of specific anthropogenic disturbances on rodent communities on a case by case basis, and when possible, incorporate sampling designs that allow studies to go beyond a purely observational nature. One type of anthropogenic modification that has been suggested as a possible disturbance in arid environments is artificial water developments that were constructed to benefit certain wildlife species (hereafter water developments).

Free water has been reported as a key limiting factor on individual species and communities (Leopold 1933), especially in arid systems (Rosenstock et al. 1999;

Krausman et al. 2006). As a result, the construction and maintenance of water developments in arid regions of the United States is a common practice. The intent of such developments is the addition of free water on a landscape in order to increase populations, influence animal movements, effect distributions of wildlife species, and facilitate livestock grazing (Simpson et al. 2011, Larsen et al. 2012). At the end of the last century, 10 of 11 western USA state wildlife agencies reported ongoing water development programs with combined annual expenditures >$1,000,000 US dollars (Rosenstock et al. 1999), and as of 2013, nearly 7000 water developments had been constructed in the western United States (Larsen et al. 2012). Furthermore, water developments have been utilized as a mitigation technique to offset the impact of military activities (Broyles 1995). Larsen et al. (2012) forecasted that the importance of water developments will increase as a conservation and management tool in the western United States. Though water developments are unique in that they are explicitly constructed to benefit certain targeted species, they also represent a potential disturbance on the landscape.

In recent years researchers have posited that water developments may negatively impact communities indirectly, by way of spreading disease, encouraging exotic species, or negatively influencing populations not dependent on free water by way of increased predation or predation risk (Broyles 1995; Rosenstock et al. 2004; DeStefano et al. 2000; Simpson et al. 2011). The notion that the addition of free water on arid landscapes negatively impacts desert adapted species is known as the indirect effect of water hypothesis (Hall et al. 2013). Investigations testing the indirect effect of water hypothesis are rare, especially at the community level. DeStefano et al. (2000) suggested the leporid community was negatively impacted in areas near water developments due to increased presence of mammalian and avian carnivores. Conversely, Cutler and Morrison (1998) found no evidence that water developments affected the relative abundance of rodents or reptiles. Carnivorous mammal and avian species known to regularly prey upon rodents have been documented to regularly visit water developments during the summer months in arid environments (Rosenstock et al. 2004; O'Brien et al. 2006; Hall et al. 2013).

To date, no investigations have tested the indirect effect of water hypothesis on rodent communities using either long-term monitoring (i.e., > 2 years) or an experimental/manipulative study design. Whether rodent communities are indirectly affected by water developments may help guide future management decisions for wildlife management agencies, sportsmen groups, and conservation organizations, especially in arid areas where the animal communities are partially comprised of rare or imperiled species, or when rodents comprise the primary food source for rare or imperiled carnivores. The overall objective of our study was to elucidate the indirect effects of water developments on the rodent community in a cold desert system. Specifically, we used both an observational and BACI design to determine 1) whether total abundance of rodents was different near to or away from active water developments, and 2) whether removal of water developments affected the total abundance of rodents.

MATERIALS AND METHODS

Study Area.—We conducted our research on 879 km^2 of the eastern portion of the U.S Army Dugway Proving Ground (DPG) located approximately 128 km southwest of Salt Lake City, in Tooele County, Utah, USA. Elevations ranged from 1302 m to 2137 m. The study site was in the Great Basin and was characterized as a cold desert (Arjo et al. 2007). Winters were cold, summers were hot and dry, with the majority of precipitation occurring in the spring. Average maximum temperatures on DPG range from 3.3°C in January to 34.7°C in July. Average minimum temperatures ranged from –8.8°C in January to 16.3°C in July. Mean annual precipitation was 20.07 cm. The study area consisted of predominately flat playa punctuated with steep mountain ranges. The lowest areas consisted of sparsely vegetated salt playa flats. Slightly higher elevation areas were less salty and supported a cold desert chenopod shrub community. At similar elevations, shrub communities dominated by greasewood (*Sarcobatus vermiculatus*) were found. Higher elevations consisted of vegetated sand dunes. Near the bases of the higher steep mountains were shrub steppe communities of big sagebrush (*Artemisia tridentata*). The highest elevation was a Utah juniper (*Juniperus osteosperma*) community including black sagebrush (*Artemisia nova*) and bluebunch wheatgrass (*Elymus spicatus*). Where wildfires had occurred along the foothills, cheatgrass (*Bromus tectorum*), tall tumble-mustard (*Sisymbrium altissimum*), and Russian thistle (*Salsola kali*) were common within communities of sagebrush, rabbitbrush (*Chrysothamnus sp.*) and juniper (Arjo et al. 2007).

The study area contained eight artificial water developments that were installed from 1970 to 1990. Impacts to vegetation near these water developments were minimal due to a lack of livestock grazing, a design that prohibited water runoff, and fencing to exclude feral horses. Resident carnivorous species regularly visiting DPG water developments included coyote (*Canis latrans*), bobcat (*Lynx rufus*), red-tailed hawk (*Buteo jamaicensis*), golden eagle (*Aquila chrysaetos*), and great horned owl (*Bubo virginianus*) (Hall et al. 2013). The small mammal fauna consisted of granivores, folivores, omnivores, and carnivores, and included Ord's kangaroo rat (*Dipodomys ordi),* chisel-toothed kangaroo rat (*D. microps*), Great Basin pocket mouse (*Perognathus parvus*) little pocket mouse (*P. longimembris*), long-tailed pocket mouse (*Chaetodipus formosus*), deer mouse (*Peromyscus maniculatus*), Piňon mouse (*P. truei*), western harvest mouse (*Reithrodontomys megalotis*), northern grasshopper mouse (*Onychomys leucogaster*), white-tailed antelope squirrel (*Ammospermophilus leucurus*), Townsend's ground squirrel (*Spermophilus mollis*), desert woodrat (*Neotoma lepida*), montane vole (*Microtus montanus*), sagebrush vole (*Lemmiscus curatus*), and desert shrew (*Notiosorex sp.*) (Arjo et al. 2007).

Design and Sampling.—We used a combination of stratified-random and paired sampling to establish eight 50m x 50m trapping grids in areas near to (hereafter proximate grids) and away from (hereafter distant grids) water developments. For proximate grids we randomly established a sampling grid centroid at a distance between 75 and 100 m from the edge of a water development's infrastructure. Similarly, for distant plots we randomly established a sampling grid centroid between 1-1.1 km from the water developments. We selected this distant zone of grid selection in an attempt to reduce the potential for overlap among plots associated with individual water developments; the minimum distance between developments was 2.3 km. Available locations for distant grids were further constrained so they were located in the same type of vegetation type (i.e., mixed shrubland, chenopod shrubland, exotic vegetation) as their respective paired proximate grid. Established trapping grids were sampled repeatedly over the course of the study; new grids were not established every session or year. In this study we were less concerned about the exact densities of rodents than the fluctuations in community abundance. Since biases in density estimates attributable to trapping method and the influence of the area outside the trapping grids are presumably consistent through time, they should not affect the correlations in fluctuations between sites or through time at the same site (Ernest et al. 2000).

Proximate and distant trapping grids were partitioned into two separate design frameworks: an observational design and a before-after control-impact (hereafter BACI) design (Morrison et al. 2001) (Figure 1). The observational grid design consisted of paired grids that underwent no experimental manipulation; proximate and distant grids were sampled over the course of the study with no reduction of water availability at developments (Fig. 2-1). For the BACI design, trapping grids were sampled for two years with no manipulation followed by a removal of water at developments (Fig 2-2). Water developments were drained using a generator (model 4000-Watt, Champion Power Equipment, Santa Fe Springs, CA) and submersible pump (model 1/2 HP, Wayne Pumps, Harrison OH), and drinking portals were covered with plywood (Fig. 2-2). Water levels were checked monthly and were re-drained if they reached >2/3 capacity. BACI designs are considered advantageous to purely observational studies as they better account for variability of response and exploratory variables attributed to temporal and spatial factors that cannot always be controlled and/or accounted for under observational field settings (Morrison et al. 2001). Our observational design complemented the BACI design by allowing for an increase in the number of observational grids when investigating the effects of disturbances, as recommended by Underwood (1994).

We sampled rodents in grids using a 7 x 7 configuration (49 traps [H.B. Sherman Traps, Inc., Tallahassee, FL, USA], 8.3-m spacing) for four consecutive nights (i.e., four capture occasions). Each four night sampling period was considered an individual trapping session. We conducted trapping sessions on each grid in early (May 01 to June 30) and late (August 1 to September 30) summer. Traps were baited with a mixture of black sunflower and mixed bird seed. All captured rodents were identified to species, tagged in each ear (Model # 1005-1, National Band and Tag CO., Newport, KY, USA), and recaptures of individuals were recorded. Capture and handling protocols were reviewed and approved by the Institutional Animal Care and Use Committees (IACUC) at the United States Department of Agriculture’s National Wildlife Research Center (QA-1734) and Utah State University (#1438). Permits to capture and handle rodents were obtained from the Utah Division of Wildlife Resources (COR #4COLL8322). In order to reduce potential sampling bias among distant and proximate grids paired with each water development, both grids were sampled on the same nights, and the order that paired plots were sampled across sessions was randomized. All capture and handling procedures were in accordance with guidelines endorsed by the American Society of Mammologists (Sikes et al. 2011) and sanctioned by Utah State University Institutional Animal Care and Use Committee (Permit #1438).

Data Analyses.**—**We used the R (R Development Core Team 2007) package RMark (Laake and Rexsatd 2008) to construct closed population capture-mark-recapture (CMR) models for program MARK (White and Burnham 1999) to derive estimates of total rodent abundance. Total abundance, a measure of the productivity of a community (i.e., the sum of abundances across all species) is a commonly utilized metric in community ecology in general, and used when investigating the impacts of disturbances on rodent communities (Ernest et al. 2000, Supp and Ernest 2014). Our candidate set of models included a null (Mo), time-varying (M_t), heterogeneity (M_h), behavioral (M_b), behavioral by group type (i.e., proximate or distant grids) (M_{bg}) and heterogeneity by group type (M_{hg}); see Chao (2001) for full explanation of models. Because our central research question was comparing proximate versus distant grids, a group effect for grid type on abundance (i.e., proximate or distant) was included in all candidate models for the abundance parameter (i.e., N_g). We used Akaike's Information Criteria with a small sample size correction factor (AIC_c) to select the model most supported by the data; model with the minimum AICc; models within 2 AICc units of the minimum AICc model were considered competitive models (Anderson and Burnhan 2002). Due to small sample sizes of captures and recaptures at each specific trapping grid, we pooled session total abundance estimates across the X proximate or X distant grids belonging to the same design framework (i.e., observational or BACI) per seasonal session.

Our design framework allowed for several types of comparisons. For observational grids we compared the total abundance of rodents on the proximate and distant grids for each seasonal session taking place prior to and then following the water manipulation. Similarly, for the BACI grids we compared the total abundance of rodents on the proximate and distant grids for each seasonal session taking place prior to (i.e., Before-Impact compared to Before-Control) and also following the water manipulation (i.e., After-Impact compared to After-Control). We tested for differences of total abundance by comparing the overlap of the 95% confidence intervals (White and Burnham 1999). Because it is possible for 95% confidence intervals to have slight overlap but still have a p-value < 0.05 (Sokal and Rolf 2012) we also used our abundance estimates and associated standard errors to conduct post hoc z-tests (Sokal and Rolf 2012) to test for differences among pooled abundance estimates. Specifically, we tested differences between 1) proximate and distant abundance estimates associated with observational grids, 2) proximate and distant abundance estimates associated with the BACI grids during the before period, and 3) proximate and distant abundance estimates associated with the BACI grids during the after period). Explanatory variables were considered significant at a probability of $P < 0.05$.

We also calculated ratios of means, defined as the mean value in a treatment group divided by the mean value in the control or alternative treatment group (Friedrich et al. 2011), as an additional method to describe the differences, if any, between proximate and distant grids for our BACI and observational design frameworks. For example, within the context of our study a mean ratio of 1.4 would suggest rodent abundance at distant grids was 40% greater than abundance at proximate grids, whereas a mean ratio of 0.6 would suggest rodent abundance was 40% greater at proximate grids. Mean ratios provide a relatively straightforward approach of comparing sample means between treatments, but should be used in addition to methods that account for measures of variance (i.e., comparison of 95% confidence intervals, z-tests).

RESULTS

Between May 2010 and September 2013 we conducted eight summer trapping sessions. We accumulated 25,088 trap nights, 5,086 captures, and captured 2,145 individual rodents. We captured twelve rodent species during the study (Table 2-1). The median number of species captured per trapping session was seven, and ranged from six to nine. Ord's kangaroo rat was the most commonly captured species and comprised 68.9% of all captures and 66.3% of all individuals captured (Table 2-1).

The M_h model was selected as the top model in 81.5% (13 of 16) of the candidate model sets; the only other selected model type was M_b (Table 2-2). There were no instances of competing models. Capture probabilities (i.e., *p*1 and *p*2) for M_h models averaged 0.85 (*SD* = 0.05) and 0.28 (*SD* = 0.07), respectively. Capture (i.e., *p*1) and recapture probabilities (i.e., *c*) for M_b models averaged 0.37 (*SD* = 0.10) and 0.65 (*SD* = 0.08), respectively (Table 2-2). Total abundance of rodents across all sessions and grids averaged 69.3 (*SD* = 25.8) rodents and ranged from 30.2 to 119.0 rodents. Estimates of total rodent abundance for distant and proximate trapping grids under the observational design framework were similar [i.e., overlap of 95% confidence intervals (CIs)] during 87.5% (7 of 8) of trapping sessions (Table 2-3). The 95% CIs for rodent abundance at the distant and proximate grids under the BACI design framework overlapped during 75% of the trapping sessions (3 of 4) prior to the water manipulation; in all three cases rodent abundance was lower at the proximate grids (Table 2-1). Following the water manipulation, non-overlap of 95% CIs occurred during 100% (4 of 4) of trapping sessions; rodent abundance estimates were lower at proximate grids in all cases. There was no evidence of a difference between abundance estimates for distant versus proximate girds under the observational framework (z = -.04796, *d.f.* = 8, P = 0.63). There was evidence of a difference between abundance estimates at distant versus proximate girds under the BACI framework during the before (z = 4.94, *d.f.* = 4, P = > 0.01) and after period (z = 5.64, *d.f.* = 4, P = > 0.01). Pooled abundance estimates under the BACI design for distant and proximate grids during the before and after periods were 70.6 (*SE* = 4.64) and 55.05 (*SE* 3.91), and 69.5 (*SE* = 4.78 and 52.00 (*SE* = 3.66), respectively. The mean ratios of abundance for distant/proximate grids within the BACI design framework prior to and following the water manipulation were 1.33 (*SD* = 0.13) and 1.35 (*SD* = 0.06), respectively. The mean abundance ratio for distant/proximate grids within the observational design framework was 0.99 (*SD* = 0.09).

DISCUSSION

We found no evidence that water developments negatively affected total rodent abundance in our study area. Though estimates of total rodent abundance varied over time and across design types, we observed a general trend of paired distant and proximate grids exhibiting similar patterns of similar or disparate estimates. For example, trapping grids that comprised the observational trapping design exhibited similar estimates of abundance during 88% of trapping sessions. Conversely, findings from trapping grids comprising the BACI design revealed distant grids consistently had higher estimates of abundance than proximate grids, and this pattern did not change following water manipulation (Table 2-1).

Researchers have posited that water developments may act as a disturbance by negatively impacting individual species or communities indirectly in several ways (Broyles 1995, Larsen et al. 2012). Our investigation was primarily focused on the specific hypothesis that increased predation or predation risk occurs in areas near water developments, and such increases engender a negative effect to desert adapted species or communities (Hall et al. 2013). Our results did not support this premise, which may be attributable to several factors. First, predation rates of rodents in areas near water developments may not be disproportionate to areas unassociated with such sites. In west Texas, Atwood et al. (2011) reported mammalian carnivores partitioned use of water resources both spatially and temporally in an attempt to reduce interspecific interactions. Investigations focused on antagonistic interactions at water sites for raptor species are lacking, but behavioral modifications, including temporal segregation, have been noted as mechanisms to reduce antagonistic interactions among a host of competing raptor species (Sergio and Hiraldo 2008). Thus, species visiting water developments that regularly prey upon rodents may face a trade-off between timely maintenance of basic physiological needs (i.e., hydration, thermoregulation) and the cost of interspecific and intraspecific interactions (Atwood et al. 2011). Such a trade-off could facilitate such predators to forego foraging in areas in proximity of water developments prior to and/or following visitations.

Alternatively, increased predation rates, or predation risk on rodents, may have occurred near water developments, but may not have affected total rodent abundance. Predation induced mortality has been suggested to be both compensatory (Mihok 1988) and additive (Meserve et al. 1993) for individual rodent species, but empirical investigations examining the numerical impact of direct predation on community level abundance or productivity are lacking; this is likely due to the inherent difficulties of monitoring cause-specific mortality and other vital rates for a myriad of species comprising a community differing in life history strategies. Increased risk to predation at water development sites may have been offset by anti-predatory behavioral strategies. Behavioral strategies have been observed as mechanisms to mitigate predation risk within a host of rodent species and communities (Kotler et al. 1994, Brown et al. 1994, Shenbrot 2014), and the magnitude of behavioral responses has been shown to be correlated to the magnitude of predation risk for some species (Orrock and Danielson 2004). However, quantifying the overall demographic response of a rodent community to changes in predation risk is a challenging endeavor, as the type of response (i.e., changes in activity patterns, foraging behavior, or space use) and the impact of response on vital rates for each species comprising a community would need to be addressed.

Rodent abundance appeared to be at least partially influenced by spatial factors other than vegetation type; distant and proximate grids within the BACI design framework consistently differed in rodent abundance despite occurring in the same type of vegetation. Findings from Clark and Kaufman (1991) and Thompson and Gese (2013) suggested that measures of vegetation structure (e.g., percent bare ground, average shrub height) may more suitably explain/predict rodent community dynamics than traditional means of spatial classification (e.g., vegetation cover type). As a result, future investigations aimed at investigating impacts of disturbances and other phenomena on rodent total abundance, and other metrics used to describe these communities (e.g., species richness, total biomass), should account for vegetation structure during the study design phase. Because we sampled small mammals on the same plots over time, our plots did not undergo any major disturbances, and because our question was explicit to water sources, we did not incorporate plot-level vegetation measurements into our analysis.

Rodent abundance appeared to be partially driven by temporal factors, as we observed a trend of higher estimates in year 3 (i.e., session 5 and 6; Table 2-1). We speculate this finding was at least partially a result of a time lag effect between precipitation and rodent abundance. Ernest et al. (2000) reported a positive correlation among precipitation, plant productivity, and rodent abundance in the Chihuahuan Desert, with plant productivity responding to precipitation during the same growing season, and rodent populations lagging at least one season behind. A similar time lag correlation between precipitation and rodent abundance was observed for rodent communities in Chihuahuan Desert shrublands (Hernandez et al. 2005). Monthly precipitation rates at DPG during the 2011 growing season were the highest observed during our study (US Army Dugway Proving Ground, West Desert Test Center Meteorological Division, unpublished data). Thus, a general trend of increased plant productivity during the growing season prior to our 2012 trapping efforts may have resulted in increased rodent reproductive output or increases in survival, facilitating an increase in overall rodent abundance. Further investigations are needed to establish the temporal drivers of rodent abundance at DPG and other portions of the Great Basin Desert.

Our study appears to be the first to incorporate a resource manipulation design in order to evaluate the potential indirect effects of water developments on a rodent community. In addition, our study appears to be one of the first investigations involving rodent communities in arid environments that accounted for capture probabilities, and thus employed an actual estimate of abundance versus an index (i.e., number of captures or capture rate). Our findings suggested that water developments do not constitute a deleterious disturbance to rodent community abundance. It is important to note that our inference is limited to areas associated with water developments on DPG, and that factors not relevant to our study (i.e., livestock or feral horse visitation to water developments) may promote negative effects at water developments for rodent and other communities. We encourage future field investigations on water developments, and other potential disturbances, to incorporate study designs that include a manipulation component. BACI designs in particular should be incorporated into field investigations more frequently. Such investigations will be more feasible when disturbances are discrete points on a landscape (i.e., water developments) and can thus be readily identified and manipulated.

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Table 2-1.—Summarized rodent capture results across all trapping grids and sessions at Dugway Proving Ground, Utah, 2010-2013. Total number of trap nights was 25,088.

Species	Number of captures	Percentage of captures	Number of individuals captured	Percentage of individuals captured
Dipodomys ordi	3507	68.95	1423	66.34
Peromyscus maniculatus	798	15.69	374	17.44
Dipodomys microps	306	6.02	133	6.20
Chaetodipus formusus	171	3.36	62	2.89
Onychomys leucogaster	95	1.87	61	2.84
Reithrodontomys megalotis	81	1.59	49	2.28
Perognathus parvus	40	0.79	16	0.75
Neotoma lepida	14	0.28	9	0.42
Peromyscus truei	12	0.24	6	0.28
Ammospermophilus leucurus	8	0.16	7	0.33
Lemmiscus curtatus	4	0.08	1	0.05
Perognathus longimembris	1	0.02	1	0.05

Table 2-2.—Summarized abundance results for closed population capture-mark-recapture models (number of parameters for top model [*K*], population estimates for grids away from [Distant Grids (*N*)] and near to [Proximate Grids (*N*)] water developments, population estimate standard errors [SE (*N*)], whether 95% confidence intervals for distant and proximate grids overlap [95% CI], and the ratio of abundance for distant grids over proximate grids [*N* Ratio]) for each trapping session and study design type, Dugway Proving Ground, Utah, 2010-2013.

Trapping Session	Design Type	Top Model	*K*	Distant Grids (*N*)	SE (*N*)	Proximate Grids (*N*)	SE (*N*)	95% CI	*N* Ratio
1	BACI	Mh	5	44.50	3.27	30.20	2.44	no overlap	1.47
1	Observational	Mh	5	94.00	2.86	101.30	2.95	overlap	0.93
2	BACI	Mh	5	86.90	3.19	71.10	3.25	no overlap	1.22
2	Observational	Mb	4	78.50	4.95	71.30	4.83	overlap	1.09
3	BACI	Mh	5	91.00	6.52	78.10	5.77	overlap	1.17
3	Observational	Mb	4	66.60	5.90	81.50	6.80	no overlap	0.82
4	BACI	Mh	5	60.70	5.60	41.90	4.16	no overlap	1.45
4	Observational	Mh	5	52.90	3.59	48.50	3.36	overlap	1.09
5	BACI	Mh	5	88.00	4.78	64.20	3.75	no overlap	1.37
5	Observational	Mh	5	119.00	6.09	115.40	5.91	overlap	1.03
6	BACI	Mh	5	95.10	2.76	76.20	2.41	no overlap	1.25
6	Observational	Mh	5	109.00	3.74	103.60	3.61	overlap	1.05
7	BACI	Mh	5	43.00	7.20	30.70	5.01	no overlap	1.40
7	Observational	Mb	4	38.00	2.23	40.00	2.29	overlap	0.95
8	BACI	Mh	5	52.10	4.40	38.40	3.47	no overlap	1.36
8	Observational	Mh	5	50.00	3.72	54.00	3.97	overlap	0.93

Table 2-3.—Summarized capture and recapture probability results for top closed population capture-mark-recapture models (number of parameters for top model [*K*], capture probabilities [*p*1 and *p*2], recapture probabilities (*c*), associated standard errors (SE), and associated lower (UCI) and upper (UCI) bounds of 95% confidence intervals for each trapping session and study design type, Dugway Proving Ground, Utah, 2010-2013.

Trapping Session	Design	Top Model	*K*	*p*1	*p1* (SE)	*p1* LCI	*p1* UCI	*p*2	*p*2 (SE)	*p2* LCI	*p2* UCI	*c*	*c* (SE)	*c* LCI	*c* UCI
1	BACI	Mh	5	0.84	0.16	0.33	0.98	0.41	0.12	0.21	0.65	na	na	na	na
1	Observational	Mh	5	0.89	0.06	0.70	0.97	0.26	0.05	0.18	0.38	na	na	na	na
2	BACI	Mh	5	0.48	0.05	0.40	0.57	na	na	na	na	0.60	0.03	0.55	0.65
2	Observational	Mb	4	0.81	0.09	0.58	0.93	0.37	0.09	0.58	0.93	na	na	na	na
3	BACI	Mh	5	0.38	0.05	0.29	0.48	na	na	na	na	0.75	0.03	0.70	0.80
3	Observational	Mb	4	0.91	0.04	0.81	0.96	0.24	0.06	0.14	0.37	na	na	na	na
4	BACI	Mh	5	0.79	0.12	0.47	0.94	0.23	0.08	0.12	0.41	na	na	na	na
4	Observational	Mh	5	0.87	0.06	0.72	0.95	0.26	0.08	0.14	0.43	na	na	na	na
5	BACI	Mh	5	0.93	0.04	0.81	0.97	0.33	0.06	0.22	0.46	na	na	na	na
5	Observational	Mh	5	0.87	0.04	0.78	0.93	0.29	0.06	0.19	0.42	na	na	na	na
6	BACI	Mh	5	0.88	0.03	0.81	0.93	0.29	0.08	0.17	0.46	na	na	na	na
6	Observational	Mh	5	0.81	0.03	0.74	0.86	0.16	0.08	0.05	0.38	na	na	na	na
7	BACI	Mh	5	0.26	0.11	0.11	0.52	na	na	na	na	0.59	0.06	0.48	0.70
7	Observational	Mb	4	0.87	0.10	0.54	0.98	0.38	0.13	0.18	0.64	na	na	na	na
8	BACI	Mh	5	0.78	0.06	0.63	0.91	0.17	0.10	0.06	0.51	na	na	na	na
8	Observational	Mh	5	0.83	0.08	0.61	0.94	0.30	0.09	0.15	0.50	na	na	na	na

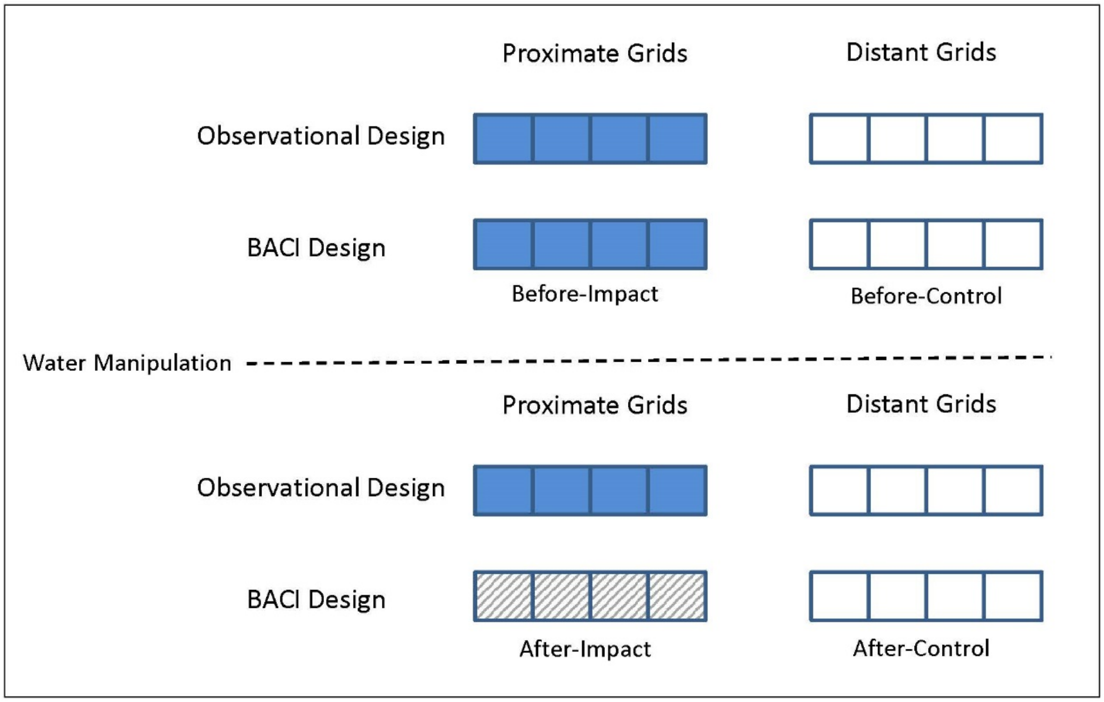

Fig. 2.1.—Diagram of study design frameworks. Each box represents a trapping grid. Water manipulation took place after the trapping sessions of 2010 and 2011 at water developments associated with the BACI design (n =4). Additional trapping sessions (n =4) took place following the water manipulation in 2012 and 2013. U.S. Army Dugway Proving Ground, USA.

Fig. 2.2.—Diagram of study design frameworks. Each box represents a trapping grid. Water manipulation took place after the trapping sessions of 2010 and 2011 at water developments associated with the BACI design (n =4). Additional trapping sessions (n =4) took place following the water manipulation in 2012 and 2013. U.S. Army Dugway Proving Ground, USA.

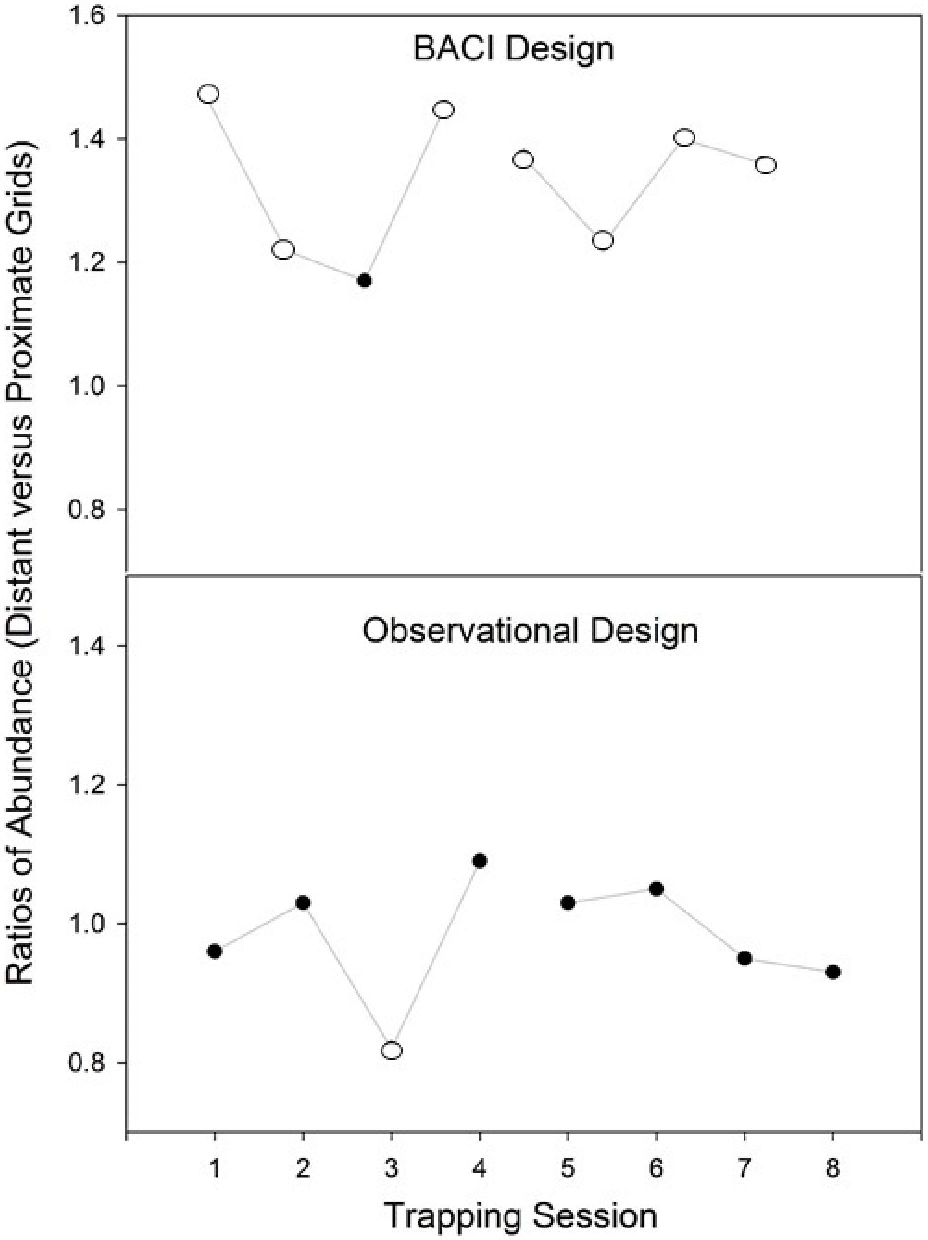

Fig. 2.3.—Ratios of abundance at trapping grids distant from and proximate to water developments at U.S. Army Dugway Proving Ground, USA 2010-2013. Water was removed from proximate grids associated with a BACI design (n =4) between sessions four and five. White circles represent non-overlap of 95% confidence intervals. An abundance ratio of 1.0 suggests no difference between abundance estimates at distant and proximate grids. White circles represent abundance comparisons derived from closed population capture-mark-recapture models that exhibited non-overlap of 95% confidence intervals.

CHAPTER 3

INFLUENCE OF WATER DEVELOPMENTS ON A DESERT CARNIVORE AND HERBIVORE[2]

Abstract

The anthropogenic manipulation of finite resources on the landscape in an attempt to benefit individual species or entire communities is commonly employed by wildlife management agencies, sportsmen groups, and conservation organizations. One such action in arid regions is the construction and maintenance of water developments (i.e., wildlife guzzlers) adding availability of free water on a landscape to increase local wildlife populations, influence animal movements, or affect distributions of certain species of interest. Despite their prevalence, the utility of wildlife guzzlers remains largely untested. We employed a before-after control-impact (BACI) design over a 3-year period on the U.S. Army Dugway Proving Ground, Utah, to determine whether water availability at wildlife guzzlers influenced the relative abundance of black-tailed jackrabbits (*Lepus californicus*) and coyotes (*Canis latrans*), and whether coyote visitations to guzzlers would decrease following elimination of water. Eliminating water availability at guzzlers did not impact jackrabbit relative abundance. However, relative abundance of jackrabbits appeared to be influenced by temporal factors such as precipitation. The relative abundance of coyotes was impacted by water availability, with elimination of water facilitating a reduction of coyote use in areas associated with our

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[2] Co-authors are Eric M. Gese and Steven J. Dempsey; chapter is formatted for PLOS One

treatment and a corresponding increase in relative abundance of coyotes in areas with no reduction of water availability. In addition, visitations of radio-collared coyotes to guzzlers declined nearly 4-fold following elimination of water availability. Our study provides the first evidence of a potential direct effect of water developments on a North American mammalian carnivore. Future investigations aimed at determining the effect of water developments on terrestrial mammals could expand on our findings by incorporating manipulations of water availability, obtaining absolute estimates of population parameters, and incorporating fine-scale spatiotemporal data.

Introduction

The manipulation of limited resources on the landscape in an attempt to benefit individual species and communities is a practice commonly employed by wildlife management agencies, sportsmen groups, and conservation organizations. One such action in arid regions is the construction and maintenance of wildlife water developments (i.e., wildlife guzzlers), which adds availability of free water on a landscape in order to buttress populations, influence animal movements, or affect the distributions of species of interest, particularly certain game species or endangered species [1, 2]. At the end of the last century, 10 of 11 western state wildlife management agencies in the United States reported ongoing water development programs with combined annual expenditures >$1,000,000 US [3], and as of 2013, nearly 7000 water developments had been constructed in the western United States [1]. Furthermore, water developments are being utilized as a mitigation technique to offset military activities [4] and are forecasted to increase as a conservation and management tool in the western United States [1].

Despite their prevalence, the utility of artificial water developments has been questioned. Researchers speculate whether increased availability of free water benefits or harms species that are adapted to desert or arid conditions [5, 6]. Essentially, the general notion that the direct uptake of free water by wildlife translates to a biological benefit [7] or that water use always equates to water need, has been challenged under certain conditions. Furthermore, others have posited that water developments may be deleterious, either by spreading disease, encouraging exotic species, hindering wilderness values, or negatively influencing populations of non-water dependent wildlife by increasing predation, predation risk, or competition [2, 4, 8-10].

Investigations lending empirical insight to the impacts of water developments on wildlife are rare, because the majority of studies have only indexed the uptake of free water at said developments without determining frequencies of use at the individual level, or the impacts of such use [11-15]. Though such studies have merit, they are unable to determine if use of free water translates to a biological or ecological effect. Adding further complexity is the notion that effects of water developments on wildlife can be either direct or indirect. Larsen et al. [1] defined the direct effects of water developments as those associated with the intake of free water [e.g., an increase in chukar (*Alectoris chukar*) survival due to chukar water intake]. In contrast, indirect effects included, but were not limited to, exploitative or interference competition with other species or conspecifics, or altered vulnerability to predation [e.g., a decrease in black-tailed jackrabbit (*Lepus californicus*) survival due to an increase in coyote (*Canis latrans*) survival, engendered by coyote water intake at water developments].

Investigations on direct effects of water developments on wildlife are sparse. Larsen et al. [16] found chukars utilized water developments in several mountain ranges but space use and diet were influenced by water development use only in certain ranges. In a seminal study including a before-after/control-impact (BACI) design [6, 17] found bighorn sheep (*Ovis canadensis*) used water developments, but reduction of water availability at developments did not influence population parameters or space use. Similarly for bighorn sheep, Broyles and Cutler [18] reported water developments did not impact population parameters, while Krausman and Etchberger [19] found no spatial affinity for water developments. Hall et al.[10] observed coyote (*Canis latrans*) relative abundance was similar in areas with and without water developments and springs despite coyote use at water sites, suggesting no direct effect of water consumption for this species.

Investigations into the potential indirect effects of water developments on wildlife are also lacking. Cutler and Morrison [20] found measures of species richness and relative abundance for small mammals and reptiles did not differ in areas adjacent to dry or wet water developments, suggesting predation rates at sites did not differ. DeStefano et al. [8] observed a negative relationship between leporid [(black-tailed jackrabbit and desert cottontail (*Sylvilagus audubonii*)] and coyote relative abundance in relation to proximity to water developments in the Sonoran Desert. This finding suggested an indirect effect of coyotes on the leporid community, either by way of predation or a lagomorph behavioral response to increased predation risk [21].Conversely, Hayden [22] observed that black-tailed jackrabbit abundance was higher near water sources in the

Mojave Desert, and believed this finding could be attributed to leaky or overflowing water tanks supporting patches of vegetation with high levels of preformed water, which was preferred forage for jackrabbits.

The limited, dissimilar, and predominantly observational findings speaking to direct and indirect impacts of water developments on wildlife species revealed the need for additional investigations, especially studies with an experimental component [1, 2, 8]. Two appropriate candidate species for such a study are the coyote and the black-tailed jackrabbit. Coyotes have been shown to regularly utilize water developments [10, 12] and it has been proposed that the species' physiological constraints and behavioral tendencies make them more likely to utilize and be dependent on free water than other desert-dwelling carnivores [23, 24]. Specifically, it has been posited that increases in anthropogenic water sources may be responsible for increases in coyote populations in arid regions of the Great Basin [25, 26]. Black-tailed jackrabbits, on the other hand, appear to persist by utilizing preformed water alone [27, 28]. Populations of these two species have been considered ecologically linked; jackrabbits often comprise the majority or a large proportion of coyotes' diet throughout areas of the western US [26, 29-31]. Coyote populations have also been shown to exhibit numerical responses to changing jackrabbit numbers in certain areas [29, 31-33].

The overall objective of our study was to elucidate the effects of water developments on the abundance of two desert-dwelling mammals, coyotes and black-tailed jackrabbits. Specifically, we used a BACI design to determine 1) whether water developments have indirect effects on black-tailed jackrabbit abundance, 2) whether water developments have a direct effect on coyote abundance, and 3) whether coyote visitations to water developments are reduced following elimination of water availability.

Methods

Ethics Statement

Fieldwork was approved and sanctioned by the United States Department of Agriculture's National Wildlife Research Center and the United States Army's Dugway Proving Ground. Permission to access land on the Dugway Proving Ground was obtained from the United States Army; permission to access Bureau of Land Management property was obtained from the Bureau of Land Management. Capture and handling protocols were reviewed and approved by the Institutional Animal Care and Use Committees (IACUC) at the United States Department of Agriculture's National Wildlife Research Center (QA-1734) and Utah State University (#1438). The Utah State University and National Wildlife Research Center IACUC committees specifically approved this study. Permits to capture, handle, and radio-collar coyotes were obtained from the Utah Division of Wildlife Resources (COR#4COLL8322).

Study Area

We conducted our research on 879 km^2 of the eastern portion of the U.S Army Dugway Proving Ground (DPG) located approximately 128 km southwest of Salt Lake City, in Tooele County, Utah, USA. Elevations ranged from 1302 m to 2137 m. The study site was in the Great Basin and was characterized as a cold desert. Winters were cold, summers were hot and dry, with the majority of precipitation occurring in the spring. Average maximum temperatures on DPG range from 3.3°C in January to 34.7°C in July. Average minimum temperatures ranged from –8.8°C in January to 16.3°C in July. Mean annual precipitation was 20.07 cm. The study area consisted of predominately flat playa punctuated with steep mountain ranges. The lowest areas consisted of salt playa flats sparsely vegetated with pickleweed (*Allenrolfea occidentalis*). Slightly higher elevation areas were less salty and supported a cold desert chenopod shrub community consisting predominately of shadscale (*Atriplex confertifolia*) and gray molly (*Kochia America*). At similar elevations, greasewood (*Sarcobatus vermiculatus*) communities were found with mound saltbrush (*Atriplex gardneri*) and Torrey seepweed (*Suaeda torreyana*). Higher elevations consisted of vegetated sand dunes including fourwing saltbush (*Atriplex canescens*), greasewood, rabbitbrushes (*Chrysothamnus* spp.), shadscale, and horsebrush (*Tetradymia glabrata*). Near the bases of the higher steep mountains were shrub steppe communities of sagebrush (*Artemisia* spp.), rabbitbrush, Nevada ephedra (*Ephedra nevadensis*), greasewood, and shadscale. The highest elevation was a Utah juniper (*Juniperus osteosperma*) community including black sagebrush (*Artemisia nova*) and bluebunch wheatgrass (*Elymus spicatus*). Where wildfires had occurred along the foothills, cheatgrass (*Bromus tectorum*), tall tumble-mustard (*Sisymbrium altissimum*), and Russian thistle (*Salsola kali*) was common within communities of sagebrush, rabbitbrush and juniper [25].

Besides several species from the families Heteromyidae and Cricetidae, the black-tailed jackrabbit was considered the most common mammalian species on DPG and surrounding areas [34]. Mountain cottontails (*Sylvilagus nuttalli*), pronghorn

(*Antilocapra americana*), mule deer (*Odocoileus hemionus*) and feral horses (*Equus ferus*) were also present. Grazing of livestock had not taken place on DPG for over 60 years. Coyotes were considered the most abundant mammalian carnivore on DPG [26]. Other resident carnivores included cougars (*Puma concolor*), bobcats (*Lynx rufus*), and kit foxes (*Vulpes macrotis*).

Experimental Design and Sampling

We established four 5-km road-based survey transects whose midpoints were adjacent to wildlife water development sites [hereafter wildlife guzzlers (model Dual Big Game, Boss Tanks, Elko, NV)]. These transects (hereafter proximate transects) served as our treatment transects because they were associated with a water development. The average distance from proximate transects to the next nearest perennial water source (i.e., pond, water development, sewage lagoon) was 4.10 km (SD = 0.54). We used ArcGIS (version 9.3, Environmental Systems Research Institute, Redlands, CA) to create four additional 5-km transects (hereafter distant transects) which were distributed randomly along available non-paved roads with the constraints of occurring on lengths of road with no angles >60 degrees, a minimum spacing distance of 2.6 km from proximate transects, and a minimum spacing of 2.6 km from a perennial water source. This minimum distance was derived from the square root of home ranges for coyotes inhabiting a semi-arid environment similar to our study area [10, 35] The square root of the home range is a linear measure used to approximate average daily movements of mammals [36] and has been encouraged and incorporated into the spatial design of water development investigations in general and within our study area [1, 10]. Black-tailed jackrabbit home ranges have consistently been reported as being smaller than coyotes [37], so we were confident that 2.6 km was a distance greater than average daily movements for black-tailed jackrabbits and coyotes. We therefore had four replicates of the two types of transects (i.e., proximate and distant).

We employed a multiple-treatment site, multiple-control site BACI design [17] where we monitored all transects prior to and after eliminating water availability at water developments. BACI designs are considered superior to observational studies because they better account for variability of response and exploratory variables attributed to temporal (e.g., annual precipitation) and spatial factors (e.g., vegetation heterogeneity across study area) that cannot always be controlled and/or accounted for under natural environmental conditions [17]. In April 2012, we drained the four wildlife guzzlers associated with proximate transects using a generator (model 4000-Watt, Champion Power Equipment, Santa Fe Springs, CA) and submersible pump (model 1/2 HP, Wayne Pumps, Harrison OH), and drinking portals were covered with plywood (Figure 3-1). Water levels were checked monthly and we re-drained them if they reached >2/3 capacity. Surveys taking place on proximate and distant transects prior to the water manipulation period were considered the pre-period, while surveys following the water manipulation were considered the post-period.

We used nocturnal vehicle-based spotlight surveys [38] to estimate relative abundance of jackrabbits along the eight 5-km transects. While driving along transects at approximately 10-15 km/hr, two observers scanned their respective side of the road and the road itself with a 3-million candlepower spotlight [39]. Surveys were conducted under clear and calm conditions between 1 h after dusk and 1 h before sunrise for three consecutive nights, resulting in a total of 24 separate spotlight counts per survey (i.e., three counts for each transect). The order of transects surveyed in a given night was randomized. Once an animal was sighted the driver stopped the vehicle and the species of leporid was identified. Species, location, distance, and bearing to the animal were recorded for each sighting. Surveys were conducted along the eight 5-km transects previously described. Surveys were temporally spaced so that we conducted one survey within each 4-month season based on energetic needs of coyotes: breeding 15 December – 14 April, pup-rearing 15 April – 14 August and dispersal 15 August – 14 December [40, 41]. When possible, we performed an additional intra-season survey, with ≥ 2 month spacing between surveys. Spotlight counts provided an index of relative abundance; the number of jackrabbits observed per transect per night. A combination of our survey effort/design and a seemingly low jackrabbit density during our study [34] did not allow for the minimum number of observations needed to robustly estimate absolute abundance using distance sampling [42] or N-mixture models [43]. We felt justified in utilizing spotlight counts as an index of relative abundance; they have been utilized to quantify hare and jackrabbit relative abundance across time and space [31, 44], and have been shown to be highly correlated with absolute abundance estimates that account for detection probabilities, when data sets are robust enough for such comparisons [45,46].

We conducted scat deposition surveys [47-49] to estimate the relative abundance of coyotes. As a passive technique, scat deposition surveys do not require the target species to behave unnaturally (e.g., investigate a scent tab). This may be beneficial, especially with species like coyotes which are wary of novel cues [50]. Surveys were conducted by initially walking the transect to clear any scat from the road surface, then returning 14 days later to walk and count the number of scats [48-49]. Following recommendations from Knowlton [47], each transect was walked in both directions to reduce missed detections of scats. Surveys were conducted along the same eight 5-km transects as the jackrabbit surveys. Scat surveys were temporally spaced in the same manner as jackrabbit surveys. Hence, each survey consisted of eight scat deposition counts (i.e., one scat deposition count per transect). We identified coyote scats based on guidelines described in [51]. Scat deposition counts provided an index of coyote abundance; the number of coyote scats per transect per survey. Scat surveys have been reported as an effective index for tracking coyote relative abundance over time and space [33, 47] and have outperformed other noninvasive surveys for mammalian carnivores [48, 52, 53].

Prior to and following our manipulation of water availability, we monitored coyote monthly visitation rates to the water developments using a sample of radio-collared coyotes inhabiting DPG. We monitored visitation of radio-collared coyotes to the water development sites with data loggers (model R4500S and model R2100/D5401, ATS, Isanti, MN) and an omnidirectional antenna following recommendations of Breck et al. [54]. We defined a visit as all data logger recordings of an individual animal occurring within 30 min at a particular water source [55]. Data loggers were calibrated to detect a signal at an average distance of 10 m from the antennae, which were placed 8 m from guzzler drinking portals. The area of signal detection uncertainty [54] was < 3 m at all data logger sites. We considered all data logger detections as visits to guzzlers.

Data Analyses

We employed generalized linear mixed models (GLMMs) [56] to test the categorical main effects of period (pre- and post-manipulation) and transect type (proximate and distant) on the continuous response variables of jackrabbit and coyote relative abundance. Specifically, we tested the impact of water development manipulation by including a period by transect type interaction in our model [57]. Within the framework of a BACI design, such an interaction tests for a differential change (i.e., non-parallelism) between impact and control sampling units following some type of manipulation [57]. Inspection of raw data revealed non-normality for both data sets. As a result we fit the following model families: lognormal, Poisson, quasi-Poisson, and negative binomial. Models that did not converge were eliminated and we assessed remaining models based on the generalized chi-square fit statistic [56]. For the jackrabbit and coyote data, the final model family used was quasi-Poisson and lognormal, respectively.

For both species, we conducted multiple surveys on each transect for both periods. In order to reduce model complexity and better account for residual variance, we collapsed our original data sets across surveys. By doing so, data were analyzed within a balanced split plot in a time model framework [58]. In order to account for variability among survey transects, and variability among survey transects within treatments, we included a survey transect (i.e., proximate or distant) by period (pre- and post-manipulation) random effect [59]. GLMM analyses were performed using the GLIMMIX procedure in SAS (version 9.4, Cary, North Carolina).

We determined if the number of radio-collared coyote visits to water developments decreased during the pre-period by comparing the number of monthly data logger visitations prior to and following our manipulation. To ensure the visitation data were not biased by sample size, we compared the number of radioed coyotes available for data logger recording each month for both periods. We used a one-way analysis of variance (ANOVA) in SAS for these comparisons. For all statistical tests we interpreted p-values in terms of relative evidence of difference [60]. Reported means, standard deviations, standard errors, and 95% CIs were derived from the raw data, rather than model driven estimates.

Results

Between September 2010 and August 2013, we conducted 7 jackrabbit surveys prior to and 5 surveys following our manipulation. Jackrabbit relative abundance across all surveys averaged 3.07 rabbits/transect/night (SD = 2.60) and ranged from 0 to 19 rabbits/transect/night. We found no evidence that elimination of water at guzzlers impacted jackrabbit relative abundance (period x transect type interaction: F = 0.41, P = 0.54, df = 1, 6; Figure 3-1). There was evidence that period influenced jackrabbit relative abundance (F = 5.76, P = 0.05, df = 1, 6; Figure 3-2). Average jackrabbit relative abundance during the before and after period for all transects was 2.68 (SE = 0.13) and 3.87 (SE = 0.35) rabbits/transect/night, respectively. There was no evidence that transect type influenced jackrabbit relative abundance (F = 1.40, P = 0.28, df = 1, 6).

Between September 2010 and August 2013, we conducted 4 seasonal coyote scat deposition surveys prior to and following our manipulation. Overall, coyote relative abundance averaged 6.01 scats/transect/survey (SD = 5.91) and ranged from 0 to 27 scats/transect/survey. We found evidence that elimination of water at guzzlers influenced coyote relative abundance (period x transect type interaction: F = 10.61, *P* = 0.02, df = 1, 6; Figure 3-3). The number of coyote scats observed on distant transects increased from 3.50 scats/transect during the pre-period (SE = 1.06) to 5.50 scats/transect (SE = 1.62) during the post-period. Conversely, the number of coyote scats observed on proximate transects decreased slightly from 9.25 scats/transect (SE = 2.79) during the pre-period to 8.50 scats/transect (SE = 1.62) during the post-period. We found some evidence that period influenced coyote relative abundance (F = 4.22, *P* = 0.09, df = 1, 6). There was no evidence that transect type influenced coyote relative abundance on its own (F = 2.58, *P* = 0.15, df = 1, 6). Average relative abundance during the pre- and post-manipulation periods for all transects was 6.35 (SE = 0.81) and 7.10 (SE = 0.86) scats/transect/survey, respectively.

For the pre-period and post-period we monitored visitations of radio-collared coyotes at wildlife guzzlers from May 2010 to April 2012 and May 2012 to August 2013, respectively. There was no evidence that the number of radio-collared coyotes (i.e., number available for monthly data logger recording) differed prior to and following water removal (F = 1.05, *P* = 0.31 df = 1, 37). The monthly sample size of marked coyotes during pre-period and post-period averaged 18.74 (SE = 0.94) and 20.25 (SE = 1.13), respectively. There was evidence that monthly visitations by radio-collared coyotes to wildlife guzzlers was influenced by the elimination of water (F = 6.19, *P* = 0.02, df = 1, 37) with the elimination of water reducing visitation by coyotes (Figure 3-4).

Discussion

Our study was the first to incorporate a resource manipulation design to evaluate the potential effects of water developments on canids and leporids in an arid environment. Overall, we found no evidence that the relative abundance of jackrabbits was influenced by anthropogenic water developments, but found support that our manipulation influenced the relative abundance of coyotes, and that coyote visitations to water developments declined following removal of water availability.

A potential indirect effect of water developments is suppressed populations of prey species of water dependent carnivores [1, 3, 8]. Our findings differ from those of DeStefano et al. [8] in that we found no evidence of an indirect impact of water developments on black-tailed jackrabbits. This disparity may be attributed to several factors. First, our experimental design may have allowed us to account for sources of bias that can go undetected with purely observational studies [57]. For example, if jackrabbit abundance had been greater near proximate rather than distant transects, our BACI design would have allowed us to determine if any such disparity was attributed to water developments, and not some other factor(s). Second, our sampling design (5-km transects) may have better captured changes in the trend of the jackrabbit population across a larger landscape. Alternatively, the spatial extent of our transects may have been too large to detect differences in jackrabbit occurring at close proximity (e.g., > 1-km) to guzzlers.

We found no evidence suggesting treatment type influenced the relative abundance of jackrabbits. This seems contradictory based on visual inspection of the data (Figure 3-2). This can be explained by the majority of the variation among treatments occurring due to variation at the survey transect level, rather than the transect type level. This likely occurred because proximate and distant transects were not established across uniform vegetation classes. That is, we did not stratify across vegetation classes, or other spatial factors that may have influenced jackrabbit relative abundance. The reason for this was two-fold. First, DPG contains high levels of vegetation heterogeneity (see methods for full description). As a result, we felt the establishment of transects partitioned by vegetation classes would not have provided sample sizes needed to adequately address our central research questions. Second, efforts aimed at discerning the role of vegetation on populations of black-tailed jackrabbits, and closely related species, have already been undertaken [31, 61, 62], though we do encourage additional such investigations. For example, previous research efforts do not appear to fully elucidate the role of exotic vegetation invasions [i.e., cheat grass (*Bromus tectorum*) invasion)] on leporid population processes in desert systems.

Jackrabbit relative abundance appeared to be partially driven by temporal factors, as we observed higher relative abundance during the post-manipulation period of the study (Figure 3-2). It was not our objective in this study to identify the suite of factors influencing the jackrabbit population at DPG, but speculate that this temporal trend was at least partially a result of a time lag effect between precipitation and jackrabbit abundance. Hernández et al. [31] reported a positive relationship between the previous

12 months of precipitation and both primary productivity and jackrabbit abundance. Similarly, Ernest et al. [63] reported rodent abundance was positively correlated to precipitation occurring during the previous season or seasons. Monthly precipitation rates at DPG during 2009 (i.e., a span potentially influencing pre-period jackrabbit relative abundance) and 2011 (i.e., a span potentially influencing post-period jackrabbit relative abundance) averaged 1.14 (SE = 0.25) and 1.96 cm (SE = 0.66), and a precipitation spike of 10.06 cm occurred in May of 2011, eleven months prior to our manipulation (US Army Dugway Proving Ground, West Desert Test Center Meteorological Division). Thus, a general trend of increased primary productivity leading up to the post-manipulation period may have resulted in increased jackrabbit reproductive output, facilitating an increase in overall jackrabbit abundance. Further analyses are needed to establish the drivers of jackrabbit abundance at DPG and other arid regions of the Great Basin.

Our data suggested the relative abundance of coyotes was impacted by water availability with the elimination of water availability at water developments facilitating a reduction of coyote use or abundance in areas associated with our treatment (Figure 3-3). This reduction coincided with an increase in coyote relative abundance in areas not associated with our manipulation (i.e., distant transects). This finding substantiates claims that additional free water on desert landscapes may have prompted coyote population increases in the Great Basin Desert [25, 26] and similar increases for other carnivores in other arid ecosystems [64]. If the DPG coyote population was predominantly driven by jackrabbit abundance we would have expected to see a similar rate of increase in coyote abundance across all transects (i.e., a numerical response to increases in prey resources). Jackrabbits have been reported as a primary food source for coyotes at DPG [26] and other western U.S. populations [30-32], and numerical responses of coyote populations to changes in hare density have been observed (31, 32, 65]. Our study may not have had a sufficient temporal span to detect a coyote population numerical response to increased jackrabbit density.

Our findings suggest a direct effect of water developments on coyotes. These findings differ from those of Hall et al. [10], despite both investigations encompassing similar spatial and temporal boundaries. Several mechanisms may be responsible for this discrepancy. First, the behavioral ecology of coyotes may be a driving factor. The sampling technique we employed (scat deposition survey) requires an animal only engage in evacuation behavior in order to be detected/counted. Other sampling techniques, such as scent-station surveys [10, 31] require that an animal behaviorally react to a novel olfactory cue. Coyotes have been shown to be wary of novel cues [50]. As a result, the use of novel cues as part of a sampling technique may introduce sources of bias, especially in a species like the coyote, where variability across the bold/shy continuum occurs [66]. In addition, human exploitation is often a predominant source of coyote mortality [67] with the use of olfactory lures at traps a commonly utilized exploitation method [68] and such efforts are not always successful [i.e., some animals encounter but escape/elude traps; [69, 70]. As a result, olfactory cues intended to serve as an attractant may actually deter a portion of coyotes in a given area due to behavioral tendencies engendered by innate and/or learned mechanisms. Hence, the use of more passive, less behavioral dependent sampling techniques (i.e., scat surveys) may reduce sampling bias. Second, the relatively large size of our sampling units (5-km transects) may be more appropriate for capturing population changes/trends of coyotes. Finally, our manipulation of water developments may have captured an effect that would often go undetected with purely observational studies. Our investigation was not designed to determine the ultimate causes responsible for the reduced relative abundance that we observed, but rather to test whether water sources are an influential factor. For example, our manipulation may have facilitated abandonment by some resident coyotes, shifts of home range boundaries and space use, increased dispersal rates of juveniles (i.e., a reduction of philopatry tolerance among packs or breeding pairs), reduced fecundity, or a combination thereof.

We demonstrated over a three-fold reduction in use of water developments by coyotes following elimination of water (Figure 4). Our data on visitations were for marked individuals only, and alone cannot fully explain our relative abundance findings, as visitations were relatively low. Our visitation results would have been buttressed if we could report the same relationship for all DPG coyotes that visited treatment water developments, rather than a radio-collared sample. In addition, determining if coyote visitations increased at other water sources within the study area following our manipulation would have helped elucidate the importance of free water to coyote populations. Marked coyotes were captured throughout the study area using several techniques (e.g., helicopter net gunning, leg-hold trapping) and efforts were made to mark only one individual per social group. In addition, from 2011 to 2012, Hall et al.

[10] recorded 869 coyote visitations (i.e., drinking events) at water developments within a study area that encompassed DPG. This investigation, however, concluded near the onset of our manipulation. Given the aforementioned, we feel that our coyote visitation findings are germane with respect to our other study findings, and provide at least partial evidence that overall coyote visitations to water developments were reduced following our manipulation.

Our study was only the second to utilize a study design with a resource manipulation component in order to determine the effect of water developments [6]. We encourage future field investigations on water developments, and other resources on the landscape, incorporate study designs that include an experimental manipulation component. BACI designs in particular should be incorporated into field investigations more frequently. Such investigations will be more feasible when potential limiting resources are discrete points on a landscape (i.e., water developments) and can thus be readily identified and manipulated. Though we did not determine an indirect effect of water developments on black-tailed jackrabbits, our findings revealed a differential change in coyote relative abundance in relation to elimination of water. In addition, we observed that visitations of coyotes at water sources were reduced following our water manipulation. Though our findings suggest that coyote populations are directly affected by water developments, caution is warranted. Falsely equating statistical significance to biological relevance is a real, if not often an ignored risk in ecological investigations [71], and despite their common validation [46, 72, 73] indices of abundance can be problematic [74, 75]. As such, we recommend future investigations on the effects of water developments on individual species and ecological interactions incorporate absolute rather than relative estimates of population parameters into their study design. Such studies should always incorporate appropriately scaled spatial data and analyses.

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Figure 3-1. **Example of water development prior to (A) and following (B) removal of available water developments on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.**

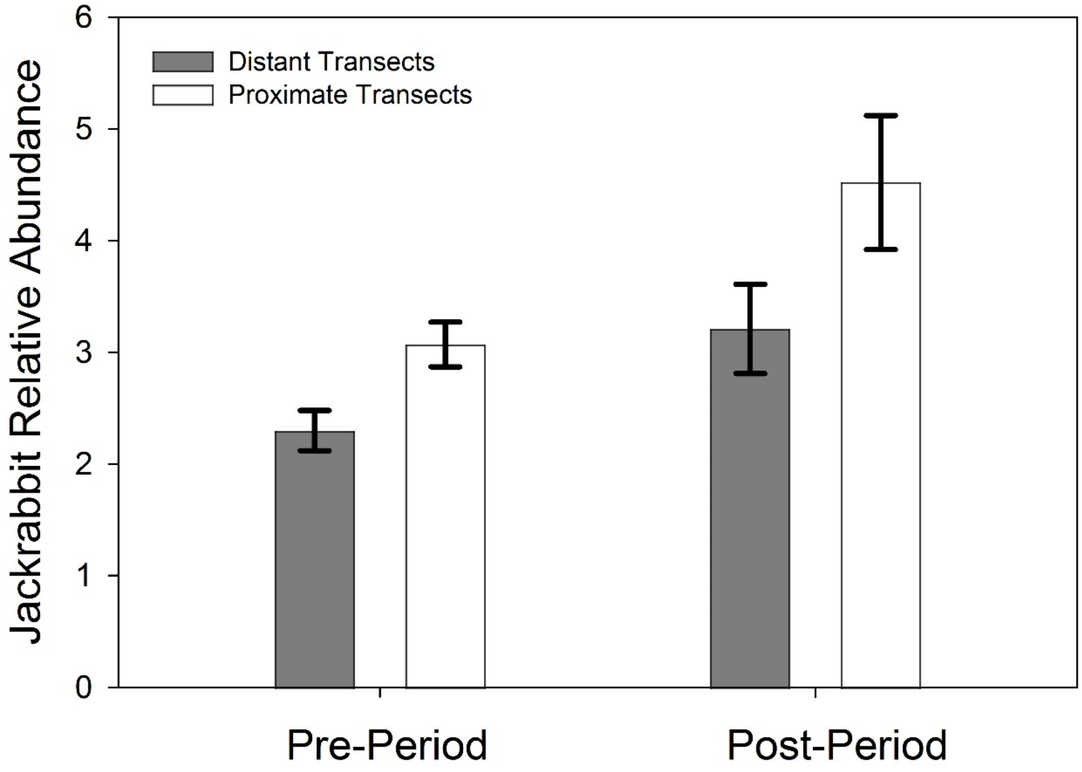

Figure 3-2. **Black-tailed jackrabbit relative abundance (± SE) observed on 5-km proximate and distant to guzzler transects prior to (pre-period) and following (post-period) removal of water availability at guzzlers on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.**

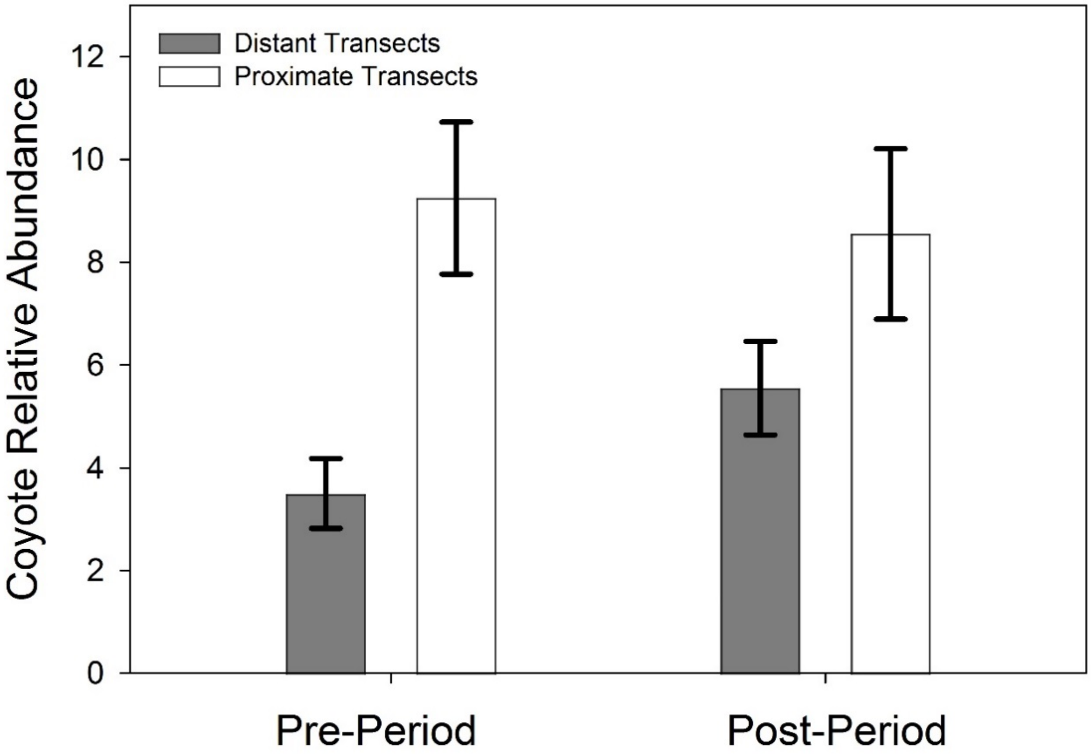

Figure 3-3. **Coyote relative abundance (± SE) observed on 5-km proximate and distant to guzzler transects prior to (pre-period) and following (post-period) removal of water availability at guzzlers on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.**

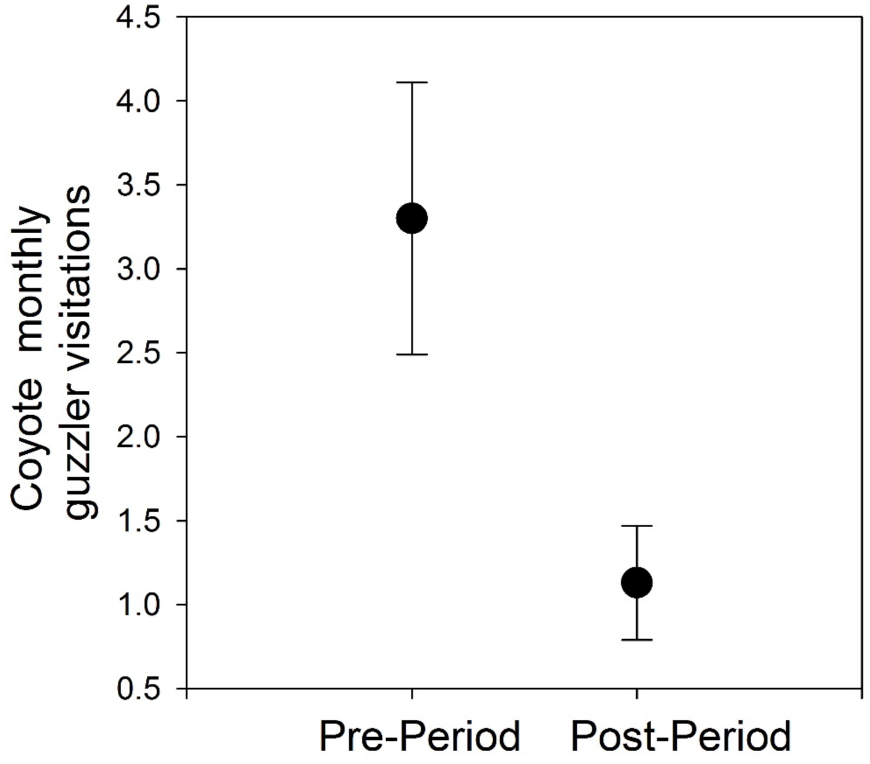

Figure 3-4. **Average monthly visitations (± SE) of a marked coyote population prior to (pre-period) and following (post-period) removal of water availability at guzzlers on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.**

CHAPTER 4

SPATIAL RESPONSE OF COYOTES TO REMOVAL OF WATER AVAILABILITY AT ANTHROPOGENIC WATER SITES[3]

Abstract

Features containing year-round availability of free water (hereafter water sites) and areas affiliated with water sites (i.e., riparian zones) occurring within arid landscapes represent a potential limiting resource for some desert dwelling vertebrates. Little is known about the relationship between water sites and mammalian carnivores. An increase of water sites in portions of the Great Basin Desert reportedly contributed to an increase in coyote (*Canis latrans*) populations. We examined frequency of visitation and spatial affinity of resident coyotes for water sites at the home range scale extent. Visitation to sites with available water averaged 13.0 visitations/season (SD = 13.5) and ranged from zero to 47. We documented no visits to water sites in 16% (10 of 64) of seasonal home ranges, <5 visits within 39% (25 of 64) of home ranges, and 25% (28 of 113) of coyote home ranges did not contain a water site. Water sites associated with riparian vegetation experienced higher visitation than wildlife water developments (no riparian vegetation present). We found no evidence that elimination of water availability influenced home range size or spatial shifting of home range areas. Water sites, especially wildlife water developments, do not represent a pivotal resource for the coyote population in our study area.

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[3] Co-authors are Eric M. Gese; chapter is formatted for Journal of Arid Environments.

Introduction

Identifying the extent to which organisms utilize certain resources on a given landscape, and the impact of such use, has become a central tenet of animal ecology. Investigations determining the spatial relationships between animals and the resources they utilize can guide conservation and management strategies (Morris, 2003; Onorato et al., 2011; Briggs et al., 2012) and predict the impacts of varying land use (Wilson et al., 2014) and climate change scenarios (Costa et al., 2010). It has been long established that resources available to animals in a given spatial mosaic are often used at variable levels (Manly et al., 2002; Begon et al., 2005). Resources can serve as a requisite component of species habitat (Shroeder et al., 2004; Cain et al., 2012; Edgel et al., 2014), while other resources may not be required, but utilized (Manly et al., 2002).

Landscape features with year-round availability of free water (hereafter water sites) and adjacent areas affiliated with water sites (i.e., riparian zones) occurring within arid landscapes represent a potential limiting resource. Many species of terrestrial vertebrates are dependent on water sites (Gill, 2006); regular intervals of free water uptake are needed to maintain metabolic functions necessary for an individual's survival (Silanikove, 1994; Larsen et al., 2012). Other species of vertebrates utilize water sites for drinking as a resource subsidy; they have the ability to persist on preformed or metabolic forms of water alone (Harrington et al., 1999; Cain et al., 2008; Hall et al., 2013). In most cases, investigations focusing on water uptake and wildlife chronicled overall use (e.g., visitations to or activity/sign at water sites) at the species or community level (e.g., an index) rather than determining patterns of individual water use (Rosenstock, et al.

2004; Morgart et al., 2005; Jennifer et al., 2010; Whiting et al., 2010). Such individual based investigations are needed to determine water site visitations per individual, the proportion of a population utilizing water sites, and to determine the relevancy of water sites as a habitat component (Shields et al., 2012). In addition to providing water uptake opportunities, water sites can facilitate establishment of riparian vegetation that provide resources that confer a reproductive, nutritional, safety, or thermoregulatory benefit to a degree greater than areas not affiliated with water sites (Bock and Bock, 1984; Doyle, 1990; Schulz and Leininger, 1991; Shalfroth et al., 2005).

Water sites influence individual space use and species habitat quality for a host of terrestrial vertebrates (Harrington et al., 1999; Allen, 2012; Cain et al., 2012; Ogutu et al., 2014), or can have little to no impact (Krausman and Etchberger, 1995; Cain et al., 2008). The majority of investigations focused on populations of large herbivores in xeric landscapes, where water sites are more influential than in mesic landscapes (Larsen et al., 2012). Such an emphasis on this group of animals is likely due to a host of factors including, but not limited to, logistical (e.g., VHF or GPS transmitter mass) and political (e.g., the disproportionate amount of research funding allocated toward game versus nongame animals) factors (Simpson et al., 2011).

Infrequent investigations have examined the relationship between water sites, water use, and the influence of such use on mammalian carnivores. Allen (2012) reported that 100% of GPS-collared dingoes (*Canis lupus dingo*) regularly visited water sites, though the frequency of visitations varied by individuals and temporal factors, and suggested the dingo population was dependent on water sites. However, determining patterns of use and the impact of water sites on many desert dwelling carnivores has not been achieved; to date, investigations have only chronicled indexes of visitations to water sites (Rosenstock et al., 2004; Atwood et al., 2011; Hall et al., 2013) or indexed activity for areas distant from and close to water sites (Hall et al., 2013).

Coyotes (*Canis latrans*) occur in a host of wildland, rural, and urban landscapes across a broad spectrum of mesic and arid environments (Bekoff and Gese, 2002, but the degree to which this species utilizes water sites, and the relationship between water sites and space use remains unexplored. Coyote populations are often managed due to issues relating to human-wildlife conflict (Knowlton et al., 1999; Conner et al., 2008; Poessel et al., 2013) or conservation of threatened or imperiled species competing with coyotes (Cypher et al., 2000; Moehrenschlager et al., 2007). It has been posited that the distribution and abundance of coyotes in the Great Basin Desert has increased in part due to the addition of water sites, by way of relaxing the limitation of arid systems to coyotes (Arjo et al., 2007; Kozlowski et al., 2008), thus increasing overall habitat quality for coyotes (Kozlowski et al., 2012). As a result, discerning the relevancy of water sites to coyotes has both management and conservation implications.

The physiological demands and behavioral characteristics of coyotes are such that water sites are more likely to be utilized than more desert-adapted carnivore species, like the sympatric kit fox (*Vulpes macrotis*) (Golightly and Omart, 1983), a species of conservation concern is several western states (Dempsey et al., 2014). For example, in the absence of water, coyotes theoretically need to consume 3.5 times the number of prey items than kit foxes to meet energetic requirements (Golightly and Ohmart, 1984). Thus, if prey items are a limiting factor on a landscape the addition of free water sites could serve as a resource subsidy to coyotes. Coyotes in the Great Basin Desert were classified as rare during the 1950s (Shippee and Jollie, 1953) and coyote abundance in this area has increased since the 1970s (Arjo et al., 2007). Further, kit fox density has been found to be negatively correlated with coyote abundance (Arjo et al., 2007), and it has been posited that a marked increase of permanent water sites in the Great Basin Desert since the mid-twentieth century may have indirectly decreased available kit fox habitat by way of increased interspecific competition and intraguild predation from coyotes, leading to reduced kit fox abundance (Arjo et al., 2007; Kozlowski et al., 2008; Kozlowski et al., 2012).

Clearly, further investigation is needed to determine the extent to which water sites are utilized by coyotes in arid landscapes, and if water sites represent a requisite habitat component for coyotes in arid regions. If water sites represent a limiting factor for a coyote population, it would be expected that coyote home ranges will overlap with water sites, water sites that overlap home ranges would be regularly utilized by coyotes, and a reduction of available water sites would prompt a spatial response by coyotes. Elucidating the relationship between water sites and coyotes has the potential to influence kit fox conservation strategies and coyote management programs, as well as increase our general understanding of the effects of free water on wildlife in arid environments. The overall objective of our study was to determine the impacts of water sites on coyotes in an arid landscape. Specifically, we aimed to determine: 1) the frequency of water site visitations by individual coyotes, 2) whether removal of water availability at water sites reduces coyote visits to water sites, 3) if the removal of water availability at water sites facilitates a change in coyote home range sizes, and 4) if removal of water availability at water sites facilitates a shift of coyote home range areas.

Methods

Study Area

We conducted our research on 1127 km^2 of the eastern portion of the U.S Army Dugway Proving Ground (DPG) and the adjoining lands managed by Bureau of Land Management (BLM), located approximately 128 km southwest of Salt Lake City, in Tooele County, Utah, USA (Fig. 4-1). Elevations ranged from 1302 m to 2137 m. The study site was located in Great Basin Desert, where winters were cold, summers were hot and dry, with the majority of precipitation occurring in the spring. Annual weather consisted of mean air temperatures of 12.7°C (range: -20.0 to 40.6°C) and mean precipitation of 150 mm (MesoWest, Bureau of Land Management & Boise Interagency Fire Center). In the study area, we identified 23 permanent water sites consisting of 10 water developments for wildlife (hereafter guzzlers), 4 natural springs, and 9 man-made ponds/catchments. Guzzlers were designed to allow no run-off or access to water by rooted vegetation. Thus, there was no riparian vegetation component associated with guzzlers. In addition, the eastern portion of the study area managed by the BLM contained 3 livestock tanks that were at times operational during winter and spring cattle grazing (November 1 to April 1). Springs and man-made ponds were often associated with riparian communities primarily comprised of tamarisk (*Tamarix ramosissima*) (Emrick and Hill, 1999). Anthropogenic water sites (i.e., guzzlers, ponds, and livestock tanks) were developed between the 1960s and 1990s (Arjo et al., 2007). Thus, the ratio of anthropogenic to natural water sites within the study area was at least 3:1, with slight seasonal variability occurring due to the turning on/off of livestock tanks. We inspected all permanent water sites (e.g., ponds, springs, guzzlers) and livestock tanks within the study area monthly to confirm water availability. Water sites were considered permanent if they contained water during ≥3 of the monthly checks for each 4-month canid biological season and year (e.g., 2011 breeding season; Dempsey et al., 2014). There was no free-flowing water present on the study area. Additional water sites (e.g., hardpans, rainfall, drainages) were ephemeral pools (<1 week); thus we assumed they were homogenous throughout the study area and did not influence overall space use of coyotes relative to water sites.

The study area consisted of predominately flat playa punctuated with steep mountain ranges. The lowest areas consisted of salt playa flats sparsely vegetated with pickleweed (*Allenrolfea occidentalis*). Slightly higher elevation areas were less salty and supported a cold desert chenopod shrub community consisting predominately of shadscale (*Atriplex confertifolia*) and gray molly (*Kochia America*). At similar elevations, greasewood (*Sarcobatus vermiculatus*) communities were found with mound saltbrush (*Atriplex gardneri*) and Torrey seepweed (*Suaeda torreyana*). Higher elevations consisted of vegetated sand dunes including fourwing saltbush (*Atriplex canescens*), greasewood, rabbitbrushes (*Chrysothamnus* spp.), shadscale, and horsebrush (*Tetradymia glabrata*). Near the bases of the higher steep mountains were shrub steppe communities of sagebrush (*Artemisia* spp.), rabbitbrush, Nevada ephedra (*Ephedra nevadensis*), greasewood, and shadscale. The highest elevation was a Utah juniper (*Juniperus osteosperma*) community including black sagebrush (*Artemisia nova*) and bluebunch wheatgrass (*Elymus spicatus*). Where wildfires had occurred along the foothills, cheatgrass (*Bromus tectorum*), tall tumble-mustard (*Sisymbrium altissimum*), and Russian thistle (*Salsola kali*) was common within communities of sagebrush, rabbitbrush and juniper (Arjo et al., 2007).

Animal Capture and Handling

Beginning in December 2009, we captured coyotes via helicopter net-gunning (Gese et al., 1987) or foothold traps (#3 Soft Catch, Oneida Victor Inc., Euclid, OH) affixed with a trap tranquilizer device (Sahr and Knowlton, 2000). Processing of coyotes included taking blood samples, affixing ear tags and recording weight, sex and morphological measurements. We aged individuals as pups (< 9 mo old), yearlings (9–21 mo) or adults based on tooth wear, tooth eruption and body size (Gese et al., 1987). We fitted adult animals with a 200 g very high frequency (VHF) radio-collar (Model M2220; Advanced Telemetry Systems, Isanti, MN). Coyotes were captured throughout the study area and efforts were made to radio collar only one individual per social group. We limited capture efforts to October through February of each year so as to not interfere with parturition and pup rearing.

Radio-telemetry and Home Range Determination

We located animals >3 times per week using a portable receiver (Model R1000; Communications Specialists, Inc., Orange, California) and a handheld 3-element Yagi antenna. We triangulated an animal's location using ≥3 compass bearings each >20° but <160° apart, for each animal within 20 minutes (Arjo et al., 2007; Kozlowski et al., 2008). We then calculated coyote locations using program Locate III (Pacer Computing, Tatamagouche, Nova Scotia). For each week, we temporally distributed telemetry sampling by collecting two nocturnal locations and one diurnal location. To reduce auto-correlation and retain temporal independence between locations, we separated each weekly nocturnal and diurnal sample by >12 hours and a difference of >2 hours in the time of day of each location (Swihart and Slade, 1985; Gese et al., 1990). All home ranges were computed using only locations with an error polygon <0.10 km^2 (Seidler and Gese, 2012). We attempted to locate each coyote >3 times weekly in order to obtain 30 locations for each coyote for each biological season as the minimum number of locations needed to adequately describe the home range of a coyote (Gese et al., 1990).

We created seasonal home ranges for all coyotes with >30 locations (Gese et al., 1990; Aebischer et al., 1993) with defined biological seasons based on the behavior and energetic needs of canids for our study area: breeding 15 December – 14 April, pup-rearing 15 April – 14 August and dispersal 15 August – 14 December (Dempsey et al., 2014). We created 95% fixed kernel density estimates (KDE) following recommendations of Walter et al. (2011) by calculating Gaussian kernels with a plug-in bandwidth estimator (cell size = 30) using the Geospatial Modeling Environment (GME) platform (Beyer, 2012). We then created home range polygons using (GME) platform and loaded these polygons into ArcGIS 10.2. (Environmental Systems Research Institute Inc., Redlands, CA). We quantified home range areas using the field calculator tool and determined the number of water sites contained within each home range using theme-intersection routines.

Water Site Visitations

We examined the relationship between coyotes and seasonal visitations to water sites within each home range by establishing data loggers (model R4500S and model R2100/D5401, ATS, Isanti, MN), following recommendations of Breck et al. (2006), at 10 wildlife water developments (hereafter guzzlers) and 3 ponds (hereafter non-guzzlers). These 13 water sites represented 54% (13 of 24) of the potential water sites within the study area and 72% (13 of 18) of anthropogenic water sites. We defined a visit as all data logger recordings of an individual animal occurring within 30 min at a particular water site (Atwood et al., 2011). For each home range we determined both the total number of intersecting water sites and the number of intersecting sites equipped with data loggers. For coyote home ranges containing water sites with data loggers, we summarized the number of visitations, which provided a visitation frequency (# of visitations to water sites/seasonal home range) for further investigation. Because we suspected non-guzzler sites might experience higher visitations than guzzler sites, we also tracked the number of visitations within each home range that occurred at guzzlers versus non-guzzlers. We did not attempt to describe visitations when home ranges contained water sites without data loggers because we had no way of determining individual coyote use of water sites without data loggers, or if visits to sources with data loggers constituted a small or large portion of overall water use within a coyote's home range.

Water Manipulation

At the conclusion of the 2012 breeding season, we drained 5 guzzlers using a generator and submersible pump, and covered drinking portals with plywood (Fig. 4-2). Guzzler water levels were checked monthly and were re-drained if they reached >2/3 capacity. In addition, one pond was excluded by affixing a 1.2 m chain-link apron to an existing surrounding chain link fence. This manipulation effort eliminated water availability at 33% (6 of 18) of perennial anthropogenic water sites within the study area. The manipulation allowed us to incorporate a multiple-treatment site, multiple-control site BACI design (Morrison et al., 2001) where we assessed home range and visitations before and after eliminating water availability at water sites. Specifically, we assigned home ranges and visitations for the temporal spans prior to and after the water manipulation into two separate periods (pre-period and post-period) and two separate classes (reference or impact). The reference class referred to all home range areas and respective visitations containing water sites not spatially associated with the manipulation. The impact class referred to all home range areas and respective visitations containing water sites slated for manipulation at the onset of the post-period. For example, a home range area and visitation frequency assigned to the pre-period and impact class would be temporally associated with the time period prior to the water manipulation and spatially associated with the manipulation (i.e., the home range contained a water site or sites that were to be manipulated at the conclusion of the pre-period). BACI designs are considered superior to observational studies as they better account for variability of response and exploratory variables attributed to temporal (e.g., annual precipitation) and spatial factors (e.g., vegetation heterogeneity across study area) that cannot always be controlled and/or accounted for under natural environmental conditions (Underwood, 1994; Morrison et al., 2001).

Spatial Separation of Home Ranges

We estimated the impact of water manipulation on spatial separation of coyote home ranges by measuring spatial overlap of 95% fixed kernel home ranges (Atwood and Gese, 2010). We used the adehabitat package in R (R Core Team, 2014) to quantify overlap by computing the proportion of a home range for each coyote that was impacted by the water manipulation (i.e., impact class) during the season just prior to the post-period (i.e., breeding 2012) covered by the home range of the same animal for the first three seasons of the post-period (i.e., pup 2012, dispersal 2012, breeding 2013). This provided us with three home range overlap values for each individual coyote that had been assigned to the impact class. We compared these overlap values with an equal number of coyotes of the same sex ratio (2 males, 2 females) affiliated with the reference class.

Statistical Analyses

Prior to analyses, we examined all data for normality and homogeneity of variances and used transformations to better meet parametric assumptions, or non-parametric tests when assumptions could not be met (Zar, 2010). We report means, medians, standard deviations and standard errors in the original scale of measurement. We employed generalized linear mixed models (GLMMs) to test the categorical main effects of period (before and after) and class (reference and impact) on the continuous response variables of coyote visitations (visitations/individual/season) and seasonal coyote home range size (km^2). The GLMM approach enables the fitting of random terms and therefore accounts for repeated sampling across error terms. Seasonal home ranges and visitations were derived repeatedly from the same individuals in different seasons and years, thus we included individual as a random effect in all models (Stroup, 2012). Specifically, we tested for an effect of water site manipulation by including a period by class interaction in our model (Underwood, 1992). Within the framework of a BACI design, such an interaction tests for a differential change (i.e., non-parallelism) between impact and reference sampling units following some type of manipulation (Underwood, 1992). Inspection of visitation data revealed non-normality that was not remedied by data transformations. As a result we fitted the following model families: lognormal, Poisson, quasi-Poisson, and negative binomial. Models that did not converge were eliminated and we assessed remaining models based on the generalized chi-square fit statistic (Stroup, 2012). For the visitations and home range area data, the final model family used was negative binomial and lognormal, respectively. We separately tested for differences in visitations by water site type (i.e., guzzlers versus non-guzzlers) using a Wilcoxon signed-rank test (Zar, 2010). For this test, we only included visitation data from home ranges that contained both guzzler and non-guzzlers that were monitored by data loggers. For all statistical tests we interpret p-values in terms of relative evidence of differences (Ramsey and Schafer, 2002). All statistical analyses were conducted using R (R Development Program Team 2014).

Results

We developed 149 seasonal home ranges from 41 coyotes spanning from the 2010 pup rearing season to the 2013 dispersal season. We excluded 23% (35 of 149) of home ranges and respective visitations from further description because they partially occurred outside of our study area. Prior to the water manipulation (i.e., the pre-period), 88% (61 out of 69) of home ranges contained at least one water site. On average, 2 water sites (min = 1, max = 9) were contained within each home range prior to removal of water sites. Following reduction of water availability, 56% (25 of 45) of home ranges contained at least one water site. Of these, 83% (19 of 23) contained only water sites monitored by data loggers that remained following water availability removal. On average, one (min = 1, max = 4) water site was contained within each seasonal home following removal of water availability.

We excluded 31% (35 of 114) of remaining home ranges and associated visitations from additional description and analyses due to home ranges either containing water sites within our study area that were not monitored by data loggers or containing zero water sites; except for home ranges and respective visitations associated with post-period and impact class (see methods for full description). We also censored remaining home ranges derived from transient coyotes (n = 6) because they were only associated with the reference class of home ranges, and could have introduced bias into further analyses (Kamler and Gipson, 2000). This left 72 home ranges from 21 individual coyotes for further description and analysis.

Coyote seasonal visitations to water sites averaged 13.0 visitations/season (SD = 13.5) and ranged from zero to 47 visits. We found suggestive evidence that elimination of water availability influenced coyote visits to manipulated water sites (period x class interaction: t = 2.06, *P* = 0.05, df = 1, 49; Fig. 4-2). Frequency of visitation to water sites affiliated with the impact class decreased from 8.46 visits/season during the before period (SE = 2.52) to 4.22 visits/season (SE = 1.69) following water manipulation. Conversely, visitations to water sites affiliated with the reference class increased from 12.17 visits/season (SE = 1.90) during the pre-period to 19.79 visits/season (SE = 1.62) during the post-period. We found no evidence that period alone influenced coyote visitations (t = -2.47 *P* = 0.35, df = 1, 49). There was evidence that class type alone influenced visitation (t = 2.58, *P* = 0.03, df = 1, 49). Average visitation within home ranges associated with the reference and impact classes were 15.58 (SE = 0.81) and 7.10 (SE = 1.78) visitations/season, respectively. We found evidence that visitation differed by water site type when coyotes had access to both types (Wilcoxon rank sum test, Z = -3.58, *P* = < 0.01, n = 19). Median seasonal visitation for coyotes whose home ranges contained at least one guzzler and non-guzzler were 0 and 7 visits, respectively (Fig. 4-3). We found no evidence that elimination of water availability influenced home range size (period x class interaction: t = -0.96 *P* = 0.34, df = 1, 49). Similarly, there was no evidence to suggest that period (t = -0.95 *P* = 0.33, df = 1, 49), or class (t = -1.37 *P* = 0.17, df = 1, 49) had an influence on home range size (Fig. 4-4). We compared seasonal home range overlap for 2 males and 1 female coyote from each home range class (i.e., impact or reference). Percent seasonal overlap of fixed kernel home ranges for coyotes assigned to the reference and impact classes were 78% (SE = 11.5) and 85% (SE = 9.2), respectively.

Discussion

Our study was the first to quantify individual based visitations to water sties for coyotes and the first to incorporate a resource manipulation design to evaluate the effects of water sites on space use of a canid species in an arid environment. Overall, we found a portion of coyotes did not utilize water sites, coyote visitations to water sites were reduced following removal of water availability, reducing water availability did not influence coyote home range size, and reducing water availability did not influence spatial shifts of home ranges.

Our visitation results provided support that the availability of perennial free water does not appear to be a requisite resource component for coyotes in our study area. We observed zero visits during 16% (10 of 64) of seasonal observations, and <5 visits during 39% (25 of 64) of observations; these null to low frequency of visitation were observed across all three season types and years of the study. Unfortunately, no studies exist that allow for comparison of these individual based visitations with other coyote populations. Allen (2012) found that all radio-collared dingoes in the Strzelecki Desert visited water sites every season, and dingoes rarely went >5 days without visiting a water site. Clearly, more investigations are needed to determine frequency of water site visitations for canids at the individual level, especially for species and populations deemed to benefit from the presence of such sites.

Our manipulation revealed visitations to water sites decreased once water was no longer available, but visitations to these sites did not altogether cease (Fig. 4-2). Coyotes may have returned to impacted water sites following the manipulation in order to re-investigate the availability of water, to engage in scent-marking (Gese and Ruff, 1997), to forage at waters' edge, or a combination thereof. We observed that coyote visitations within home ranges containing water sites unassociated with the water manipulation (i.e., the reference class) experienced higher visitations (Fig. 4-2). This finding may have been caused by the disparity of water site types among design classes. Manipulated water sites consisted of only one non-guzzler, while water sites affiliated to the reference class contained two non-guzzlers that were associated with riparian vegetation, which was rare on our study site (Emrick and Hill, 1999). Such riparian vegetation may have provided coyotes with foraging and/or bedding opportunities at the waters' edge that were not available at guzzlers, which ostensibly resulted in non-guzzlers providing additional resources when compared to guzzlers. De Boer et al. (2010) observed that lion (*Panthera leo*) prey availability was higher at ponds and rivers when compared to random locations, but investigations that test whether water sites with riparian vegetation provide additional resources for coyotes or other desert canids are lacking. We did not visually monitor coyote behavior at water sites during this study, nor did we assess coyote prey availability at water sites or non-water sites. Thus, we cannot make any firm conclusions regarding the mechanisms driving this finding.

Our assessment of home ranges in relation to water sites provided additional support that access to water sites was not an obligatory resource component for coyotes.

For example, 25% of home ranges occurring exclusively within our study area did not contain a single water site. Further, 33% (36 of 108) of the home ranges we investigated either did not contain a water site or contained a water site that was not visited for an entire season (e.g., approx. 120 day period). In addition, our water site manipulation did not appear to influence home range sizes (Fig. 4-4) or the overlap of home ranges.

We were only able to track seasonal overlapping of home ranges for three coyotes that belonged to the impact class for the three consecutive seasons following the manipulation. Such a small sample size and resulting lack of statistical inference warrants caution. However, all three of these animals maintained a spatial affinity to their home ranges following the manipulation (i.e., they did not die, disperse, or abandon their home ranges after water was no longer available), and none of the animals adjusted their movements in a manner where post-manipulation home ranges included a water site. Anthropogenic modifications to landscapes can influence coyote home range selection, which is considered a second-order selection process (Johnson, 1980). Boisjoly et al. (2010) determined that clear-cutting activities in boreal forests increased coyote habitat quality by increasing food accessibility, and posited that such anthropogenic activity may have allowed coyotes to establish home ranges. Hidalgo-Mihart et al. (2004) found coyotes utilizing landfills had home ranges less than half the size of coyotes that occurred in vegetation zones predominantly unaltered by anthropogenic modification. Conversely, Atwood et al. (2004) found coyote home range sizes were largest in areas with the highest levels of anthropogenic modification to the landscape. Clearly, the influence of anthropogenic factors on coyote selection processes is highly variable and contingent upon myriad factors that can differ across time and space.

It has been revealed or postulated that anthropogenic water sites can directly alter the distributions and densities of ungulates, birds, and mammalian carnivores (de Leeuw et al., 2001; Kristian and Boarman, 2003; Cain et al., 2012; Allen, 2012), but empirical evidence of water sites engendering indirect effects are sparse. Harrington et al. (1999) documented a population crash of roan antelope (*Hippotragus equinus*) following an increase of water developments in the northern portion of Kruger National Park. They speculated that these water sites served as a subsidized resource facilitating a population increase of more water dependent species [i.e., zebra (*Equus quagga*) and wildebeest (*Connochetes taurinus*)], which engendered increased lion numbers and lion predation on roan antelope. These claims were substantiated when the removal of water developments coincided with a roan antelope population recovery (Harrington et al., 1999). An indirect effect of water sites has been proposed as a factor contributing to reduced kit fox abundance and distribution in the Great Basin Desert; additions of water sites in the mid to late twentieth century coincided with increased abundance of coyotes (Arjo et al., 2007; Kozlowski et al., 2008). Depressing coyote populations has been shown to positively alter the abundance of gray foxes (*Urocyon cinereoargenteus*) (Henke and Bryant, 1999) and swift foxes (Kamler et al., 2003). Similarly, Kamler et al. (2013) found that areas void of black-backed jackals (*Canis mesomelas*) experienced higher densities of Cape foxes (*Vulpes chama*) when compared to areas occupied by jackals.

Thus, the proposition that water sites have impacted the carnivore community in our study area is rational.

Our results suggest that under the environmental conditions present during our study, water sites did not appear to constitute a requisite resource for adult resident coyotes, or a resource that influenced home range size. We found no spatial shifts in home ranges, no increase in home range size, as well as no abandonment of their home range or reduced survival (i.e., no coyotes died) following the cessation of water availability in their home range. We speculate that the observed increase of coyotes (Arjo et al., 2007) may be more attributable to changes in coyote management practices, or habitat change (i.e., invasion of cheatgrass). Within a study area that encompassed our own, Egoscue (1956) argued that coyote abundance was suppressed by way of intensive coyote control efforts, including regular usage of baited toxicants spaced at intervals aimed to maximize lethality to coyotes rather than carnivores with smaller home ranges (i.e., kit foxes). The use of baited toxicants was a common predator control tactic in Utah, including DPG, during the mid-twentieth century (Shippee and Jollie, 1953). In 1972, the enacting of Executive Order 11643 banned the use of baited toxicants and additional restrictions have been placed on the use of toxicants for predator control by the Environmental Protection Agency (Mitchell et al., 2004). Dorrance and Roy (1976) and Nunley (1986) suggested that coyote control programs that relied heavily on toxicants were more effective at suppressing coyote populations than contemporary methods. Therefore, the observed increase of coyote numbers observed by Arjo et al. (2007) over the latter half of the twentieth century may be in part tied to changes in coyote management practices that temporally coincided, but were largely unrelated, to the additions of anthropogenic water sites.

Our study was the first to incorporate a manipulation design to test the effects of water sites on canids. Our findings provide evidence that water sites, especially guzzlers, do not represent a pivotal resource for coyotes in our study area, during the temporal span the investigation was conducted. That being said, we recommend some caution be exercised in relation to our findings. Our investigation focused on one study area, spanned only a 4 year period, and focused primarily on second and fourth-order selection processes (Johnson, 1980). We recommend future investigations on the impact of free water on coyotes, and other species of interest, should consider designs with replication at the study site level (Cain et al., 2008), a longer temporal span (i.e., several generation times of the species of interest), an examination/comparison of population state variables and/or vital rates, and selection processes at other orders (i.e., first and third-order).

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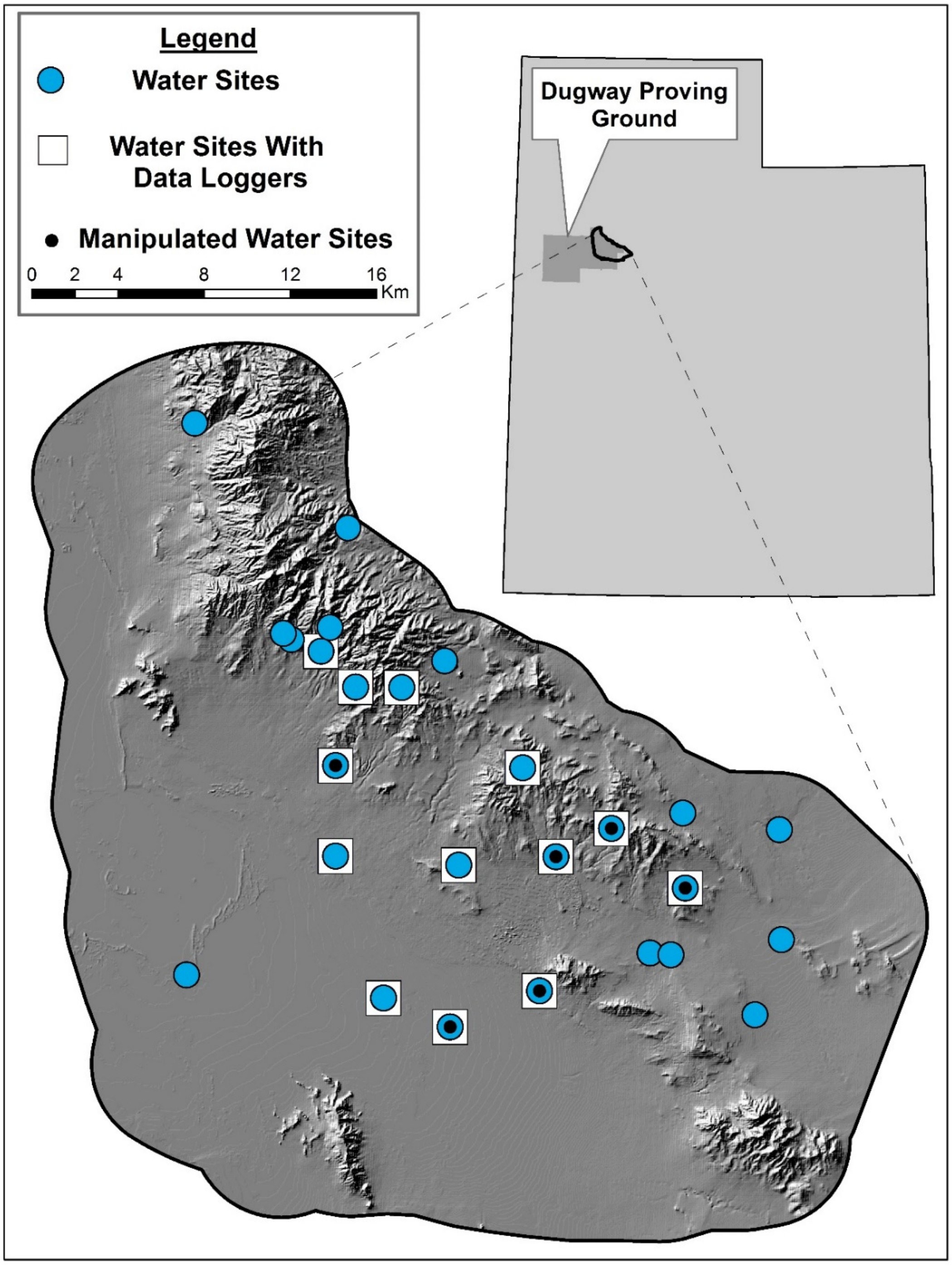

Fig. 4-1. Study area (1127 km^2) and permanent/ephemeral free water sites (n = 26) within and adjacent to the U.S. Army Dugway Proving Ground, USA, 2010–2013.

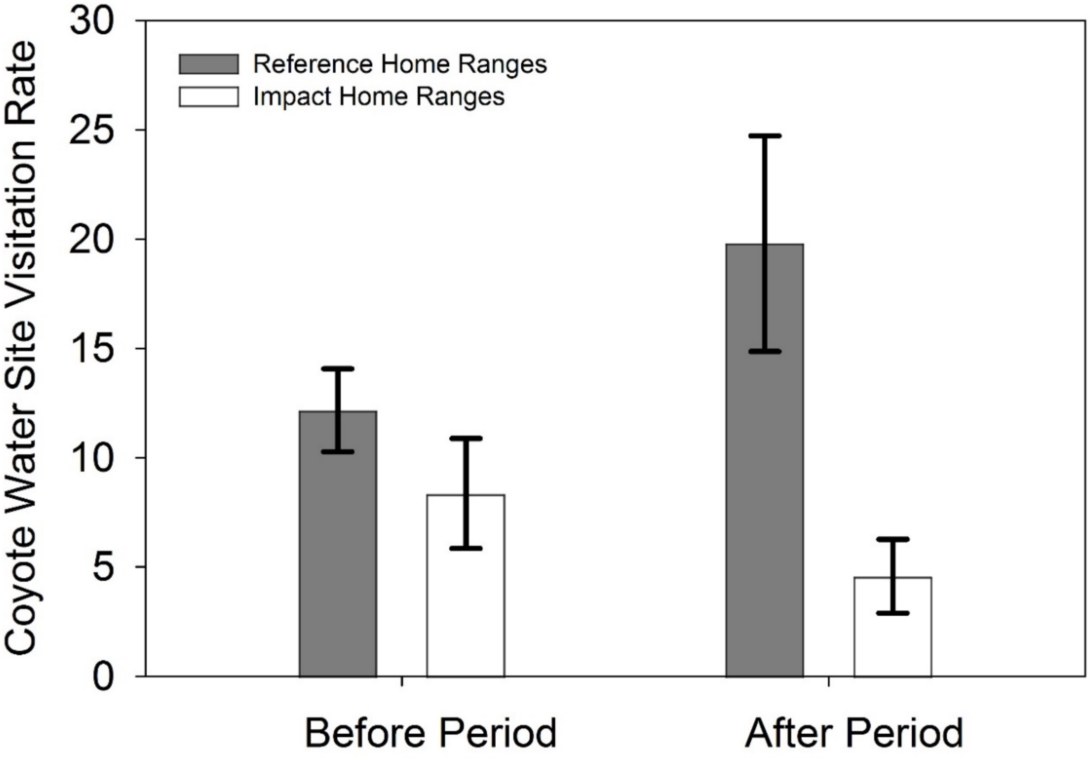

Fig. 4-2. Mean coyote seasonal water site visitations (± SE) observed within seasonal home ranges prior to (Pre-Period) and following (Post-Period) removal of water availability at a subset of water sites on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013 (n = 69).

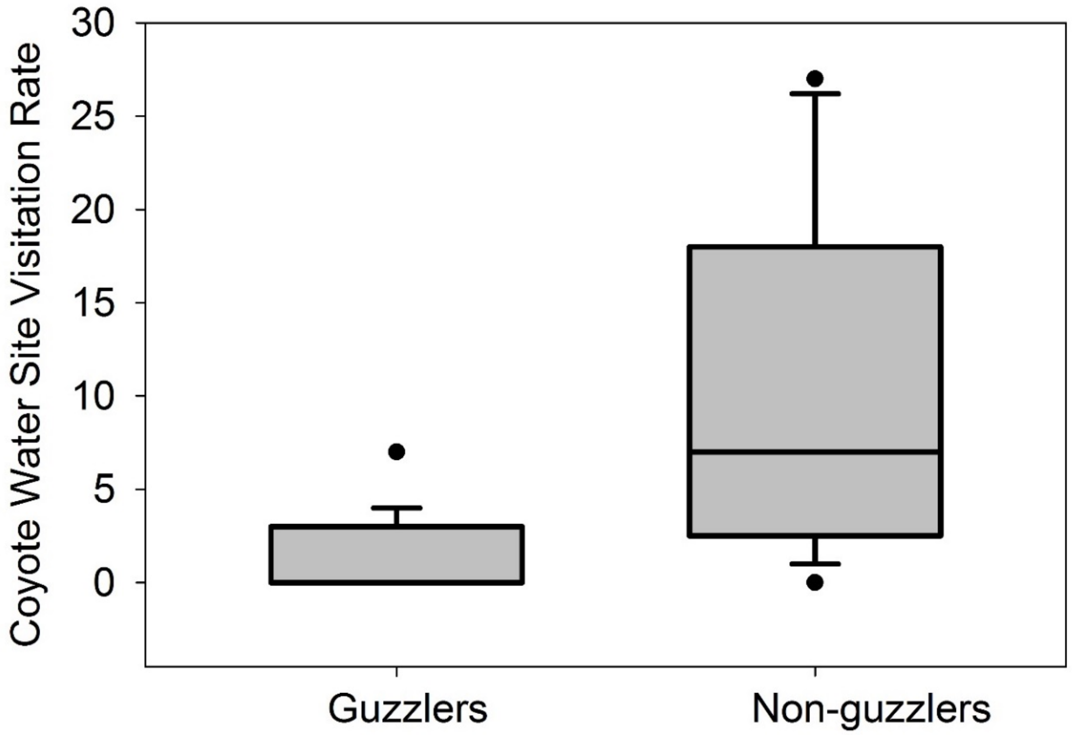

Fig. 4-3. Box plot of coyote seasonal water site visitations observed within home ranges that contained guzzler and non-guzzler water sites with water availability and monitored by data loggers (n = 21) on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.

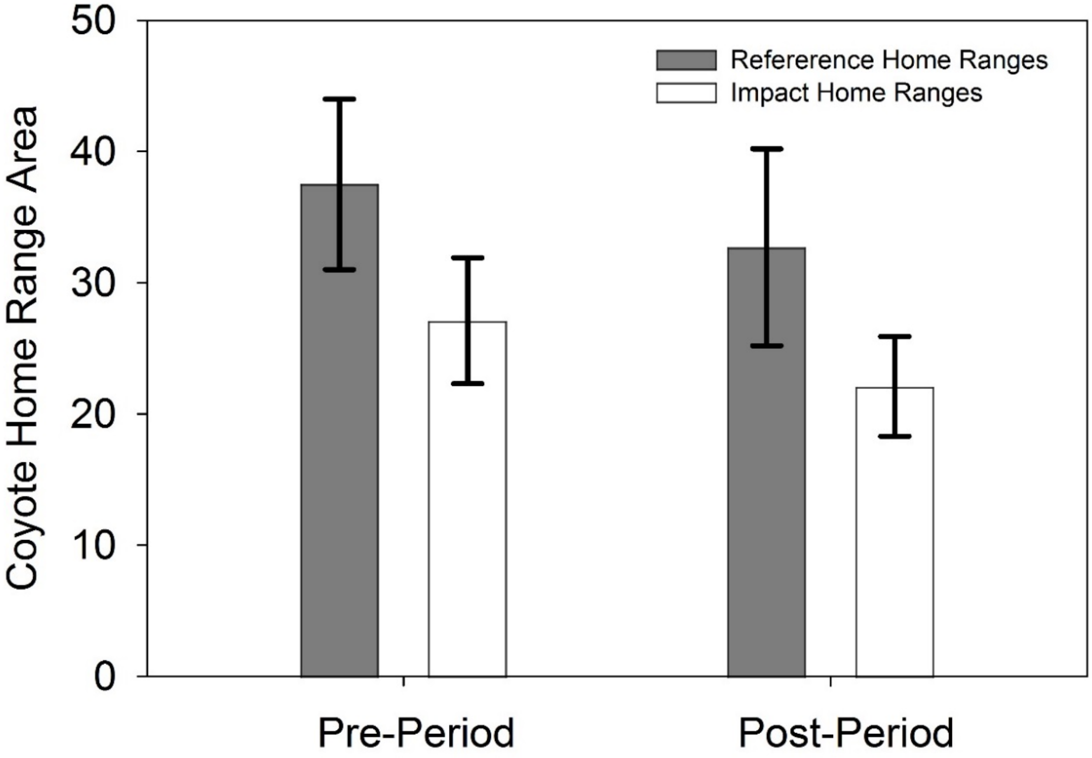

Fig. 4-4. Mean coyote seasonal home range area (km^2) (± SE) observed prior to (Pre-Period) and following (Post-Period) removal of water availability at a subset of water sites on a study area encompassing U.S. Army Dugway Proving Ground and adjacent BLM land, Utah, USA, 2010-2013 (n = 69). Reference home ranges refers to coyote home ranges that contained water sites with water availability during both periods. Impact home ranges refers to coyote home ranges that contained water sites until water availability was removed at the onset of the Post-Period.

CHAPTER 5

INFLUENCE OF WATER DEVELOPMENTS ON DEMOGRAPHY AND SPATIAL ECOLOGY OF A SMALL DESERT CARNIVORE: THE KIT FOX[4]

ABSTRACT

Understanding the impacts of human activity on wildlife species and communities has become a central tenet of wildlife management and conservation. The addition of water developments to support urbanization, ranching, and wildlife is pervasive across many arid portions of the western United States. Despite their prevalence, the impact of water developments on wildlife remains unclear for the majority of species living in arid areas. It has been hypothesized that the addition of water developments to the Great Basin Desert may have contributed to a decrease in kit fox abundance, though this viewpoint remains largely untested. From 2010 to 2013, we examined survival rates, relative abundance, and habitat characteristics of kit foxes on the U.S. Army Dugway Proving Ground (DPG), Utah. Using both a before-after and before-after-control-impact design, we collected 2 years of pre-manipulation data, then changed water availability on half the study area, and continued data collection for an additional 2 years post-manipulation. We found no evidence that removal of water availability at water developments influenced kit fox survival or relative abundance. In addition, we found that areas associated with the majority of water developments differed from kit fox

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[4] Co-authors are Eric M. Gese; chapter is formatted for Journal of Wildlife Management

territories in terms of elevation, soil type, and dominant cover type (e.g., percent shrub, herbaceous, barren). Our study was only the second in North America to include a manipulation component in order to elucidate the effects of water developments on a desert adapted species. We suspect the observed changes in the Great Basin kit fox population and canid community may be attributed to a combination of factors, including changes in coyote management practices, fire suppression, invasion of exotic herbaceous vegetation, and subsequent impacts to prey resources and other predators, that temporally coincided with, but were largely unrelated to significant increases of free water availability.

Understanding the impacts of the human ‘footprint’ on wildlife species and communities has become a central tenet of ecology and related sciences (Leu et al. 2008). In addition, the pervasiveness of man-made infrastructure on the majority of terrestrial landscapes has helped spur several subfields of scientific investigation, such as road ecology (Forman et al. 2002, van der Ree et al. 2015) and urban wildlife management (Adams and Lindsey 2009, McCleery et al. 2014). Determining the impact of anthropogenic activities on terrestrial wildlife can be arduous, as these activities can have variable impacts on a myriad of species, and the impacts of such activities are not always all-encompassing for the same species across space and/or time (Larsen et al. 2010, Morrison and Mathewson 2015). Furthermore, impacts of anthropogenic activities on wildlife can be indirect, and as a result more difficult to detect and quantify (Krausman and Cain 2013, Morrison and Mathewson 2015). Though extensive research has investigated the impacts of select anthropogenic activities on wildlife habitat relationships as well as population state variables and vital rates [e.g., energy development (Wilson et al. 2013, Brittingham et al. 2014, Jones et al. 2015, Ramirez and Mosley 2015)], many others remain largely unexplored.

Adding anthropogenic water sites (hereafter water developments) to arid environments of the western United States is commonplace. These water developments can have several aims, including promoting urban development (i.e., sewage and catchment ponds) (Kristan and Boarman 2007), improving grazing habitat for livestock (Brooks et al. 2006, Holecheck et al. 2010, LaBaume 2013), and benefiting target wildlife species (Larsen et al. 2012, Krausman and Cain 2013). Adding water developments to arid environments was historically deemed ubiquitously beneficial to wildlife in areas where water was lacking, as such developments increased the availability of a critical limiting resource (Leopold 1933). The practice of adding water developments as a wildlife management tool and mitigation strategy has been adopted by public land management agencies (Simpson et al. 2011, Larsen et al. 2012), natural resource extraction companies (Haynes and Klopatek 1979), and branches of the United States military (Broyles and Cutler 1999, Arjo et al. 2007, Hall et al. 2013). However, a growing body of literature, both conceptual and empirical in nature, has suggested the impacts of water developments on certain species can be adverse (Broyles 1995, DeStefano et al. 2000, Arjo et al. 2007) and/or not in accordance with management objectives (Krausman and Etchberger 1995, Broyles and Cutler 1999, Cain et al. 2008). In addition, it has been reported that our overall understanding of the impacts of water developments on wildlife is seriously lacking (Simpson et al. 2011, Larsen et al. 2012, Krausman and Cain 2013).

Impacts of water developments on wildlife can be categorized as being direct or indirect (Larsen et al. 2012). Direct effects of water developments are those associated with the intake of free water [e.g., an increase in chukar (*Alectoris chukar*) survival due to chukar water intake]. In contrast, indirect effects include, but are not limited to, exploitative or interference competition with other species or conspecifics, or altered vulnerability to predation [e.g., a decrease in black-tailed jackrabbit (*Lepus californicus*) survival due to an increase in coyote (*Canis latrans*) survival, engendered by coyote water intake at water developments]. Investigations on direct effects of water developments on wildlife are sparse. Larsen et al. (2010) found chukars utilized water sites in several mountain ranges but space use and diet were influenced by water development use only in certain ranges. In a seminal study including a before-after/control-impact (BACI) design, Cain et al. (2008) found bighorn sheep (*Ovis canadensis*) used water developments, but reduction of water availability at developments did not influence population parameters or space use. Hall et al. (2013) found that the relative abundance of coyotes (*Canis latrans*) was similar in areas with and without free water, despite coyote use of water sites, suggesting no direct effect of water consumption on the abundance of this species.

Investigations of the indirect effects of water developments on wildlife are also lacking. Cutler and Morrison (1998) found measures of species richness and relative abundance for species not known to consume free water, including rodents and reptiles, did not differ in relation to areas adjacent to dry or wet water developments, suggesting predation rates at sites did not differ, or that increased predation was compensatory. Conversely, DeStefano et al. (2000) observed a negative relationship between leporid [(black-tailed jackrabbit and desert cottontail (*Sylvilagus audubonii*)] and coyote relative abundance in relation to proximity to water developments in the Sonoran Desert. Similarly, Kristan and Boarman (2007) suggested water developments contributed to population increases of common ravens (*Corvus corax*) in the Mohave Desert, which may have led to increased mortality rates for several species of conservation concern.

A specific "indirect effect of water" has been hypothesized for the canid community in portions of the Great Basin Desert (Larsen et al. 2012). Specifically, it has been argued that water developments constructed during the 1970s-1990s on and near the US Army Dugway Proving Ground (DPG) removed the arid system limitations of coyotes, which compete with kit foxes (*Vulpes macrotis*) for habitat, space, and food (Arjo et al. 2007, Nelson et al. 2007, Kozlowski et al. 2008). The justification for this theory is largely premised on the differential physiological demand for free water by coyotes and kit foxes. Golighlty and Omhart (1984) reported that the amount of wet prey biomass required per day to meet energy and water requirements (i.e., preformed water) is 504 and 1780 g for coyotes, and 101 and 175 g for kit foxes, respectively. Put in simpler terms, in the absence of available free water, coyotes must consume over 3x the wet prey biomass to meet water versus energy requirements, while kit foxes need consume less than 2x the amount. This disparity in kit foxes' and coyotes' abilities to utilize preformed water, coupled with behavioral differences for the two species

(Golightly and Omhart 1983), and the addition of anthropogenic water developments has theoretically led to an expansion of coyote habitat on DPG, which may have in turn contributed to an increase in the coyote population and reduction of kit foxes by increased interference competition and intraguild predation (TRIES 1997, AGEISS 2001, Arjo et al. 2007, Kozlowski et al. 2012).

However, recent investigations do not lend support to water developments being a contributing factor to changes in the canid community at DPG. Hall et al. (2013) observed no difference in coyote or kit fox activity in areas with and without free water at DPG, but observed that kit foxes regularly visited water sources in the Mojave Desert. The latter finding suggests that kit foxes will utilize water sources if they are located in areas containing environmental characteristics that constitute kit fox habitat, but to date, no studies have been able to chronicle the frequency of water development visitations by individual kit foxes. In addition, preliminary findings from a recent investigation suggest that only a portion of coyotes on DPG utilize water sources, and that removal of water availability at developments does not result in territory abandonment or large shifts in territories (Kluever and Gese, in review).

Clearly, further investigation is needed to parse the influence of wildlife water developments on kit foxes. Elucidating this relationship will increase our understanding of the effects of free water on wildlife in arid environments and has the potential to influence kit fox conservation strategies. For this investigation we specifically aimed to determine: 1) the frequency of water development visitations by individual kit foxes, 2) whether removal of water availability at water developments influences kit fox survival,

3) if the removal of water availability at water developments facilitates a change in kit fox relative abundance, and 4) if environmental variables at areas associated with water developments differ from those in kit fox home ranges.

STUDY AREA

We conducted our research on the eastern portion of the U.S Army Dugway Proving Ground (DPG) and the adjoining lands managed by Bureau of Land Management (BLM), located approximately 128 km southwest of Salt Lake City, in Tooele County, Utah, USA. Elevations ranged from 1302 m to 2137 m. The study site was located in Great Basin Desert, where winters were cold and summers were hot and dry, with the majority of precipitation occurring in the spring. Annual weather consisted of mean air temperatures of 12.7°C (range: -20.0 to 40.6°C) and mean precipitation of 150 mm (MesoWest, Bureau of Land Management and Boise Interagency Fire Center). In the study area, we identified 19 permanent free water sites consisting of 10 wildlife waterers, 4 natural springs, and 5 man-made ponds/catchments. Four additional ponds were run-off based and ephemeral. In addition, the eastern portion of the study area managed by the BLM contained 3 livestock tanks that were at times operational during winter and spring cattle grazing (November 1 to April 1). Springs and man-made ponds were often associated with riparian communities primarily comprised of tamarisk (*Tamarix ramosissima*) (Emrick and Hill 1999). Water developments (i.e., guzzlers, ponds, and livestock tanks) were developed between the 1960s and 1990s (Arjo et al. 2007). Guzzlers were primarily placed among or at the base of mountainous areas in order to buttress populations of chukar and mule deer (*Odocoileus hemionus*), and ponds were primarily located in flat land areas to support urban development (Hall et al. 2013). Thus, the ratio of water developments to natural water sites within the study area was at least 4:1, with slight seasonal variability occurring due to the turning on/off of livestock tanks and ephemeral catchment ponds. There was no free-flowing water present on the study area. Additional water sites (e.g., hardpans, rainfall, drainages) were ephemeral pools lasting <1 week; thus we assumed they were homogenous throughout the study area.

The study area consisted of predominately flat playas punctuated with steep mountain ranges. The lowest areas consisted of salt playa flats sparsely vegetated with pickleweed (*Allenrolfea occidentalis*). Slightly higher elevation areas were less salty and supported a cold desert chenopod shrub community consisting predominately of shadscale (*Atriplex confertifolia*) and gray molly (*Kochia america*). At similar elevations, greasewood (*Sarcobatus vermiculatus*) communities were found with mound saltbrush (*Atriplex gardneri*) and Torrey seepweed (*Suaeda torreyana*). Higher elevations consisted of vegetated sand dunes including fourwing saltbush (*Atriplex canescens*), greasewood, rabbitbrushes (*Chrysothamnus* spp.), shadscale, and horsebrush (*Tetradymia glabrata*). Near the bases of the higher steep mountains were shrub steppe communities of sagebrush (*Artemisia* spp.), rabbitbrush, Nevada ephedra (*Ephedra nevadensis*), greasewood, and shadscale. The highest elevation was a limited Utah juniper (*Juniperus osteosperma*) community including black sagebrush (*Artemisia nova*) and bluebunch wheatgrass (*Elymus spicatus*). Previous kit fox investigations had taken place within our study area during 1955-1958 (Egoscue 1956, 1962), 1966-1969 (Egoscue 1975), 1996

(TRIES 1997), and 1999-2001 (Arjo et al. 2003, 2007, Kozlowski et al. 2008, 2012). DPG had undergone a dramatic decrease in black-tailed jackrabbit abundance when compared to the mid-twentieth century (Arjo et al. 2007), which was likely caused by encroachment of exotic herbaceous vegetation; 40% of historical juniper woodland and shrub communities had been replaced by exotic herbaceous vegetation (Emrick and Hill 1999). Home range sizes for kit foxes at DPG were consistently the largest recorded for the species throughout its range (Arjo et al. 2003, Dempsey et al. 2014) and rodents, especially kangaroo rats, were the primary kit fox prey item (Kozlowski et al. 2008). In addition coyotes occurred throughout DPG, kit fox distribution was limited (Kozlowski et al. 2012, Dempsey et al. 2014), and habitat use by DPG kit foxes represented spatial and behavioral strategies designed to minimize spatial overlap with coyotes while maximizing access to resources (Kozlowski et al. 2012)

METHODS

From December 2009 to March 2012, we captured, radio-collared, and monitored kit foxes for 2 years as the “baseline” monitoring period. At the conclusion of the 2012 breeding season (April), we initiated the “manipulation” period when we drained 5 guzzlers using a generator and submersible pump and excluded one pond by affixing a 1.2 m chain-link apron to an existing surrounding chain link fence. This manipulation effort eliminated water availability at 31% (6 of 19) of the permanent water developments within the study area. An investigation on kit fox space use and abundance by Dempsey et al. (2014) revealed that, despite intensive trapping and survey efforts, kit foxes were not detected in the northern and northeastern areas of DPG where the majority of water sources were located; kit foxes were only captured and found to utilize the western and southern portions of DPG. As a result, we chose to eliminate water availability at a subset of water sources that experienced a high frequency of coyote visitations, were logistically feasible to manipulate, and included water sources that appeared to be located on the periphery of kit fox habitat as determined by Dempsey et al. (2014). This design allowed us to test whether our overall water manipulation influenced kit foxes at the study site level using a before-after design. In addition, we tested whether individual manipulated water sources influenced kit foxes using a Before-After Control-Impact (BACI) design (Morrison et al. 2001) by monitoring areas associated with and unassociated with manipulated water developments prior to and following the water manipulation.

Animal Capture and Handling

Beginning in December 2009, we captured kit foxes via road based transect trapping Schauster et al. 2002, Dempsey et al. 2014) and at known den sites (Kozlowski et al. 2008, Dempsey et al. 2014) using box traps (25 x 25 x 80 cm; Model 107; Tomahawk Live Trap LLC, Hazelhurst, Wisconsin) baited with hot dogs. Trapping transects were distributed to provide maximum coverage of the area and allow for increased likelihood of capturing most of the kit foxes occupying the study area (Dempsey et al. 2014). We deployed traps in the evening and checked them early morning each day. We coaxed captured foxes into a canvas bag placed at the edge of the trap, then restrained them by hand while wearing thick leather gloves (Kozlowski et al. 2008). We weighed, sexed, ear tagged, and fitted each fox with a 30–50 g radio-collar

(Model M1930; Advanced Telemetry Systems, Isanti, Minnesota). Collars included a mortality sensor that activated after 4 hours of non-motion and weighed <5% of body mass. We handled all foxes without the use of immobilizing drugs and released them at the capture site. Upon detecting a mortality signal we immediately recovered the transmitter and remains of the kit fox. The possible cause of mortality was determined by examining the carcass for external and internal injuries, puncture wounds, and hemorrhaging. Physical evidence at the site of mortality, such as tracks, scat, or hair, also assisted us in determining the possible cause of death. If we did not observe any gross trauma we sent animals to the Utah State (Logan, Utah) or Wyoming State (Laramie, Wyoming) Veterinary Diagnostic Laboratory for diagnosis.

Radio-telemetry and Home Range Determination

We located animals >3 times per week using a portable receiver (Model R1000; Communications Specialists, Inc., Orange, California) and a handheld 3-element Yagi antenna. We triangulated an animal's location using ≥3 compass bearings each >20° but <160° apart, for each animal within 20 minutes (Arjo et al. 2007, Kozlowski et al. 2008). We then calculated coyote locations using program Locate III (Pacer Computing, Tatamagouche, Nova Scotia). For each week, we temporally distributed telemetry sampling by collecting two nocturnal locations and one den (resting) location. To reduce auto-correlation and retain temporal independence between locations, we separated each weekly nocturnal and diurnal sample by >12 hours and a difference of >2 hours in the time of day of each location (Swihart and Slade 1985, Gese et al. 1990). All home ranges were computed using only locations with an error polygon <0.10 km^2 (Seidler and Gese

2012). We attempted to locate each kit fox >3 times weekly in order to obtain 30 locations for each kit fox for each biological season as the minimum number of locations needed to adequately describe the home range (Gese et al. 1990).

We created seasonal home ranges for all kit foxes with >30 locations (Dempsey et al. 2014) with defined biological seasons based on the behavior and energetic needs of canids for our study area: breeding 15 December – 14 April, pup-rearing 15 April – 14 August and dispersal 15 August – 14 December (Dempsey et al. 2014). We created 95% fixed kernel density estimates (KDE) following recommendations of Walter et al. (2011) by calculating Gaussian kernels with a plug-in bandwidth estimator (cell size = 30) using the Geospatial Modeling Environment (GME) platform (Beyer 2012). We then created home range polygons using GME and loaded these polygons into ArcGIS 10.2. (Environmental Systems Research Institute Inc., Redlands, CA). We quantified home range areas (km^2) using the field calculator tool and determined the number of water developments contained within each home range using theme-intersection routines.

Prey Base

We sampled rodents in 16 trapping grids using a 7 x 7 configuration (49 traps [H.B. Sherman Traps, Inc., Tallahassee, FL, USA], with 8.3-m spacing) for four consecutive nights (i.e., four capture occasions). Each four night sampling period was considered an individual trapping session. We conducted trapping sessions on each grid in early (May 01 to June 30) and late (August 1 to September 30) summer. Traps were baited with a mixture of black sunflower and mixed bird seed. All captured rodents were identified to species, tagged in each ear (Model # 1005-1, National Band and Tag

Company, Newport, KY, USA), and recaptures of individuals were recorded. Capture and handling protocols were reviewed and approved by the Institutional Animal Care and Use Committees (IACUC) at the United States Department of Agriculture's National Wildlife Research Center (QA-1734) and Utah State University (#1438). Permits to capture and handle rodents were obtained from the Utah Division of Wildlife Resources (COR #4COLL8322). All capture and handling procedures were in accordance with guidelines endorsed by the American Society of Mammalogists (Sikes et al. 2011).

Water Development Visitations

We examined frequency of kit fox seasonal visitations to water developments within each home range by establishing data loggers (model R4500S and model R2100/D5401, ATS, Isanti, MN), following recommendations of Breck et al. (2006), at 13 water developments (10 wildlife water developments and three ponds). These 13 sites represented 68% (13 of 19) of the permanent water developments within the study area. We defined a visit as all data logger recordings of an individual animal occurring within 30 min at a particular water development (Atwood et al. 2011). For kit fox home ranges containing water developments with data loggers, we summarized the number of visitations, which provided a visitation frequency (# of visitations to water developments/seasonal home range) for further investigation. We did not attempt to describe visitations when home ranges contained water developments without data loggers because we had no way of determining individual kit fox use of water sites without data loggers, or if visits to sources with data loggers constituted a small or large portion of overall water use within a kit fox home range.

Relative Abundance Surveys

We established four 5-km road-based survey transects whose midpoints were adjacent to wildlife water development sites [hereafter wildlife guzzlers (model Dual Big Game, Boss Tanks, Elko, NV)]. These transects (hereafter proximate transects) served as our treatment transects because they were associated with a water development. The average distance from proximate transects to the next nearest perennial water source (i.e., pond, water development, sewage lagoon) was 4.10 km (SD = 0.54). We used ArcGIS (version 9.3, Environmental Systems Research Institute, Redlands, CA) to create four additional 5-km transects (hereafter distant transects) which were distributed randomly along available non-paved roads with the constraints of occurring on lengths of road with no angles >60 degrees, a minimum spacing distance of 2.6 km from proximate transects, and a minimum spacing of 2.6 km from a perennial water source. We did not establish survey transects associated with two manipulated water sources (i.e., one guzzler and one pond) due to lack of road coverage. Surveys taking place on proximate and distant transects prior to the water manipulation period were considered the pre-period, while surveys following the water manipulation were considered the post-period.

For survey transects, we employed a multiple-treatment site, multiple-control site BACI design where we monitored all transects prior to and after eliminating water availability at water developments. BACI designs are considered superior to observational studies because they better account for variability of response and explanatory variables attributed to temporal (e.g., annual precipitation) and spatial factors (e.g., vegetation heterogeneity across study area) that cannot always be controlled and/or accounted for under natural environmental conditions (Morrison et al. 2001). As described in Study Design, in April 2012, we drained the four wildlife guzzlers associated with proximate transects.

We conducted scat deposition surveys (Knowlton 1984, Schauster et al. 2002, Dempsey et al. 2014) along eight 5-km transects to estimate the relative abundance of kit foxes. As a passive technique, scat deposition surveys do not require the target species to behave unnaturally (e.g., investigate a scent tab). Surveys were conducted by walking the transect to clear any scat from the road surface, then returning 14 days later to walk and count the number of scats deposited (Schauster et al. 2002). Following recommendations from Schauster et al. (2002) and Knowlton (1984), each transect was walked in both directions to reduce missed detections of scats. Hence, each survey consisted of eight scat deposition counts (i.e., one scat deposition count per transect). We identified kit fox scats based on guidelines described in Murie and Elbroch (2005). Scat deposition counts provided an index of kit fox abundance; the number of kit fox scats per transect per survey. Scat surveys have been reported as an effective index for tracking kit fox abundance over time and space (Dempsey et al. 2014) and have outperformed other noninvasive surveys for mammalian carnivores (Knowlton 1984, Harrison et al. 2002, Long et al. 2007, Dempsey et al. 2014).

We also conducted scent station surveys as a second estimate of relative abundance of kit foxes. We placed scent stations at 0.5 km intervals on alternating sides along each 5 km transect (Warrick and Harris 2001, Schauster et al. 2002), resulting in eleven stations/transect/day for four consecutive days. A scent station consisted of a cleared 1-m circle of lightly sifted sand (Linhart and Knowlton 1975) with a Scented Predator Survey Disk (SPSD; United States Department of Agriculture's Pocatello Supply Depot, Pocatello, Idaho) with Fatty Acid Scent (FAS) placed in the center. The SPSD with FAS was recommended for ‘‘ease of use, attractiveness to kit fox, and their low cost’’ (Thacker et al. 1995). FAS saturated SPSD's are preferred over the use of liquid lures because they allow for control of a consistent attractiveness between batches (Roughton and Sweeny 1982). We checked stations each morning for tracks of kit foxes, coyotes, bobcats (*Lynx rufus*), leporids, small mammals, and other potential prey species. We then resifted each station and replaced the SPSD. To help maintain consistent attractiveness, we removed SPSD's from use once they were noticeably deteriorated, broken, or after a full season of use. We resampled inoperable station nights (due to inclement weather) for an additional 1–2 days when necessary to complete the 4 full days of surveying. This survey provided a count of scent station visits (i.e., total number of visits, with a maximum possible number of visits of 44) as a measure of relative abundance. We elected not to convert count data to proportions due to excessive zeros in the data set (Zar 2010).

Home Range and Water Zone Characteristics

We delineated circular buffers equal in area to the average kit fox home range around each data logger monitored water development (Fig. 5-1). This allowed us to compare several environmental characteristics of kit fox home ranges with areas associated with water developments at a spatial extent germane to our focal study species, as recommended by Larsen et al. (2012). We only assessed environmental characteristics of water zones that were monitored with data loggers; these 13 sites represented 69% of the permanent free water sites within our study area. We compared areas around water developments to kit fox home ranges for possible differences in environmental variables previously reported as important habitat components for kit fox: elevation (McGrew 1976, Fitzgerald 1996), dominant vegetation type (Kozlowski et al. 2008), and soil type (Egoscue 1962, Fitzgerald 1996, Robinson et al. 2014). Elevation and soil type data were available from GIS databases (Utah Automated Geographic Reference Center; http://gis.utah.gov). Soils in the study area were classified into 4 major classes: silt, fine sand, blocky loam, and gravel. We eliminated the gravel soil type from future analyses because it constituted <5% of the area associated with kit fox home ranges and water zones. Dominant vegetation class data were obtained from the Landfire database (http://landfire.cr.usgs.gov/) and were classified into 3 major classes: herbaceous, shrub, or barren. These three classes comprised 94% of the total area encompassed within kit fox home ranges and water zones. We used the GME platform (Beyer 2012) to obtain mean elevation for each home range and water zone and the proportion of area encompassing each home range and water zone for the soil type and vegetation classes.

Data Analyses

We used the R (R Development Core Team 2007) package RMark (Laake and Rexstad 2007) to call program MARK (White and Burnham 1999), and construct closed population capture-mark-recapture (CMR) models to estimate total rodent abundance for each trapping session (i.e., 2 sessions per year). Pooled annual rodent abundance estimates exhibited a non-linear trend, with consistent abundance over the first two years of the study, an increase the following year, and a precipitous drop the final year (Fig. 5-2). We did not sample rodents during the breeding season for any year, and thus assumed that small mammal abundance estimates pooled and averaged from the dispersal and pup rearing for each year were representative of small mammal abundance during the respective breeding season. This allowed us to create a time-varying covariate for kit fox analyses, referred to hereafter as “prey base”, that represented a qualitative trend of small mammal abundance over the course of the study (e.g., Prey base covariate = Years 1 and 2: moderate, Year 3: high, Year 4: low).

We estimated survival rates using the known fate model option in RMark. We developed encounter histories at the seasonal temporal scale and used the Delta method to approximate variances of annual survival rates (Powell 2007). The model was age-structured, allowing juveniles to graduate into the adult cohort after surviving through April of the year following their birth (Gese and Thompson 2014).

Using a list of individual-based covariates (i.e., age, sex) and temporal variables (i.e., season, year, manipulation, prey base), we developed a candidate set of 15 *a priori* models containing univariate, two-way additive, and two-way interactive combinations based on our primary research question and previous investigations on kit fox ecology (Burnham and Anderson 2002). We used Akaike’s Information Criteria with a small sample size correction (AIC_c) to select the model or models most supported by the data. Models within 2 AICc units of that with the minimum AICc were considered competing models, and we used model averaged estimates in the case of competing models (Burnham and Anderson 2002).

We employed generalized linear mixed models (GLMMs) (Stroup 2012) to test the categorical main effects of period (pre- and post-manipulation) and transect type (proximate and distant) on the continuous response variables of kit fox relative abundance: kit fox scats/transect/survey and kit fox scent station visits/transect/survey. Specifically, we tested the impact of water development manipulation by including a period by transect type interaction in our model (Underwood 1992). Within the framework of a BACI design, such an interaction tests for a differential change (i.e., non-parallelism) between impact and control sampling units following some type of manipulation (Underwood 1992). Inspection of raw data revealed non-normality and a high frequency of zeros for both data sets. First, we fit the following model families: lognormal, Poisson, quasi-Poisson, and negative binomial. Models that did not converge were eliminated and we assessed remaining models based on the generalized chi-square fit statistic (Stroup 2012). We compared remaining model families with zero inflated models of the same model family using a Vuong test; zero inflated regression models outperform traditional models of the same family when excess zeros are generated by a separate process from the count values (Everitt and Hothorn 1997). For the scat and scent station data, we selected the Poisson model family for our final models.

For both measures of relative abundance, we conducted multiple surveys on each transect for both periods. In order to reduce model complexity and better account for residual variance, we collapsed our original data sets across surveys. By doing so, data were analyzed within a balanced split plot in a time model framework (Aho 2014). In order to account for variability among survey transects, and variability among survey transects within treatments, we included a survey transect (i.e., proximate or distant) by period (pre- and post-manipulation) random effect (Demidenko 2013). All statistical analyses for relative abundance were performed using the glmm and pscl packages in R (R Development Core Team 2013).

In order to better meet the assumption of independence of observations, we collapsed summary data of kit fox home range environmental characteristics across individual foxes. Inspection of environmental variable data revealed skewness and unequal variances that could not be remedied with data transformations. As a result, we used single response two-tailed permutation tests with 20,000 resamples to test for differences between elevation (continuous variable) and categories of soil type and dominant vegetation (interval/ratio variables). Permutation tests are distribution free in the sense that probabilities of obtaining extreme test statistic values given the truth of the null hypothesis (Type I errors) are based on permutations of the data from randomization theory and are not based on an assumed population distribution (Manly 2006). Permutation tests were performed using the blossom package in R. For all statistical tests we interpreted p-values in terms of relative evidence of differences (Ramsey and Schafer 2002).

RESULTS

Between 15 January 2010 and 1 November 2013, 84 kit foxes were captured and fitted with radiocollars. Of these, 43 and 41 were classified as adult or juvenile at time of capture, respectively. Throughout the study, 7256 locations were recorded on the 84 collared foxes, allowing for the calculation of 114 seasonal home ranges (37 in breeding, 30 in dispersal, and 47 in pup rearing). The mean number of days a fox was monitored, from radio-collaring to either death, loss of signal, or conclusion of the study, was 246 days (SD = 292.7).

A total of 50 kit foxes died during the study (25 adult, 25 juvenile). Of these deaths, 24 (48%) were confirmed coyote predation, 7 (14%) were eagle predation, 6 (12%) were suspected predation, 5 (10%) were unknown cause, 4 (8%) were vehicle collision, 1 (2%) was bobcat predation, 1 (2%) was esophageal feed impaction, 1 (2%) was suspected rattlesnake bite, and 1 (2%) was study influenced. The study-influenced death was censored in survival analyses. Many of the suspected predation events involved recovery of a torn, bloody, or buried radio-collar and only remnants of a carcass. We were unable to conduct a necropsy on these individuals. Thus, suspected and confirmed predation accounted for 76% of the kit fox deaths with coyote predation being the leading cause of death. The percentage of deaths caused by coyotes prior to and following our water manipulation was 41% (12 of 27) and 48% (12 of 23), respectively.

The most supported survival models, which we considered competing, included an additive or interaction effect between age and annual rodent prey base (Table 5-1). Age appeared to have the strongest influence on survival rates (Fig. 5-3). Model averaged annual adult and juvenile survival rates averaged across all years were 55.50 (SD = 2.73) and 27.93% (SD = 3.99) respectively. Other variables that we considered (i.e., sex, season, water manipulation) did not appear to influence survival (Table 5-1).

Examination of model averaged survival estimates suggested a pattern of juvenile survival being more influenced by changes in rodent prey base than adult survival (Fig. 5-2) but comparison of 95% CIs revealed overlap across all years for juveniles and adults. We developed 114 seasonal home ranges from 38 kit foxes spanning from the 2010 pup-rearing season through the 2013 dispersal season. Prior to the water manipulation, 12 of 72 (17%) kit fox home ranges contained a water development. Following reduction of water availability, 1 of 42 (2%) kit fox home ranges contained a water development. Overall, kit fox seasonal visitations to water developments averaged 2.8 (SD = 3.1). Zero water development visitations took place following the water manipulation.

Between September 2010 and August 2013, we conducted 5 seasonal kit fox scat deposition surveys prior to and following our manipulation. On average, we observed 3.26 scats/transect/survey (SD = 5.99), with a range of 0 to 29 scats/transect/survey. We found no evidence that elimination of water at guzzlers influenced kit fox relative abundance (period x transect type interaction: t = 0.42, *P* = 0.44, df = 6). We found convincing evidence that kit fox relative abundance differed by transect type (t = -2.42, *P* < 0.01, df = 6; Fig. 5-4), but found no evidence that relative abundance differed by period (t = -0.82, *P* = 0.41, df = 6). The number of kit fox scats observed on distant transects during the pre-period was 5.50 (SE = 1.77), and 6.35 (SE = 1.66) scats/transect during the post-period (Fig. 5-4). The number of kit fox scats observed on proximate transects during the pre-period and post-period was 0.55 (SE = 0.17) and 0.65 (SE = 0.22) scats/transect, respectively (Fig. 5-4).

Between September 2010 and August 2013, we conducted 5 seasonal scent station surveys prior to and following our manipulation. On average, 2.27 visits/transect/survey (SD = 3.15) were observed and counts ranged from 0 to 15 visits/transect/survey. We found no evidence that elimination of water at guzzlers influenced kit fox relative abundance (period x transect type interaction: t = 1.123, $P = 0.26$, df = 6). We found convincing evidence that kit fox relative abundance differed by transect type (t = -1.85, $P < 0.01$, df = 6; Fig. 5-5), but found no evidence that relative abundance differed by period (t = -0.11, $P = 0.48$, df = 6). The number of kit fox visits observed at distant transects during the pre-period and post-period was 3.54 (SE = 0.77) and 4.05 (SE = 0.88) visits/transect during the post-period, respectively (Fig. 5-5). The number of scent station visits observed on proximate transects during the pre-period and post-period was 0.60 (SE = 0.19) and 0.79 (SE = 0.22) respectively (Fig. 5-5).

We found seasonal 95% KDE home range sizes for kit foxes averaged 19.45 km^2 (n = 114, SD = 15.1). For all years combined, average home range size of kit foxes during the breeding season was 24.25 km^2 (n = 37, SD = 20.91), followed by the dispersal season ($\bar{x}$ = 19.56 km^2, n = 30, SD = 10.34) and pup-rearing ($\bar{x}$ = 15.93 km^2, n = 47, SD = 11.32). Average elevation within kit fox home ranges and water zones averaged 1387 (SE = 18.62) and 1491 (SE = 35.84), respectively (Fig. 5-6). We found convincing evidence that elevation of kit fox home ranges was greater than water zones (n = 51, $P < 0.001$). We also found convincing evidence that home ranges and water zones consisted of different proportions of silt (n = 51, $P < 0.001$) and blocky loam (n = 51, $P < 0.001$), where home ranges contained a greater proportion of silt and water zones contained a greater proportion of blocky loam (Fig. 5-7) .We found no evidence of a difference for fine sand (n = 51, $P = 0.19$) (Fig. 5-7). We found suggestive evidence that fit fox home ranges and water zones contained different proportions of barren land cover (n = 51, $P < 0.08$), and convincing evidence of differences for shrubland (n = 51, $P < 0.001$) and herbaceous (n = 51, $P < 0.001$) dominant cover types (Fig. 5-8).

DISCUSSION

Our study was the first to quantify individual based visitations to water sites for kit foxes, the first to incorporate a resource manipulation design to evaluate the effects of water developments on kit fox survival and abundance, and the first to compare environmental characteristics of areas associated with kit foxes and water developments. We found that overall kit fox use of water developments was rare. We also found that the majority of kit foxes whose home ranges contained free water would occasionally visit water developments, and that the majority of water developments occurred in areas that varied from kit fox home ranges in elevation, soil and vegetation.

Our visitation findings provided additional support that kit foxes at DPG are an arid adapted species that rarely utilizes free water (Hall et al. 2013). However, Hall et al. (2013) also observed regular visitations for kit foxes in the Mojave Desert; they speculated that this difference was likely attributed to disparities in nighttime temperatures between the two deserts which resulted in a reduced thermal gradient for kit foxes to dissipate heat from thermal activates in the Mojave. Further, Rosenstock et al. (2004) recorded 76 kit fox drinking events at water developments in Arizona, which suggested kit foxes will utilize free water when available in the Sonoran Desert. Given these disparate findings, and the limited number of undertaken investigations regarding kit foxes and free water, we caution against range-wide generalizations regarding kit fox use of free water.

Neither adult nor juvenile kit fox survival were influenced by reductions of free water availability at DPG. Age class clearly had the largest influence on survival, as juvenile survival was markedly lower than adult survival (Fig. 5-3). Juvenile survival has not been estimated often for kit foxes, but existing estimates have ranged from 0.05 to 0.78 for juvenile swift foxes (Rongstad et al. 1989, Karki et al. 2007, Gese and Thompson 2014). We found qualitative but not statistical evidence that juvenile survival may be more influenced by rodent abundance than adult survival. When this resource is abundant, juvenile kit foxes may transfer from natal dens in better body condition and/or adult kit foxes may allow juveniles to remain near natal dens for a longer temporal span [i.e., delayed dispersal (Sparkman et al. 2012)], thus increasing the likelihood that juveniles will survive to the adult age class. We did not explicitly test for differences in dispersal for this study because only 5 juvenile foxes emigrated from the study area (two prior to the manipulation and three following the manipulation). We highly recommend that future kit fox investigations examine juvenile kit fox survival rates, as this appears to be an area that has received little attention.

Our estimate of annual adult survival (0.56) falls within the range reported across the species' range (0.35 – 0.85; White and Garrott 1997, Arjo et al. 2007). For DPG, annual adult survival rates for 1996, 1999, and 2000 were 0.65, 0.85, and 0.71, respectively, but all estimates were associated with high sampling variance (White and Garrott 1997, Arjo et al. 2007). Unfortunately, the estimation of annual survival per se was not possible in the kit fox investigations that occurred prior to the marked increase of free water at DPG (Egoscue 1956, 1962). Golden eagle predation had not previously been recorded on DPG, yet we found it accounted for nearly 15% of all kit fox mortalities. Golden eagle predation of kit foxes has only been reported once (Cypher and Scrivner 1992), and our investigation appears to be the first where golden eagle predation contributed >5% of kit fox mortality. Golden eagle predation has been reported as the most common source of mortality for swift foxes (Moehrenschlager et al. 2007) and island foxes (Roemer et al. 2002). We suspect kit foxes at DPG may be foraging more often during crepuscular periods in order to compensate for limited resources (Kozlowski et al. 2012). Such activity would put individual kit foxes at greater risk of predation from golden eagles, a diurnal predator. If eagle predation is additive, then increases in eagle predation may be lowering overall kit fox survival at DPG, which could have population level ramifications. Alternatively, documented reductions of shrubland vegetation and expansions of exotic herbaceous vegetation at DPG may be increasing golden eagle lethality of kit foxes if kit foxes are more readily detected and easier to capture in open grasslands than mosaic shrublands. Further, documented reductions of leporid abundance at DPG (Arjo et al. 2007, B. M. Kluever, unpublished data) may be prompting golden eagles to select for kit foxes more than they would in the presence of a robust leporid population (Moehrenschlager et al. 2007).

The percentage of kit foxes killed by coyotes during our study did not appear to change following the reduction of water availability (i.e., 41% prior to manipulation, 48% following manipulation). These numbers were similar to those observed at DPG by Arjo et al. (2007), and they fall within the lower range of coyote death rates observed for kit foxes across their range (White and Garrott 1997). In a meta-analysis of kit fox populations, White and Garrott (1997) suggested that prey abundance and behavioral spacing mechanisms likely regulate kit fox populations, but coyote predation may be a limiting or partially regulating factor. Whether coyotes represent a regulating or limiting factor on kit fox population dynamics has yet to be clearly determined at DPG, as well as throughout the majority of the species' range.

Our scent station and scat deposition surveys provided no evidence that removal of water availability at developments influenced kit fox abundance (Figs. 5-4 and 5-5), even though the BACI design associated with this component of our study has repeatedly been reported as being superior to purely observational studies (Underwood 1992, Morrison et al. 2001). Our kit fox relative abundance findings resembles that of an investigation at DPG by Hall et al. (2013), where an observational, non-road based scent station survey design was utilized. We were unable to derive actual estimates of kit fox abundance using our survey transect design because we did not incorporate a noninvasive genetic sampling component into our study design. In addition, our low kit fox capture rates (< 1 fox captured for every 100 trap nights) did not allow for robust use of invasive capture-mark-recapture estimation methods. We urge future studies on small carnivores to incorporate designs that allow both relative and absolute population parameters to be obtained and compared.

We found that kit fox home ranges and areas associated with water developments varied by elevation, dominant vegetation, and most soil types, providing some evidence that the majority of water developments at DPG were constructed in areas that do not currently represent kit fox habitat, and may not have historically constituted kit fox habitat. Portions of DPG and surrounding areas have undergone exotic herbaceous vegetation encroachment (Emrick and Hill 1999, Arjo et al. 2007), but the impact of this change on the kit fox population at DPG, and others, is unclear. Herbaceous vegetation encroachment reduces jackrabbit abundance, but preliminary findings from our small mammal sampling effort suggested that moderate levels of exotic species encroachment may facilitate an increase of rodent abundance, which would provide additional evidence of the intermediate disturbance hypothesis regarding exotic vegetation encroachment (Malick et al. 2012, Fischer et al. 2012, Hu et al. 2013). Further, though shrublands at DPG contain more plentiful food resources than grasslands and barren to semi-barren vegetation communities, these areas also increase the risk of intraguild predation for kit foxes, and as a result are utilized less often by kit foxes (Kozlowski et al. 2012). Similar findings have been documented for populations of kit foxes (Robinson et al. 2014) and swift foxes (Thompson and Gese 2007). Thus, the encroachment of exotic herbaceous vegetation may actually buttress the creation of additional kit fox habitat where resources are poorer than native shrub communities, but coyote predation risk is depressed.

Elevation has been reported as an important habitat component for kit foxes through its indirect influence on vegetation assemblages (McGrew 1976, Fitzgerald 1996), and we found that kit fox home ranges were consistently associated with areas of low elevation when compared to water zones. Kit foxes have traditionally been reported as a species that primarily utilizes lowland flat areas (Egoscue 1975, Zoellick and Smith 1992), and it is unlikely that elevation gradients at DPG have changed during the past half century. We found that kit foxes home ranges and water zones varied by proportions of blocky loam soil and silt. Using a resource selection function, Dempsey (2013) found that kit fox distribution at DPG was influenced by soil type, where kit foxes rarely occurred in areas with large blocky soils, which would be difficult for den excavation. Den sites are considered to be important to kit foxes as they “provide shelter from temperature extreme, moist microclimate, escape from predators, and a place to rear young” (Arjo et al. 2003) and are a critical part of the survival strategy of kit foxes (Gerrard et al. 2001). Proper denning conditions (i.e., soil type) may therefore be required to support kit foxes at DPG. Although kit foxes are highly mobile and capable of traveling away from denning areas to forage, they still tend to occur on soils where dens are easily dug, suggesting kit foxes may stay within ‘den friendly’ soils because of the use of dens for refuge from predation (White et al. 1994, Koopman et al. 2000, Arjo et al. 2003). Our findings appear to provide evidence that water development construction may not have greatly reduced kit fox habitat at DPG.

Support for the indirect effect of water hypotheses for the canid community at DPG is partially predicated on observed and hypothesized changes in coyote and kit fox populations and canid-habitat relationships following a period of marked increases in water developments (Arjo et al. 2007, Kozlowski et al. 2008, 2012). We posit that other factors may have contributed to such changes. Within a study area that encompassed our own, Egoscue (1956) argued that coyote abundance was suppressed by way of intensive coyote control efforts, including regular use of baited toxicants spaced at intervals aimed to maximize lethality to coyotes rather than carnivores with smaller home ranges (i.e., kit foxes). In addition, Shippee and Jollie (1953) reported that coyotes were historically controlled on and near DPG using a host of methods including spring den hunting, shooting, trap sets, poison pellets, cyanide guns, and poisoned sheep carcasses; >80 coyotes were harvested in 1951-1952 alone. It seems intuitive that this suite of control factors may have been a driving force behind the seemingly low coyote numbers reported at DPG in 1953 (Shippee and Jollie 1953). In 1972, the enacting of Executive Order 11643 banned the use of baited toxicants and additional restrictions have been placed on the use of toxicants for predator control by the Environmental Protection Agency (Mitchell et al. 2004). Dorrance and Roy (1976) and Nunley (1986) suggested that coyote control programs that relied heavily on toxicants were more effective at suppressing coyote populations than contemporary methods, though non-target species can also be impacted. In addition, the banning of baited toxicants may have engendered a population increase of golden eagles in and around DPG (Millsap et al. 2013). Therefore, the observed commonality of kit foxes observed by Egoscue (1956, 1962) and rarity of coyotes reported by Egoscue (1956), Shippee and Jollie (1953), and Arjo et al. (2007) during the mid-twentieth century may be at least in part be tied to changes in coyote control management practices that temporally coincided, but were largely unrelated to, the additions of water developments. However, changes in fire suppression, invasion of exotic grasses, and subsequent changes in prey abundance and composition, plus changes in abundance of other predators (e.g., golden eagles), cannot be ruled out as having influenced kit fox distribution and abundance over the last 50 years. The combination of all these changes may in fact have predicated a “perfect storm” causing a continued decline in kit fox habitat and population size in the Great Basin.

Our study was the first to report individual based frequencies of water site visitations for kit foxes, examine environmental characteristics of kit fox home ranges and water developments, and incorporate a manipulation design to test the indirect effect of water hypothesis (Larsen et al. 2012) for kit foxes. Our findings provide evidence that water developments do not appear to be a factor driving the canid community at DPG during our investigation. That being said, we recommend some caution be exercised in relation to our findings. Our investigation focused on one study area and spanned only a 4-year period. We recommend future investigations on the impact of free water on kit foxes, canid communities, or other species or communities of interest should consider designs with replication at the study site level (Cain et al. 2008, Krausman and Cain 2013), a longer temporal span (i.e., several generation times of the species of interest) and selection processes at orders not examined in our study [i.e., first and third-order; (Johnson 1980)].

MANAGEMENT IMPLICATIONS

Based on our findings, we do not recommend that water manipulation be utilized as a tactic to enhance kit fox populations in the Great Basin Desert. Along the same vein, we do not recommend that water developments be considered as a strategy to suppress populations of kit fox competitors and/or predators. Efforts to increase kit fox populations should focus on improving kit fox resources by increasing populations of prey species or direct long-term predator control efforts in areas where predation on kit foxes is determined to be a limiting factor.

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Table 5-1. Results from age-structure known fate survival models for radio-collared kit foxes on and adjacent to the U.S. Army Dugway Proving Ground, Utah, 2010-2013. Models shown are those that outperformed the null (age-structured) model.

Model	AICc	ΔAIC*c*	*w*
age * prey base	241.64	0.00	0.35
age + prey base	242.68	1.04	0.21
age	243.67	2.02	0.11
age * water manipulation	244.04	2.40	0.10
age + water manipulation	244.62	2.98	0.08
age * year	245.36	3.72	0.05
age + year	245.81	4.17	0.04
age + sex	247.41	5.77	0.02
season + age	248.58	6.95	0.01
prey base	251.45	9.81	> 0.01
season + prey base	252.21	10.57	> 0.01

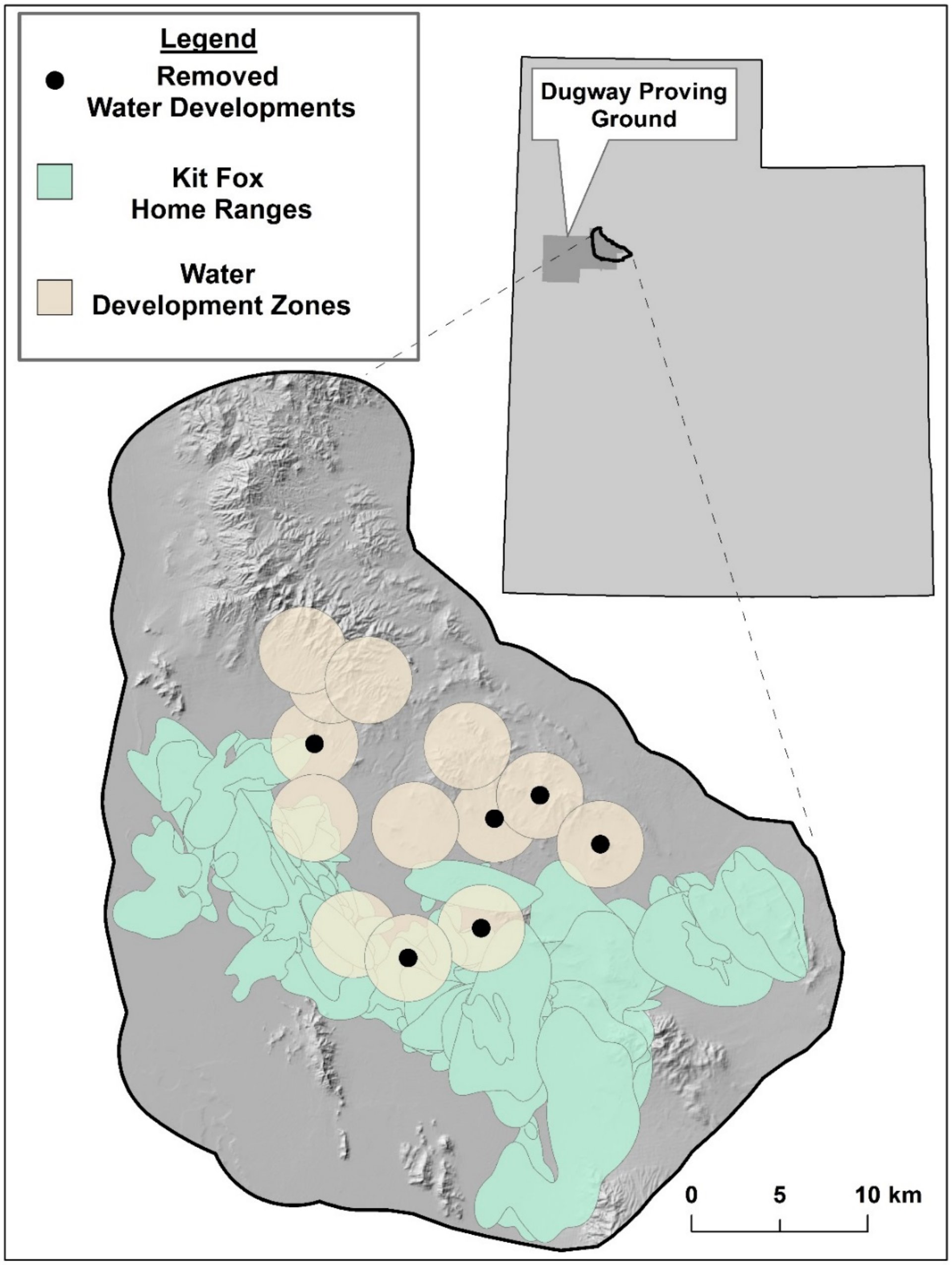

Figure 5-1. Map of 95% fixed kernel seasonal home ranges for kit foxes and water development zones equal in area to the average kit fox home range size on and around the U.S. Army Dugway Proving Ground, Utah. USA, 2010-2013.

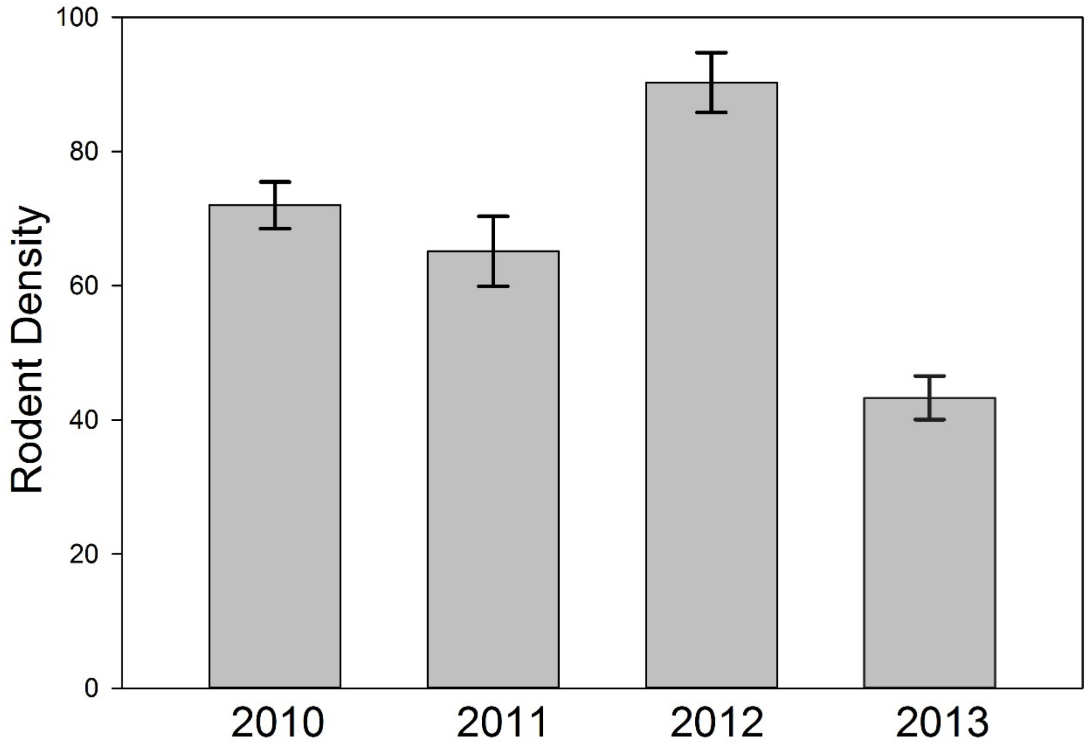

Figure 5-2. Pooled rodent density (rodents/0.04 km^2) (± SE) from sixteen 50 x 50 m trapping grids on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.

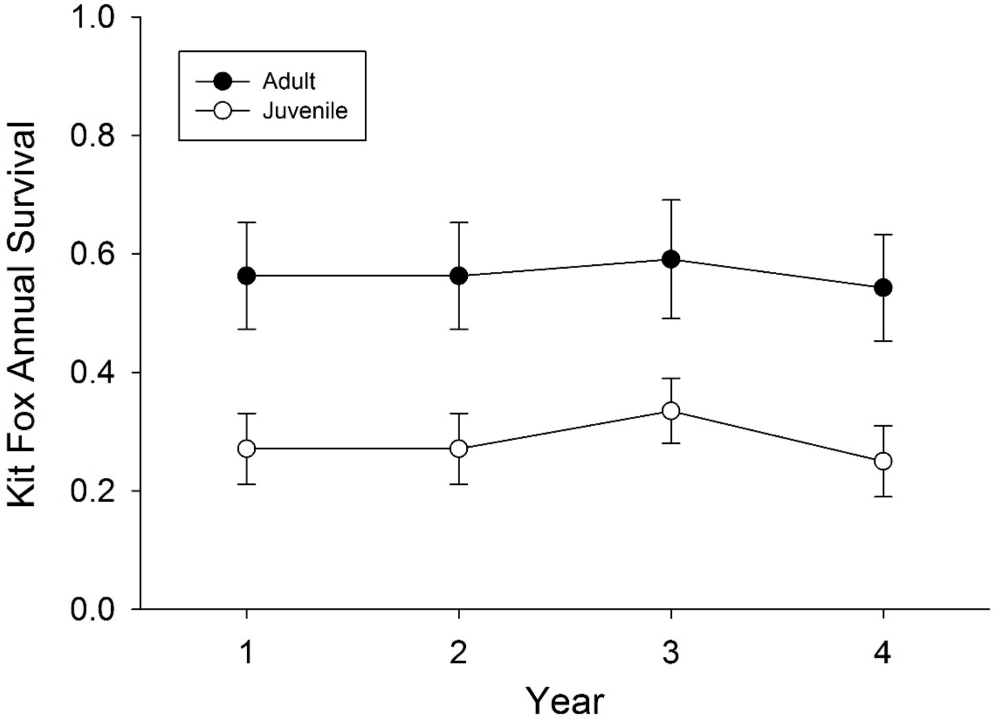

Figure 5-3. Model averaged kit fox annual survival rates (± SE) derived from competing top models (age*prey base, age+prey base) on and around the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.

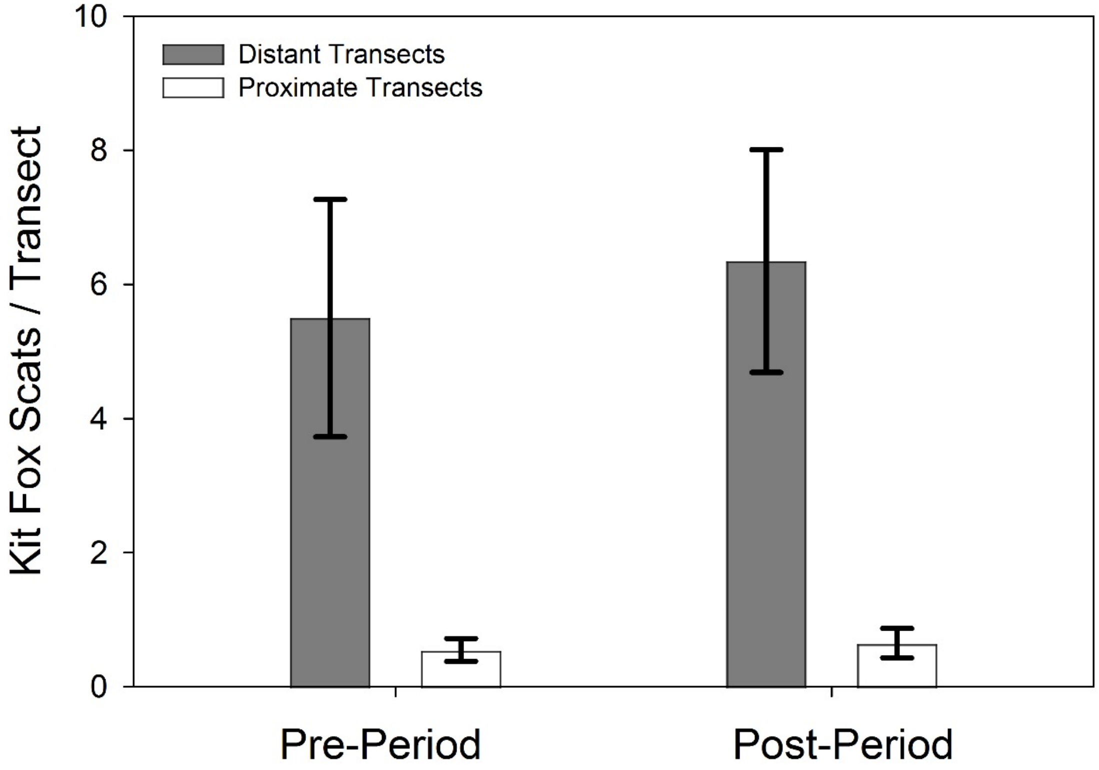

Figure 5-4. Kit fox scats (± SE) observed on 5-km proximate and distant to guzzler transects prior to (pre-period) and following (post-period) removal of water availability at guzzlers on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013

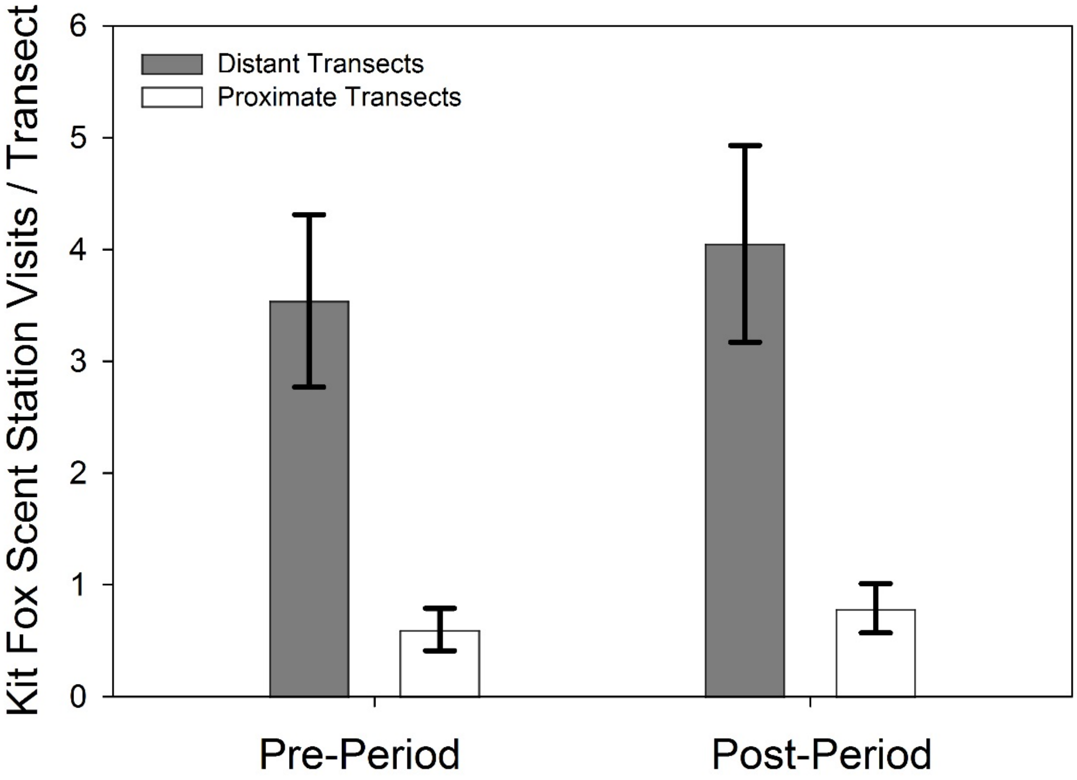

Figure 5-5. Kit fox scent station visits (± SE) observed on 5-km proximate and distant to guzzler transects prior to (pre-period) and following (post-period) removal of water availability at guzzlers on the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.

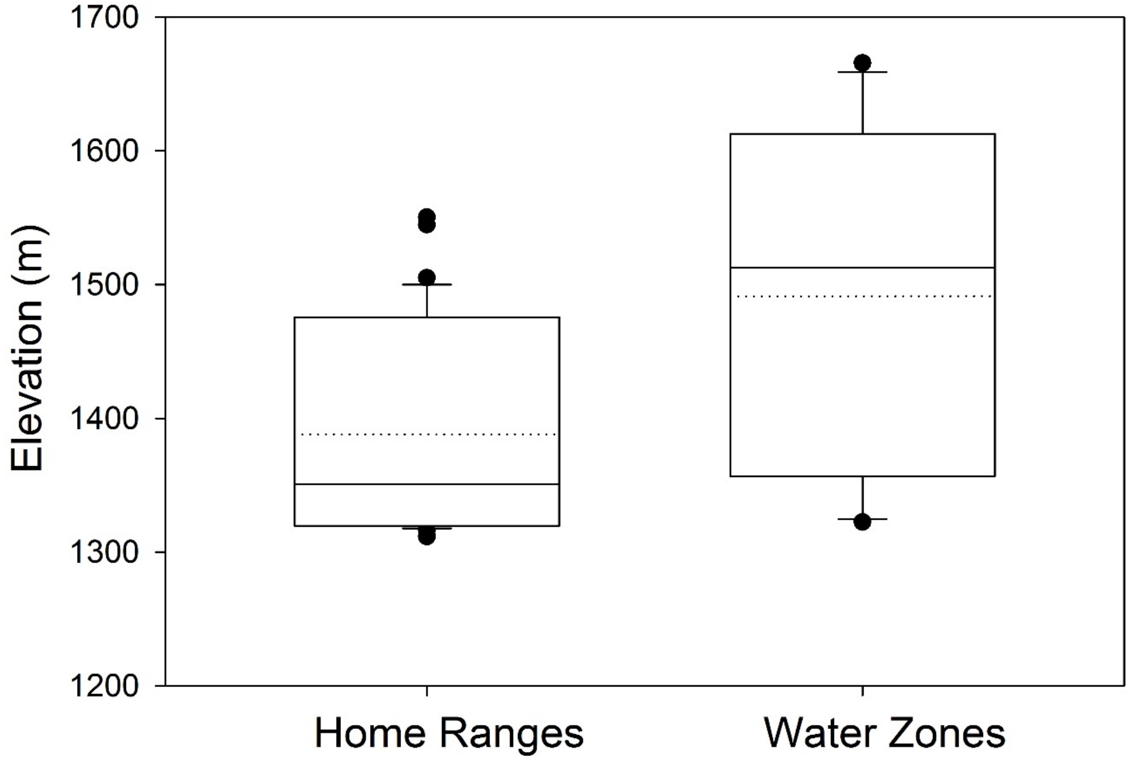

Figure 5-6. Box plots of average elevation (meters) observed within seasonal kit fox home ranges pooled by individual foxes (n = 38) and areas associated with water developments (n = 13) on and near the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013

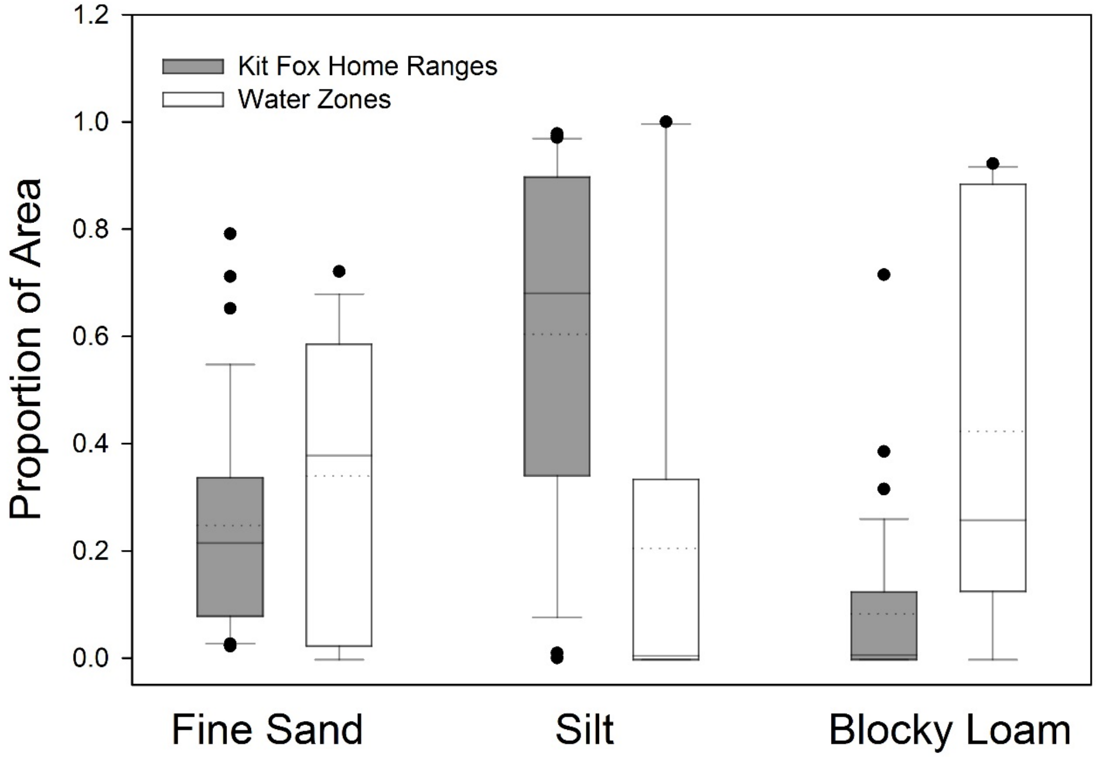

Figure 5-7. Box plots of the proportions of soil types (fine sand, silt, and blocky loam) observed within seasonal kit fox home ranges pooled by individual foxes (n = 38) and areas associated with water developments (n = 13) on and near the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.

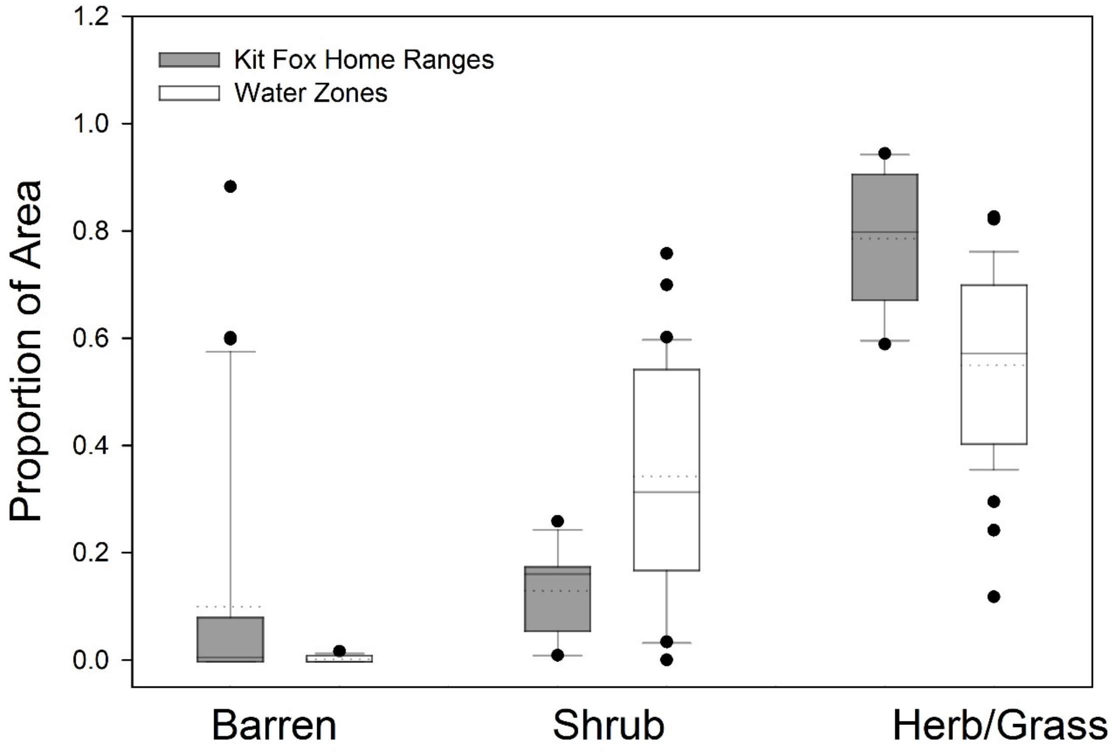

Figure 5-8. Box plots of the proportions of dominant vegetation classes (barren, shrubland, herbaceous) observed within seasonal kit fox home ranges pooled by individual foxes (n = 38) and areas associated with water developments (n = 13) on and near the U.S. Army Dugway Proving Ground, Utah, USA, 2010-2013.

CHAPTER 6

CONCLUSIONS

This dissertation evaluated the impact of water developments on mammalian species and communities in an arid environment over a 4-year period. Chapter 2 focused on employing an observational and BACI design over 4 summers to determine whether water developments influenced rodent abundance, a key prey source of carnivores at this site. Chapter 3 examined coyote and black-tailed jackrabbit relative abundance in relation to water availability at water developments using a BACI design. Chapter 4 focused on coyote frequency of visitations to water developments and spatial affinity of resident coyotes for water sites at the home range scale extent within a manipulation design framework. Chapter 5 examined the role of water developments and removal of water availability on kit fox demography, kit fox frequency of visitations to water developments, and the relationship of kit fox home range characteristics to areas associated with water developments.

In chapter 2, I found no evidence that water developments negatively impacted rodent abundance. Under the observational design framework, estimates of total rodent abundance for trapping grids that were distant and proximate to water developments were similar [i.e., overlap of 95% confidence intervals (CIs)] in 87.5% (7 of 8) of the trapping sessions. The 95% CIs for rodent abundance at distant and proximate grids under the BACI design framework overlapped for 75% (3 of 4) of the trapping sessions prior to the water manipulation; in all three cases abundance was lower at proximate grids. Following the water manipulation, non-overlap of 95% CIs occurred in 100% (4 of 4) of the trapping sessions; rodent abundance estimates were lower at proximate grids in all cases. Results from z-tests validated the 95% CI interval comparisons. My investigation was the first to incorporate a manipulation component to determine the indirect effects of water developments on rodent communities. These findings were similar to those reported by Cutler and Morrison (1998) and Burkett and Thompson (1994). Rodent abundance appeared to be driven by a combination of spatial [e.g., vegetation structure heterogeneity (Clark and Kaufman 1991, Thompson and Gese 2013)] and temporal [e.g., variations in seasonal precipitation (Ernest et al. 2000, Hernandez et al. 2005)] factors.

Increased predation rates, or predation risk on rodents, may have occurred near water developments, but may not have affected total rodent abundance. Predation induced mortality has been suggested to be both compensatory (Mihok 1988) and additive (Meserve et al. 1993) for individual rodent species, but empirical investigations examining the numerical impact of direct predation on community level abundance or productivity are lacking. Furthermore, behavioral strategies have been observed as mechanisms to mitigate predation risk within a host of rodent species and communities (Kotler et al. 1994, Brown et al. 1994, Shenbrot 2014), and the magnitude of behavioral responses has been shown to be correlated to the magnitude of predation risk for some species (Orrock and Danielson 2004).

My results should help guide future management decisions for wildlife management agencies, sportsmen groups, and conservation organizations, especially in arid areas where the animal communities are partially comprised of rare or imperiled species, or when rodents comprise the primary food source for rare or imperiled carnivores. It is important to note that this spatial inference is limited to areas associated with water developments on DPG, and that factors not relevant to our study (i.e., livestock or feral horse visitation to water developments, riparian vegetation near water developments) may promote negative or positive effects at water developments for rodent and other communities. Future field investigations on water developments, and other potential disturbances, would benefit from the incorporation of study designs that include a manipulation component.

In chapter 3, I found no evidence that water availability at guzzlers indirectly impacted jackrabbit relative abundance. There was evidence that the temporal period associated with my water manipulation coinciding with increased jackrabbit relative abundance, but I am confident that this increase was not associated with free water availability because of the BACI design (Morrison et al. 2001). I suspect that the observed temporal trend was at least partially a result of a time lag effect between precipitation and jackrabbit abundance. Hernandez et al. (2011) reported a positive relationship between the previous 12 months of precipitation and both primary productivity and jackrabbit abundance. Similarly, Ernest et al. (2000) reported that rodent abundance was positively correlated to precipitation occurring during the previous season or seasons. Monthly precipitation rates at DPG during 2009 (i.e., a span potentially influencing pre-period jackrabbit relative abundance) and 2011 (i.e., a span potentially influencing post-period jackrabbit relative abundance) averaged 1.14 (SE = 0.25) and 1.96 cm (SE = 0.66), and a precipitation spike of 10.06 cm occurred in May of 2011, eleven months prior to our manipulation (US Army Dugway Proving Ground, West

Desert Test Center Meteorological Division). Thus, a general trend of increased primary productivity leading up to the post-manipulation period may have resulted in increased jackrabbit abundance.

Relative abundance of coyotes was impacted by water availability, with the elimination of water availability at water developments facilitating a reduction of coyote use or abundance in areas associated with the experimental treatment. Specifically, I observed a 30% increase in coyote relative abundance on transects distant from water during the post period, but a 10% decrease on transects proximate to water developments once water availability was removed. Though these findings suggest that coyote populations are directly affected by water developments, caution is warranted in regards to this study's implications. The 10% reduction of coyote relative abundance may have been the result of a portion of coyotes abandoning territories spatially affiliated with water developments, but rather fine-scale changes in coyote movement patterns within territories that did not impact coyote abundance. Falsely equating statistical significance to biological relevance is a real and often an ignored risk in ecological investigations (Martinez-Abrain 2008), and despite the common validation of indices of abundance (Barrio et al. 2010, Hopkins and Kennedy 2004, Kays et al. 2008), they can be problematic (White 2005, Edwards et al. 2014). As such, I recommend future investigations on the effects of water developments on individual species and ecological interactions to incorporate absolute estimates of population parameters or spatial (e.g., habitat selection) analyses into their study design, as was done in chapter 4.

Results from chapter 4 showed that visitation to water developments with available water averaged 13.0 visitations/season (SD = 13.5), and ranged from zero to 47. In addition, water developments associated with riparian vegetation did experience higher visitation than wildlife water developments (no riparian vegetation present) and removal of water availability facilitated reduced visitations to water developments. However, I documented no visits to water sites in 16% (10 of 64) of seasonal home ranges, <5 visits within 39% (25 of 64) of home ranges, and no water sites in 25% (28 of 113) of coyote home ranges. Thus, 33% (36 of 108) of the home ranges that were investigated either did not contain a water site or contained a water site that was not visited for an entire season (e.g., approx. 120 day period). Furthermore, I found no evidence that elimination of water availability influenced home range size, spatial shifting of home range areas, or facilitated coyote mortality or territory abandonment.

This visitation study reveals that free water does not appear to be an obligatory (e.g., population wide) habitat component for coyotes in the study area during the temporal span investigated. No other studies have explored individual based visitations by coyotes to water developments, so comparisons with other populations are not possible. Allen (2012) found that all radio-collared dingoes in the Strzelecki Desert visited water sites every season, and dingoes rarely went >5 days without visiting a water site. In three different arid landscapes, Hall et al. (2013), Rosenstock et al. (1999), and Attwood et al. (2011) reported that coyotes regularly visited water developments, but data were reported as indices, and thus cannot be used to determine the proportion of coyotes on the landscapes that were visiting water developments. Future investigations are needed to determine both the frequency and relevancy of coyote visitations to water sites.

My assessment of home ranges in relation to water sites provided additional support that access to water sites was not an obligatory resource component for coyotes. I speculated that availability of free water at DPG serves as a non-pivotal resource subsidy that is utilized frequently by some members of the population, and little to null by others. I was only able to track seasonal overlapping of home ranges for three coyotes that belonged to the impact class for the three consecutive seasons following the manipulation. Such a small sample size and resulting lack of statistical inference warrants caution. However, all three of these animals maintained a spatial affinity to their home ranges following the manipulation (i.e., they did not die, disperse or abandon their home ranges after water was no longer available). Regardless of these results, anthropogenic modifications to landscapes have been shown to influence coyote home range selection, which is considered a second-order selection process (Johnson 1980). Boisjoly et al. (2010) determined that clear-cutting activities in boreal forests increased coyote habitat quality by increasing food accessibility, and posited that such anthropogenic activity may have allowed coyotes to establish home ranges. Hidalgo-Mihart et al. (2004) found coyotes utilizing landfills had home ranges less than half the size of coyotes that occurred in vegetation zones predominantly unaltered by anthropogenic modification.   Conversely, Atwood et al. (2004) found coyote home range sizes were largest in areas with the highest levels of anthropogenic modification to the landscape. Clearly, the influence of anthropogenic factors on coyote selection processes is highly variable and contingent upon myriad factors that can differ across time and space.

An indirect effect of water site availability has been proposed as a contributing factor to reduced kit fox abundance and distribution at DPG; addition of water sites in the mid to late twentieth century coincided with increased abundance of coyotes (Arjo et al. 2007; Kozlowski et al. 2008). Such an indirect effect of water sites was observed for an ungulate community in Africa (Harrington et al. 1999). Furthermore, depression of coyote populations has been shown to positively alter the abundance of gray foxes (*Urocyon cinereoargenteus*) (Henke and Bryant 1999) and swift foxes (Kamler et al. 2003). Despite the rationality behind the notion that increase in water developments altered the canid community at DPG, I speculate that the observed population increase observed at DPG beginning in the mid-twentieth century may have been attributed more to changes in coyote management practices. Egoscue (1956) argued that coyote abundance at DPG was suppressed by way of intensive coyote control efforts, including regular usage of baited toxicants spaced at intervals aimed to maximize lethality to coyotes rather than carnivores with smaller home ranges (i.e., kit foxes). The use of baited toxicants was a common predator control tactic in Utah, including DPG, during the mid-twentieth century (Shippee and Jollie 1953). In 1972, the enacting of Executive Order 11643 banned the use of baited toxicants and additional restrictions have been placed on the use of toxicants for predator control by the Environmental Protection Agency (Mitchell et al. 2004). Dorrance and Roy (1976) and Nunley (1986) suggested that coyote control programs that relied heavily on toxicants were more effective at suppressing coyote populations than contemporary methods.

Results from chapter 4 should allow wildlife managers and conservation strategists to better determine management plans for coyotes and conservation efforts for species of concern threatened, at least partially, to high levels of coyote predation or competition, like the kit fox. Based on these findings, I do not recommend that water manipulation be utilized as a tactic to attempt modifications or suppressions of coyote populations in arid environments. Along the same vein, I do not recommend that water developments be considered as a strategy to enhance populations of species whose distributions and/or populations are lessened by the presence of coyotes. That being said, it is important to note that this chapter did not explicitly investigate the impacts of water developments on kit fox demography and/or space use. Such investigations are needed to validate this viewpoint that water developments have largely not shaped canid community dynamics at DPG.

The results from chapter 5 showed that kit fox annual survival was not influenced by water manipulation, but was markedly influenced by age class and to a lesser extent by rodent abundance. Surprisingly, I also found that kit foxes at DPG experienced the highest levels of golden eagle predation ever recorded. Despite utilizing two well established data types for relative abundance (e.g., scat deposition surveys and scent station surveys) and a BACI design, I found no evidence that removal of water availability impacted kit fox abundance. Finally, I found that kit fox home ranges differed from areas associated with the majority of water developments in the study area based on elevation, most soil types, and vegetation cover type (e.g., shrubland, herbaceous, barren). Taken in tandem, I believe these findings provide convincing evidence that water developments do not indirectly affect kit foxes at DPG.

The 0.56 adult annual survival estimate falls within the range of 0.35 to 0.85 reported across the species' range (White and Garrott 1997, Arjo et al. 2007). For DPG, annual adult survival rates for 1996, 1999, and 2000 were 0.65, 0.85, and 0.71, respectively, but all estimates were associated with high variance (White and Garrott 1997, Arjo et al. 2007). Unfortunately, annual survival estimates were not derivable from kit fox investigations that occurred prior to the marked increase of free water at DPG (Egoscue 1956, 1962). The percentage of kit foxes killed by coyotes during my study did not appear to change following the reduction of water availability (i.e., 41% prior to manipulation, 48% following manipulation). These numbers were similar to those observed at DPG by Arjo et al. (2007), and they fall within the lower range of coyote death rates observed for kit foxes across their range (White and Garrott 1997).
Golden eagle predation had not previously been recorded on DPG, yet I found that it accounted for nearly 15% of all kit fox mortalities. Golden eagle predation of kit foxes has only been reported once (Cypher and Scrivner 1992), and this investigation appears to be the first to show golden eagle predation contributing to >5% of kit fox mortality.

The kit fox relative abundance findings resemble that of an investigation at DPG by Hall et al. (2013), where an observational, non-road based scent station survey design was utilized. I was unable to derive actual estimates of kit fox abundance using my survey transect design because a noninvasive genetic sampling component was not incorporated into the study design. In addition, low kit fox capture and recapture rates (< 1 fox captured for every 100 trap nights) prevented the use of an invasive capture-mark-recapture framework.

I found that kit fox home ranges were consistently associated with areas of low elevation when compared to water zones. Kit foxes have traditionally been reported as a species that primarily utilizes lowland flat areas (Egoscue 1975, Zoellick and Smith 1992), and it is unlikely that elevation gradients at DPG have changed during the past half century. I found that kit foxes home ranges and water zones varied by proportions of blocky loam soil and silt. Using a resource selection function, Dempsey (2013) found that kit fox distribution at DPG was influenced by soil type, where kit foxes rarely occurred in areas with large blocky soils which would be difficult for den excavation. It is unlikely that the distribution of soil types at DPG have undergone drastic changes in the past half century. Hence, if soil type (e.g., denning substrate) restricts kit fox habitat at DPG, it seems unlikely that the establishment of water developments in areas previously unsuitable for kit foxes had a marked effect on the population once developments were constructed. It is estimated that 40% of historical juniper woodland and shrub communities at DPG have been replaced by exotic herbaceous vegetation (Emrick and Hill 1999). Though shrublands at DPG contain more plentiful food resources than grasslands and barren to semi-barren vegetation communities, these areas also increase the risk of intraguild predation for kit foxes, and as a result are utilized less often by kit foxes (Kozlowski et al. 2012). The role that exotic herbaceous vegetation encroachment has played on kit fox demography and habitat relationships in the Great Basin Desert is an area deserving further exploration.

Support for the indirect effect of water hypotheses was not garnered by chapter 5. As discussed in chapter 4, coyote management practices may have played a role in observed changes to the canid community at DPG. Shippee and Jollie (1953) reported that coyotes were historically controlled on and near DPG using a host of methods including spring den hunting, shooting, trap sets, poison pellets, cyanide guns, and poisoned sheep carcasses; >80 coyotes were harvested in 1951-1952 alone. I was unable to determine exactly when coyote management practices changed at DPG, but by 1991, the majority of the study area was no longer open to coyote hunting (U.S. Army Dugway Proving Ground 1991). Whether or not the intense coyote management practices reported by Egoscue (1956) and Shippee and Jollie (1953) were suppressing the coyote population at DPG is unknown; the effectiveness of coyote removal in suppressing coyote populations varies (Connolly 1995, Blejwas et al. 2002, Kilgo et al. 2014). Nonetheless, the observed commonality of kit foxes observed by Egoscue (1956, 1962) and rarity of coyotes reported by Egoscue (1956), Shippee and Jollie (1953), and Arjo et al. (2007) during the mid-twentieth century may be at least in part tied to changes in coyote control management practices that temporally coincided, but were largely unrelated, to the additions of water developments. However, other changes such as fire suppression, invasion of herbaceous vegetation, and subsequent changes in prey abundance and composition, plus changes in abundance of other predators (e.g., golden eagles), might have also been influencing kit fox distribution and abundance over the last

50 years. The combination of all these changes may in fact have predicated a “perfect storm” causing a continued decline in kit fox habitat and population size in the Great Basin.

The investigations comprising this dissertation represent only the second attempt in North America to incorporate a manipulation framework in order to better understand the impacts of water developments on wildlife. Overall, I found no evidence of indirect effects of water developments for any of the arid adapted species observed. I found marginal evidence of a direct effect of water developments on coyote relative abundance, but when considering the findings as a whole, I feel it unlikely that the water manipulation facilitated an ecologically relevant decrease in coyotes at the population level in the study area. Finally, my findings suggest that observed changes to the canid community at DPG were not primarily driven by water developments.

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APPENDIX

VITA

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EDUCATION

2009-present Ph.D. Candidate, Wildlife Biology
Utah State University (USU) - Wildland Resources Department
Dissertation: The role of water developments on mammals in an arid landscape.
Advisor: Dr. Eric M. Gese, GPA: 3.8

2005-2007 M.S. in Renewable Natural Resources
University of Arizona - School of Natural Resources
Thesis: The role of social factors, environmental variables, and predators on behavior of range cattle.
Advisors: Dr. Stewart W. Breck and Dr. Larry D. Howery, GPA: 4.0

1999-2003 B.S. in Fisheries and Wildlife Management
University of Nebraska - School of Natural Resources, GPA: 3.66

EMPLOYMENT (past ten years) (* *denotes part-time*)

8/2009-present **GRADUATE RESEARCH ASSISTANT**
USU, Logan, UT
Supervisor: Dr. Eric Gese, 435-760-5270, eric.gese@usu.edu

Responsible for the conceptualization and implementation of study design, data collection, data management, data analyses, and production of high quality research products of a large scale research project. Served as hiring agent and supervisor for all technicians, field assistants, and volunteers. Procured funding beyond original budget. Trained, managed, and provided leadership for 20+ technicians. Served as a liaison with cooperators, shareholders, and agency personnel.

1/2011-3/2012 **WILDLIFE BIOLOGIST***
Frontier Corporation USA, Providence, UT
Supervisor: Dennis Wegner, 435-757-7022, dwenger@frontiercorp.net

Conducted field and office aspects of biological surveying in order to collect data and prepare documentation necessary for regulatory compliance, permitting, and wildlife mitigation. Field duties included surveys for raptors, point counts for migratory songbirds, greater sage-grouse habitat assessment and lek surveys. Developed progress reports and regularly met with clients/cooperators.

9/2007-1/2010 **SENIOR WILDLIFE BIOLOGIST**
Hayden-Wing Associates, Laramie, WY
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Served as project manager for several coal-bed methane planned areas of development. Assisted in field and office aspects of biological surveying in order to collect data and prepare documentation necessary for permitting, licensing, and conservation area planning. Prepared NEPA documentation, including environmental impact statements and environmental assessments. Field duties included surveys for raptors, prairie dog surveys, greater sage-grouse habitat assessment and lek surveys. Regularly met with clients and regulatory agency personnel.

8/2005-8/2007 **GRADUATE RESEARCH ASSISTANT**
University of Arizona, Tucson, AZ
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Responsible for the conceptualization and implementation of research budget, study design, data collection techniques, data management, data analyses, hiring and supervising of technicians, and production of high quality research products. Served as hiring agent and supervisor for all technicians, field assistants, and volunteers. Procured additional funding beyond original budget.

3/2005-5/2007 **FIELD BIOLOGIST***
Environmental Planning Group, Tucson, AZ
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Assisted in field and office aspects of biological surveying in order to collect data and prepare documentation necessary for permitting, licensing, NEPA compliance and wildlife and conservation area planning. Specific field duties included surveys for rare plants and wildlife inventories along proposed power line routes.

11/2004-1/2005 **LAB TECHNICIAN**
Nebraska Veterinary Diag. Center, Lincoln, NE
Supervisor: Judi Galeota, 402-472-9416, judith.galeota@unl.edu

Utilized modern laboratory techniques to test for chronic wasting disease in white-tailed deer and Rocky Mountain elk in order to track the distribution and potential spread of the disease in several Midwestern and Western U.S. states.

4/2004-10/2004 **FIELD TECHNICIAN**
Tern & Plover Conserv. Partnership, Lincoln, NE
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Reduced conflicts with gravel mines in eastern Nebraska and least terns and piping plovers by maintaining communications with mine personnel, encouraging birds to nest in areas away from mine activity using fladry, and monitoring population parameters of bird colonies.

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1. **Kluever, B. M.,** E. M. Gese, S. J. Dempsey. 2015. The influence of road characteristics and species on detection probabilities of carnivore faeces. Wildlife Research. 42 75-82. doi: 10.1071/WR14244
2. Lonsinger, R. C., E. M. Gese, S. J. Dempsey, **B. M. Kluever**, T. R. Johnson, and L. P. Waits. 2014. Balancing sample accumulation and DNA degradation rates to optimize noninvasive genetic sampling of sympatric carnivores. Molecular Ecology Resources doi: 10.1111/1755-0998.12356
3. Dempsey, S. D., E. M. Gese, and **B. M. Kluever**. 2014. Finding a fox: an evaluation of survey methods for detecting a small desert carnivore. PLOS ONE 9(8). doi:10.1371/journal.pone.0105873
4. Mohammadi, S., **B. M. Kluever**, T. Tamashiro, Y. Amano, and J. G. Hill III. 2014. Spatial and thermal observations of a Malayan Krait (*Bungarus canidus*) from Thailand. Tropical Natural History 14(1) 21-26.
5. **Kluever, B. M.,** E. M Gese, S. J. Dempsey, and R. N. Knight. 2013. A comparison of methods for monitoring kit foxes at den sites. Wildlife Society Bulletin 37(2) 439-443.
6. Breck, S. W., P. Clark, L. D. Howery, D. Johnson, **Kluever, B. M**, S. Smallidge, and A. Ciblis. 2012. A perspective on livestock-wolf interactions on western rangelands. Rangelands 34(5) 6-11.
7. **Kluever, B. M**., L. Lagos, S. W. Breck, L. D. Howery, M. Sanmartín D. L. Bergman, and F. Bárcena. 2012. Integrity and retention of ear-tag radio transmitters in domestic cattle and feral horses. Wildlife Society Bulletin 36(1) 189-193.
8. Breck, S. W., **B. M. Kluever**, M. Panasci, J. Oakleaf, T. Johnson, W. Ballard, L. D. Howery, and D. L. Bergman. 2011. Domestic calf mortality and producer detection rates in the Mexican wolf recovery area: implications for livestock management and compensation schemes. Biological Conservation 144(2) 930-936.
9. **Kluever, B. M**., L. D. Howery, S. W. Breck, and D. L. Bergman. 2009. Predator and heterospecific stimuli alter behaviour in cattle. Behavioural Processes 81(1) 85-91.
10. **Kluever, B. M**., S. W. Breck, L. D. Howery, P. R. Krausman and D. L. Bergman. 2008. Vigilance in range cattle: the influence of predation, social interactions and environmental factors. Journal of Rangeland Ecology and Management 61(3) 321-328.
11. **Kluever, B. M.** 2007. The influence of social interactions, environmental factors, And predators of behavior of range cattle. Graduate Thesis. University of Arizona.

MANUSCRIPTS

1. Dempsey, S. D., E. M. Gese, **B. M. Kluever,** and R. C. Lonsinger. *In prep*. Using a noninvasive survey technique to develop a species distribution model for a rare desert

carnivore.

2. **Kluever, B. M.,** E. M. Gese and S. J. Dempsey. *In prep.* Response of jackrabbits and coyotes to removal of water developments.
3. **Kluever, B. M.** and E. M. Gese. *In prep*. The impact of wildlife water developments on a rodent community in the Great Basin Desert.
4. Smith, T., **B. M. Kluever** and E. M. Gese. *In prep.* Differential impacts of invasive plant species on diversity and biomass of microherbivores.
5. **Kluever, B. M.** and E. M. Gese. *In prep.* Spatial response of coyotes to removal of water sites in a desert landscape.
6. **Kluever, B. M.** and E. M. Gese. *In prep.* Impact of removal of water sites on kit foxes.

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TEACHING EXPERIENCE

Courses Taught as Teaching Assistant

** indicates USU, † University of Nebraska*

2003	Wildlife Management Techniques †	Dr. Larkin Powell
2003	Fundamentals of Communication †	Dr. Adam Jones
2014	Wildland Resources Techniques *	Dr. Dan MacNulty
2015	Animal & Plant Populations *	Dr. David Koons

Guest Lectures

2011	Avian Ecology *	Dr. Frank Howe
2013	Avian Ecology *	Dr. Frank Howe
2014	Mammalogy *	Dr. Morgan Ernest
2015	Animal & Plant Populations*	Dr. David Koons

Mentoring

2013-present	Victoria Holman, Biology Undergraduate, USU
2013-present	Trinity Smith, Wildland Resources Undergraduate, USU Undergraduate Honors Thesis Committee Member

GRANTS AND AWARDS

2013	Research Grant, The Wildlife Society - Utah Chapter, *$1500*
2009-2012	S. J. and J. E. Quinney School of Natural Resources PhD Fellowship, USU, $*60000*
2012	Wildlife and Conservation Research Grant, T & E Inc., *$2400*
2012	Research Grant, American Society of Mammalogists, *$1500*
2012	Graduate Student Senate Travel Award, USU, *$500*
2011	Wildlife and Conservation Research Grant, T & E Inc., *$2200*
2010	Graduate Student Senate Travel Award, USU, *$500*
2010	Wildlife and Conservation Research Grant, T & E Inc., *$2200*

2010	Graduate School Senate Research Grant, USU, *$1000*
2006	Wildlife and Conservation Research Grant, T & E Inc., *$2400*
2006	Ray Cowden Memorial Scholarship, Univ. of Arizona, *$500*
2003	John Holling Memorial Scholarship, Univ. of Nebraska, *$700*
2002	Platte Valley Scholarship, Univ. of Nebraska, *$300*

GRANTS TO MENTEES

2014	USU GIS & Remote Sensing Laboratory UtahView Research Grant, Victoria Holman, *$1700*
2014	USU Undergraduate Research & Creative Opportunity Grant Trinity Smith, *$1000*

PRESENTATIONS (* *denotes mentee author)*

Conference Presentations

1. *Smith, T., E. M. Gese, and **B. M. Kluever**. 2015. The influence of invasive plant species on the small mammal community in a cold desert. Poster at the Utah Chapter Wildlife Society Annual Meeting-Moab, UT.
2. *Holman, V., and **B.M. Kluever**. 2015. The effects of cheat grass on black-tailed jackrabbit relative abundance at Dugway Proving Ground, UT. Poster at the Maps on the Hill Annual Meeting-Salt Lake City, UT.
3. **Kluever, B. M.**, E. M. Gese, and S. J. Dempsey. 2013. A comparison of methods for monitoring kit foxes at den sites. Presented at the Utah Chapter Wildlife Society Annual Meeting-Bryce Canyon, UT.
4. Dempsey, S. J., E. M. Gese, **B. M. Kluever** and R. N. Knight. 2013. Finding a fox: an evaluation of detection methods for censusing a small desert carnivore. The Wildlife Society Annual Meeting-Portland, UT. Presented at The Utah Chapter Wildlife Society Annual Meeting-Bryce Canyon, UT.
5. Dempsey, S. J., E. M. Gese, **B. M. Kluever** and R. N. Knight. 2012. Evaluation of survey methods for kit fox on the U.S. Army Dugway Proving Ground, Utah. Poster at the Wildlife Society Annual Meeting-Portland, UT.
6. **Kluever, B. M**., E. M. Gese, and S. J. Dempsey. 2012. Are cameras or observers more effective at monitoring kit foxes at den sites. Presented at the Wildlife Society Annual Meeting-Portland, OR.
7. **Kluever, B. M**., E. M. Gese, R. N. Knight, S. J. Dempsey, G. B. Smith., N. Mesce, and A. Hodge. 2010. Spatial patterns and individual water use of coyote and kit fox in relation to wildlife water developments and other free standing water sources at the U.S. Army Dugway Proving Ground, Utah: preliminary findings. Poster at the Wildlife Society Annual Meeting-Snowbird, UT.
8. Howery, L. D., **B. M. Kluever**, and S. W. Breck. 2009. Overview of indirect impacts of predators on livestock foraging behavior. Presented at the International Society for Range Management Conference. Wolf-Cattle Interactions Symposium. Albuquerque, NM.
9. **Kluever, B. M**., L. D. Howery and S. W. Breck. 2007. The influence of social and environmental factors on vigilance of range cattle. Presented at the International

Society for Range Management Conference-Reno, NV.

10. Breck, S. W., **Kluever, B. M**. and D. L. Bergman. 2007. Carnivore predation on calves and producer detection rates on east eagle drainage of the Mexican wolf recovery area. Presented at the Wildlife Society Annual Meeting-Tucson, AZ.
11. **Kluever, B. M**., Howery. L. D., and S. W. Breck. 2007. The effects of predator stimuli on behavior of range cattle. Presented at the International Union of Game Biologists 27th Congress-Uppsala, Sweden.
12. **Kluever, B. M**., Howery. L. D., and S. W. Breck. 2007. The effects of predator stimuli on behavior of range cattle. Presented at the Wildlife Society Annual Meeting-Tucson, AZ.
13. **Kluever, B. M**., S. W. Breck and L. D. Howery. 2006. Integrity and retention of transmitters used to monitor livestock depredation. Poster at the Arizona and New Mexico Chapters, Wildlife and Fisheries Society Annual Meeting-Flagstaff, AZ.
14. **Kluever, B. M**., and C. L. Welch. 2004. A Synopsis of the tern and plover conservation partnership and preliminary results for 2004 nesting data. Annual meeting. Poster at the Wildlife Society-Nebraska Chapter-Hastings, NE.

Agency Presentations

1. **Kluever, B. M.,** and E. M. Gese. 2012. Research update: water manipulation for kit fox and coyote project on Dugway Proving Ground. Annual meeting, DoD Integrated Natural Resource Management Plan, Salt Lake City, Utah.
2. **Kluever, B. M.**, and E. M. Gese. 2011. Research update: water manipulation for kit fox and coyote project on Dugway Proving Ground. Annual meeting, DoD Integrated Natural Resource Management Plan, Salt Lake City, Utah.
3. **Kluever, B. M.**, and E. M. Gese. 2010. Research update: water manipulation for kit fox and coyote project on Dugway Proving Ground. Annual meeting, DoD Integrated Natural Resource Management Plan, Salt Lake City, Utah.
4. **Kluever, B.M.** 2006. Learning from cattle-carnivore interactions in Arizona. Annual Arizona Wildlife Services Annual Retreat.

RELATED PROFESSIONAL EXPERIENCE AND TRAINING

2014	Wildland Resources Department Representative, College of Natural Resources Graduate Student Council. USU
2012	Wildlife Capture and Handling Workshop. USU
2009-2013	Volunteer, 3,581 hours. USDA APHIS Wildlife Services
2009	Hantavirus Safety Training. USU
2002	Secretary, Wildlife Club. University of Nebraska

MEDIA

2010	Contributor, Wild About Utah - Utah's Desert Fox - 2010 http://www.bridgerlandaudubon.org/wildaboututah/
2010	Featured Professional, The Leonardo - Faces of Biotechnology http://blog.theleonardo.org/search/label/STEM%20Works http://www.youtube.com/watch?v=jA70TlV2bbo

2010	Interviewee, Utah Public Radio Science Questions -The Infamous Coyote http://sqradio.org/

SERVICE & AFFILIATIONS

Reviewer	Journal of Wildlife Management
Reviewer	Journal of Ecology & the Environment
Reviewer	International Journal of Zoology
Reviewer	Behavioural Processes
Reviewer	PLOS ONE
Member	The Wildlife Society
Member	American Society of Mammalogists