

MEXICAN GRAY WOLVES AND THE ECOLOGY OF FEAR: A COMPARATIVE
ASSESSMENT OF COMMUNITY ASSEMBLAGES IN ARIZONA

by

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DEDICATION

This master's thesis is dedicated to two species in particular – the Mexican gray wolf (*Canis lupus baileyi*), and the coyote (*Canis latrans*). These two species have been subjugated to the intense pressure of environmental anthropocentrism, in which the wishes of humans were violently placed onto them and other predators. This anthropocentrism continues, although not as violently, and thus hopefully one day wolves, coyotes, humans, and other predators will be able to live in ecological balance.

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ABSTRACT

The dramatic ecological influence of northwestern gray wolves (*Canis lupus occidentalis*) on the landscape of Yellowstone National Park, termed a trophic cascade, has been well-documented and a marvel of reintroduction efforts of apex predators in the United States. These wolves have initiated both a traditional trophic cascade, which spans across trophic levels (predator-prey-vegetation), and a carnivore cascade, which spans across a predator guild (wolf-coyote-fox). The present study asks whether or not the Mexican gray wolf (*Canis lupus baileyi*) can influence the distribution of coyotes, gray foxes, elk, and mule deer in the Blue Range Wolf Recovery Area (BRWRA). This was completed through means of a camera trap survey, with two experimental plots within the core home range of Mexican gray wolves in the BRWRA (i.e. the “north” and “south” plots) and one plot west of the BRWRA (i.e. the “west” plot) without sustained Mexican gray wolf activity. After the camera survey, detection rates for the coyotes, gray foxes, elk, and mule deer were calculated. Through both a generalized linear mixed modelling (GLMM) approach, and a Kruskal-Wallis analysis, detection rates were compared across the north, south, and west plots. No statistically significant differences were found in detection rates between the plots, which means there is no indication that the Mexican wolves are influencing the distribution of coyotes, gray foxes, elk, or mule deer in the region. However, since the Mexican wolf population in Arizona is heavily managed, as well as issues with missing data in the camera survey itself, it would be unwise to make broad claims about the ecological implications of Mexican gray wolf reintroduction from this study.

INTRODUCTION

Apex Predators, Trophic Cascades, and Non-Trophic Cascades

Apex predators, or those species that occupy the highest trophic level in a given environment, have been repeatedly shown to have strong effects upon their respective communities and ecosystems (Ripple et al 2014b). Attributes that are frequently ascribed to these apex predators are: body size of larger than 18-34 kg, K-selected reproductive strategy, cooperative social behaviors, “family planning” (e.g. female reproductive suppression and infanticide), and territoriality; these characteristics facilitate apex predators to intrinsically regulate their own population sizes (Wallach et al 2015). Whilst having this ability, apex predators also hold a keystone role in communities and ecosystems by extrinsically regulating the population sizes and densities of prey species as well as other species within the predator guild; this property can influence the occurrence of an ecological phenomenon known as an ecological cascade.

Ecological cascades are defined as reciprocal effects that alter the abundance, biomass, or productivity of a population, community, or trophic level across more than one link in a food web (Pace et al 1999). Two broad categories of ecological cascades occur – trophic cascades and non-trophic cascades. Trophic cascades are those ecological cascades that occur across trophic interactions, or those interactions that involve the consumption of one species by another (i.e. predator-prey, plant-herbivore, etc), whereas non-trophic cascades would be defined as ecological cascades that occur across other types of species interactions, such as mutualism and competition (O’Connor et al 2012, Sanders et al 2013). For this discussion, it is important to note the different types of ecological cascades, as apex predators have the propensity to initiate ecological cascades across trophic levels (e.g. predator-herbivore-vegetation) or across non-trophic levels, such as within a predator guild (e.g. wolf-coyote-fox).

Ecological cascades have two mechanisms of operation, both integral to the overall cascade effect: N-driven cascades and μ -driven cascades. In N-driven ecological cascades the apex predator can directly influence the census population size of organisms. In μ -driven ecological cascades the apex predators can influence the foraging behavior of either prey species or other carnivores within their guild and thus more

indirectly influence the population density and distribution across a landscape (Brown, Laundre, and Gurung 1999). It is namely this last mechanism that can be more important in terms of optimal foraging strategy of either other carnivores or prey species, and this instills a “landscape of fear” for these species in which these species incorporate predation risk into their optimal foraging strategies (Brown, Laundre and Gurung 1999; Ripple and Beschta 2004a). Both of these mechanisms have been demonstrated to be equally important, with the μ -driven mechanism perhaps being stronger due to predation risk factor being incorporated into the optimal foraging strategy with the presence of an apex predator (Lima and Dill 1990; Schmitz, Beckerman, and O’Brien 1997).

When these apex predators are missing from their communities or ecosystems, either through natural extinction or anthropogenic local extirpation, irruptions of herbivores likely result, which would increase browsing rates and cause trophic downgrading of local environments (Estes et al 2011). This would also cause the release of mesopredators to potentially assume an “apex predator-like” role in the environment, although in some cases the mesopredator cannot attain the apex predator ecological niche (Ritchie et. al 2009; Ripple et. al 2013; Wallach et al 2015; Jones et. al 2016). Thus, the importance of maintaining the presence of apex predators within communities and ecosystems is to keep the ecosystems stable and prevent trophic downgrading.

Case Study: Gray Wolves (*Canis lupus occidentalis*) and Yellowstone National Park

Perhaps one of the most well-known apex predators in the United States is the gray wolf (*Canis lupus sbspp.*). The gray wolf was persecuted during the early 1900s due to ubiquitous misunderstanding and fear of the wolf, as well as the perceived threat of livestock depredation. Following this fear, misunderstanding, and anthropocentrism, the United States Congress established the United States Bureau of Biological Survey with the Division of Predator and Rodent Control (PARC) in 1915 which had expressed the purpose of eliminating the wolves and other large predators from the United States (Brown 1983; Mech and Boitani 2003).

As a result, not only was trophic downgrading caused due to increased browsing rate by elk (*Cervus canadensis*) and deer (*Odocoileus virginianus* and *Odocoileus hemionus*), but the tide was turning as American culture gained a more favorable opinion of wolves (Mech and Boitani 2003). As such, wolf conservation programs were initiated and wolves (*Canis lupus occidentalis*) were captured in Canada (Chambers et al 2012) and released into Yellowstone National Park in 1995 and 1996 (Merkle et al 2009). Research following the reintroduction of gray wolves into Yellowstone National Park studied how the wolves affected lower trophic levels and the abiotic environment (i.e. trophic cascade), and also included carnivore cascades (i.e. non-trophic cascade).

Researchers attempted to find evidence for occurrence of a trophic cascade in Yellowstone National Park resulting from wolf reintroduction. Ripple and Larsen(2000) documented a historic trend of aspen (*Populus tremuloides*) recruitment in Yellowstone National Park and found that aspen recruitment was diminished during the years of wolf extirpation from Yellowstone National Park, and the age class that would have been recruited during this time was missing (Larsen and Ripple 2003). Following the wolf reintroduction, studies found that reintroduction of the gray wolf contributed to increased recruitment of aspen (Ripple et al 2001), cottonwoods (*Populus spp.*) (Beschta 2003), willow (*Salix spp.*) (Ripple and Beschta 2004b), and alders (*Alnus incana tenuifoli*) (Ripple, Beschta, and Painter 2015). This trend of increased recruitment is still maintained 15 years after the initial wolf reintroduction into Yellowstone National Park (Ripple and Beschta 2012). In addition, studies have found that due to the increase tree recruitment the morphology of Yellowstone rivers have changed (Beschta and Ripple 2006; Beschta and Ripple 2012), leading to the inference that wolves had an effect on the abiotic environment of Yellowstone National Park. Thus, in terms of trophic cascades, the northwestern gray wolf effect on the Yellowstone National Park ecosystem has been quite remarkable.

This trophic cascade in Yellowstone National Park resulting from wolf reintroduction was primarily a μ -driven cascade; that is to say, it was primarily behavior-driven. Through reintroducing the wolf back into the environment, the elk (*Cervus canadensis*) changed its optimal foraging strategy by

incorporating the risk of predation and thus leaving “plant refugia” in locales that have high predation risk (Ripple and Beschta 2004a). It is these “plant refugia” locales where the greatest amount of tree recruitment (aspen, cottonwood, willow, and alder) occurred within Yellowstone National Park. Therefore, the major mechanism of trophic cascade occurrence was not through population reduction, but rather through behavior modifications that redistributed the density of elk post-reintroduction (Ripple and Beschta 2004a).

Carnivore Cascades

In addition to the remarkable trophic cascades, carnivore cascades occur within the carnivore guild that lives in sympatry with the apex predator (e.g. wolf-coyote-fox). Studies have shown that the mechanisms are the same except behavior modification is achieved by competitors of the same guild rather than at different trophic levels. For example, in Yellowstone National Park as a result of the wolf reintroduction, interference competition between coyotes (*Canis latrans*) and the northwestern gray wolves (*Canis lupus occidentalis*) causes a change, not necessarily in the population sizes of the coyotes, but in the population density and abundance of coyotes in a particular locale (Berger and Gese 2007; Berger, Gese, and Berger 2008; Merkel et al 2009).

These systems influence the distribution and abundance of coyotes, but the wolves can also extend their influence to other systems. For instance, by influencing the distribution and abundance of coyotes, wolves can affect pronghorn distributions (Berger, Gese, and Berger 2008), allow for irruptions in small mammal populations such as rodents (Miller et al 2012), and influence the distribution of foxes (*Vulpes spp.* and *Urocyon cinereoargenteus*) at the local scale (Levi and Wilmers 2012) and the continental scale (Newsome and Ripple 2014). Wolves can also allow recovery of grizzly bear (*Ursus arctos*) populations by limiting browsing of berry plants by elk (Ripple et al 2014a). Therefore, wolf reintroduction could assist in the recovery and conservation of other predator species by proxy. It is important to note that wolves involved in the carnivore cascades discussed here were *Canis lupus occidentalis*, one of the larger wolf subspecies.

Mexican Gray Wolves in Arizona and New Mexico

Mexican gray wolves (*Canis lupus baileyi*), the smallest subspecies of gray wolf, was historically distributed in the American Southwest and Mexico (Brown 1983; Beschta and Ripple 2010; Hendricks et al 2016). Beschta and Ripple (2010) was the only study to examine whether or not Mexican gray wolves initiated a trophic cascade after their reintroduction to the Apache-Sitgreaves National Forest in 1998. They analyzed aspen recruitment in the BRWRA in the Apache National Forest near Alpine, Arizona in three locales: a refugium site inaccessible to both elk and cattle, an old growth site accessible to elk, and a thinned site accessible to elk. They found aspen in the refugium site was consistent with climactic variation, and no significant difference from what aspen recruitment was expected from climactic variation post-reintroduction of the Mexican gray wolf. As this was contradictory to what Ripple and Beschta observed in previous studies with wolves in Yellowstone National Park, they concluded there was no trophic cascade occurring in the BRWRA of 2010.

Curiously, there have been no studies of whether or not N-driven or μ -driven cascades have occurred with regard to the Mexican gray wolf. In other words, there have been no studies on whether Mexican gray wolves have influenced either the population size or population density of other sympatric carnivores such as coyote or fox. In addition, there have been no studies on the interactive behavior between Mexican gray wolf and these sympatric canids. If the Mexican gray wolf can influence the population size and/or distribution of coyote and fox, it would be important to estimate whether or not Mexican gray wolves have established an ecologically effective density to initiate a carnivore cascade (Soulé et al 1999). In addition, data used in the Beschta and Ripple (2010) study is approaching 10 years old, and updated information to reassess the question of Mexican gray wolf involvement in any ecological cascades would be useful.

The Present Study

For the present study, the primary objective is to estimate whether or not the Mexican gray wolf is influencing the distribution of prey species, such as Rocky Mountain Elk (*Cervus canadensis*), mule deer (*Odocoileus hemionus*), as well as competitor species, specifically coyote (*Canis latrans*) and gray fox (*Urocyon cinereoargenteus*) in the BRWRA. In concordance with previous studies, the hypothesis of this study is that Mexican wolves have influenced the distribution of both prey and competitor species, and therefore we predict that areas without Mexican gray wolves would have a higher abundance of prey and competitor species than areas with Mexican gray wolves.

METHODS

Study Site Description

The present study was conducted within the confines of the BRWRA and adjacent areas, which are all within the larger Mexican Wolf Experimental Population Area (MWEPA). The MWEPA is defined by the geographical area that encompasses the states of Arizona and New Mexico, bounded by Interstate 40 (I-40) to the north and the international United States-Mexico border to the south (US Fish and Wildlife Service, 2017). The MWEPA itself contains a variety of biotic communities as defined by Brown and Lowe (1994), ranging from Subalpine Grassland in the White Mountains of Arizona, to the Sonoran Desertscrub that is characteristic of Southern Arizona.

This study primarily dealt with the Rocky Mountain (Petran) Montane Conifer Forest and all cameras are at elevations between 1900m-2900m. This biotic community is semi-ubiquitous across the Mogollon Rim of Arizona – this is a major biotic community that is found within the BRWRA, and as such a majority (sixteen) of camera sites are within this biotic community, specifically located in pure stands of Ponderosa Pine (*Pinus ponderosa*). Other adjacent biotic communities present in the BRWRA are Rocky Mountain (Petran) Subalpine Conifer Forest, a mixed conifer forest including aspen (*Populus tremuloides*), and Subalpine Grassland, and two cameras are placed within these biotic communities (Figure 1). It is useful to mention that not only wolves, but Rocky Mountain Elk (*Cervus canadensis*), mule deer (*Odocoileus hemonius*), coyotes (*Canis latrans*), and gray foxes (*Urocyon cinereoargenteus*) are ubiquitous throughout this area (Anderson and Wallmo 1984, Bekoff 1977, Fritzell and Haroldson 1982, Nowak 1999).

Wildlife Camera Experimental Design

The present study consisted of three plots (two experimental plots with high wolf activity and one control plot with no wolf activity), with six (6) cameras per plot for a total of eighteen (18) camera sites. The experimental plots, hereafter referred to as “north” and “south” plots and within the boundaries of the BRWRA, were in areas of sustained Mexican wolf activity across multiple years. The “north” experimental

plot consisted of: Burro Mountain, Mamie Creek, Roger's Marsh, Rudd Creek, Sherlock Draw, and Sizer Knoll camera sites. The "south" experimental plot consisted of: Crow Poison, Double Cienega, Foote Creek, Highline Trail, Reservation Creek, and Tenney Mountain camera sites. The control plot, hereafter referred to as the "west" plot, was located west of the experimental plots in an area of no Mexican wolf activity. The "west" control plot consisted of: Canyon Creek, Hess Draw, Hog Wash, Potato Field Draw, Upper Cherry Creek, and Willow Creek camera sites. All three plots were within the Apache Sitgreaves National Forest in Arizona, and in the same habitat type (Figure 1).

Remote digital wildlife trail cameras (Covert Deuce) were placed within experimental plots based on Mexican wolf pack home range data for 2014, 2015, and 2016 provided by wolf biologists with the Mexican Wolf Interagency Field Team (IFT). The home ranges included 90% kernels, which represented areas Mexican wolves occupied 90% of the time, and 50% kernels, which represented areas Mexican wolves occupied 50% of the time (50% kernels represent the "core" home range of each pack and were always within the boundaries of the 90% kernels for a given wolf pack). Home range maps across years were compared, and locations where the 50% kernel for a given pack overlapped across multiple years were candidates for camera placements for the experimental plots. The same home range maps were referred to when deciding the camera locations for the control plot. Areas that were excluded from any 90% kernel of any given Mexican wolf pack over multiple years were candidates for the control plot. Once candidate locations were identified, and since the wolf biologists of the Mexican Wolf IFT knew each of the locations with certainty, the wolf biologists decided on exact locations for wildlife cameras in all three plots and provided GPS coordinates for each camera.

Camera settings were: 1 photo per event, 5MP photo size, 1 minute interval between photos, normal PIR sensitivity, xenon flash for night photos, and normal range for flash unless a particular camera site required adjustment in PIR sensitivity or flash range.

Each of these cameras were baited by placing scent lure (Mark June Cherry Red commercial bait) approximately 5m from the camera at ground height each time the camera was checked. This was done to maximize the number of carnivores that walked in front of the camera. Each camera was checked by the wolf biologists of the Mexican wolf IFT approximately every 2-2.5 months, for a total of one year, starting approximately December 1, 2016, and ending by November 30, 2017. In this way, data was collected for each off our seasons and for one full year.

Data Analyses

Analyses of this data was performed in two parts. First, the photos downloaded from each camera were sorted using a hierarchical method as described in Sanderson and Harris (2013). This hierarchical method involved sorting photos by plot (i.e. north, south, or west), and within each of these plots into location folders. There was a location folder for each camera, and within each location folder was a folder for each species photographed. Following photo sorting, the total number of photographs captured at each location, trap-nights (i.e. camera trap effort) for each camera location, number of independent photos of each species captured, and detection rates of each species captured were calculated with an algorithm in the program (Sanderson and Harris 2013). Photographs were considered independent if 60 minutes passed without the camera capturing another photograph of the same species – if there was another photo of the same species within that 60 minute interval, it was discounted. For further analyses, photographs that contained no species (i.e. “ghost” photographs), or photographs that contained domestic species (e.g. domestic dog *Canis lupus familiaris*, domestic cow *Bos taurus*, or humans *Homo sapiens*), were discounted. In this way, the number of independent photographs of wildlife was calculated. The independent photographs of all wildlife species, which included the independent photographs of the species of interest (i.e. coyotes, gray foxes, elk, and mule deer), were used for further statistical analyses, which consisted of analyzing differences in detection rates for species of interest between plots.

Statistical analyses were performed in two ways to provide multiple measures to determine significance for detection rates between plots. First, generalized linear mixed modelling analyses were completed using a GLMM package in the statistical program R (Knudson 2017). Secondary analyses were the Shapiro-Wilk normality test, the Kruskal-Wallis test, and Conover's test of multiple comparisons with a Holm-Bonferroni correction using the PMCMR package in the statistical program R (Thompson et al 2010; Bates et al 2011; Pohlert 2014; Li et al 2014; Gustavo Pinoargote, personal communications).

RESULTS

Results of Camera Trap Analysis

In the north plot, there were a total of 10,034 photographs captured, 313 of which were independent photographs of wildlife species, across a total of 1,679 trap nights. In the south plot, there were a total of 7,342 photographs captured, 295 of which were independent photographs of wildlife species, across a total of 1056 trap nights. Lastly, in the west (control) plot, there were a total of 21,444 photographs captured, 616 of which were independent photographs of wildlife species, across a total of 1,782 trap nights. This results in a total of 38,820 total photographs captured across all 18 camera locations, 1,224 of which were independent photographs of wildlife species, across a total of 4,517 trap nights (Table 1, Table 2, Table 3).

Further analyses were completed by the software program in Sanderson and Harris (2013). These analyses calculated detection rates of each species (i.e. coyotes, gray fox, elk, and mule deer) by dividing species abundance (i.e. the number of photos of that species) by camera trap effort (i.e. trap-nights). In this way, we are able to analyze the abundance of each species per unit camera effort. In the north plot, the average detection rate was 1.44 coyotes/unit effort for coyotes, 0.05 gray foxes/unit effort, 12.52 elk/unit effort, and 3.16 mule deer/unit effort. In the south plot, the average detection rate was 0.788 coyotes/unit effort, 0.16 gray foxes/unit effort, 24.03 elk/unit effort, and 5.05 mule deer/unit effort. Lastly, for the west (control) plot, the average detection rate was 5.38 coyotes/unit effort, 0.97 gray foxes/unit effort, 14.01 elk/unit effort, and 8.95 mule deer/unit effort (Table 4). These detection rates were then statistically analyzed.

Results of Statistical Analysis

Statistical analyses consisted of first using a generalized linear mixed modelling (GLMM) approach to test for significant differences in the detection rates amongst the north, south, and west plots. The GLMM analyses for coyotes ($p = 0.604$), gray foxes ($p = 0.259$), elk ($p = 0.480$), and mule deer ($p = 0.270$) all resulted in non-significant differences between the three plots for any species of interest.

Supplementary statistical analyses were performed to provide additional measures for significant differences between the three plots. A Shapiro-Wilk normality test was performed to see if the distribution of detection rates was normally or non-normally distributed, and the distributions of detection rates for coyotes ($p = 1.402e-07$), gray foxes ($p = 2.33e-07$), elk ($p = 0.0002399$), and mule deer ($p = 0.002226$) were all determined to be non-normal due to all p-values being below the threshold of 0.05. Next, a Kruskal-Wallis test was performed for each species, and the differences between plots for coyote ($p = 0.2686$), gray fox ($p = 0.2668$), elk ($p = 0.8948$), and mule deer ($p = 0.1923$) were all determined to be non-significant. Lastly, a Conover's test for multiple comparisons, with Holm-Bonferroni corrections, was performed between each plot for each species of interest to provide a post-hoc analysis of each pairwise comparison. For coyote, the north-west ($p = 0.64$), south-west ($p = 0.37$), and north-south ($p = 0.64$) pairwise comparisons all resulted in non-significant differences between plots. For gray fox, the north-west ($p = 0.36$), south-west ($p = 0.65$), and north-south ($p = 0.65$) pairwise comparisons all resulted in non-significant differences between plots. For elk, the north-west ($p = 1.00$), south-west ($p = 1.00$), and north-south ($p = 1.00$) pairwise comparisons all resulted in non-significant differences between plots. Lastly, for mule deer, the north-west ($p = 0.24$), south-west ($p = 0.50$), and north-south ($p = 0.51$) pairwise comparisons all resulted in non-significant differences between plots.

DISCUSSION

According to these analyses, it is a reasonable suggestion that Mexican gray wolves have yet to be successful in achieving their ecological role as top predator in the BRWRA. This conclusion is reached by the lack of significant difference of detection rates between plots with Mexican gray wolves (as measured by overlap of 50% kernels, which represent the sustained presence of a core home range for one or multiple packs of Mexican gray wolves) compared to plots without Mexican gray wolves. These non-significant differences were also true for coyotes, gray foxes, elk, and mule deer. From statistical analyses, another reasonable suggestion is that Mexican gray wolves do not affect the distribution of coyotes, elk, or mule deer. Given that the Mexican gray wolves have been consistently been in the area for 20 years, there is some question of why they have not yet achieved their ecological role as top predator. This result is contradictory to the patterns observed in Yellowstone National Park and other areas of wolf reintroduction, yet the Mexican wolves not having an effect on ecological cascades was observed with prior data collected for trophic cascade (Beschta and Ripple 2010). Though, this relationship between the north, south, and west plots only implies a correlation, not a causation, between wolf presence/absence and detection rates of elk, mule deer, coyote, and gray fox. From this study, we can say that there is no relationship between wolf presence/absence and the detection rates of the species of interest, but we cannot say that the lack of significant difference in detection rates between the plots was because of wolf presence/absence. This lack of significant difference between the plots may be due to a variety of factors.

One plausible explanation for the lack of significance may be due to the Mexican gray wolf population size and range. In 2016, the Mexican gray wolf population was estimated to be around 113 wolves occupying 41,735 km², primarily in New Mexico (US Fish and Wildlife Service, 2016) whereas the wolf population of approximately the same size in Yellowstone National Park occupied 8989 km² (Smith et al 2017). For a landscape of fear to be instituted in coyotes, gray foxes, elk, and mule deer, these species would have to encounter the Mexican wolves, in either a competitive or preyed upon manner, often enough for the Mexican wolves to be incorporated into each species' optimal foraging strategy (Lima and Dill 1990,

Brown, Laundré, and Gurung 1999) as happened in the Yellowstone National Park system (Ripple and Beschta 2004a). If these species were to encounter the Mexican gray wolf on a consistent basis, it would be reasonable to assume a modified distribution of these species, as measured by detection rate, such that a lower detection rate is observed for coyotes, elk, and mule deer, but a higher detection rate for gray foxes, in areas consistently occupied by Mexican wolves. If, however, a Mexican gray wolf is encountered only occasionally, not consistently, then the risk of predation/competition by the Mexican gray wolf would not be incorporated into the species' optimal foraging strategy. Therefore, in this case, we should not see differences in distribution of these species as measured by detection rate. According to our analyses there is no statistically significant difference in detection rate between plots with Mexican wolves and plots without, thus we suggest that a landscape of fear has not been instituted in the Blue Range Wolf Recovery Area.

Another reasonable explanation for the lack of significance is the reintroduced habitat itself. Brown (1983) states that the Mexican gray wolves historically have been associated with montane forests and woodlands within vegetation such as oak woodlands, pinyon-juniper forests, and adjacent grasslands while avoiding the Mohave, Sonoran, and Chihuahuan deserts and adjacent semi-desert grasslands. These habitats would be consistent with the habitats of Madrean Evergreen Woodland, Interior Chaparral and Montane Grasslands of Brown and Lowe (1994), which are found in the Sky Islands of Southern Arizona and New Mexico and extends well into the Sierra Madre Occidental and Sierra Madre Oriental mountain ranges of Mexico. Historically, there was perhaps another subspecies of wolf (*Canis lupus youngi* or *Canis lupus mogollonensis*) that occupied Petran Montane Conifer Forests of the Blue Range Wolf Recovery Area (Brown 1983). Both Brown (1983) and Hendricks (2016) support the notion that current territory occupied by the reintroduced Mexican gray wolves would have been the extreme farthest north of the historical distribution. In addition, according to Brown (1983) the presumed main prey item of the Mexican wolf would have been Coues' White-tailed deer (*Odocoileus virginianus couesi*) and possibly the mule deer (*Odocoileus hemionus*), not elk (*Cervus canadensis*). Thus, the lack of significance could be partially due

to the reintroduction effort being in non-historical habitat, which may affect hunting behavior, as well as presumably a non-historical prey base.

It is also worth mentioning that as part of the management plan of the Mexican wolf, supplementary food caches (i.e. road-killed native prey carcasses provided to wolf packs to assist in feeding young) and diversionary food caches (i.e. road-killed native prey carcasses provided to wolf packs to reduce potential conflicts with livestock) are provided to aid the Mexican wolves (US Fish and Wildlife Service, 2016). This provision of food caches may affect the hunting behavior of the wolves, and thus may have some effect on the ecological implications of the reintroduction effort. Though, this has not been directly studied, and so an ecological study needs to be done to assess the ecological implications of these food caches.

In any case, the Mexican gray wolf could be argued to have no significant effect on coyotes, gray foxes, elk, or mule deer for any of these reasons, or for reasons that we have not considered here. Though, due to these concerns, one should be cautious about making such claims, as this study only indicates that Mexican gray wolves have yet to achieve their top predator status. The present study is not complete in its assessment of the ecological implications of the reintroduction efforts of the Mexican wolves and further examination of a variety of questions need to be addressed before making any kind of claim as to the ecological implications of the Mexican wolf.

These further examinations of effects could take a variety of forms. First, there could be further analyses on the effects across the prey guild, which would include the prey base of not only the Mexican wolves (e.g. ungulates), but also the prey base of competitors (e.g. coyotes, pumas), as well as a vegetation assessment to analyze bottom-up trophic effects. Second, there should be a more complete analysis on other potential prey items of Mexican wolves, rather than solely focusing on elk and mule deer. One notable instance that would be necessary to look at would be the effects that Mexican gray wolves have on pronghorn (*Antilocapra americana*), as they were detected at three cameras: Hess Draw (1.32 pronghorn/unit effort), Potato Field Draw (3.21 pronghorn/unit effort), and Willow Creek (0.36 pronghorn/unit effort), which results in an average detection rate of 0.79 pronghorn/unit effort. Pronghorn

were only found in the west plot, and thus were not able to be analyzed in the present study. Though, pronghorn are known to be ubiquitous both historically and currently (Hall and Kelson 1959; Nowak 1999), and so it would be interesting to look at potential reasons as to why no pronghorn were detected at all within the 50% kernels of the Mexican gray wolf home range.

In addition to these assessments, it would be ideal to compare the results of this study to that of historical records and other Mexican gray wolf reintroduction efforts. For the historical assessment, it would be crucial to know how the Mexican wolf has historically affected the distribution of coyotes, gray foxes, mule deer, and white-tailed deer (*Odocoileus virginianus*) through trapping records (Brown 1983) in the American Southwest and Mexico. In this way, we would develop a metric by which to compare current distributions of these species of interest to their historical distributions. As for other Mexican gray wolf reintroduction sites, through a binational effort there were Mexican wolves reintroduced in the northern Sierra Madre Occidental of Chihuahua and Sonora in Mexico, and 41 wolves have been reintroduced in this area. As of 2017, these wolves are still in the establishment phase and approximately 31 wolves inhabit the northern Sierra Madre Occidental (US Fish and Wildlife, 2017). With this particular reintroduction site being in historical Mexican wolf territory (Brown 1983, Hendricks et al 2016), this would be perhaps a more suitable location for studies into the ecological effects of Mexican wolves. After all, this is the similar habitat that Roy T. McBride trapped the seven Mexican wolves in 1980 which founded the captive breeding program the ancestors to all modern Mexican wolves (Brown 1983).

In any case, whether it's continuing this study in the northern Sierra Madre Occidental in Mexico, to analyzing historical records in both Mexico and the American Southwest, to broadening the scope and looking at other competitors, prey bases, and vegetation assessments, there are far more questions to pursue before a proper conclusion on the ecological impact of the Mexican wolf reintroduction can be obtained.

FIGURE

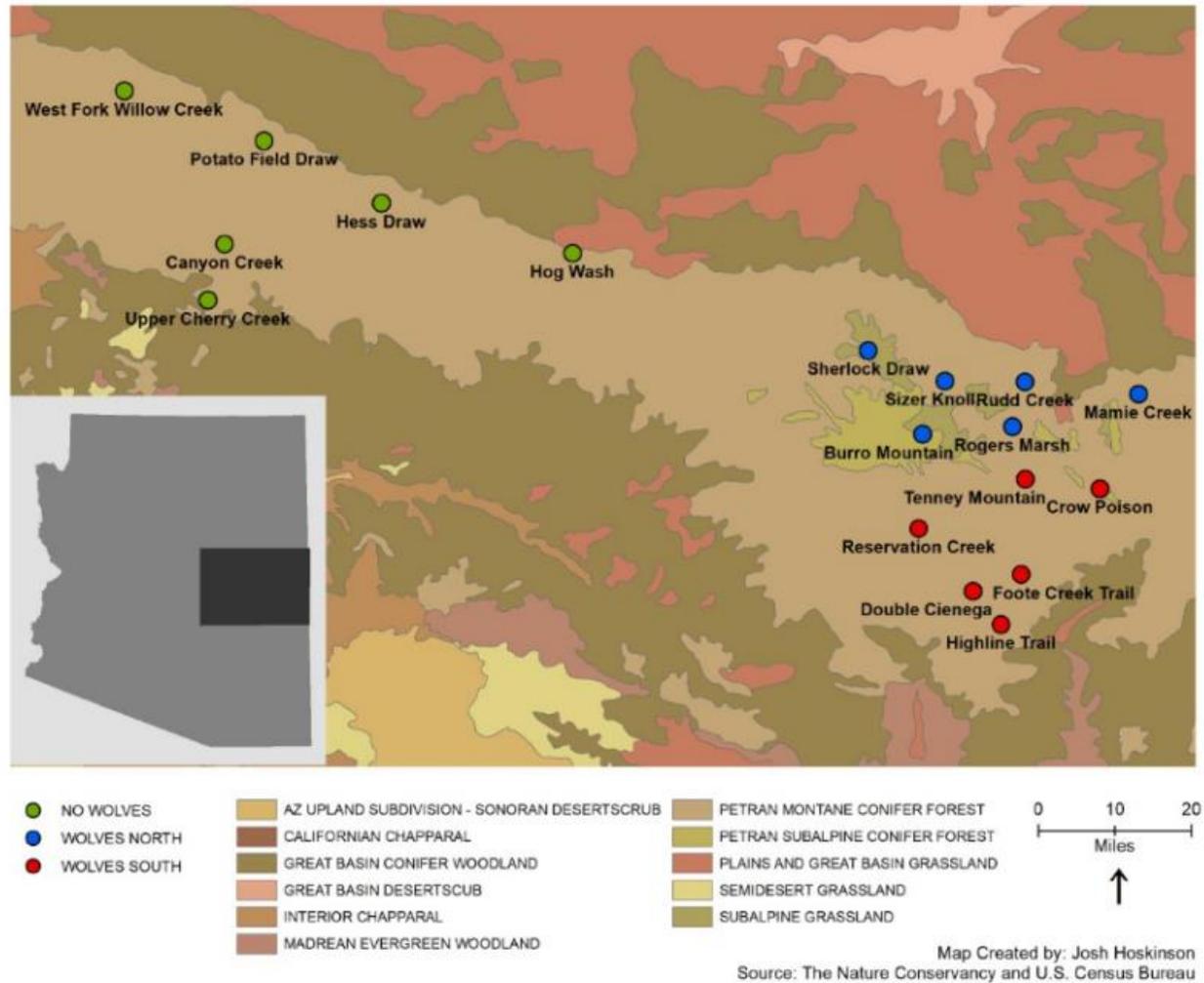


Figure 1. Map of camera locations overlaid onto the biotic communities map from Brown and Lowe (1994). The study area in relation to the entire state of Arizona is depicted in the insert map in the bottom left hand corner. Legend includes the biotic communities, camera locations for the no wolves plot (i.e. west plot) labeled in green, the north plot labeled in blue, and the south plot labeled in red.

TABLES

Table 1. North study plot wildlife pictures. Description of the start/end dates, number of trap-nights, the total number of pictures, and the total number of independent pictures of wildlife for the north plot. Four of the six cameras were down for a period of time, and so each start/end date for each period that the camera was actively capturing photographs is described. Trap-nights, or camera effort, are the number of 24 hour periods that the camera was actively capturing photographs. Total number of pictures includes all of the photographs obtained by that particular camera. The total number of independent pictures of wildlife excludes multiple photographs of the same species taken within a 60-minute period, photographs with no species (i.e. “ghost” photos), and photos of domesticated species (e.g. domestic dogs, domestic cows, and humans). This column includes, but isn’t limited to, the species of interest.

Location	Start Date	End Date	Trap Nights	Total Number of Pictures	Total Number of Independent Pictures of Wildlife
Burro Mountain	1. 11/6/2016 2. 7/5/2017	1. 4/19/2017 2. 8/10/2017	268	908	22
Mamie Creek	1. 11/15/2016 2. 6/8/2017	1. 5/15/2017 2. 10/12/2017	332	3155	43
Roger’s Marsh	1. 11/5/2016 2. 7/3/2017	1. 3/28/2017 2. 10/26/2017	349	2539	49
Rudd Creek	11/25/2016	6/24/2017	97	113	46
Sherlock Draw	11/6/2016	10/16/2016	344	1688	91
Sizer Knoll	1. 11/6/2016 2. 8/1/2017	1. 4/15/2017 2. 8/18/2017	289	1634	62
Total	-	-	1679	10034	313

Table 2. South study plot wildlife pictures. Description of the start/end dates, number of trap-nights, the total number of pictures, and the total number of independent pictures of wildlife for the south plot. Two of the six cameras were down for a period of time, and so each start/end date for each period that the camera was actively capturing photographs is described. Trap-nights, or camera effort, are the number of 24 hour periods that the camera was actively capturing photographs. Total number of pictures includes all of the photographs obtained by that particular camera. The total number of independent pictures of wildlife excludes multiple photographs of the same species taken within a 60-minute period, photographs with no species (i.e. “ghost” photos), and photos of domesticated species (e.g. domestic dogs, domestic cows, and humans). This column includes all wildlife photos, not just the species of interest.

Location	Start Date	End Date	Trap Nights	Total Number of Pictures	Total Number of Independent Pictures of Wildlife
Crow Poison	1. 1/13/2017 2. 7/26/2017	1. 3/1/2017 2. 8/27/2017	178	215	11
Double Cienega	11/5/2016	7/26/2017	257	527	167
Foot Creek	11/16/2016	5/19/2017	179	3774	73
Highline Trail	11/22/2016	6/23/2017	157	2260	14
Reservation Creek	1. 11/30/2016 2. 7/26/2017	1. 3/24/2017 2. 9/9/2017	282	514	28
Tenney Mountain	11/30/2016	1/13/2017	3 [†]	52	2
Total	-	-	1056	7384	295

† There was a malfunction in the camera at Tenney Mountain, South Plot. Thus, even though it was out longer than three nights, it only had pictures for three nights.

Table 3. West study plot wildlife pictures. Description of the start/end dates, number of trap-nights, the total number of pictures, and the total number of independent pictures of wildlife for the west plot. Three of the six cameras were down for a period of time, and so each start/end date for each period that the camera was actively capturing photographs is described. Trap-nights, or camera effort, are the number of 24 hour periods that the camera was actively capturing photographs. Total number of pictures includes all of the photographs obtained by that particular camera. The total number of independent pictures of wildlife excludes multiple photographs of the same species taken within a 60-minute period, photographs with no species (i.e. “ghost” photos), and photos of domesticated species (e.g. domestic dogs, domestic cows, and humans). This column includes all wildlife photos, not just the species of interest.

Location	Start Date	End Date	Trap Nights	Total Number of Pictures	Total Number of Independent Pictures of Wildlife
Canyon Creek	1. 2/16/2017 2. 6/9/2017	1. 5/23/2017 2. 10/5/2017	232	6065	193
Hess Draw	1. 11/27/16 2. 6/8/2017	1. 5/18/2017 2. 10/13/2017	303	2149	102
Hog Wash	1. 11/30/2016 2. 6/8/2017	1. 5/15/2017 2. 11/2/2017	332	3159	101
Potato Field Draw	11/27/2016	11/9/2017	312	767	80
Upper Cherry Creek	11/26/2016	11/10/2017	302	2849	45
Willow Creek	11/26/2016	9/23/2017	301	6455	95
Total			1782	21444	616

Table 4. Description of the camera effort and the detection rates for coyotes, gray foxes, elk, and mule deer for each of the locations at each of the study plots. Camera effort, or trap-nights, is the number of 24 hour periods that the camera was actively capturing photographs. Detection rate is the number of photographs of that species, at that location, per unit camera effort.

Plot	Location	Camera Effort	Coyote Detection Rate	Gray Fox Detection Rate	Elk Detection Rate	Mule Deer Detection Rate
North	Burro Mountain	268	0.00	0.00	7.46	0.00
	Mamie Creek	332	0.90	0.00	10.84	0.00
	Roger's Marsh	349	0.29	0.29	12.61	0.29
	Rudd Creek	97	4.12	0.00	17.53	13.40
	Sherlock Draw	344	2.62	0.00	10.76	1.45
	Sizer Knoll	289	0.69	0.00	15.92	3.81
South	Crow Poison	178	1.69	0.00	3.93	0.00
	Double Cienega	257	2.33	0.39	52.53	2.33
	Foot Creek	179	0.00	0.56	15.08	7.26
	Highline Trail	157	0.00	0.00	0.64	0.00
	Reservation Creek	282	0.71	0.00	5.32	0.00
	Tenney Mountain	3	0.00	0.00	66.67	0.00
West	Canyon Creek	232	25.43	0.86	23.28	18.1
	Hess Draw	303	1.32	0.00	23.1	6.93
	Hog Wash	332	1.51	0.00	7.23	20.48
	Potato Field Draw	312	1.28	0.00	18.27	0.32
	Upper Cherry Creek	302	1.32	0.66	6.62	4.3
	Willow Creek	301	1.44	4.31	5.57	3.59

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