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Influence of Landscape Factors on Wildlife Presence and Road Mitigation Structure Performance

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INFLUENCE OF LANDSCAPE FACTORS ON WILDLIFE PRESENCE

AND ROAD MITIGATION STRUCTURE PERFORMANCE

A Thesis

by

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ABSTRACT

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There are roughly 80 ocelots (*Leopardus pardalis*) remaining in the United States, with the entire population constrained to south Texas, with roadkill being a predominant source of mortality. To prevent additional roadkill and maintain wildlife movement, Texas Department of Transportation constructed 11.9 kilometers of wildlife exclusion fencing, 5 wildlife crossing structures (WCS), and 18 wildlife guards on State Highway 100. This thesis focused on determining the effort required for a control-impact monitoring study, the influence of biotic and abiotic factors around the roadway on wildlife presence, and the performance of mitigation structures and the road mitigation corridor. This research shows that control-impact studies are important for road ecology projects and their design strongly influences survey effort. Additionally, felid presence is likely influenced by vegetation and distance to WCS, and will most likely use WCS with a small box-culvert design.

DEDICATION

The completion of my master's studies would not have been possible without the love and support of my friends and family. My mother and father, Keni and Ralph Hopkins, who never stopped caring, and provided the encouragement to pursue my career and apply to graduate school. I wish I could do more than just thank them for the consistent support they have shown over the last three decades. My partner, Denise Bickford, and emotional support pets Kiva, Paria, and Rocket (the best boy), provided a caring home life, without which I could not have maintained the motivation required to graduate. Thank you for your love, support, and patience. Finally, my Dungeons & Dragons group consisting of Bryan McLaughlin, Jaimie Olle, and Levi Heikkila, who refused to split the party and helped push me to the end of my academic quest. I am truly grateful for their weekly dose of bardic inspiration.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

On a worldwide scale, natural habitats are shrinking due to the expansion of human infrastructure into once isolated areas, increasing the anthropogenic footprint (Jakes et al. 2018). Roadways are one of the most common and expansive forms of infrastructure, with more than 14 million lane-kilometers of paved roads within the United States alone (Benitez-Lopez et al. 2010, Andis et al. 2017). The influence of roadways goes well beyond their asphalt surface, Forman (2000) estimated that roads cover 1% of the landmass within the United States, but show some influence on 20% of the surrounding area. Roads and other linear infrastructure have a large impact on biodiversity loss, as they increase wildlife mortality and fragment habitats and ecosystems (Spellerberg 1998, Forman et al. 2003). By fragmenting habitats, roads reduce habitat amount and quality and divide wildlife populations into less viable subpopulations, increasing their likelihood of extinction (Spellerberg 1998, Jaeger and Fahrig 2004b). Once a road is constructed, its negative influence does not remain constant (Benitez-Lopez et al. 2010). Benitez-Lopez et al. (2010) reported a correlation between traffic volume and nearby mammal populations, as traffic increases mammal populations decrease. Therefore, even roads that have a relatively low impact on surrounding wildlife may become increasingly disruptive as traffic patterns increase with urbanization.

The negative effects of roadways may be classified into two categories: direct and indirect (Benitez-Lopez et al. 2010). Direct effects involve physical contact between wildlife and roadways or vehicles (Jaeger and Fahrig 2004b, Litvaitis and Tash 2008, Benitez-Lopez et al. 2010, Livingston 2019). The most well-known and most detrimental direct effects are wildlife-vehicle collisions, which may result in roadkill, human injury, and vehicle damage (Benitez-Lopez et al. 2010, Carvalho and Mira 2011). Within some mammal populations wildlife-vehicle collisions may be responsible for 30-40% of all mortalities (Jaeger and Fahrig 2004b). Beyond these direct effects, indirect effects have a more subtle but widespread influence, such as noise pollution, light pollution, and invasive plant dispersal (Grigione and Mrykalo 2004, Flory and Clay 2006, Brown et al. 2012, Nega et al. 2012, D'Amico et al. 2015, Gaston and Holt 2017). These indirect effects may be observed far beyond the roadway, extending up to five kilometers for certain taxa in sub-optimal habitats (Benitez-Lopez et al. 2010).

This area where populations are influenced by direct and indirect road effects is aptly named the road-effect zone (Forman and Deblinger 2000, Bissonette and Rosa 2009, Clevenger and Huijser 2011). Road-effect zones influence different species in a variety of ways, either discouraging or encouraging congregation around the linear barrier (van der Ree et al. 2007, Shanley and Pyare 2011, Peaden et al. 2016, Mata et al. 2017). However, it seems most species are negatively influenced by road presence (Pocock 2005, Fahrig and Rytwinski 2009). Fahrig and Rytwinski (2009) investigated 79 studies of 131 species and found 114 of those species responded negatively to road presence. Road avoidance may have multiple negative impacts on the long-term viability of wildlife populations, including obstructing movement, fragmenting habitat, and reducing landscape connectivity (Jaeger et al. 2005, Benitez-Lopez et al. 2010, Clevenger and Huijser 2011). Road avoidance may also divide populations into subpopulations, further reducing a species' long-term viability (Jaeger and Fahrig 2004b, Janečka et al. 2008).

Beyer et al. (2016) labeled and described four types of linear impediments to wildlife movement, with distinctions based on whether the impediment may be crossed or circumnavigated. These four distinctions are important, as all four of these impediments may be found along a single stretch of roadway (Beyer et al. 2016). A 'barrier' is an obstacle that may be crossed, but not circumnavigated; while an 'obstacle' can be circumnavigated, but not crossed. An 'impedance' can be crossed *or* circumnavigated, requiring the animal to make a choice of how to cross. Finally, 'constraints' cannot be crossed nor circumnavigated (Beyer et al. 2016).

Mitigating direct effects of roadways on wildlife is relatively simple, as a constraining linear barrier may be placed between wildlife habitat and the road surface (Kenneth Dodd et al. 2004, Allen et al. 2013, Cserkész et al. 2013, Jakes et al. 2018). This usually takes the form of wildlife exclusion fencing which, depending on design and maintenance, can greatly reduce wildlife access to the road surface, effectively removing the direct influence of roadways on wildlife (Clevenger and Waltho 2000, Forman et al. 2003, Craighead et al. 2009, Jakes et al. 2018). Jaeger and Fahrig (2004b) found that effective wildlife exclusion fencing may reduce wildlife access to a road surface by 93%. Though not as effective, wildlife guards may be placed at gaps in the fence, or in areas where vehicles will frequently travel, reducing the effects of gaps in the constraining barrier (Belant et al. 1998, Glista et al. 2009, Allen et al. 2013, Flower 2016). However, the addition of wildlife exclusion fencing increases the negative indirect effects of the roadway because it transforms the porous-but-dangerous linear barrier the road posed pre-construction into a constraint post-construction (Clevenger et al. 2001, Jaeger and Fahrig 2004b, Beyer et al. 2016, Jakes et al. 2018). If the ends of the wildlife exclusion fencing are not within the home ranges of protected wildlife, animals cannot cross to the other side or circumnavigate (Jaeger and Fahrig 2004b, Beyer et al. 2016). This change in the barrier effect increases population persistence as wildlife are no longer struck by vehicles, but may negatively impact species over time by further reducing movement between populations (Jaeger and Fahrig 2004b, Beyer et al. 2016).

Wildlife crossing structures (WCS) are the most popular mitigation method for reducing the movement barrier roadways create, especially in constraining areas with wildlife exclusion fencing (Mata et al. 2003, Bissonette and Adair 2008, Mata et al. 2008, Corlatti et al. 2009, van der Grift et al. 2013). Regardless of design, WCS are meant to provide a safe path for wildlife to cross an unsafe linear barrier (Forman and Alexander 1998, Lesbarreres and Fahrig 2012). Within the United States, most WCS are either large animal bridges that cross over a roadway or culverts that pass below the road surface (van der Grift et al. 2013). Wildlife-proof fencing and WCS are often combined to maximize the effectivness of both mitigation structure types (Clevenger et al. 2001, Dodd et al. 2007, McCollister and van Manen 2010). Fencing may be used to funnel wildlife along roadways toward WCS, increasing the likelihood individuals will choose to cross safely and decreasing the chance they will attempt to pass onto the roadway (McCollister and van Manen 2010).

Due to the expense of WCS, they usually cannot be placed along the entire length of the roadway, and are therefore placed to link patches of critical habitat (Clevenger 2005, Andis et al. 2017). The limiting aspect of WCS expense requires structures to be placed and designed to maximize wildlife use (van der Grift et al. 2013). Placement is extremely important, as animals are more likely to use WCS in areas with more ideal habitat, and might avoid structures in less ideal habitat (Haddad et al. 2003, Beyer et al. 2016, Abrahms et al. 2017, Andis et al. 2017).

Additionally, if animals are unlikely to utilize the area a WCS is placed in, they may never learn of the existence of the WCS on the landscape (Abrahms et al. 2017). Design is equally important, as design may encourage, discourage, or prevent usage by some species (Fahrig and Rytwinski 2009, Beyer et al. 2016). Due to the wide variety of WCS designs, a single structure may serve as a barrier for some species but a constraint to others (McDonald and St Clair 2004, van Vuurde and Van der Grift 2005, Ford and Clevenger 2010, Kintsch and Cramer 2011, Beyer et al. 2016).

The degree to which wildlife utilize a WCS determines the structure's permeability, as well as the permeability of the entire mitigation corridor (Beyer et al. 2016, Andis et al. 2017). Since wildlife exclusion fencing serves as a movement constraint, WCS may serve as the only crossing point for multiple kilometers (Beyer et al. 2016). If they are not designed to maximize visitation and use by target species, a single structure may not be able to fully mitigate the length of road or fencing where it was installed (Andis et al. 2017). Additionally, if an area of wildlife mitigation fencing is long enough with a low frequency of WCS, it may provide a greater ecological barrier than the original roadway before mitigation structure construction (van der Grift et al. 2013, Beyer et al. 2016). To ensure that structures are mitigating habitat fragmentation, it is important to understand the permeability of the road mitigation corridor (van der Ree et al. 2007, van der Grift et al. 2013, Andis 2016, Andis et al. 2017).

Determining permeability may only be done through an empirical study of actual use at the wildlife crossing structures compared to wildlife movement in the surrounding habitat (Andis et al. 2017). Andis et al. (2017), van der Grift et al. (2013), and van der Ree et al. (2015) recommend that a control-impact study is one of the most accurate methods for determining the permeability of a road mitigation corridor. Control-impact studies on road mitigation structures focus on comparing actual crossing rates at the structure (impact) to occupancy observed within the surrounding habitat (control) (Torres et al. 2011, van der Grift et al. 2013, van der Ree et al. 2015). Control-impact studies allow researchers to remove the influence of structure location (placement) on crossing rates, allowing additional studies to explore the impact of structural attributes (design) on wildlife use (Andis et al. 2017).

The negative direct and indirect effects of roadways are magnified on endangered and elusive species, such as the United States ocelot population (*Leopardus pardalis*) (Janečka et al. 2011, van der Ree et al. 2011, Janečka et al. 2016). Elusive species are more likely to show strong road avoidance and often require optimal habitat to maintain their populations, something the United States ocelot population does not have (Boarman and Sazaki 2006, Haines et al. 2006a, Cypher et al. 2009). With the advent of urbanization in south Texas, more than 95% of ocelot rangeland was converted to agricultural and urban land (Haines et al. 2005b). Ocelots were listed as endangered within the United States in 1982, with roughly 80 individuals within the Texas population today (Ascensão et al. 2019). These 80 individuals are dispersed between two separate populations, both of which are confined to south Texas (Haines et al. 2006b). The first is the Willacy County population, which survives on a conservation easement and the second is the Cameron County population, which primarily lives on or near the Laguna Atascosa National Wildlife Refuge (Haines et al. 2006b). Though the two populations are less than 30 kilometers apart, there has been no documented dispersal between the two populations and they are genetically isolated (Haines et al. 2006b, Janečka et al. 2011). Ocelot populations in Cameron County, Willacy County, and Mexico are genetically distinct, indicating isolation between populations, with the Cameron County ocelot population having the lowest genetic diversity

(Grigione et al. 2009, Janečka et al. 2011). It is believed that this separation is due to a loss of connecting habitat and road-mortalities (Haines et al. 2006b).

Wildlife-vehicle collisions accounted for 40% of known Texas ocelot mortalities between 1986 and 2002, and are considered a direct threat to ocelot survival (Haines et al. 2005a, Haines et al. 2006b). To reduce the detrimental effect of busy roadways on the Texas ocelot population, Texas Department of Transportation (TxDOT) created a wildlife road mitigation corridor in eastern Cameron County, Texas (Figure 1). State Highway 100 (SH100) is a four-lane highway extending from Interstate 69E east, serving as the only access to South Padre Island, a popular tourist destination (Transportation 2016). SH100 has an average daily traffic volume of 7,152 vehicles/day, with traffic speed limits posted at 65 miles per hour (Transportation 2016). SH100 was selected for a road mitigation corridor because it crosses the Laguna Atascosa National Wildlife Refuge, has a high traffic volume when compared to other Refuge roads, and was historically a source of mortality for ocelots (Tewes and Hughes 2001). TxDOT constructed 11.9 km of contiguous wildlife exclusion fencing and 18 wildlife guards (WG) to prevent additional ocelot mortalities. To maintain wildlife movement through the SH100 mitigation corridor, TxDOT also constructed or modified five WCS with four distinct designs (Figure 2). The SH100 road mitigation project is focused on ocelots as the primary target species, however, based on suggestions from Clevenger (2005), mitigation structures were also meant to promote ecosystem health. Therefore, all species that may use a wildlife crossing structure were considered secondary target species.

Cogan (2018) conducted research focused on establishing a camera trap array to monitor actual crossing use at the WCS and WGs within the SH100 mitigation corridor. However, a control array had not yet been implemented. The goal of my thesis research was to design and implement a control-impact study for the SH100 mitigation corridor to determine the potential influence of placement on mitigation structure performance. However, studying ocelot presence within the road-effect zone is difficult, as ocelots are not dispersed evenly across the landscape (Harveson et al. 2004). Ocelots in south Texas are specialists that prefer Tamaulipan thornscrub, a dense, thorny, diverse composition of woody plants (Harveson et al. 2004, Gavin and Komers 2006). Any control-impact study would need to focus monitoring within this rare vegetation type, without reducing monitoring effort for any species that may utilize the wildlife crossing structures. Therefore, the second chapter of this thesis explores placement methodology for a control array in south Texas ecosystems. The third chapter determines the influence of environmental factors on wildlife presence within the SH100 mitigation corridor road-effect zone. Finally, the fourth chapter compares the findings of the control array to actual crossing rates at mitigation structures.

CHAPTER II

MINIMIZING FALSE CAPTURES IN A RANDOMIZED CAMERA TRAP ARRAY

Introduction

Due to the extreme expense of constructing wildlife crossing structures (WCS), studies focused on the effectiveness and betterment of their design are almost as important as the structures themselves (van der Grift et al. 2013). On the State Highway 100 (SH100) mitigation project, quantifying the influence of crossing structure design and location on felid crossing rates may be directly correlated with the survival of the south Texas ocelot population, and therefore is of primary concern (Haines et al. 2006b). Currently, wildlife camera traps are proving to be a reliable, effective, and low-cost solution for monitoring WCS and ocelots (Trolle and Kéry 2005, Kays et al. 2010, Cramer 2012, Burton et al. 2015). Camera traps are appropriate for road mitigation projects as they are non-invasive, have minimal effects on wildlife behavior, comparatively low labor costs, and yield robust, comparable data (Kays et al. 2010). These benefits allow researchers to create a minimal footprint at structures, allowing for more robust data and possibly a reduced effect on wildlife visitation rates (Gill et al. 2001, Beale and Monaghan 2004). Additionally, camera traps are already widely used in wildlife biology and road mitigation monitoring studies and literature (Gagnon et al. 2011, Huijser et al. 2011, Cramer 2013, Welbourne et al. 2016, Andis et al. 2017, O'Connor et al. 2017).

While camera traps have proven to be an effective monitoring tool, their operation is often misunderstood. As noted by Welbourne et al. (2016), multiple scientific articles incorrectly describe wildlife camera trap operation, especially the sensing mechanism. Most camera trapping studies use camera traps with passive infrared sensors (PIR) (Meek et al. 2014, Welbourne et al. 2016). Wildlife cameras with a PIR sensor are triggered when the internal pyroelectric elements differ from one another, which occurs when they are exposed to objects emitting different wavelengths of electromagnetic radiation (Welbourne et al. 2016). Under ideal conditions, this triggering occurs when an animal with a different surface temperature than surrounding background objects moves into or within the detection zone of the passive infrared sensor, leading to a successful capture of the individual (Welbourne et al. 2016). However, work in the field is rarely ideal; different types of inanimate objects and vegetation exhibit different thermal properties, often leading to a combination of objects within the sensor range that possess different surface temperatures (Kaplan 2007, Welbourne et al. 2016). During typical field conditions, such as a windy day, vegetation that is a different temperature than other background objects may move enough to trigger the camera when there is no wildlife present, leading to a false capture (Meek et al. 2012, Welbourne et al. 2016). In certain circumstances, the number of false captures at a site can outweigh the number of successful captures, leading to dead batteries or memory cards filled to capacity before the service interval.

Due to the operation of PIR sensors in wildlife camera traps, the environment within the sensor's field of view is often controlled to minimize the number of false captures while maximizing the likelihood of capturing an individual when they enter the field of view (Kays et al. 2009, Rowcliffe et al. 2011). When camera arrays are erected to monitor a specific location or subject, controlling the environment in front of the camera is relatively simple because researchers may pick and choose the best camera location (Kays et al. 2010). This is generally the case with WCS cameras, where cameras are placed to effectively cover the structure entrance and exit, generally in areas where vegetation cannot grow (Gagnon et al. 2011). However, cameras placed in random locations provide greater complexity for camera setup because researchers have significantly less control over camera sites (Kays et al. 2010, Rowcliffe et al. 2011). To accurately estimate average daily wildlife occurrence within surrounding area, such as within a control array in a control-impact camera trapping study, placement methods may strongly impact wildlife observation (Gompper et al. 2010, Kays et al. 2010, Colyn et al. 2018). Many camera trapping studies control the vegetation immediately in front of the camera traps, ensuring that grass, brush, or other vegetation is not tall enough to trigger the PIR sensor, however this sharply increases visitation frequency and survey effort of the investigator (Gagnon et al. 2011).

It is unusual for studies to provide detailed setup descriptions for their camera trap arrays (Rowcliffe and Carbone 2008, Hamel et al. 2013, Meek et al. 2014). For motion-activated wildlife cameras, changes in camera placement and setting may influence the chance of capturing a present animal by 30 to 70% (Hamel et al. 2013). The majority of literary sources that use camera traps omit valuable camera placement information, such as camera height, direction, vertical angle, or objects or landscape features within camera field of view (Hamel et al. 2013, Meek et al. 2014, Kolowski and Forrester 2017). Within the current literature, it is far more common to indicate sites where camera traps are placed in reference to a specific landscape feature, such as near a trail or road (Heilbrun et al. 2010, Kays et al. 2010, Cusack et al. 2015). Colyn et al. (2018) and Kolowski and Forrester (2017) suggest that, while this information is valuable, it is important to include more information regarding site features, placement, and camera facing, information traditionally absent from published papers.

Beyond camera placement, camera settings may have a large impact on capture rates (Rowcliffe et al. 2011). Position of wildlife in front of the camera sensor may influence camera trap effectiveness, as wildlife farther from the sensor's core field of view have a lower chance of triggering a capture (Rowcliffe et al. 2011). For most camera models, this core field of view ranges for a short distance directly in front of the camera, and does not include the side or farther distances (Rowcliffe et al. 2011). In an effort to increase the core field of view, researchers may increase the "sensitivity" of wildlife camera trap sensors, however, in some camera models changing the sensitivity in the settings may only alter the angle of the sensor (Rowcliffe et al. 2011). By increasing sensor sensitivity wildlife camera traps may miss wildlife that pass close to the camera because they may pass below or between the field of view of the sensor (Rowcliffe et al. 2011). However, this is not true of all models. According to an instruction manual for Reconyx wildlife camera models, altering the sensitivity of their cameras influences the range of frequencies required to trigger the camera (Reconyx Corporation, Holmen, WI, USA). Other than sensitivity, the number of captures per trigger, the number of photographs taken per unit time, and the quality of the photographs taken may all influence the effectiveness of the camera trap and the subsequent survey effort to sort captures (Rowcliffe et al. 2008, Rowcliffe et al. 2011). Due to the relatively inexpensive cost of wildlife camera traps, multiple models have begun to appear on the market (Ahumada et al. 2019). Glover-Kapfer et al. (2019) predict that camera traps will soon go through major technological changes, increasing the difficulty of standardization or understanding model differences.

Objectives & Hypotheses

To quantify the influence of road mitigation structure design and location, a control array would need to be designed for comparison against the impact array detailed by Cogan (2018) that has been monitoring road mitigation structures since construction completion in January of 2018. This control array needed to follow many of the same design traits as the impact array to allow accurate comparison, while maintaining a low survey effort by minimizing false captures. South Texas is a volatile environment for the use of camera traps, and often maintains weather conditions that are known for creating difficult camera trapping conditions or for increasing camera failure (Glover-Kapfer et al. 2019). The results of testing this control array will be used to design and implement a more manageable control array for future research on the SH100 mitigation corridor, and therefore had to account for several external factors. Understanding the influence of camera placement, settings, and a suite of environmental factors will help future arrays minimize survey effort. The hypotheses tested were as follows: 1) Time of year will have a significant influence on number of false triggers in the control array, and days with the highest daily temperature will also have the highest number of false captures. 2) The direction that a camera is placed will influence the number of false captures, with cameras facing east or west having a higher rate of false captures than cameras facing north or south. 3) Since wildlife cameras use a similar detection system, a particular model will not have a significant influence on number of false captures.

Methods

To employ a control array around the State Highway 100 (SH100) mitigation corridor, emphasis was placed on camera location. The primary target species for the SH100 mitigation corridor was ocelots, a rare and elusive felid that is strongly associated with dense thornscrub, an equally rare habitat type in south Texas (Harveson et al. 2004, Jackson et al. 2005, Connolly 2009). The Harveson et al. (2004) study estimated that dense thornscrub accounted for 3% of their study area on Laguna Atascosa National Wildlife Refuge in Cameron County, Texas. Secondary target species included any south Texas species that may use a wildlife crossing structure as its primary means for safely crossing the roadway. Many of these species were not associated with a specific vegetation cover, therefore, the control array needed to equalize survey effort between multiple vegetation types that were not evenly distributed across the surrounding landscape. The vegetation surrounding SH100 was characterized, allowing the vegetation cover at potential control sites to be predicted prior to visitation on the ground. Predetermining expected vegetation reduced initial field validation as sites would not need to be investigated prior to site placement.

Determining Influence of Vegetation & Measuring Vegetation Factors for a Control Array

To determine vegetation cover in a similar manner to the Harveson et al. (2004) ocelot study, three vegetation categories were created. Land cover within the available control study area was mapped by conducting a supervised classification of 1-meter National Agriculture Imagery Program (NAIP) imagery, primarily from 2016 with some supplemental data from 2014, in ArcMap 10.5.1 (United States Department of Agriculture 2016, ESRI 2017). Using reflectivity recorded within the NAIP imagery, ground cover was broken into ten distinct classes that represented the most common land cover type within the 1-meter square. These classes were combined based on the primary vegetation type at each site, resulting in five vegetation classes (Figure 3). Bare dirt accounted for areas with dirt roads or where vegetation was extremely low, including sea oxeye daisy (*Borrichia frutescens*) salt flats. Grasses included a variety of species, which ranged in height from a few centimeters to over a meter, with gulf cordgrass (*Spartina spartinae*) predominant. Cactus included a variety of native species, with Texas prickly pear (*Opuntia engelmannii*) being the most widespread within the study area. Open was all areas where bare dirt, grass, or cactus was not predominant within the one-meter resolution, but was instead a diverse mix of two or more of these land cover types. Trees consisted of native woody species that have an average growth higher than 3 meters, the majority being thorn-forest species, including huisache (*Vachellia farnesiana*), honey mesquite (*Prosopis glandulosa*), Texas ebony (*Pithecellobium flexicaule*), and granjeno (*Celtis pallida*) (Jahrsdoerfer and Leslie Jr. 1988). Based on Harveson et al. (2004), canopy cover was considered the most influential factor on potential felid presence; therefore percent of tree class was used to categorize potential sites into three vegetation categories. Sites with greater than 70% tree canopy cover were categorized as dense thornscrub, sites with 40% to 70% tree canopy cover were classified as mixed thornscrub, and sites with less than 40% tree canopy cover were considered open grassland (Figure 3). Sites with greater than 40% grasses were discarded as potential sites, due to the high survey effort required to maintain camera function within this vegetation type (Figure 3). Each of the three wildlife crossing structures (WCS) within the available study area were within one of the three vegetation categories, with WCS3 in mixed thornscrub, WCS3A in dense thornscrub, and WCS4 in open grassland (Figure 4).

To test the accuracy of the ArcGIS vegetation map and confirm its usage for predicting vegetation categories at potential control sites, the vegetation category at 77 locations was predicted in ArcGIS, visited on the ground, and validated with field measurements. Percent canopy cover was used as the primary factor for validating expected vegetation categorization.

Canopy cover was measured ten times using a GRS densitometer as instructed by the manufacturer, allowing each measurement to account for 10% of the determined percent canopy cover (Geographic Resource Solutions, Arcata, CA, USA). Measurements were first taken at the camera site, then five meters in each cardinal direction, followed by measurements two and a half meters to the northwest, northeast, southwest, southeast, and in the direction of camera facing. These ten measurements provided an estimate for the percent canopy cover within a five-meter square centered on the camera placement. Of these 77 test sites, 76 were within the predicted vegetation category, providing a successful prediction rate of 98.7%. This success rate was considered high enough to design the control array based on predetermined vegetation categories calculated using ArcGIS.

Two additional vegetation metrics were measured at control sites for use in comparing vegetation categories. Ground cover was calculated using the GRS densitometer and with the same methods as canopy cover measurements. Maximum vegetation height was measured using a Robel pole delineated in 10-centimeter increments, up to a maximum of five meters. Any vegetation higher than five meters was listed as five meters. The tallest vegetation within 10 centimeters of the Robel pole was used for determining maximum height (Simpson et al. 1996, Pitman et al. 2005). These measurements were recorded at the same 10 locations that canopy cover was recorded. Vegetation heights were averaged to determine a single averaged vegetation height for each site and each vegetation category.

The three metrics describing the sites, canopy cover, ground cover, and average vegetation height, were factored to vegetation category and compared individually with Kruskal Wallis tests. For significant tests, a Dunn’s multiple comparison test with Bonferroni correction was conducted. All analyses were conducted in SPSS V 25 (IBM Corp.)

Study Design: Control Array

To determine control site locations, a 50-meter grid was overlaid across the available study area on US Fish and Wildlife Service (USFWS) land on Laguna Atascosa National Wildlife Refuge, Cameron County, Texas. A 50-meter grid was chosen because it maintained 25-meter site spacing, which was recommended by Kays et al. (2010) for ocelot studies in woody, tropical environments. Based on the USFWS specialized use permit, sites could not be placed more than 150-meters from the road corridor, limiting the study area to 2.28-kilometers^2. A stratified random design was used with the three vegetation categories (open grassland, mixed thornscrub, dense thornscrub) determined in ArcGIS as the strata. Vegetation within a 15-meter buffer of each control site was classified into one of the three vegetation categories, providing the vegetation expected to be found at that site (Figure 3).

To set up the camera array in the field, 12 sites were randomly chosen within each of the three vegetation categories as predicted by the vegetation map in ArcGIS (Figure 4). These 36 sites were evenly distributed between the north and south sides of SH100, with 18 on either side of the roadway (Table 1). Camera manufacturer and facing were both controlled factors determined before visitation to the field (Table 1). Sites were located in the field using a Garmin GPSMap 64st handheld GPS (Garmin Ltd., Olathe, KS, USA), which had an accuracy of five to 10 meters. A camera placement protocol was used according to recommendations in Colyn et al. (2018) to reduce the bias of placing sites to increase capture probabilities of specific species (Table 2). If the pre-determine random site chosen prior to visitation did not match placement protocol, a spiral track was followed in a clockwise direction, with one meter between each spiral track, until all requirements for the placement protocol were met or to a maximum of 10-

meters from the pre-determined site (Table 2). If sites did not meet placement protocol requirements they would be abandoned and moved to a backup site, however this did not occur as all pre-determined sties met placement protocols. All cameras were placed 50-centimeters off the ground.

Two sites had to be moved during the study period, one within dense thornscrub and the other in mixed thornscrub, and both on the north side of SH100. In April 2019, a mulcher cleared a trail to allow access to a powerline within the study area. These sites were relocated using the original placement point as the start of the spiral method described above until all protocol requirements were the same as the original site.

Cameras

Wildlife cameras used within the array consisted of 16 Moultrie MCG-13270 (EBSCO Industries, Birmingham, AL, USA) and 20 Bushnell 119874 (Bushnell Corporation, Overland Park, KS, USA) cameras. Cameras were set to take a single picture per trigger and to take pictures as quickly as possible based on camera brand (Table 3). Bushnell cameras were set to the lowest interval setting of 0.6 seconds, while Moultrie were set to the lowest interval setting of "None," which was provided by the manufacturer as 1.3 seconds. Vegetation was not cleared more than three meters in front of cameras within the control array, therefore cameras within this array were set to "Low" PIR setting in accordance with manufacturer recommendations (Hofmeester et al. 2017). If a camera malfunctioned, it was removed from the field immediately and replaced with a camera of the same manufacturer and model. Sites were visited every four weeks to exchange SD cards, check and/or replace batteries, and clear vegetation.

Temperature Measurements

To determine if the influence of ambient air temperature had an impact on the number of false captures recorded, temperature was measured at the three WCS within the study area using Kestrel DROP D3 Wireless Temperature, Humidity & Pressure Data Loggers (Nielsen-Kellerman Company, Boothwyn, PA, USA). Temperature data loggers were placed on the wildlife-proof fencing above the WCS and on the south side of the roadway. All data loggers were placed in direct sunlight, as directed by the manufacturer. Temperature was measured every two hours starting at midnight, with data collection every week. The temperature data recorded at WCS3, WCS3A, and WCS4 were very similar, therefore, they were averaged together for analysis.

Photo Processing and Statistical Analyses

All pictures collected form wildlife cameras in the control array were renamed using the program ReNamer to change photo file names to the date and time the capture was taken. Captures were organized by site and sorted by taxa, with false captures treated as a separate category. The total number of false captures was recorded at each site and labeled by hour, day, week, and month.

A linear regression was conducted to determine the influence of average daily air temperature on the natural log transformed number of false captures within the control array, using the following equation:

$$y = 20.054 \times 10^{0.0749x}$$

An analysis of variance (ANOVA) with Type III correction was ran to determine if camera facing, north or south side of SH100, camera model, vegetation category, or month had a significant influence on the number of false captures within the control array. Normality and heteroscedasticity assumptions were not violated based on a histogram and Q-Q plot. Interactions between factors were also compared. A Tukey's HSD post-hoc test was used to compare means within factors that were found to be significant. All analyses were run in Program R 3.4.3 using the car package and the broom package (R Development Team 2019).

Results

The control array was installed from December 2018 through October 2019, resulting in 317 trap nights, or operating for 11,022 of the 11,142 available camera trap nights, indicating all cameras were functional 98.9% of the available nights. The control array took 939,944 false captures, with Bushnell cameras accounting for 261,144 false captures and Moultrie taking 678,732 false captures (Table 4).

Average daily temperature had a consistent pattern, rising from 15℃ in the coldest month of January, up to 31℃ in the hottest month of August (Figure 6). The rate of false captures had two distinct peaks, the first during the south Texas spring growing season, and the second during the hottest month of August at the end of growing season (Figure 5). Based on findings from the linear regression, average daily temperature was determined to be associated with increased false capture rates ($R^2 = 0.3201$, $p < 0.0001$), as average daily temperature increased so did the number of false captures (Figure 7). The peak in number of false captures during the growing season likely influenced these results, however, month was not found to be a significant factor on average number of false captures (F = 0.622, degrees of freedom = 386, $p = 0.5768$).

Based on the results of the ANOVA, camera facing was not a significant factor (F = 2.738, df = 386, $p = 0.2020$) but facing did have a slight effect on average number of false captures (Figure 8). West had the lowest average number of false captures (mean = 944 ± 166 standard error), while south had the highest (3007 ± 569). North and east had a relatively similar average number of false captures with 2695 ± 558 and 2491 ± 812, respectively.

According to the results of the ANOVA, Bushnell and Moultrie models took a significantly different number of false captures ($F = 7.806$, $df = 386$, $p = 0.0001$). On average, each Bushnell camera took 1,187 ± 326 false captures per month, while Moultrie cameras took 4,848 ± 1,144 false captures per month. This discrepancy was primarily due to large peaks in the number of false captures at Moultrie locations during the growing season and hottest months of the year, July and August (Figure 9). While Bushnell cameras had slight peaks during these times, it was lower compared to those seen at Moultrie sites.

Results of the ANOVA revealed that vegetation category was also a significant factor on the total number of false captures ($F = 7.645$, $df = 386$, $p = 0.0347$). Within vegetation, dense thornscrub (811 ± 239) had fewer false captures than mixed thornscrub (2,633 ± 424), which had fewer false captures than open grassland (3408 ± 702) (Figure 10). Results from a post-hoc Tukey test revealed dense thornscrub and mixed thornscrub were not significantly different in the number of false captures ($p = 0.0769$). Mixed thornscrub and open grassland were also not significantly different ($p = 0.5240$); however, the number of false captures in dense thornscrub and open grassland differed significantly ($p = 0.0035$). The large spikes in average number of false captures during the spring south Texas growing season were primarily seen in open grassland and mixed thornscrub, with little change in the average number of false captures recorded in dense thornscrub (Figure 10). While dense thornscrub had a relatively mild spike, mixed thornscrub and open grassland each had large spikes in average number of false captures.

Vegetation Factors

The results of the Kruskal-Wallis tests showed that percent canopy cover, percent ground cover, maximum vegetation height and minimum vegetation height differed between at least two of the three vegetation categories (Fig 11). Percent canopy cover differed significantly between all vegetation categories ($p = 0.02$) with average canopy cover 89% ± 3.4 for dense thornscrub, 55% ± 3.4 for mixed thornscrub, and 0% ± 0.0 for open grassland. Percent ground cover at dense thornscrub sites (41% ± 9.9) was significantly different than that found at mixed thornscrub (78 ± 6.0) ($p = 0.013$). However, open grassland (68 ± 6.6) did not differ significantly from dense thornscrub ($p = 0.200$) or mixed thornscrub ($p = 0.931$) categories. Average vegetation height different significantly between vegetation categories, with average vegetation height found at open grassland sites (40 ± 2.6) being significantly different than that found at dense thornscrub (378 ± 12.6) ($p < 0.001$) and mixed thornscrub (196 ± 12.3) ($p = 0.009$). Average vegetation height at dense thornscrub sites was also significantly different than that at mixed thornscrub ($p = 0.044$).

Discussion

Multiple factors attributed to the number of false captures taken by cameras within the control array. They hypothesis that higher temperatures would result in a higher rate of false captures was supported. Although Welbourne et al. (2016) stated that ambient air temperature had no effect on internal passive-infrared (PIR) camera sensors, in this study temperature had a significant influence on false capture rates. These results support those found by Meek et al. (2012) and Glover-Kapfer et al. (2019), which showed that PIR camera trap performance was influenced by weather, especially extreme heat. On this project, the association between increasing average daily temperature and rate of false captures appears to be primarily influenced by Moultrie camera models, as well as sites within open grassland. While the influence of temperature is a strong contributing factor to the rate of false captures, vegetation cover above the camera also influenced the effect of temperature as well. As canopy cover increased, false captures during the growing season and times of extreme temperatures was decreased.

The hypothesis that camera direction would influence the number of false captures was not supported as direction of camera facing did not significantly influence the average number of false captures; However, it may still be an important factor for reducing survey effort. Pictures taken at sites with east and west facing cameras were the most difficult to sort as pictures taken during sunrise and sunset were often difficult or impossible to analyze due to the sun's intensity overexposing each picture. This problem was only prevalent at open grassland sites, where vegetation was often so short that given no technological limits, effective camera range would be to the horizon. Sunrise and sunsets did not have the same effect on cameras in mixed thornscrub and dense thornscrub because vegetation blocked most of the light during these times.

The third hypothesis that camera model would not have a significant influence on number of false captures was not supported as Moultrie camera took significantly more false captures than Bushnell cameras. This effect may be mitigated by using cameras of the same or comparable models or by equally distributing models between treatment groups, as done in the present study. Glover-Kapfer et al. (2019) suggested that Bushnell and Moultrie brand cameras shared a similar camera trap rating, but in this study Moultrie models took a significantly higher number of false captures, potentially encouraging the use of Bushnell cameras for a control array. However, both Bushnell and Moultrie brand cameras were plagued with issues during this study. Nine Moultrie cameras malfunctioned and had to be replaced with other wildlife cameras of the same model. These malfunctions included both hardware and software issues; ranging from one Moultrie taking pictures exactly every 10 seconds, to cameras shutting down and refusing to restart. Five Bushnell cameras also malfunctioned, with issues from times and dates randomly changing to cameras not powering up or down. Multiple Bushnell cameras struggled to change between day and night settings. Pictures were taken at night with the daytime filter over the camera lens, causing pictures to be extremely dark. These pictures had to be lightened before sorting, greatly increasing sorting effort.

The influence of vegetation cover should be considered when designing and implementing a control camera trap array. Results from the Kruskal-Wallis test showed that measurements taken at control sites presented significant differences between the predetermined vegetation categories based on canopy cover. This was expected as canopy cover was used as a treatment factor when placing control sites. Average canopy height was significantly different between all categories and showed the same decreasing trend as canopy cover. This was expected, as in south Texas cacti and grasses do not grow as tall as trees, therefore as percent canopy cover (woody cover) decreases at a site, average vegetation height is also expected to decrease (Ewing and Best 2004). Finally, percent ground cover did not show the same pattern as percent canopy cover but had a parabolic pattern with the highest value occurring at mixed thornscrub. This was likely due to available light within the understory and the variety of soil types present on the landscape, which has been found to have a significant influence on vegetation (Archer 1995, Ewing and Best 2004). Soils that do not promote vegetation growth tend to have lower ground covers, while mixed thornscrub, a mid-succession vegetation type, tends to develop on soils that retain water and promote growth (Archer 1995). These mid-succession areas usually possess extensive ground cover (grasses and cacti) with a woody overstory developing, but not thick enough to significantly reduce light and exclude understory species (Archer 1995). Patches of dense thornscrub represent late-succession areas, and were likely developed on good soils with low-frequency disturbance (Archer 1995, Kazmaier et al. 2001). These expectations are anecdotally confirmed using aerial photography of the SH100 area from the 1960s, which showed patches of dense woody cover in areas that still support dense thornscrub.

While cameras within dense thornscrub took significantly fewer false captures than those within mixed thornscrub and open grassland, sites within dense thornscrub were also less influenced by the growing season and high average daily temperatures. Increased vegetation cover likely maintains more consistent temperatures and light intensity on vegetation within camera facing (Kaplan 2007). Additionally, higher canopy cover reduced the impact of winds on vegetation within the camera field of view, as wind velocity is exponentially reduced below the canopy layer. During the hottest time of the year, solitary trees within mixed thornscrub habitat became the primary trigger for false captures at mixed thornscrub sites. This was likely due to the different thermal properties exhibited by the leaves when warmed by direct sunlight compared to the surrounding shaded vegetation (Welbourne et al. 2016). Without other trees as a windbreak, leaves were shaken by strong, gusting summer winds, leading to a high number of false captures. Open grassland sites were strongly influenced by the growing season and high temperatures, with two large peaks in the average number of false captures during these times. Grasses grew quickly, and had an extensive seed bed, creating multiple waves of sprouts during the growing season. Some grasses reached camera height less than a week after visitation.

To minimize false captures, researchers could erect small preliminary control arrays, which may be used to identify factors that may have the greatest influence on number of false captures. Using data from this preliminary control array, researchers can create a placement protocol that can be used for designing and placing the larger control array. Placement protocols could be reported within the appendix of published papers, allowing other researchers in similar ecosystems to model study designs to reduce survey efforts while maximizing successful captures. Though this practice is not yet common, it is slowly becoming encouraged (Meek et al. 2014, Colyn et al. 2018). Meek et al. (2014) and Colyn et al. (2018) provide multiple guiding principles for reporting camera trap research to allow researchers to minimize bias of successful captures between projects, and the same principles may be used for reducing survey effort to minimize the number of false captures. While camera traps are already pervasive throughout wildlife biology and road ecology, Glover-Kapfer et al. (2019) report that within the next 10 to 20 years camera traps will go through a major technological shift, where models will have more sensitive PIR sensors, faster trigger speeds, and potentially new trigger technologies. While these technological changes will create a more empirical camera trapping environment, new models and techniques will require extensive in the field testing by researchers. Published placement protocols could streamline the transition process and provide a record of relevant study design information that will allow projects using the current camera trap designs to be comparable to the more efficient camera traps on the horizon.

CHAPTER III

WILDLIFE PRESENCE AND FREQUENCY OF OCCURRENCE NEAR STATE HIGHWAY 100

Introduction

To reduce the impact of roadways on wildlife populations, many transportation and conservation agencies construct wildlife crossing structures (WCS) and road mitigating wildlife exclusion fencing to reduce the number of wildlife-vehicle collisions (van der Grift et al. 2013). However, road mitigation structures for wildlife are expensive, accounting for 10% or more of the total road construction budget (van der Grift et al. 2013). Without intense evaluation of the effectiveness of road mitigation projects, researchers may unintentionally endanger the long-term viability of wildlife populations by installing structures that are less effective than is necessary to maintain the population, wasting valuable time for the species and financial resources (van der Grift et al. 2013). Determining the effectiveness of WCS in the field requires long-term comprehensive monitoring projects (van der Grift et al. 2013, Andis et al. 2017). Many studies use a decrease in roadkill and increase in actual crossing rates at the wildlife crossing structures to demonstrate the value of mitigation measures; however, these studies fail to examine population-level effects (van der Grift et al. 2013, Andis et al. 2017). Calculating the performance of mitigation structures by determining the influence of location and design on species use is essential for determining if mitigation project goals were met (van der Grift et al. 2013, van der Ree et al. 2015).

Post-construction of the mitigation corridor, control-impact arrays are the best type of monitoring design for determining the effect of WCS location on actual crossing rate (Henke and Bryant 1999, van der Ree et al. 2015). While the end goal of this method is to use results from the control array to compare against the impact array to create an estimation of structure performance, it may also be used to determine possible influences within the road-effect zone where wildlife might congregate on a micro scale (Mysterud and Ims 1998, Andis et al. 2017). Wildlife movement may be influenced on a fine scale within the road-effect zone due to several biotic and abiotic factors, such as vegetation cover, distance to nearest available crossing structure, and distance to the roadway. Abrahms et al. (2017) suggests wildlife movement and available resources should inform corridor conservation, and Forman and Deblinger (2000) suggest that, to maximize WCS effectiveness, it is important to place structures in areas of the road-effect zone where wildlife have the strongest presence near the roadway. Before determining future WCS location, it is important to understand wildlife behavior within the road-effect zone based on multiple landscape factors (Clevenger and Wierzchowski 2006).

Understanding influencing factors within the road-effect zone is especially important for the State Highway 100 (SH100) mitigation project and ocelot conservation measures. Harveson et al. (2004) found that ocelots are strongly associated with dense thornscrub, a vegetation cover type that is comparatively rare in south Texas, especially near the SH100 mitigation corridor. Harveson et al. (2004) conducted a land cover analyses and found that dense thornscrub only covered 3% of their 182-km^2 study area on Laguna Atascosa National Wildlife Refuge. Open areas (<75% canopy cover for their study) accounted for the other 97% of their study area, with no moderate cover (75-95% canopy cover for their study) found (Harveson et al. 2004). According to Forman and Deblinger (2000) and Tewes and Hughes (2001), if WCS are constructed to promote ocelot crossings, they would likely be constructed in areas with the densest thornscrub near the roadway and of a design that promotes felid use. However, no species functions in isolation but are single components of complex ecosystems (Clevenger and Wierzchowski 2006). Managing for the longevity of an ecosystem requires diverse conservation efforts, with the inclusion of multiple mitigation structures with a variety of designs (Clevenger and Wierzchowski 2006). Therefore, any single-species mitigation system may have effects on other species that were not the target of the mitigation project (Clevenger and Wierzchowski 2006). Understanding the influence of ocelot-preferred factors on non-target taxa may allow researchers to design and install a mitigation corridor that promotes the longevity of ecosystems and in turn the viability of individual species (Clevenger and Wierzchowski 2006).

Objectives & Hypotheses

The object of this research was to examine the presence of target wildlife within natural vegetation surrounding the State Highway 100 (SH100) mitigation area, with a specific focus on ocelots. Control sites were stratified based on vegetation cover to determine expected crossing frequencies at mitigation structures. Determining the influence of vegetation factors, distance to nearest wildlife crossing structure (WCS), distance to nearest wildlife guard (WG), and distance to the roadway provides researchers with additional tools for estimating wildlife visitation to a structure, possibly encouraging or discouraging animals from crossing. Analysis for this objective used number of captures of wildlife from a vegetation-based control array. The hypotheses tested were as follows: 1) Presence and richness of target species at control sites on either side of the roadway (north and south SH100) will not be significantly different. 2) Presence and richness of target species will be highest in dense thornscrub, followed by mixed thornscrub, and open grassland will have the lowest presence and richness of target species. 3) Canopy cover will be positively associated with species richness and number of independent events, while ground cover will be negatively associated with target species richness and number of independent events. 4) Proximity to WCS will be positively associated with species richness and number of occurrences of target species. 5) Proximity to WG will have no effect on occupancy and richness for any target species at control sites.

Methods

A control-impact study design was used on the State Highway 100 (SH100) mitigation corridor to determine the influence of structure location on performance, and compared against wildlife crossing structures (WCS) to calculate the permeability of the road corridor as a whole. Determining the influence of structure location requires the identification and study of biotic and abiotic factors around the roadway, and how they influence observed species richness and occurrence. To reduce bias when comparing the control and impact arrays, the control array needed to be placed in vegetation that was similar to that measured at the WCS, requiring a stratified random study design that would encompass vegetation as strata.

Study Design: Control Array

The control array (detailed in Chapter II) consisted of 36 sites, placed using a stratified random design, within three vegetation categories of dense thornscrub, mixed thornscrub, and open grassland, with canopy cover used as the primary factor to determine groups. Dense thornscrub included sites with greater than 70% canopy cover; mixed thornscrub included sites with 40-70% canopy cover, and open grassland were all sites with less than 40% canopy cover. The available study area was limited by a United States Fish and Wildlife Service (USFWS) special use permit to within 150-meters of the SH100 road corridor and to the Laguna Atascosa National Wildlife Refuge. A grid of potential sites with 50-meter spacing was overlaid across the available study area. To ensure sites were placed within the intended vegetation category, the vegetation category surrounding each site was determined using ArcMap 10.5.1 prior to visitation on the ground (United States Department of Agriculture 2016). Site selection was conducted using supervised classification using 1-meter National Agriculture Imagery Program (NAIP) imagery, primarily from 2016 with some supplemental data from 2014 (United States Department of Agriculture 2016, ESRI 2017). Twelve sites were placed in each vegetation category and were evenly distributed on the north and south sides of SH100. Cameras were a mix of 16 Moultrie MCG-13270 wildlife cameras and 20 Bushnell 119874 wildlife cameras. The control array was installed in December 2019 and removed in October 2020. The planned study period for the control array was twelve months, but was reduced to 10 months due to vegetation removal and controlled burns initiated by USFWS.

Camera placement protocol was based on the recommendations from Colyn et al. (2018). If the pre-determined site did not meet all the characteristics required to match the intended vegetation category, a spiral transect with 1-meter spacing was followed until all requirements for the placement protocol were found (Table 2). Cameras were not faced toward any existing road, water, or trails. No bait or attractants were used. Vegetation was cleared or cameras were placed in areas where the maximum detection distance for mid-sized species was three meters. Terrain also had to be flat for three meters directly in front of the camera. All cameras were angled so that the area within the maximum detection distance of the camera (three meters) occupied half of the camera field of view.

Camera Settings

Settings within the control array were chosen to minimize the number of false captures while ensuring cameras would capture animals present at the site. Both camera types were set to a single capture per trigger with no delay between triggers. For Bushnell, the lowest interval between triggers was 0.6 seconds, for Moultrie it was 1.3 seconds (Bushnell Corporation, KS, USA, EBSCO Industries, Birmingham, AL, USA). Both brands were set to the lowest PIR sensitivity. This choice was made based on recommendations from the manufacturers, as the sensor detection zone was limited to three meters from the camera (Bushnell Corporation, KS, USA, EBSCO Industries, Birmingham, AL, USA). Sites were visited once a month to exchange SD cards, check and/or replace batteries, and clear vegetation immediately in front of the camera.

Photo Processing

The program ReNamer was used to rename each capture file to the date and time the capture was taken. Captures were then organized by site and sorted by taxa. Except for flying birds and rodents, all species were classified to the lowest level possible. The program DataOrganize was used to create a text file with camera location, species, date, and time. Finally, the camtrapR package in Program R 3.4.3 was used to create summary statistics from the camera trap data. Independent occurrences were defined as one observation of a single species within a 30-minue period, based on the last capture of an individual of the same species (O'Brien et al. 2003, Niedballa et al. 2016). Only multiple individuals caught in a single photo were counted as multiple individuals.

Statistical Analyses

To determine if species communities changed based on which side of SH100 the control site was on and the vegetation category it was placed in, species communities were compared using non-metric multidimensional scaling (nMDS) plots based on results from bootstrap averaging, with 100 bootstraps per group. Prior to any analyses, a Bray-Curtis similarity matrix was applied to data that had been square root transformed with no dummy variable used. These analyses were conducted using Primer 7 (PRIMER 2015). A permutational multivariate analysis of variance (PERMANOVA) with 9,999 permutations was used to further explore the influence of side of SH100 and vegetation category might have on species richness and occurrence. Side of SH100 and vegetation category were both treated as fixed factors. Results of the PERMANOVA were further investigated using a test for homogeneity of multivariate dispersions (PERMDISP) within each factor. If results of the PERMANOVA and PERMDISP showed significance, a SIMPER analysis would be used to determine the dissimilarity created by each species between groups.

To investigate other potential influencing factors on species richness (calculated as S= the total number of species at each site) and occurrence, an analysis of variance (ANOVA) with Type III correction was ran to determine if vegetation category had a significant influence on species richness. Normality and heteroscedasticity assumptions were not violated based on a histogram and Q-Q plot. A Tukey's HSD post-hoc test was used to compare means within factors. A negative binomial generalized linear model was calculated to determine any association between species richness and the factors north or south side of the control array, percent canopy cover, percent ground cover, distance to SH100, distance to nearest WCS, or distance to nearest wildlife guard (WG) (Lindén and Mäntyniemi 2011). This generalized linear model was run in Program R 3.4.3 using the MASS package, following this equation:

$$species(x) = glm.nb(x \sim Side + Nearest\ WCS + Nearest\ WG + Canopy\ Cover + Ground\ Cover + Distance\ to\ SH100)$$

The model was then analyzed using the dredge function within the MuMIn package in Program R 3.4.3. The dredge function created and compared multiple models by manipulating included factors until all factor combinations were tested, then returned the ten models with the lowest ΔAIC. Models less than two ΔAIC from the best model were averaged together using model.avg in the MuMIn package in Program R 3.4.3 (Burnham et al. 2011, Planillo et al. 2017, Planillo and Malo 2018). Results were reported from the averaged model. Model fit was determined using McFadden pseudo-R^2 scores calculated for each model averaged (McFadden and Domencich 1975, McFadden 1977, Kim et al. 2019). The same model averaging process was used to model the total sum of wildlife occurrences and by target species individually. Species were only included if they had greater than 100 occurrences in the control array and observed in multiple vegetation classes, these species included bobcats, coyotes, eastern cottontail, collared peccary, northern bobwhite, nilgai, nine-banded armadillo, striped skunk, Virginia opossum, and white-tailed deer.

Results

The camera array took 1,058,263 total pictures, with 118,319 successful captures, constituting an 11% success rate. Of those successful captures, 43,529 were either of humans, non-target species, or contained a species that could not be identified, and were therefore excluded from analysis. After removal of these pictures, 74,790 captures were of target species, accounting for 7% of the total dataset. Of those successful captures of target species, 32,610 were at sites within dense thornscrub, 35,838 were within mixed thornscrub, and 6,342 were within open grassland. The north and south sides of State Highway 100 (SH100) had a similar number of successful captures of target species, with the north side having 37,584 and the south side having 37,206. These 74,790 successful captures equated to 5,793 occurrences, which were defined as one observation of a single individual within a 30-minue period, based on the last capture of an individual of the same species (Table 5). Of those occurrences, 3,189 were within dense thornscrub, 1,882 within mixed thornscrub, and 722 within open grassland. The half of the array on the north side of SH100 had a total of 2,687 occurrences while the south side had 3,106.

Results of the PERMANOVA showed that significant differences in the observed wildlife community composition were found between side (pseudo-F = 5.647; $p = 0.0001$), vegetation (pseudo-F = 10.914; $p = 0.0001$), and the interaction of side and vegetation (pseudo-F = 2.0817; $p = 0.0101$). Results of the PERMDISP for side of SH100 was not significant ($p = 0.8922$) suggesting that the differences observed were not due to dispersion. This finding was clear in the nMDS plot based on bootstrap averaging (Figure 12A), as there was a large gap between the two groups. The PERMDISP analysis for vegetation also indicated no dispersion effects for the differences between dense thornscrub and mixed thornscrub animal communities ($p = 0.6025$). However, the PERMDISP did detect dispersion effects when comparing the wildlife communities from dense thornscrub and open grassland ($p = 0.002$), and mixed thornscrub to open grassland ($p = 0.0006$). These results are easily visualized in the nMDS plot (Figure 12B), as all three groups are distinctly separate, with open grassland having a higher dispersion than the other two classes. The distance between open grassland and the clustering of dense thornscrub and mixed thornscrub shows that the significant difference found between these wildlife communities is not strictly due to dispersion, but also due to the observed location differences around the centroid. The SIMPER analysis found that eastern cottontail created the most dissimilarity (17.97%) between dense thornscrub (mean = 90 ± 35.3 standard error) and mixed thornscrub (62 ± 12.1). Virginia opossum (15.91%) and bobcats (10.86%) also had a strong contribution to the dissimilarity between the wildlife observed in these two groups, with both being observed more in dense thornscrub than in mixed thornscrub. A similar pattern was found between dense thornscrub and open grassland, where eastern cottontail created the most dissimilarity between dense thornscrub and open grassland (2 ± 1.7), accounting for 19.62% of the dissimilarity between wildlife observed in these two groups. Virginia opossum (13.48%), white-tailed deer (10.82%), and bobcats (10.56%) also accounted for a large portion of the dissimilarity between dense thornscrub and open grassland and were all observed more in dense thornscrub than open grassland. Eastern cottontail was also the predominant species driving dissimilarity (24.96%) between mixed thornscrub and open grassland. Two other species strongly contributed to the dissimilarity, white-tailed deer (13.54%) and nine-banded armadillo (10.13%), both of which were observed more in mixed thornscrub.

According to the results of the ANOVA ran on species richness within the three vegetation categories, significantly more species were observed as canopy cover increased (F = 37.800, degrees of freedom = 31, $p < 0.0001$). Results of the Tukey's HSD post hoc test indicated that species richness within dense thornscrub (13 ± 0.3) was significantly higher (Figure 13A) than species richness in mixed thornscrub (11 ± 0.6) ($p = 0.0036$) and species richness in open grassland (7 ± 0.6) ($p < 0.0001$); species richness in mixed thornscrub was also significantly higher than species richness in open grassland ($p < 0.0001$).

Based on results of the generalized linear model ran on species richness data, as percent canopy cover increased the number of species captured increased as well ($p < 0.0001$). Two factors were shown to be associated with species richness: percent canopy cover (Figure 13B) and distance to wildlife crossing structure (WCS) (Figure 14). Sites closer to WCS had significantly higher species richness ($p = 0.0088$). Based on the generalized linear model for species richness, interaction between canopy cover and distance to WCS was not shown to be significant ($p = 0.4745$). McFadden pseudo-R^2 indicated a model using percent canopy cover and distance to WCS provided a strong indicator of wildlife diversity within SH100 area (range of McFadden pseudo-R^2 scores from 0.2525 to 0.2606).

Based on the results of the generalized linear model of all species combined, percent canopy cover was the only significant factor associated with wildlife occurrences (Table 6). As seen in species richness, sites with higher percent canopy cover had a higher number of occurrences than sites with lower canopy covers ($p = 0.0002$). Distance to WCS and wildlife guard (WG), percent ground cover, and north or south side of SH100 were all included in the final model averaging; however, the influence of these factors was calculated as non-significant. The all species combined model was far less suited for prediction as McFadden pseudo-R^2 scores ranged from 0.0651 to 0.0707.

Side was a significant factor ($p < 0.004$) in the models for abundance in four species: bobcats, coyotes, northern bobwhite, and nilgai (Table 6). Side improved model fit but was not significant for four additional species models (Table 6). Bobcats, coyotes, northern bobwhite, and nilgai were observed more on the south side of SH100 than on the north side. For the coyote and nilgai averaged generalized linear models, side was found to be the only significant factor, indicating these species were not associated with any variable other than the south side of SH100. This south side biased distribution of individuals was especially prevalent for bobcats, where 83% of occurrences were at control sites on the south side of SH100.

Percent canopy cover was calculated to be a significant factor ($p < 0.0004$) for seven species, more than any other factor, and had a positive association with all seven species (Table 6). As canopy cover increased, the total number of bobcats, collared peccary, eastern cottontail, northern bobwhite, nine-banded armadillo, Virginia opossum, and white-tailed deer occurrences also increased. Percent canopy cover was included in the generalized linear models for nine of the ten species; making it the factor included in the highest number of species models (Table 6). For eastern cottontail and white-tailed deer, percent canopy cover was the only significant associated factor (Table 6). Bobcat occurrence was strongly associated with vegetation cover. There were only eight individual bobcat occurrences within all open grassland sites, while mixed thornscrub had 133 occurrences, and dense thornscrub had 296 (Table 5).

Percent ground cover was the only factor which was not shown to be a significant associating factor for any species (Table 6). However, it was included within the generalized linear models for five of the ten species, including coyotes, eastern cottontail, nilgai, and nine-banded armadillos (Table 6). These species were not restricted to only herbivores and/or wildlife that could find some camouflage or shelter from low ground cover.

Distance to SH100 was included in generalized linear models for seven of the ten species (Table 6) but was only found to be significant for one species, nine-banded armadillos ($p = 0.0113$). Nine-banded armadillos occurred more at sites closer to SH100 than sites farther away (Table 6).

Distance to the nearest WCS was also included within the generalized linear models for seven of the ten species (Table 6) and was only shown to be significant for one species, bobcats (bobcats, $p = 0.0072$). Bobcats occurred more at sites closer to WCS than at sites farther away (Table 9).

The generalized linear models ran for each species calculated that distance to WG was significant for two species, collared peccary (*p* = 0.0036) and Virginia opossum (*p* = 0.0072). Collared peccary and Virginia opossum occurred more at sites closer to WGs than at sites farther away (Table 9). Distance to wildlife guard was included within the generalized linear models of eight of the ten species, making it the factor included in the second highest number of the generalized linear models ran for each species (Table 9).

Discussion

The hypothesis that the presence and richness of target species at control sites on either side of State Highway 100 (SH100) would not be significantly different was not supported. The significant difference in the number of occurrences of bobcat, coyote, norther bobwhite, and nilagi in the north and south sides of SH100 indicated these species may not have an equal distribution within the habitat on either side of the roadway. These differences in species occurrence may be due to several reasons; potentially because there may be higher quality habitat on one side of the roadway. Alternatively, the populations on either side of SH100 may not yet have equalized following construction of the road mitigation corridor (van der Grift et al. 2013). This south side biased distribution of individuals was especially prevalent for bobcats, where 83% of occurrences were at control sites on the south side of SH100. This bias is important for the south Texas ocelot population, as most of the population is north of SH100, and individuals that are seen in the SH100 area are usually young dispersing males (Blankenship et al. 2006). Ocelots have yet to be observed at a wildlife crossing structure (WCS) on SH100 (Cogan 2018); but the strong presences of bobcats on the south side of the mitigation corridor may indicate there is available felid habitat on the south side of the highway or that the habitat is saturated with bobcats. Alternatively, a high number of resident bobcats may inhibit ocelot movement

(Tewes and Hughes 2001). However, until ocelots are regularly seen moving through the WCS, the area south of SH100 will likely be underutilized by this endangered species.

The hypothesis that canopy cover would be positively associated with species richness and number of independent events while ground cover would be negatively associated was only partially supported. Slight changes in surrounding vegetation cover had a strong influence on the wildlife communities observed at control sites, and showed a strong impact on number of occurrences, demonstrating that canopy cover was positively associated with species richness. However, ground cover seemed to have little association with wildlife occurrences and was the most variable among the vegetation categories. Moreover, results from analyses including ground cover did not support the hypothesis. Occurrences of 7 of the 10 species (Table 6) were associated with higher canopy cover. The hypothesis that presence and richness of target species in dense thornscrub would be higher than in mixed thornscrub, and both would be greater than open grassland, was supported. Species richness was greatest in dense thornscrub, followed by mixed thornscrub, and finally open grassland. While dense thornscrub appears to be important for local wildlife populations, it is a comparatively rare vegetation type within the study area (Bradley and Fagre 1988). While studying canopy cover on Laguna Atascosa National Wildlife Refuge, an area just north of SH100, Harveson et al. (2004) found that only 3% of their 182-km^2 study area was dense thornscrub (>95% canopy cover for the purposes of that study). In this study, of the 659 potential camera placement sites, only 25 (3.8%) of the available sites had a predicted canopy cover greater than 75%. For bobcats, collared peccary, eastern cottontail, northern bobwhite, nine-banded armadillo, and Virginia opossums, more than 50% of occurrences were at sites within the dense thornscrub vegetation category (Table 5). These findings indicate that this vegetation cover is important to the wildlife communities surrounding

SH100 and reflects findings from Forman and Deblinger (2000), which found that in the road-effect zone, mammal abundance was typically highest in protected forest. Abundance is likely higher in areas with dense vertical vegetation due to its dampening influence on indirect road effects (Benitez-Lopez et al. 2010, Brown et al. 2012). The influence of vegetation emphasizes the importance of WCS placement on actual crossing rates; WCS in areas where animals are more likely to occur will have a higher rate of use than structures in areas where animals are less likely to occur (Bond and Jones 2008, Abrahms et al. 2017). The association of target species with dense thornscrub is best shown using bobcat occurrences at three control sites on the south side of SH100, two sites within dense thornscrub and one within mixed thornscrub (Figure 15). Primary placement for these sites was at the minimum distance which still allowed independence, 50-meters, where the three sites formed an equilateral triangle (within the dotted yellow circle in Figure 15). Bobcats were seen regularly at the two dense thornscrub sites (n = 26 and 34) while at the mixed site, bobcats were only observed six times. Though these cameras were at the minimum distance for camera placement, they had a disproportionate number of occurrences. The disproportionate number of bobcat occurrences on the south side of SH100 may be a function of preference of habitat selection on a very fine scale (Mysterud and Ims 1998, Corlatti et al. 2009, Thurmond 2014). The differences in bobcat occurrences indicate a strong association with vegetation cover, which may be used to focus on promoting bobcat (and potentially ocelot) presence or absence at a site.

Fortin et al. (2013) reported that wildlife may aggregate around anthropogenic features to use habitat edge found near these areas. Additionally, after crossing a barrier, individuals often redistributed themselves over limited distances due to the high costs of moving to new locations (Fortin et al. 2013). This redistribution of individuals is usually not equal, as wildlife aggregate in areas with better habitat (Mysterud and Ims 1998, Fortin et al. 2013). This seems to be true for the SH100 mitigation corridor, as the hypothesis that there would be a higher number of occurrences and greater species richness closer to wildlife crossing structures than farther away, was not supported. Distance to SH100 was only a significant factor for nine-banded armadillos, and distance to nearest WCS was only significant for bobcats. Bobcats seemed to primarily congregate near WCS3A, supporting findings of Tewes and Hughes (2001) that bobcats prefer smaller culverts with more vegetative cover. Thurmond (2014) also found that bobcats in high plains Texas ecosystems preferred areas ≤ 1 km from anthropogenically-impacted areas, and preferred areas near a mixture of habitat types. Since bobcats were the only species that occurred in higher numbers closer to WCS this may indicate that, two years after construction completion, WCS may not be known to enough individuals to create an aggregating effect around structures. As time goes on this will likely change, as more individuals learn about the access provided by the WCS and remain nearby.

The hypothesis that proximity to WG will have no effect on occupancy or richness of target species at control sites was supported, with only Virginia opossum and collared peccary seeming to be associated with WGs. While the association of Virginia opossum near wildlife guards (WGs) is well supported by data collected from cameras at WG, collared peccary clustered around a single structure, WG12 (Figure 4) (Kline et al. 2019). Dense thornscrub directly hedges against WG12 and the wildlife exclusion fencing on the edge of the roadway. Control sites within this area had the highest number of collared peccary occurrences. A permanent gate was installed at WG12 before the control array was implemented. This gate is effective at blocking mid- and large-sized wildlife from using or entering the WG (Peterson et al. 2003). Collared peccary were not observed at WG12 during the study period, are not expected to be able to circumnavigate the gate, and were not observed crossing or interacting with any WG during the study period. Therefore, the association of collared peccary and WG is due to the habitat immediately surrounding the mitigation structure, and not the structure itself. Virginia opossum can squeeze through the gate and are regularly seen accessing the site. Virginia opossum were also the most frequently captured species at WG, with 477 crossings at all the WGs between December 2018 and October 2019 (Kline et al. 2019).

While ground cover appeared to have little impact on encouraging or discouraging wildlife, canopy cover showed a strong association with increasing species richness and the presence of most species. These results indicate that canopy cover may be a possible indicator for using vegetation as a primary factor for future placement of mitigation structures within south Texas (Abrahms et al. 2017). While vegetation may not be a strong indicator that bobcat, collared peccary, eastern cottontail, northern bobwhite, nine-banded armadillo, Virginia opossum, and white-tailed deer will be seen at mitigation structures, it does indicate that these are the areas where these species might congregate and therefore increase the likelihood that individuals will discover the presence of structures. These findings may also be used at existing mitigation structures, where vegetation may be planted at existing structures to increase the available canopy cover, possibly encouraging new species to use those areas (Tewes and Hughes 2001). The inverse may also be applied, if canopy cover is reduced to a minimum around WGs it may serve as an additional barrier for wildlife, discouraging them from approaching the mitigation structure in the first place. These results support the suggestions made by Tewes and Hughes (2001), that to promote ocelot use dense vegetation should surround the entrance and exit to WCS. Additionally, by designing road mitigation corridor for ocelots, researchers will likely promote use by the majority of south Texas species.

CHAPTER IV

DETERMINING PERMEABILITY OF THE STATE HIGHWAY 100 MITIGATION CORRIDOR

Introduction

Monitoring wildlife movement at wildlife crossing structures (WCS) is relatively simple. Usually, cameras are placed facing the entrances and exits of the structure, ensuring full coverage of wildlife movement and calculation of the actual crossing rate. The actual crossing rate equals the number of times the structure was successfully crossed by an individual, and is usually calculated on a species basis (Andis et al. 2017). While knowing the actual crossing rate provides evidence that the structure is being utilized by wildlife, it provides no empirical evidence for understanding the factors effecting crossing rate (Andis et al. 2017). However, actual crossing rate is necessary for determining these factors, and may be used to calculate structure performance (Andis et al. 2017). Knowing structure performance allows researchers to compare potential road mitigation structure designs and placements to maximize ecological and monetary investments of road mitigation projects (van der Grift et al. 2013). Road mitigation structure performance is based on the influence of location and design on actual crossing rates (Andis et al. 2017). The influence of location is important as it accounts for the number of times a structure is expected to be visited by a species during a given period. Once an individual has approached a crossing structure, design influences how likely that individual is to utilize that structure. These two factors are added together to determine structure performance (Andis et al. 2017).

Roedenbeck et al. (2007) and Popescu et al. (2012) stated that before-after-control-impact (BACI) study designs have the highest inferential strength for assessing the influence of road effects and how population and landscape factors influence crossing frequency at road mitigation structures. However, in many studies, pre-construction monitoring is often limited or may not be available at all (Clevenger 2005, van der Grift et al. 2013). In the case of the State Highway 100 (SH100) mitigation corridor in Cameron County Texas, before-during-after data was collected at wildlife crossing structure locations and reported by Cogan (2018); but a control-impact study was not implemented at that time. The primary target species for the SH100 mitigation corridor was ocelots, and prior to the construction of the road mitigation corridor SH100 was a known source of ocelot mortalities. The preventive structures within the corridor consist of 11.9-kilometers of wildlife exclusion fencing and 18 wildlife guards (WGs). To maintain wildlife movement across the constraining barrier of the road corridor, five WCS with four different designs were modified or installed, with construction completing in January 2018 (Figure 2). Any species that may utilize a WCS to safely cross SH100 was considered a secondary target species. Understanding the performance of this road mitigation corridor for primary and secondary target species was a necessary next step in project monitoring.

In some cases, a control-impact study design may be implemented post-construction to effectively determine road mitigation structure performance (Hardy et al. 2003, van der Grift et al. 2013, Rytwinski et al. 2015, Andis et al. 2017). Andis et al. (2017) and van der Ree et al. (2015) have suggested that, once construction has been completed, comparing crossing rates of wildlife at the mitigation structure to wildlife abundance in the surrounding habitat is the most empirical method for determining the influence of location on crossing rates (Andis et al. 2017). If actual crossing rate is determined by monitoring the crossing structure (impact) and the influence of location is determined using a control array in the surrounding habitat, the remaining structure performance is the influence of structure design (Andis et al. 2017). Armed with this knowledge, researchers may determine which known design and location is the most effective for target species so that future structures can emulate and incorporate those factors.

Andis et al. (2017) suggested a control-impact design where expected crossing frequencies are calculated by using a control array placed around and compared to a road mitigation structure (impact). If a roadway were not present, wildlife would be expected to utilize the area at the same rate they are observed moving within the surrounding habitat. When compared to movement at a WCS, if the structure is completely mitigating the roadway wildlife should move through the structure at a similar rate to surrounding habitat (Andis et al. 2017). The comparison of expected crossing frequencies to actual movement rates at the WCS eliminates the influence of location and provides performance differentials (PD) for the structure, calculating the influence of design. Positive PD indicates animals utilize the structure more often than expected; therefore, the design of the structure encourages wildlife to cross. Negative PD indicates animals cross less often than expected, likely meaning that the design discourages or physically prevents wildlife from crossing (Andis et al. 2017). If wildlife were observed passing through the structure and the performance differential is close to zero, individuals are moving through the structure at similar rates to that seen in the surrounding area. Multiple road ecology studies have suggested that when WCS are placed in conjunction with mitigation fencing, fencing will funnel wildlife from the surrounding area to the structure (Huijser et al. 2016, Jakes et al. 2018, Seidler et al. 2018). Under this assumption, if road mitigation structures are functioning as intended and are creating additive effects, each structure should have a positive performance differential for all target species within the area.

The goal of many road ecology projects is to reduce wildlife mortalities while maintaining wildlife movement and genetic connectivity across the barrier(s) (van der Grift et al. 2013). Understanding the minimum level of wildlife movement necessary to maintain viable populations requires in-depth genetic studies (Clevenger and Waltho 2005); however, performance differentials may be used to estimate how permeable the roadway is for each species. Researchers may then determine if the permeability of a roadway needs to be increased to ensure mitigation goals are achieved and provide a relative indication of the number of essential WCS on a future road mitigation corridor. Andis et al. (2017) used percent difference to determine the permeability of the road mitigation corridor. Percent difference is calculated by dividing the performance differential by the total number of occurrences of a single species within the control array and multiplying by 100. A negative percent difference indicates the roadway is a constraining barrier for the species, any value between 1 and 100 indicated the roadway is permeable for the species but does not allow full connectivity, while more than 100% indicates the species is likely drawn to the roadway as they use structures more often than the surrounding habitat (Andis et al. 2017). Finally, the amount of permeable roadway (or wildlife crossing structures) required to ensure full connectivity may be determined by comparing the actual crossing rate at the wildlife crossing structures to the total occurrences in the control array (Andis et al. 2017).

When conducting a control-impact study on a homogenous landscape, the surrounding vegetation accurately reflects vegetation observed at the crossing structure and the control array may be placed randomly within the surrounding area (van der Ree et al. 2015, Andis et al. 2017).

If sites are spaced to maintain independence, precise control site placement may vary depending on the research question and desired survey effort. However, in heterogeneous landscapes or in fragmented habitats, placing control sites randomly around WCS may increase bias when determining the expected crossing frequencies of some species (van der Ree et al. 2015).

The primary target species of the State Highway 100 (SH100) road mitigation project was ocelots and the presence of ocelots in south Texas is strongly associated with the presence of Tamaulipan thornscrub (Haines et al. 2006b, Booth-Binczik et al. 2013).The vegetation around SH100 was not homogenous but a diverse ecosystem consisting of native grasslands, Tamaulipan thornscrub, sea oxeye daisy flats, and mixed cactus, with vegetation cover varying significantly on a fine scale (Ewing and Best 2004). Placing a random control array based on proximity to the crossing structure was not feasible, and a new placement method had to be developed. van der Grift et al. (2013) noted that, in areas where a proximity-based control-impact array may not be feasible, one based on an important landscape factor may be used. To better estimate expected crossing frequencies within this heterogeneous landscape, control sites were placed in areas with vegetation similar to that surrounding WCS.

Objectives & Hypotheses

The primary goal of the State Highway 100 (SH100) mitigation corridor was to reduce ocelot road mortalities. A secondary goal of the SH100 mitigation corridor was to maintain ocelot movement, reducing the barrier effect of the mitigation structures. Though this project was focused on ocelots, structures were also meant to benefit the wider variety of wildlife within south Texas. The objective of this research was to determine the permeability of the SH100 road mitigation corridor for wildlife species within the surrounding habitat. The hypotheses tested were as follows: 1) Performance differentials based on proximity to mitigation structures will not be significantly different than performance differentials based on vegetation categorization surrounding the mitigation structures. 2) Performance differentials calculated will be positive for all target species at wildlife crossing structures. 3) Performance differentials will be negative for all target species at wildlife guards. 4) The SH100 mitigation corridor serves as a "movement barrier" and not a "movement constraint" for all target species and allow full connectivity for all target species, as defined by Beyer et al. (2016).

Methods

Following a control-impact design described by van der Ree et al. (2015) and Andis et al. (2017), two arrays were used to calculate performance differentials for State Highway (SH100) mitigation structures. The impact array monitored actual wildlife use at wildlife crossing structures (WCS) and wildlife guards (WGs). Only complete crossings of target wildlife species from one side to the other were considered a successful use of the structure. Complete crossings were then compared to wildlife occurrences within the surrounding area, which were calculated using a control array. Occurrences within the control array were summed for comparison using two methods, proximity and vegetation. The proximity dataset used occurrences from control sites closest to the compared impact site. The vegetation dataset used occurrences from control sites placed in similar vegetation to that seen at the compared impact site. Due to structure design, not all species were expected to be observed at all WCS. Based on their abundance within the surrounding area, 13 target species were expected to be observed at mitigation structures, black-tailed jackrabbits, bobcats, collared peccary, coyote, eastern cottontail, nilgai, nine-banded armadillo, northern bobwhite, northern raccoon, striped skunk, Texas tortoise, Virginia opossum, and white tailed deer. The design of WCS3 and WCS4 is large enough to allow crossings these species. However, WCS3A was considered too small for either nilgai or white-tailed deer passage. Therefore, these two species were not expected to utilize WCS3A. Instead, only 11 species were expected to be observed crossing at WCS3A. Other species were observed in the control and impact arrays but were not included as target species due to their low occurrences within the complete dataset. Due to the rarity of ocelots on the south Texas landscape, bobcats were considered the closest approximation for ocelot in terms of body size for use around SH100.

Study Design: Impact Array

Cameras monitoring the mitigation structure were designated the impact array. Prior to the instillation of the mitigation corridor, sites of future WCS were monitored using wildlife camera traps until construction completion in January 2018. During construction, Cogan (2018) determined optimal camera placement and settings for monitoring actual rate of use at all mitigation structures. The impact array was specifically placed to capture the complete entrance and exit of each WCS and WG, as well as a small portion of the surrounding habitat. Cameras were aimed and spaced to fully cover the entrance and exits of WCS, allowing researchers to confidently determine actual wildlife use at each structure. Entrances to WGs were monitored with two cameras, both facing the entirety of the WG. All cameras were placed 30 to 50 cm above the ground, a height chosen to maximize the probability of detecting ocelots, bobcats, and medium-sized mammals (Cogan 2018).The impact array was composed of 84 Reconyx PC900 Hyperfire Professional Covert Camera Traps, with an array of settings from single picture to video (Reconyx Corporation, Holmen, WI, USA). Cameras were checked every two weeks to replace batteries and SD cards, and to remove any problem vegetation within camera facing.

Study Design: Control Array

The control array was a mix of 36 wildlife cameras (also used in Chapters II and III): 16 Moultrie (Moultrie MCG-13270, Manufacturer, city, state, country) and 20 Bushnell (Bushnell Aggressor 119874 Bushnell Inc, city state country). One camera was placed at each site. All sites were within 150-meters of SH100 and on Laguna Atascosa National Wildlife Refuge, Cameron County, Texas, USA. Sites were distributed equally on the north and south sides of SH100, resulting in 18 cameras on the north and south sides of the highway. A stratified random design was used for camera placement, with potential placement sites determined in ArcGIS prior to visitation. A 50-meter grid was overlaid across the entire study area, and each potential site was categorized into three vegetation categories: dense thornscrub which required > 70% tree cover measured on the ground, mixed thornscrub with 40 – 70% tree cover measured on the ground, and open grassland had < 40% tree cover when measured on the ground. Sites with greater than 40% grasses were discarded as potential sites due to the high camera trap maintenance as this vegetation cover poorly represented vegetation surrounding mitigation structures. Vegetation was used as the predominant placement factor with the 36 control sites randomly picked but equalized between the three vegetation categories, resulting in 12 sites per class with six sites on either side of SH100. All sites were visited once a month to replace batteries and SD cards and to trim any vegetation within three meters of camera facing.

Camera Settings

Cameras within the impact array were programmed to take a burst of three photos per trigger, with one second intervals between pictures and no time between trigger events. Using a photo burst allows wildlife to fully enter the frame when the second or third picture are taken, potentially increasing the number of successful triggers (Cogan 2018). Cameras in the control array were set to take a single picture per trigger with no delay between triggers. Bushnell cameras were set to the lowest interval setting of 0.6 seconds, while Moultrie were set to the lowest interval setting of “None,” which was provided by the manufacturer as 1.3 seconds.

Cameras within the impact array were set to their highest PIR setting. Choosing this setting was recommended by the manufacturer, as vegetation at impact sites could be carefully controlled and removed for roughly five meters directly in front of the camera (Reconyx Corporation, Holmen, WI, USA). Vegetation was not cleared more than three-meters in front of cameras within the control array, therefore cameras within this array were set to “Low” PIR setting in accordance with manufacturer recommendation (Bushnell Corporation, KS, USA, EBSCO Industries, Birmingham, AL, USA). Due to the differences in effective sensor range between the control and impact arrays, capture distance was included within the equation for determining expected crossing frequencies.

Photo Processing

Photos taken for the control and impact arrays from December 2018 to October 2019 (317 camera trap nights each array) were used in analyses. All photo data from the control and impact arrays were processed in the same manner. Photos were renamed using the program ReNamer to rename capture file names to the date and time the capture was taken. Captures were then organized by site and sorted by taxa. DataOrganize was used to create a text file with camera location, species, date, and time. Finally, the camtrapR package in Program R 3.4.3 was used to create summary statistics from the camera trap data, which determined an occurrence as a

30-minue period based on the last capture of an individual of the same species (O'Brien et al. 2003, Niedballa et al. 2016). Only multiple individuals caught in a single capture were counted as multiple individuals. Number of occurrences was the data used for the control array, while the impact array was further categorized based on wildlife interaction with the structure.

From the impact array, only data from full crossings were used in analysis. A completed crossing at a WCS occurred when at least one individual entered one side of the roadway, crossed completely under the road corridor, and exited the other side of the structure. A completed crossing at WGs was considered when at least one individual entered the WG in any manner (either crossing above the surface of the WG or in the well below grade), crossed the barrier, and exited the other side. This included crossings both into and out of the road corridor, as WG are expected to work as an effective barrier regardless of direction of travel.

Calculation of Expected Crossing Frequencies and Performance Differentials

To compare the crossing performance for each structure, performance differentials (PDs) were calculated for the most common species at each structure based on the work by Andis et al. (2017), Clevenger and Waltho (2000), and van der Ree et al. (2015). To find the performance differential, expected crossing frequencies (ECFs) for each species were first calculated, as these are the expected movement of each species within the road-effect zone and the baseline for determining performance (Andis et al. 2017). Expected crossing frequencies were determined for each species and for each crossing structure. Two types of ECFs were calculated using photo data collected from the control array; these two ECFs used different methods for determining which sites from the control array would be compared to the WCS. The first method used the total number of occurrences for each species at control sites within the same vegetation category as the structure, regardless of distance to the structure. These are the vegetation-based expected crossing frequencies. The second method calculated expected crossing frequencies using data collected from control sites closest to the WCS they were being compared to, regardless of vegetation category at the control site. These are the proximity-based expected crossing frequencies. ECFs were calculated using an equation derived from Andis et al. (2017), but modified to include the expected sensor range of cameras within the control array and width of compared structure:

$$ECF_i = \frac{\sum X_i}{(\sum d_c \div d_w)}$$

The sum of the number of independent occurrences for species i (X_i) was divided by the total area visible by the subset of control sites compared to that crossing (d_c) divided by the compared structure width (d_w) (actual values may be found in Appendix III, Table 5). To provide ECFs with some biological importance, it was subtracted from the observed crossing rates at the structure, providing the performance differential for a specific species (PD_i) (Andis et al. 2017).

Performance differentials (PDs) were calculated using expected crossing frequencies from both the vegetation (VPD) and proximity methods (PPD) by subtracting the ECF for species i (ECF_i) from actual crossing rates of the species at the structure (ACR_i),as follows:

$$PD_i = ACR_i - ECF_i$$

Performance differentials were calculated for the three wildlife crossing structures within the bounds of the control array, WCS3, WCS3A, and WCS4. The nearest control sites were multiple kilometers from WCS1 and WCS2 because these structures lie west of the study area, therefore it was impossible to compare PPDs against VPDs. However, due to the nature of vegetation-based expected crossing frequencies, VPDs could be calculated. The control array was also compared to seven wildlife guards, all of which were close to or within the control array study area, WG11, WG12, WG13, WG14, WG15, WG16, and WG17 (Figure 4, listed west to east). Structure widths were measured at the base using a meter tape.

To determine if both methods for calculating expected crossing frequencies estimated similar results, PPDs and VPDs were compared by WCS using an ANOVA with type III correction. If the results of the ANOVA were significant a Tukey's HSD post-hoc test was used to compare means for PPDs and VPDs for each crossing structure but was not compared between crossing structures. For example, the PPD for WCS3 was compared to the VPD for WCS3 and was not compared to the VPD for any other structure. All statistical analyses were conducted in Program R 3.4.3 using the stats package.

Permeability Estimates of State Highway 100

Permeability estimates were based on methods developed by Andis et al. (2017). To create a permeability estimate for the entire road mitigation corridor, analysis was conducted only using VPDs. First, the percent difference was calculated for each species. The percent difference between rates at the wildlife crossing structures and the control array was calculated by dividing the vegetation-based performance differential for species i (VPD_i) by the sum of occurrences of the species in the control array ($\sum X_i$) and multiplying by 100, as follows:

$$\%\ Difference\ for\ Species_i = \left({}^{VPD_i}\!/\!_{\sum X_i}\right) \times 100$$

This percent difference provides permeability estimates for the SH100 mitigation corridor. If the ratio resulted in the undefined value zero divided by zero, it was reported as zero. These calculations were not statistically compared but were used to infer whether the species was observed more at the mitigation structures than in the surrounding habitat, or vice versa.

The percent of permeable roadway (or wildlife crossing structures) predicted for 100% connectivity, where wildlife may traverse the mitigation corridor as if the roadway were not present, was also calculated. This analysis was based on a formula from Andis et al. (2017):

$$\%\ Permeable\ Road\ for\ Species_i = \left({\sum Y_i \div d_1}\Big/{\sum X_i \div d_2} \right) \times 100$$

Where: $\sum Y_i$; = the total number of occurrences of the species within the impact array; d_1 = the summed width of the entrances to all of the WCS in meters; $\sum X_i$ = the total number of occurrences of a species within the control array; and d_2 = the total usable range of the control array in meters. This analysis was conducted on data using the vegetation-based occurrences from the control array and was not compared using any statistical analysis but for inferring how effective the road mitigation corridor is for each species.

Results

Results of the ANOVA comparing performance differentials calculated for all wildlife crossing structures (WCS) combined using the number of occurrences based on proximity and vegetation showed that the results of these methodologies did not differ significantly (F = 0.043, degrees of freedom = 74, p = 0.8389) although there were numerical differences in PD depending on method (Table 8). Thus, all performance differentials reported through the rest of this section are based on the vegetation-based expected crossing frequencies only. Performance differentials based on the vegetation-based expected crossing frequencies allow for the greatest comparison, since they may be used for structures that lie outside the boundaries of the control array.

Northern bobwhite, northern raccoon, Virginia opossum, and white-tailed deer were observed utilizing WCS3 more frequently than surrounding vegetation (Table 8). Therefore, 31% of the species expected to use WCS3 had a positive performance differential (PD). The only positive white-tailed deer performance differential was observed at WCS3 (PD = 48.8). Of the species expected to be observed utilizing WCS3A, 54% of them had a positive performance differential at the structure. Bobcats, collared peccary, coyote, nine-banded armadillo, northern raccoon, and Virginia opossum were seen more often at WCS3A than in the surrounding vegetation. The only positive bobcat performance differential was recorded at WCS3A (88.83; WCS3 = -2.42; WCS4 = -0.33). Seven species, including being coyote, eastern cottontail, nine-banded armadillo, northern bobwhite, northern raccoon, striped skunk, and Virginia opossum, were observed more frequently at WCS4 (Table 8) than in the surrounding area. Of the 13 species that were expected to use WCS4, 54% of them had positive performance differentials. Black-tailed jackrabbits, nilgai, and the state-listed Texas tortoise had negative performance differentials at all WCS and WG.

WCS1 and WCS2 did not have vegetation within 50 meters of the structure entrance and were therefore considered in open grassland. Bobcats, northern raccoons, and Virginia opossum were seen at these structures more often than in areas with the same vegetation cover. For WCS2, these were the only species observed crossing at the structures, while WCS1 also had 5 coyote crossings, however, this was lower than the expected crossing rate.

Wildlife guards appear to be an effective tool at discouraging and preventing wildlife from crossing them, as 54% of all species only had negative performance differentials (Table 9).

Seven species, black-tailed jackrabbits, bobcats, collared peccary, nilgai, striped skunk, Texas tortoise, and white-tailed deer, only had negative performance differentials at all wildlife guards (WGs) (Table 9). Three other species had more negative values than positive values, which included coyote, eastern cottontail, and nine-banded armadillo, suggesting wildlife guards were moderately effective for these species. Performance differentials for northern raccoons and Virginia opossum were almost entirely positive, with northern raccoons only having two negative values and Virginia opossum only having one. Based on their positive performance differentials, WG are permeable to northern raccoons and Virginia opossums.

Only WG12 had completely negative values (Table 9). However, during the time of this study, a permanent gate was installed at WG12 that greatly reduced wildlife use. The next lowest, WG13, had three species with positive performance differential. Three structures, WG14, WG15, and WG17, had four species with positive differentials and WG11 had five.

Of the 13 target species compared using performance differentials, collared peccary (62%), coyote (19%), northern raccoon (1,830%), and Virginia opossum (324%) had positive percent differences, indicating current mitigation structures allow these species some permeability within the mitigation corridor (Table 10). For northern raccoon and Virginia opossum, the percent difference was above 100%, indicating they are using mitigation structures more than the surrounding area. Eastern cottontail (116%), nine-banded armadillo (112%), Texas tortoise (1,594%), and white-tailed deer (105%) had values that indicated SH100 needs to be more than 100% permeable to allow complete connectivity, indicating these species would require WCS of different designs (Table 10). Northern raccoon (2%) and Virginia opossum (9%) were the closest to having full connectivity (Table 10). Bobcats would require 70% of the roadway to be permeable to allow full connectivity.

Discussion

Andis et al. (2017) and van der Ree et al. (2015) suggested a study design that placed control sites within average daily animal movement range of the wildlife crossing structures (WCS) and within similar vegetation. While this control array design is likely the most empirical, it creates a limiting factor that cannot be overcome for many studies that would strongly benefit from a control array. The hypothesis that performance differentials based on proximity to mitigation structures will not be significantly different than performance differentials based on vegetation categorization surrounding the mitigation structures was supported, suggesting that a vegetation-based array may be used for calculating performance differentials. Placing a control array based on vegetation-based factors provides studies with additional freedom when attempting a control-impact study design (van der Grift et al. 2013). For some studies, where the proximity-based method is unusable, control arrays based on other methods may provide equal or superior results (van der Grift et al. 2013).

The vegetation-based method is especially important for projects like this one, with limited land access and a specialist primary species. Available study area only included land managed by US Fish and Wildlife Service, which did not include two of the five WCS and excluded a third of the mitigation fencing (Figure 4). A control-impact study with control sites placed near WCS1 and WCS2 was not possible due to restrictions on land access. For studies without land access at certain sites, control-impact studies based on vegetation or other important landscape factors may be the best alternative. A stratified random design is very effective for these study limitations, as it allows researchers to place control sites in important locations but remove site placement bias (van der Ree et al. 2015). In the case of this project, a randomly placed control-impact study based on proximity would not place enough control sites in dense thornscrub to be able to accurately predict ocelot presence or use (Appendix 1), or the wildlife communities associated with that vegetative cover (Harveson et al. 2004). These findings provide researchers with the flexibility to attempt a wider variety of control-impact studies on road mitigation projects that may provide important empirical results for future conservation measures.

Not all species had positive performance differentials at all wildlife crossing structures thus not supporting hypothesis that performance differentials will be positive for all target species at WCS. However, the performance of all five WCS could be compared to determine which structure provided the most permeability for each species (Table 10). These performance differentials provide support for the use of multiple structure designs on a single mitigation project. WCS1, WCS2, and WCS3A all seemed to specifically support bobcat movement above what would be expected within the same vegetation category. Based on body size similarities, perhaps ocelots would use these structures as well. Based on Tewes and Hughes (2001) the culvert design of WCS1 and WCS2 are not ideal for encouraging ocelots however,Cain et al. (2003) found the presence of catwalks promoted felid use. WCS3 also plays an important role for the mitigation corridor, as it is the only structure where white-tailed deer have been observed crossing (Table 8). This is likely due to its height and width, which have been found to encourage ungulate use (Clevenger and Waltho 2000). As this crossing structure is fairly centrally located, it may provide enough connectivity for deer to remain genetically viable on both sides of the State Highway (SH100) mitigation corridor (Figure 4), though this cannot be determined without an in-depth long-term genetics study. WCS4 proved valuable with the highest number of species that utilize the crossing structure more than expected. However, none of these species were a primary target species for this project (felids).

Though all five crossing structures are important, and the four designs appear to suit a variety of species, performance differentials were not positive for all species at all WCS; resulting in the rejection the hypothesis that the SH100 mitigation corridor serves as a barrier and not a constraint for all target species. Design or age of the structure has been shown to have a large impact on actual crossing rates at the structure (Clevenger et al. 2003, Ford and Clevenger 2010). The difference in performance differentials between various structures of the same type was sometimes different by an order of magnitude. For 8 of the 13 species, more than half of their actual crossing use was at WCS3A (Table 8 and Appendix III, Table 6). For bobcats, collared peccary, coyotes, eastern cottontail, nine-banded armadillo, northern bobwhite, northern raccoon, striped skunk, and white-tailed deer, most of their crossings were observed at a single structure. Such high numbers at a single structure provide evidence that there were subpopulations around these WCS that are using these structures at a high rate. While this may benefit the species on a local level, it does little to provide the genetic connectivity required to maintain isolated populations (Epps et al. 2005, Beyer et al. 2016). Without the capacity to identify individual animals at crossing structures, it is impossible to know whether most crossings at these structures are by local subpopulations or dispersing individuals. If these are subpopulations utilizing the structures instead of dispersing individuals, the mitigation corridor along SH100 may currently be serving as a constraining linear barrier for many species.

While having five WCS with four designs may not promote increased use by a single species as much as expected, it allowed a diverse group of species to utilize at least one structure more often than expected. Clevenger (2005) suggested planning road mitigation structures for ecosystem health as opposed to promoting use by a single species. Having diverse crossing structure designs supports the wildlife community around the roadway more than if there were multiple crossing structures of a single design. For example, WCS3A had the highest bobcat performance differential. If the four other crossing designs utilized within the SH100 mitigation corridor were instead replaced with designs similar to WCS3A, the performance differential equation would calculate that felid movement would be expected to quadruple. However, without diverse designs, three species would likely be constrained by the road mitigation corridor. Eastern cottontail, northern bobwhite, and white-tailed deer are currently crossing at other structures but not at WCS3A. While designing a corridor based only on WCS3A may increase felid movement, it may have unintended cascading effects that could decrease the viability of an ocelot population. Lagomorphs are an important food source for coyotes in south Texas (Anderson and Pelton 1976, Andelt 1985). If lagomorph populations suffer due to the constraining barrier of a roadway, coyotes may be forced to seek alternative food sources (Henke and Bryant 1999, Tewes and Hornocker 2007). Stressed coyotes have been known to attack and kill felids (Blankenship 2000). The potential for cascading effects forces researchers to strongly consider the goals for a mitigation project; especially whether to manage for ecosystem health or the benefit of a single species.

Performance differentials were not negative for all species at all wildlife guards (WG) therefore the hypothesis that performance differentials would be negative for all target species at WG was rejected. While species such as northern raccoon and Virginia opossum seem to actively seek out and utilize WG, many species avoided WG, such as collared peccary, nilgai, Texas tortoise, and white-tailed deer. These results were not surprising when compared to the results of other studies into the effectiveness of WG, such as Allen et al. (2013) and Cogan

(2018), which indicated pawed species may be able to easily cross WG. However, the most significant findings are those related to bobcats and coyotes, as they are carnivores with padded feet and are likely surrogates for estimating the ability of an ocelot to cross a WG (Cogan 2018). Both bobcats and coyotes regularly crossed wildlife guards; however, bobcats had no positive performance differentials. Most bobcat crossings occurred at WG13, where an individual was observed crossing the guard from habitat to roadside, spending a short time within the road corridor, and then crossing WG13 back to the habitat side. These results indicate that, while WG likely do little for discouraging bobcats from crossing into the road corridor, bobcats may choose not to cross for other reasons. Bobcats may be learning of alternative safe crossing paths and are only choosing to cross WG to specifically enter the road corridor without intending to cross.

Most of these findings are preliminary, as the SH100 mitigation corridor is only two years old, and it is unlikely that it would be a complete success so soon after construction. van der Grift et al. (2013) estimates that to have an 80% or higher probability of detecting an effect of road mitigation structures, the structures should be at least three years old, and researchers may not see an effect for 12 years or more. van der Grift et al. (2013) also suggests that if the assessment endpoint of the study is population viability, it is extremely important to start or continue monitoring for multiple years after project completion. The worst-case scenario is to study for too short a time to detect any real effect and drawing incorrect conclusions (van der Grift et al. 2013). Haines et al. (2006b) predicted that dispersing ocelots will likely be young males, which usually do not disperse until the carrying capacity in an area is met. Additionally, they appear to follow dispersal corridors consisting of connected patches of dense thornscrub (Beier and Noss 1998, Berger 2004). It may take years for the ocelot population to become stable enough to encourage enough dispersing individuals to travel from the Cameron County population on Laguna Atascosa National Wildlife Refuge to SH100. Now that the results of this research have shown that the control-impact study based on vegetation is effective at monitoring the area around the WCS, the SH100 monitoring project could continue to utilize this study design for multiple years, allowing researchers to observe the real effect of the road mitigation structures to better design and implement ocelot-focused road mitigation projects.

CHAPTER V

CONCLUSIONS

This project supports the argument made in van der Grift et al. (2013) and Andis et al. (2017) that the empirical benefits of a control-impact study outweigh the additional effort. Though effort for control arrays may be high, it provides invaluable information for future road mitigation projects, providing a long-term investment for researchers. The first of which is creating a database of expected crossing frequencies that may be compared to a smaller, longer-lasting observational study at the crossing structure (Clevenger 2005). While not ideal, this study design would allow researchers to use a relatively short-term control array for comparison against an impact array that lasts for many years, determining if performance differentials change over time. Control arrays also estimate the health and viability of surrounding wildlife populations, which may require additional management practices before road mitigation projects become fully effective (van der Ree et al. 2015). For example, if the population around the road mitigation project is distributed differently than predicted, researchers may be able to change the landscape factors surrounding the structures (such as vegetative cover or available water sources) to better reflect the features where wildlife are present. This may promote visitation on a micro scale, redistributing individuals across the landscape, potentially closer to conservation efforts and away from threatening areas.

Control-impact studies also promote the road ecology field by providing an empirical and necessary step into understanding structure design. By removing the influence of location on actual crossing use, additional projects may determine the influence of structure design (Andis et al. 2017). Optimizing structure design for target species allows researchers to maximize the immense investment they are making when installing road mitigation corridors. van der Grift et al. (2013) suggested that without a larger data pool and relevant understanding of the factors that control wildlife crossing structure use, effective designs and mitigation measures cannot be determined on a species basis. Until this data pool is created, future research projects will likely unnecessarily waste funds on designs that do not maximize the potential conservation effort for the target species of the mitigation project.

For example, to promote ocelot use at underpass-style wildlife crossing structures (WCS), Tewes and Hughes (2001) suggested designing underpasses with no standing water. While ocelots were not seen around or at structures during this study period, the surrogate for ocelots, bobcats, were abundant. Bobcats utilized WCS1 and WCS2 at higher numbers than expected (Appendix III, Table 6), both of which have permanent water flow (Figure 2). The WCS1 and WCS2 are of the same design and have catwalks higher than average water level, however these are often difficult to access without doing some wading. Bobcats were frequently documented utilizing these dry paths. During periods of heavy rain WCS3 would contain standing water with dry side paths and bobcats were still seen crossing, though less frequently than expected (Figure 2). These findings indicate that below-grade underpass style crossings do not need to be designed to preclude standing water if a dry path is provided, greatly reducing design costs (Cain et al. 2003, Niemi et al. 2014). By including structures near moving bodies of water, researchers may create a cascading positive effect for felids that may be seen during stressful periods. For example, Jaeger and Fahrig (2004a) suggested that during times when prey species are scarce (such as during a drought), ocelots will likely follow the optimal foraging theory and seek out less profitable prey. Additionally, during times when prey is less frequent Texas felids have been known to increase their home range by 100% (Elizalde-Arellano et al. 2012, Bliss-Ketchum et al. 2016). Areas with running water provide diverse habitats, allowing for a greater variety of less profitable ocelot prey (Nielsen et al. 2010, Bliss-Ketchum et al. 2016, Paolino et al. 2018). While a WCS with permanent running water may be visited less by felids than a dry structure in dense thornscrub, the structure with permanent water may be designed to provide a positive performance the majority of the time and a link to habitat used during times of extreme stress (Planillo et al. 2017). By understanding and utilizing both design types, ocelot researchers can maximize the investment of future road mitigation structures.

The results from this study also support the concept that the difficulty and effort required for a control array is likely directly correlated with the size of the entire project. The impact array for this project contained 84 cameras and took about 230,000 photos each month, requiring a high amount of survey effort. By comparison, the control array contained 36 cameras and took about 130,000 photos each month, with heavy skewing during the summer months. While this is a substantial sum, this was also a test array, not designed to completely minimize survey effort. During the cooler parts of the study period and outside the growing season, the total number of pictures from the control array never rose above 50,000 pictures per month. Additionally, researchers could face cameras in the direction that would reduce the total number of false captures to a more manageable level. For the empirical benefits of a control-impact study design, the additional effort required from the control array was acceptable and manageable. However, most road ecology projects are much smaller than 84 cameras, and an additional array would make up a much larger percentage of the total cameras within the field. This may be avoided, however, by taking a similar approach to Andis et al. (2017). Instead of placing a large array into the field for extended periods, cameras may be rotated from one location to another, allowing for fewer sites in the field at a time. This method may work well for studies that lack the funds or workforce to sort a high number of pictures in a short amount of time, instead sub-setting those pictures into more manageable amounts and periods.

Road mitigation projects will hold an important role in maintaining and preserving the United States ocelot populations (Haines et al. 2007). However, the role road mitigation projects will need to play may be different than goals for most road ecology projects (Pullinger and Johnson 2010, Patten and Burger 2018). While many projects focus on maintaining transient wildlife movement between subpopulations, this may not be the most productive route for ocelot conservation. Haines et al. (2006b) advised against attempting to connect the Cameron County and Willacy County ocelot populations, determining that it would not provide a lasting benefit to ocelot conservation. However, Haines et al. (2006b) does suggest the incorporation of additional road mitigation projects and an increased study of ocelots near roadways and crossing structures. Most of the ocelots remaining within the United States are in isolated subpopulations of ≤ 10 individuals and reside within smaller patches of habitat within the highly fragmented landscape (Haines et al. 2006b). These subpopulations do pose a higher risk of extinction than subpopulations within areas of larger continuous habitat; however this habitat type is quickly disappearing in the Lower Rio Grande Valley. Lombardi et al. (2020) predicts that by 2050, urban development will become the predominant land cover type in the Lower Rio Grande Valley, which will require denser road networks and higher traffic volumes. This change will be especially detrimental to ocelots, as Lombardi et al. (2020) also forecasts the loss and fragmentation of existing woody cover.

Instead of attempting to connect the larger ocelot populations, Haines et al. (2006b) suggested connecting areas with large patches of remaining cover, such as Laguna Atascosa National Wildlife Refuge (LANWR), to areas with sub-optimal cover where multiple subpopulations might survive; essentially, connecting areas with larger ocelot populations to areas with sub-optimal habitat that are not likely to become developed in the near future (Mabry and Barrett 2002). Maintaining these genetic corridors could increase the viability of dispersed populations (Beier 1993, Epps et al. 2005, Haines et al. 2006b, Figueiredo et al. 2015). While this increases the chances of survival for these resident populations, transient ocelots, individuals who travel from one population to another maintaining genetic diversity, only have a 57% survival rate (Cook et al. 2006, Zerinskas and Pollio 2013). Haines et al. (2006b) predicted that the construction of road mitigation projects on dispersal corridors would reduce transient ocelot mortalities by 50%. However, these structures would need to provide a high crossing rate for ocelots, indicating researchers must use optimal designs and locations (Haines et al. 2006b, Dillon and Kelly 2008).

Results from this project support the theories put forth by Tewes and Hughes (2001) and Cain et al. (2003), which suggested that optimal structure design for ocelots would be box culverts with screening woody vegetation surrounding the entrances. This is the exact description of WCS3A, the structure with the highest performance differential for bobcats. However, Tewes and Hughes (2001), de Oliveira et al. (2010), and Caso (2013) all suggested that high bobcat usage rates may create a biological barrier and discourage ocelot visitation. This assumption is supported by Sánchez-Cordero et al. (2008) and Nordlof (2015), who found that in areas cohabited by ocelots and bobcats, ocelots changed their activity patterns to avoid times with highest bobcat activity, while bobcats did not exhibit such behavior. The structures most likely to be used by ocelots may have a high enough saturation of bobcats creating a biological barrier is present (Horne 1998, Tewes and Hughes 2001).

The duration of the control-impact array conducted as a part of this thesis was cut short due to a prairie restoration project conducted by the US Fish and Wildlife Service (FWS) on Laguna Atascosa National Wildlife Refuge (Moczygemba 2019). The current conservation effort of the aplomado falcon (*Falco femoralis*) in coastal Texas is based on increasing available falcon habitat, which consists of open prairie with widely scattered woody vegetation (Hector 1981, Hunt et al. 2013). Falcons are also limited due to predation by great horned owls (*Bubo virginianus*) which primarily hunt from dense brush (Hunt et al. 2013). McClure et al. (2017) suggested that the most effective method for increasing the aplomado falcon population would be to add additional territories by increasing available habitat. To that end, in October 2019 the FWS began mechanically removing mesquite and huisache vegetation along SH100 and northeast of WCS3A (Moczygemba 2019). This project specifically avoided FWS designated thornscrub habitat and would not thin near crossing structures (Blihovde 2019, Moczygemba 2019). However, it did include sites considered dense thornscrub (greater than 70% canopy cover) for the purposes of this project and is expected to clear much of the vegetation corridor north of WCS3A. As of the publication of this thesis this restoration project is still ongoing, however preliminary vegetation measurements indicate control site locations previously documented as dense or mixed thornscrub are now converted into open grassland. This prairie restoration is important for aplomado falcon conservation, but it will have an unknown impact on wildlife use within the surrounding area. However, based on the results found in Chapter III of this thesis, it is expected that decreases in canopy cover will reduce species richness and felid occupancy on the north side of SH100 near WCS3A. These changes may potentially reduce the likelihood of ocelot presence near productive wildlife crossing structures, decreasing expected use. This conflict of interest highlights the need for extensive communication between researchers and managers and the need for coordination in future conservation efforts.

TABLE 1. The number of cameras within the control array around State Highway 100 mitigation corridor, Cameron County, Texas. and how cameras were distributed between subfactors within each factor. Bushnell 119874 model cameras and Moultrie MCG-13270 model cameras are referenced by manufacturer name.

		SIDE		FACING				VEGETATION		
	Total	North	South	N	S	E	W	Dense Thornscrub	Mixed Thornscrub	Open Grassland
BUSHNELL	20	10	10	4	6	4	6	8	6	6
MOULTRIE	16	8	8	5	3	5	3	4	6	6

TABLE 2. Protocol used for camera placement requirements in the control array around State Highway 100 mitigation corridor, Cameron County, Texas. These factors are listed in the order they were encountered in the field.

1	If canopy cover at the pre-determined site does not align with the vegetation class expected, the site may be placed within 10 meters of the pre-determined site. Starting from the pre-determined placement point, begin moving in a clockwise spiral direction, with each pass 1-meter away from the previous track. The first site reached that matches all criteria is where the site will be placed. NOTE: If no site is found within the spiral, this site is unusable, and the next available alternate site should be used.
2	The camera cannot face any existing roads, fencing, or water. Do not place cameras in reference to existing trails, however if on a trail is the first point cameras may be placed, avoid facing the camera towards the trail.
3	Place cameras to maximize visibility of both sensor and camera. Terrain must be flat for 3 meters in front of the camera. Additionally, vegetation within 3 meters of the camera must be clearable with a weedwhacker (i.e. not woody or taller than 1.5 meters). Trees and woody vegetation should not be removed aside from minimal clearing of branches from camera sight line.
4	Plant temporary post so camera box is 0.5 meters from ground level at the camera. Post should be placed in a vertical alignment that ensures camera field of view is oriented similarly between all cameras. Ground out to 3 meters should make up ½ of camera field of view. Vegetation/sky then account for the other ½ of the camera field of view.

TABLE 3. Camera settings for both camera types within the control array around State Highway 100 mitigation corridor, Cameron County, Texas. Factors containing (-) indicate that manipulation of the indicated setting was not available within the camera model. Multi-Shot indicates how many pictures were taken per trigger. Time Between Pictures is the amount of time waited between each capture when capturing multiple photos per trigger. Interval indicates the amount of time the camera will wait between taking pictures. Interval was set to the lowest possible value for each camera brand. If settings are not listed, they were left at the default setting from the manufacturer.

	Image Size	Image Format	Multi-Shot	Time Between Pictures	LED Control	Interval	PIR Sensor Level	Night Vision Shutter
BUSHNELL	HD	Full Screen	1 Photo	N/A	Low	0.6	Low	High
MOULTRIE	-	-	1 Photo	N/A	-	None	Low	-

TABLE 4. Total number of false captures by side, vegetation class, and facing within the control array around the State Highway 100 mitigation corridor, Cameron County, Texas. Average number of false captures by camera model is included, also categorized by month. Highlighted cells indicate values that are higher than the grand average for that factor. All values provided are representative of false captures taken due to normal survey conditions.

		DEC	JAN	FEB	MAR	APR	MAY	JUNE	JULY	AUG	SEP	OCT	TOTAL
SIDE	North	12,270	14,028	4,924	11,235	39,178	60,389	29,921	93,146	165,220	21,068	683	452,062
	South	3,722	6,178	9,564	41,553	70,977	59,916	40,862	109,651	78,252	25,537	6,089	452,302
VEGETATION	Dense Thornscrub	1,184	2,418	2,363	14,186	12,902	7,440	4,697	21,726	29,574	9,936	492	106,917
	Mixed Thornscrub	9,627	12,793	7,107	22,484	56,533	32,036	31,710	73,764	72,119	23,634	5,757	347,564
	Open Grassland	5,181	4,995	5,018	16,118	40,721	80,829	34,376	107,307	141,779	13,035	523	449,882
FACING	N	4,415	9,766	4,083	8,664	30,536	25,353	16,877	50,778	67,072	3,150	429	221,123
	S	5,608	2,366	1,624	5,769	23,200	40,806	15,023	68,936	72,774	8,141	2,403	246,650
	E	2,938	3,740	4,983	29,800	40,043	46,805	34,549	72,221	76,690	28,515	2,979	343,263
	W	3,031	4,334	3,798	8,555	16,630	7,341	4,334	10,862	26,936	6,799	961	93,581
MODEL	Bushnell	195	190	254	956	1,817	1,475	1,247	2,208	3,653	870	192	13,057
	Moultrie	806	1,094	627	2,245	4,937	6,054	3,056	10,576	11,361	1,943	195	42,894

TABLE 5. Counts for all target species captured in the control array around State Highway 100 mitigation corridor, Cameron County, Texas, in relation to north or south side of the mitigation corridor and three vegetation classes.

	SIDE OF SH100		VEGETATION CLASS		
	North	South	Dense Thornscrub	Mixed Thornscrub	Open Grassland
Bobcat	75	362	296	133	8
Collared peccary	81	61	126	16	0
Coyote	43	172	62	18	135
Eastern cottontail	1,002	847	1,076	749	24
Nilgai	41	159	41	90	69
Nine-banded armadillo	115	214	176	152	1
Northern bobwhite	109	381	310	157	23
Striped skunk	104	98	54	20	128
Virginia opossum	358	236	547	46	1
White-tailed deer	705	453	501	501	156
TOTALS	2,687	3,106	3,189	1,882	722

TABLE 6. Factors included in model averaging for each species captured in the control array around the State Highway 100 mitigation corridor, Cameron County, Texas. The range of McFadden pseudo-R^2 scores for each model are provided, and the factors that were found to be significant (< 0.05) are labeled. Side of SH100 was the only categorical variable (north or south), all other factors were continuous. Canopy Cover (%) and Ground Cover (%) are negatively associated if species were observed more in areas with low cover, and positively associated if observed in areas with high cover. Distance to SH100, WCS, and WG are negatively associated if there were more observations closer to the factor, and positive if there were more observations farther. Factors that were included in at least one model but were not found to be significant are represented with (-). Coefficients are listed in the Appendix.

	McFadden pseudo-R^2	Canopy Cover	Distance to SH100	Distance to WCS	Distance to WG	Ground Cover	Side of SH100
Species Richness	0.2525 – 0.2606	Positive		Negative	-		-
All Species Combined	0.0651 – 0.0707	Positive		-	-	-	-
Bobcat	0.1826	Positive		Negative			South
Collared peccary	0.2972 – 0.3068	Positive	-	-	Negative		
Coyote	0.0482 – 0.0753	-	-	-	-	-	South
Eastern cottontail	0.0490 – 0.0643	Positive	-		-	-	-
Nilgai	0.0780 – 0.0955		-		-	-	South
Nine-banded armadillo	0.0664 – 0.0822	Positive	Negative	-		-	-
Northern bobwhite	0.0928 – 0.1089	Positive		-	-	-	South
Striped skunk	0.0209 – 0.0285	-		-	-		-
Virginia opossum	0.1724 – 0.1849	Positive	-		Negative		
White-tailed deer	0.0410 – 0.0603	Positive	-	-	-		-

TABLE 7. Total counts of wildlife occurrences and successful crossings for all target species used in Objective 3 and during the study period in both the control array and impact array of the State Highway 100 mitigation corridor, Cameron County, Texas. The Control Array section contains the total number of individuals observed within the control array. The control array is categorized by vegetation class; dense thornscrub (Dense), mixed thornscrub (Mixed), and open grassland (Open). The Impact Array section contains the total number of successful crossings at sites within the impact array. The impact array is categorized by wildlife crossing structure and wildlife guard.

	CONTROL ARRAY			IMPACT ARRAY												
	Vegetation Class			Wildlife Crossing Structures					Wildlife Guards							
SPECIES	Dense	Mixed	Open	WCS1	WCS2	WCS3	WCS3A	WCS4	WG11	WG12	WG13	WG14	WG15	WG16	WG17	WG18
Black-tailed jackrabbit	0	0	177	0	0	0	0	0	0	0	0	2	0	0	1	0
Bobcat	296	133	8	16	22	19	185	0	0	0	29	1	0	0	0	0
Collared peccary	126	16	0	0	0	0	230	0	0	0	0	0	0	0	0	0
Coyote	62	18	135	5	0	1	184	66	30	1	28	30	7	1	15	13
Eastern cottontail	1,076	749	24	0	0	36	3	577	4	0	8	15	0	0	5	1
Nilgai	41	90	69	0	0	0	0	0	1	0	0	0	0	0	0	0
Nine-banded armadillo	176	152	1	0	0	6	69	39	4	0	0	0	1	0	0	1
Northern bobwhite	310	157	23	0	0	221	0	19	13	0	5	0	6	0	30	0
Northern raccoon	10	2	14	213	225	26	22	16	18	0	90	74	21	2	50	12
Striped skunk	54	20	128	0	0	1	1	275	2	0	1	6	4	0	6	27
Texas tortoise	18	19	4	0	0	0	1	0	0	0	0	0	0	0	0	0
Virginia opossum	547	46	1	8	830	375	692	619	199	12	225	6	16	1	18	138
White-tailed deer	501	501	156	0	0	425	0	1	0	0	0	0	0	0	0	0

TABLE 8. Vegetation-based performance differentials and proximity-based performance differentials for wildlife crossing structures (WCS) in the State Highway 100 mitigation corridor, Cameron County, Texas. A positive performance differential indicates wildlife were seen more frequently at the structure than expected. A negative performance differential indicates wildlife were seen less frequently at the structure than expected. Zero indicates the species was observed the number of times as expected, which may be not at all (indicated by a *). Nilgai and white-tailed deer were unlikely to use WCS3A due to their size, however performance differentials were determined for comparison purposes.

	Proximity-Based Performance Differentials			Vegetation-Based Performance Differentials		
SPECIES	WCS3	WCS3A	WCS4	WCS3	WCS3A	WCS4
Black-tailed jackrabbit	0	-4.7	-9.3	0*	0*	-7.4
Bobcat	-4.7	87	-4.9	-2.4	88.8	-0.3
Collared peccary	-0.7	119.2	0*	-0.7	120.5	0*
Coyote	-0.8	95.4	10.8	-0.6	98	16
Eastern cottontail	-21.5	-61.2	166	-25.3	-43.2	188.3
Nilgai	-2	(-4.6)	-9	-3.8	(-1.7)	-2.9
Nine-banded armadillo	-1.7	23.3	12.6	-5.4	30.4	12.8
Northern bobwhite	33.4	-16.9	-6.1	29.7	-12.9	5.3
Northern raccoon	4.2	11.4	3.8	4.2	11.6	4.7
Striped skunk	-0.7	-1	72.5	-0.7	-1.7	84.9
Texas tortoise	-0.6	0	-2.6	-0.8	-0.2	-0.2
Virginia opossum	55	358.5	191.3	59.6	355.6	203
White-tailed deer	42.2	(-34)	-10	48.8	(-20.9)	-6.2
TOTAL	102	665	472	103	624	498

TABLE 9. Vegetation-based performance differentials for target species observed at wildlife guards (WG) in the State Highway 100 mitigation corridor, Cameron County, Texas. A positive performance differential indicates wildlife were observed crossing the structure more frequently than expected. A negative performance differential indicates wildlife were observed crossing the structure less frequently than expected. Zero indicates the species was observed the number of times as expected, which may be not at all (indicated by a *).

SPECIES	WG11	WG12	WG13	WG14	WG15	WG16	WG17
Black-tailed jackrabbit	-7.4	0*	0*	-6.9	-7.4	-7.4	-7.2
Bobcat	-0.3	-12.3	-2.4	-0.1	-0.3	-0.3	-0.3
Collared peccary	0*	-5.3	-0.7	0*	0*	0*	0*
Coyote	0.9	-2.4	2.3	0.9	-4	-5.4	-2.3
Eastern cottontail	-0.1	-44.8	-30.3	2.3	-1	-1	0
Nilgai	-2.7	-1.7	-3.8	-2.9	-2.9	-2.9	-2.9
Nine-banded armadillo	0.8	-7.3	-6.3	0*	0.2	0*	0*
Northern bobwhite	1.9	-12.9	-6	-1	0.4	-1	5.6
Northern raccoon	3.4	-0.4	9.8	15.6	4	-0.2	10.4
Striped skunk	-4.9	-2.3	-0.7	-4	-4.5	-5.3	-4
Texas tortoise	-0.2	-0.8	-0.8	-0.2	-0.2	-0.2	-0.2
Virginia opossum	43.5	-20.2	22.7	1.3	3.5	0.2	3.9
White-tailed deer	-6.5	-20.9	-20.9	-6.5	-6.5	-6.5	-6.5
TOTAL	28	-131	-37	-2	-19	-30	-3

TABLE 10. The sum of all successful crossing at wildlife crossing structures (Impact Total) and the number of total individuals observed within the control array (Control Total) on State Highway 100, Cameron County, Texas. The percent difference (Control-Impact % Difference) provides an indication of how permeable the mitigation structures on SH100 are: a negative value indicates the roadway is a completely impermeable barrier for the species, any value between 1 and 100 indicates the roadway is permeable for the species but does not allow full connectivity, while more than 100 indicates the species is likely drawn to wildlife crossing structures. The percentage of road that would need to be permeable to allow full connectivity for the species (Predicted % Road for Full Connectivity) is also included. If the value is less than 100%, than that is the percent of roadway that must be permeable (a WCS) to allow full connectivity for that species. If it is higher than 100% than the roadway will likely serve as a barrier to the species, regardless of the amount of permeable structure constructed within the mitigation fencing. This estimation assumes that future structures would be as successful as current designs. The WCS where each species had the highest crossing rate (Best WCS) and its performance differential (Performance Differential) are provided. These estimates could not be determined for black-tailed jackrabbits and nilgai, as they were never observed utilizing a WCS during the study period.

SPECIES	Impact Total N	Control Total N	Control-Impact % Difference	Predicted % Road for Full Connectivity	Best WCS	Performance Differential
Black-tailed jackrabbit	0	177	-100	-	-	-
Bobcat	242	437	-44	70%	WCS3A	88.8
Collared peccary	230	142	62	24%	WCS3A	120.5
Coyote	256	215	19	32%	WCS3A	98
Eastern cottontail	616	1,849	-66	116%	WCS4	188.3
Nilgai	0	200	-100	-	-	-
Nine-banded armadillo	114	329	-65	112%	WCS3	30.4
Northern bobwhite	240	490	-51	79%	WCS3	29.7
Northern raccoon	502	26	1,830	2%	WCS3A	11.6
Striped skunk	277	202	37	28%	WCS4	84.9
Texas tortoise	1	41	-97	1,594%	WCS4	-0.2
Virginia opossum	2,524	594	324	9%	WCS3A	355.6
White-tailed deer	426	1,158	-63	105%	WCS3	48.8

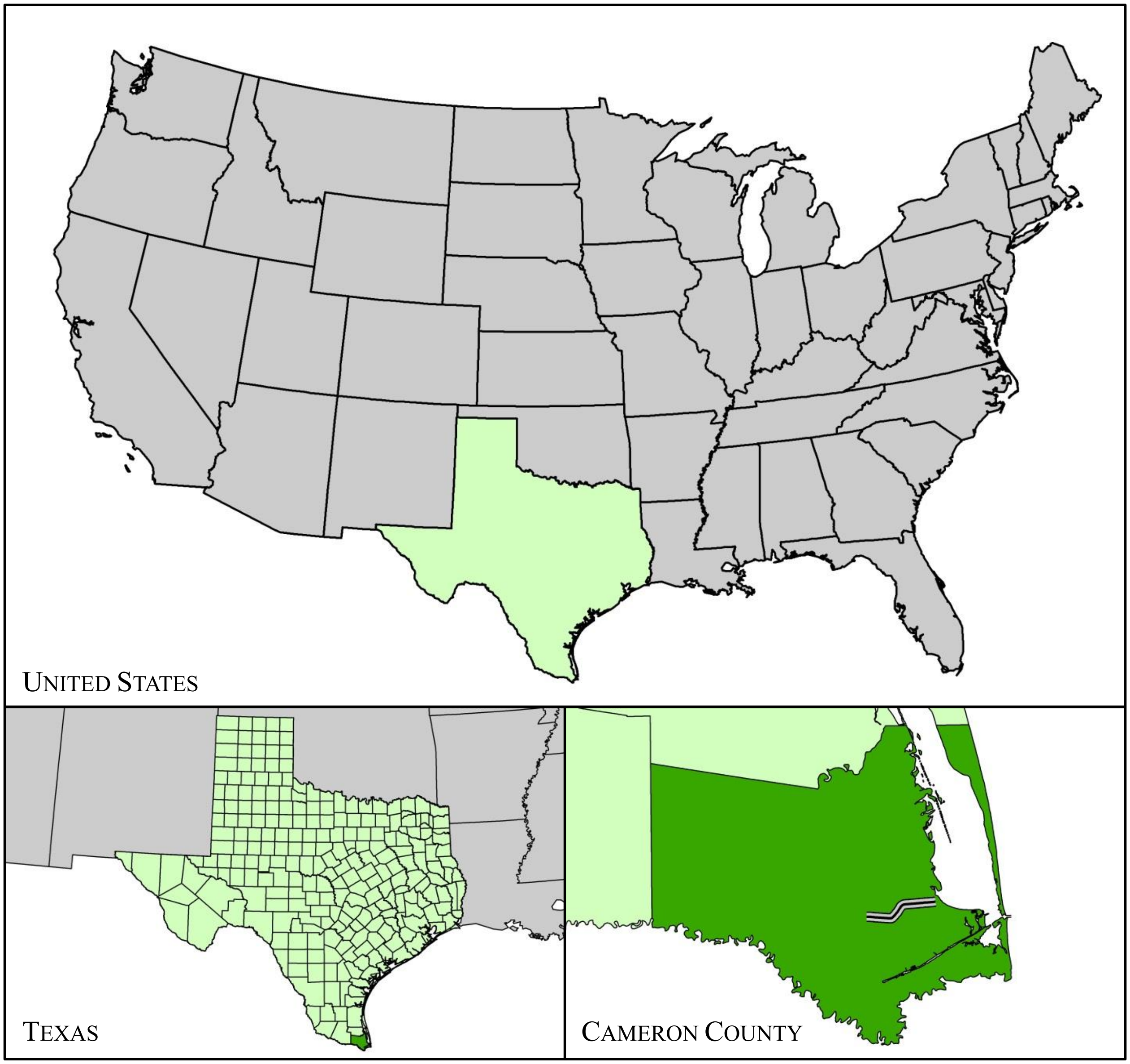


FIG. 1. The study area along State Highway 100 (SH100) mitigation corridor in Cameron County, Texas, United States.

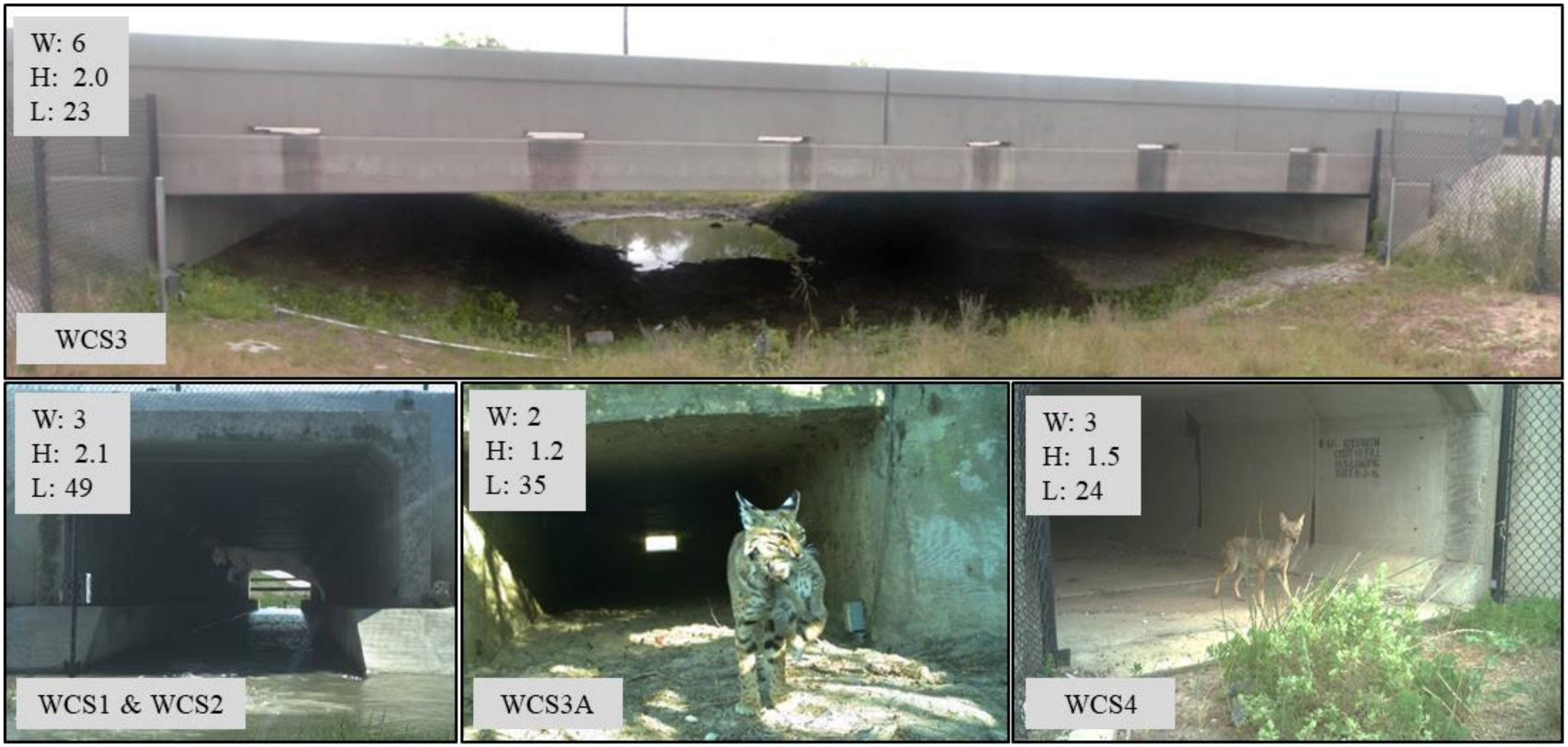


FIG. 2. The four wildlife crossing designs used within the State Highway 100 mitigation corridor, Cameron County, Texas. Construction on WCS1, WCS2, WCS3, and WCS4 was completed in January 2018. WCS1 and WCS2 are of the same design, below-grade culvert crossings with elevated catwalks and permanent water flow. WCS3 is a below-grade bridge-type crossing, with occasional pooling water. WCS3A is a below-grade culvert crossing that was constructed during the mid-1990s, however did not have wildlife-proof fencing until January 2018. WCS4 is a below-grade culvert. Measurements are provided in meters.

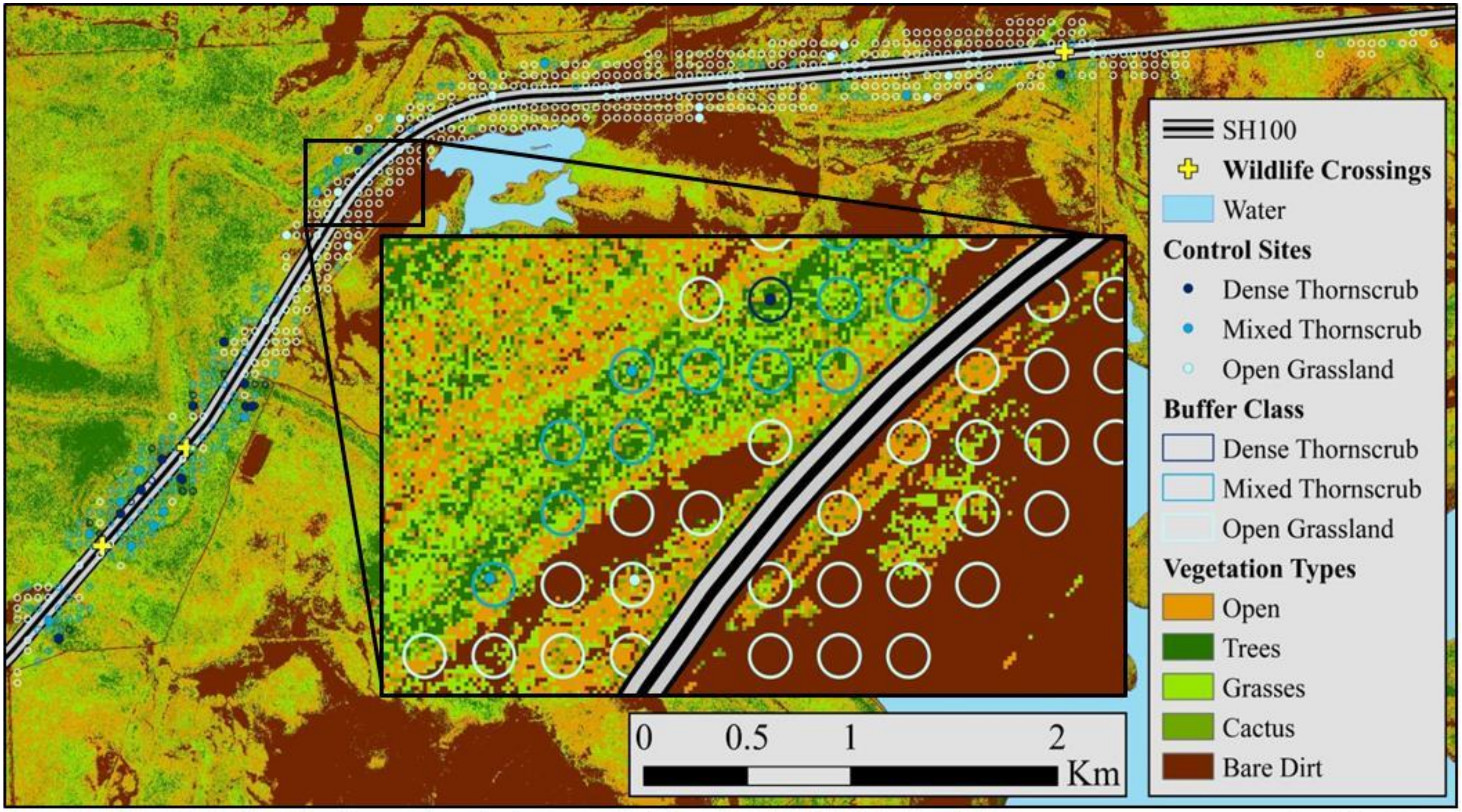


FIG. 3. Vegetation classification conducted along State Highway 100 mitigation corridor, Cameron County, Texas. Classification was conducted in ArcGIS utilizing 1-meter National Agricultural Imagery Program (NAIP) imagery, primarily from 2016, with some supplemental data from 2014. Vegetation was broken into 10 classes based on reflectance, then simplified into similar factors. Vegetation was quantified within each of the 15-meter buffers around all potential sites, and site type was determined by the percent of each vegetation class. Sites were excluded if grass accounted for 40% or more of the vegetation within the 15-meter buffer.

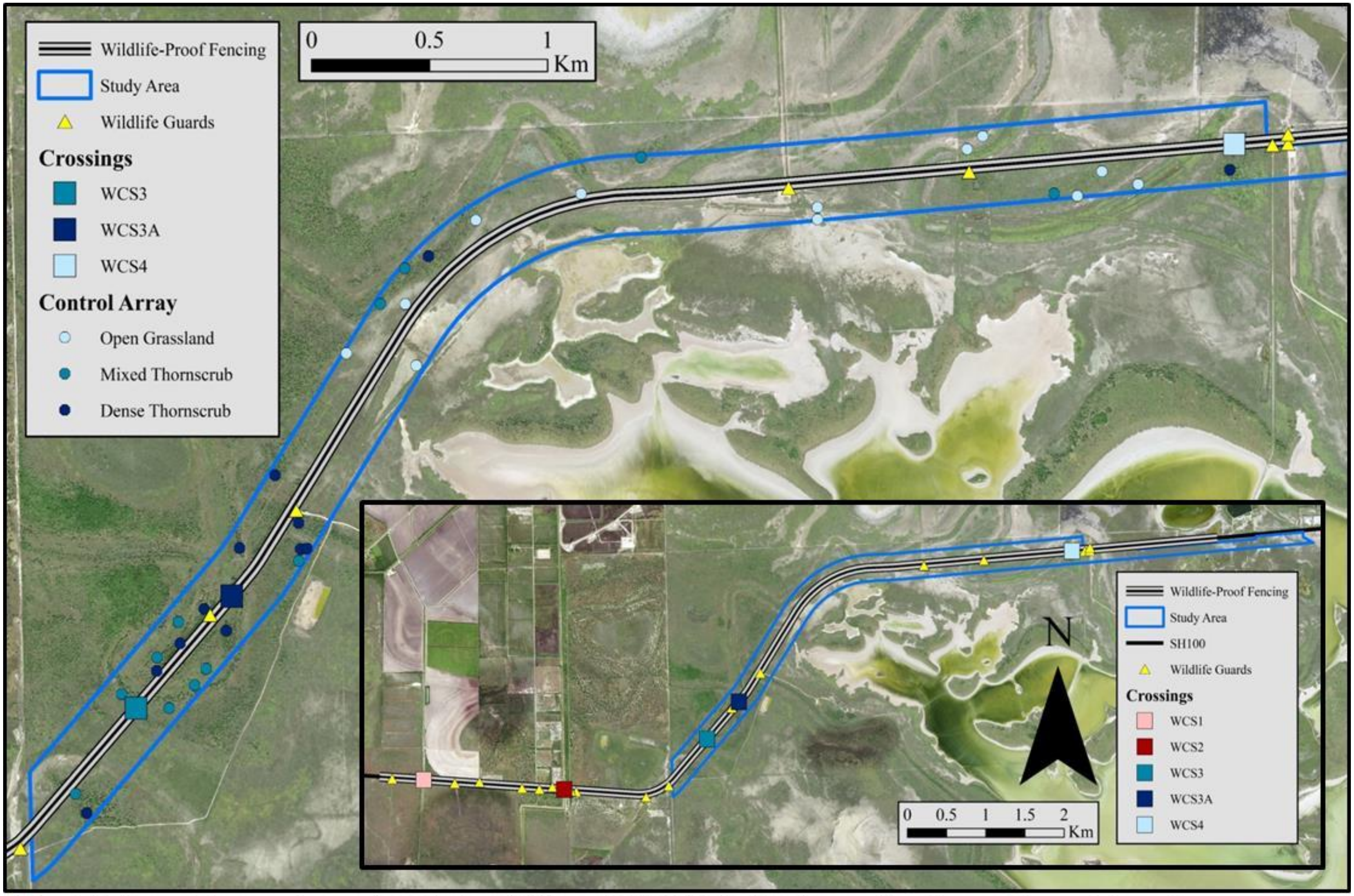


FIG. 4. Entirety of the control array along the State Highway 100 (SH100) mitigation corridor, Cameron County, Texas. Sites were compared to WCS within similar vegetation type, which is shown using color-coordination. The blue outline was the available study area within 150-m of the roadside and on US Fish and Wildlife property. Inset is the entire length of wildlife-proof fencing on SH100.

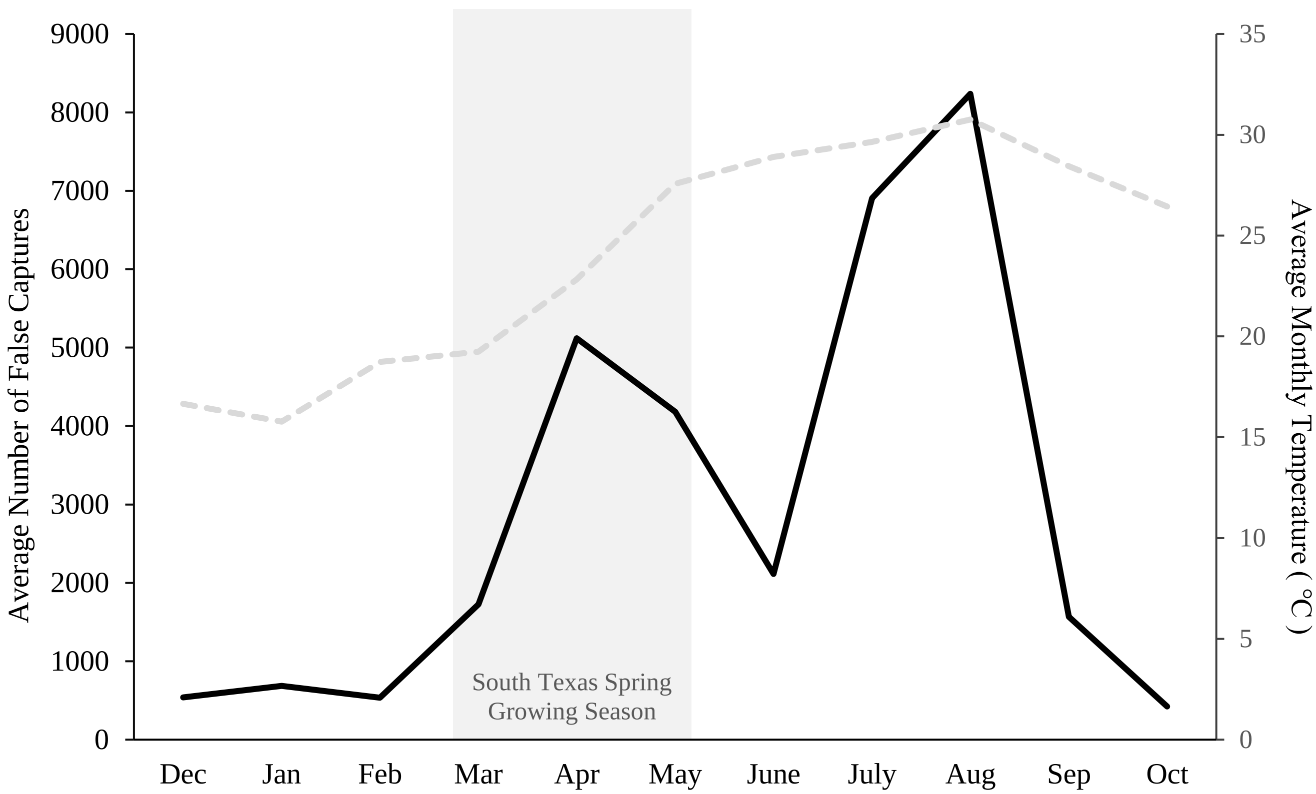


FIG. 5. The average number of false captures within the control array around the State Highway 100 mitigation corridor, Cameron County, Texas and average monthly temperature at mitigation structures. Months range from December 2018 to October 2019. Average number of false captures are represented by a black line and temperature is represented by a dotted grey line.

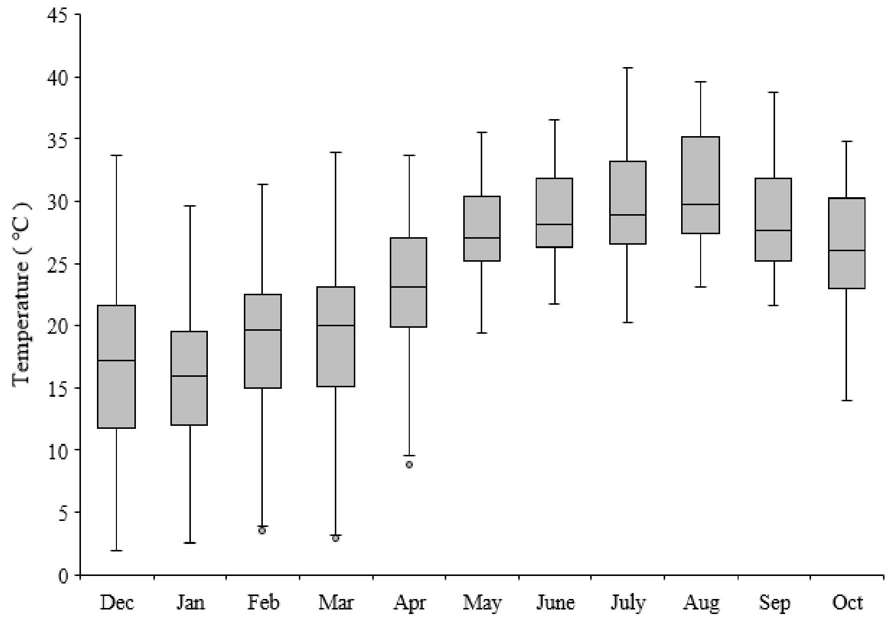


FIG. 6. Box plot of temperatures averaged from Kestrel data loggers placed at WCS3, WCS3A, and WCS4 in the State Highway 100 mitigation corridor, Cameron County, Texas. Each box is the first and third quartiles, the median, and whiskers are the full range. Months range from December 2018 to October 2019.

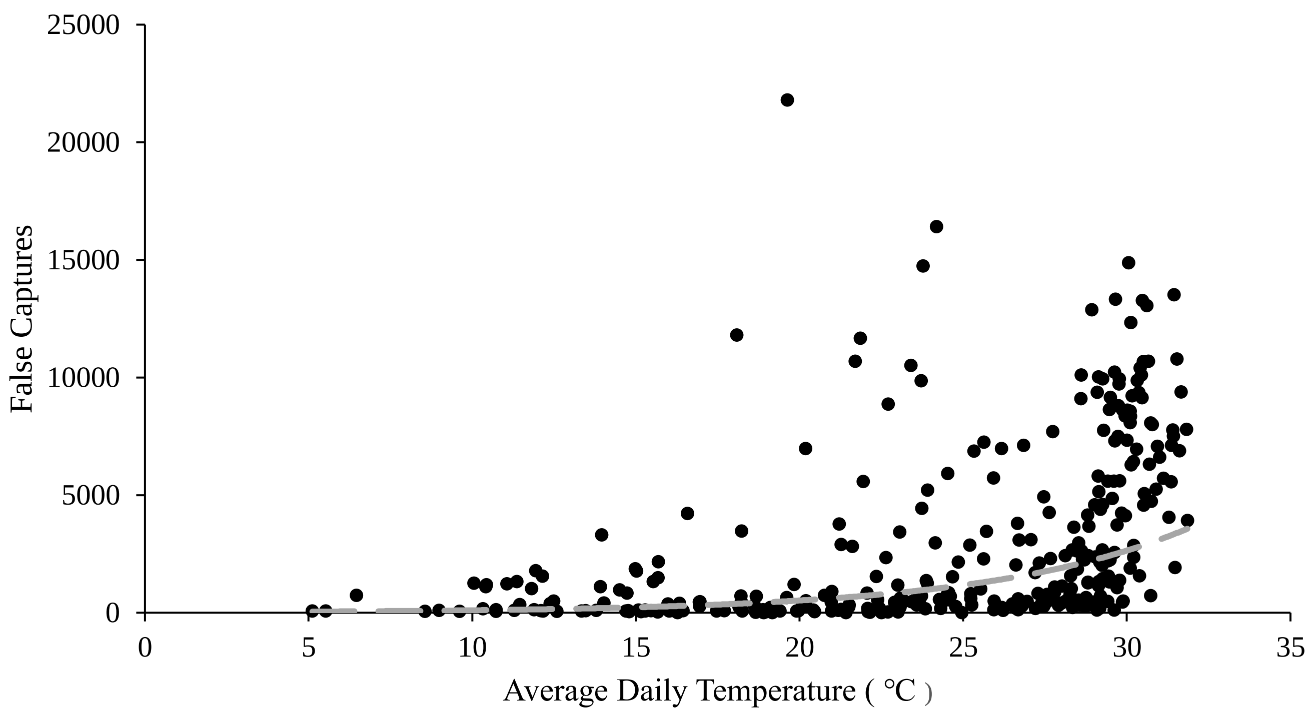


FIG. 7. Frequency of daily false captures by average daily temperature collected within the control array around the State Highway 100 mitigation corridor, Cameron County, Texas. A linear regression conducted on the natural log transformed number of false captures indicated there was a significant positive association between average daily temperature and number of false captures ($R^2 = 0.3201$; $p < 0.0001$).

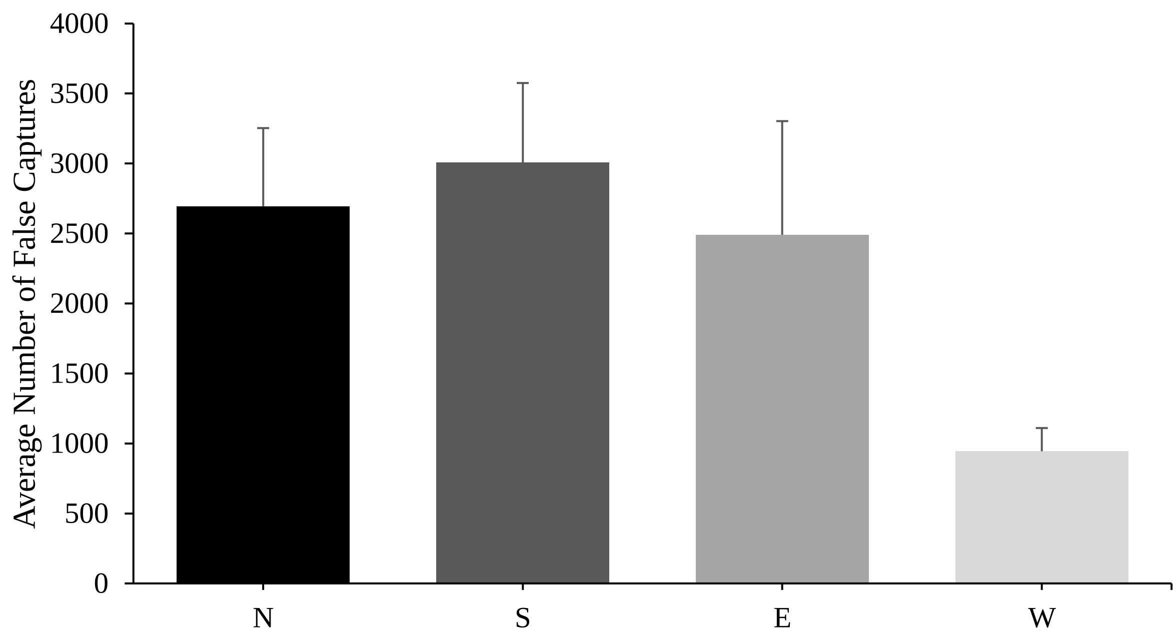


FIG. 8. Average number of false captures by direction of camera facing (± 1 S.E.) for the control array around State Highway 100, Cameron County, Texas. Based on the results of an ANOVA, there was no significant difference between the four directions (p = 0.2020).

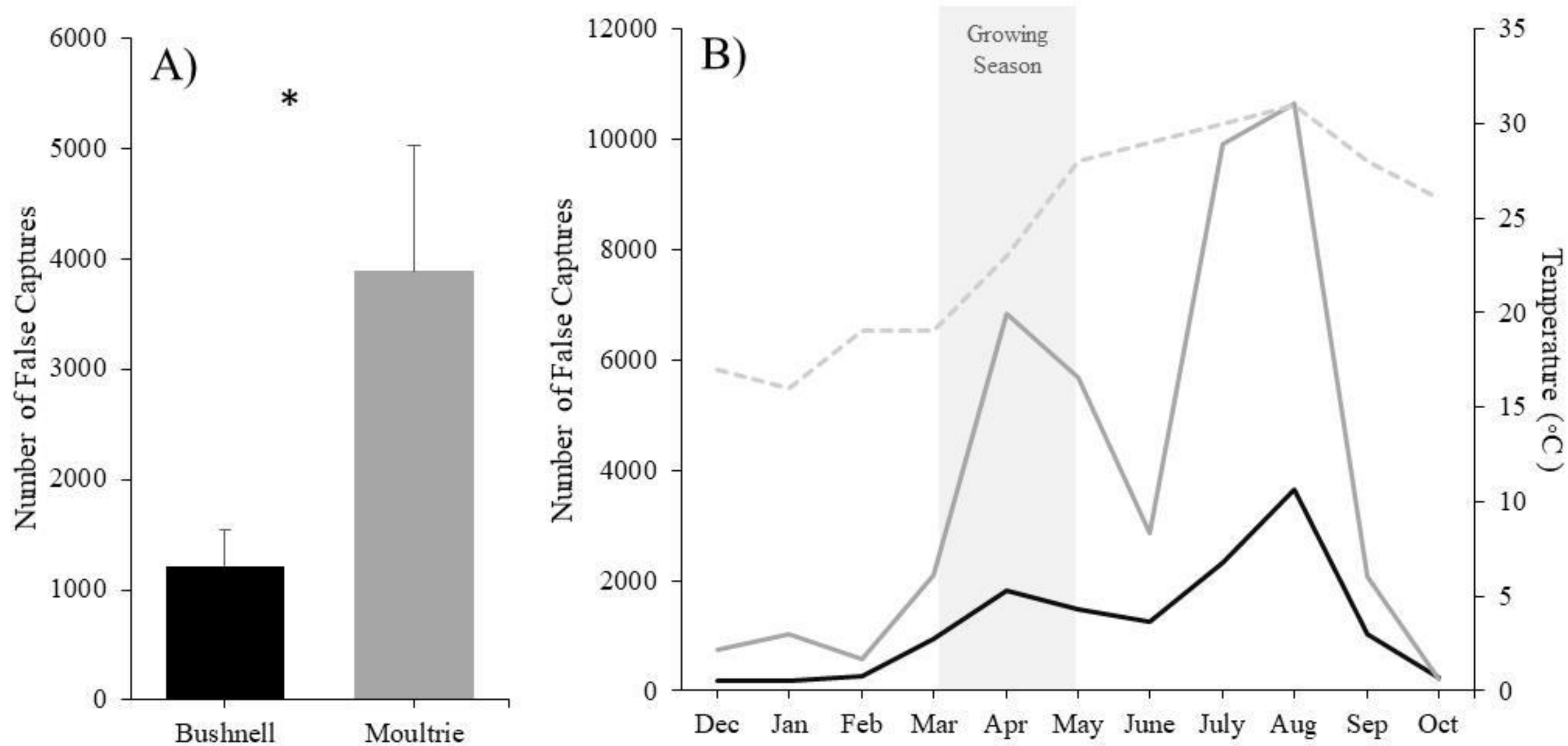


FIG. 9. A) Average number of false captures by camera model (± 1 S.E.) within the control array around State Highway 100, Cameron County, Texas, which was found to be a significant factor on average number of false captures ($p = 0.0001$). B) Average number of false captures per month for the entire study period, from December 2018 to October 2019. Colors denoting manufacturer are the same in both graphs. Within the line graph, temperature is delineated by a dotted grey line and the south Texas spring growing season is represented with a grey field.

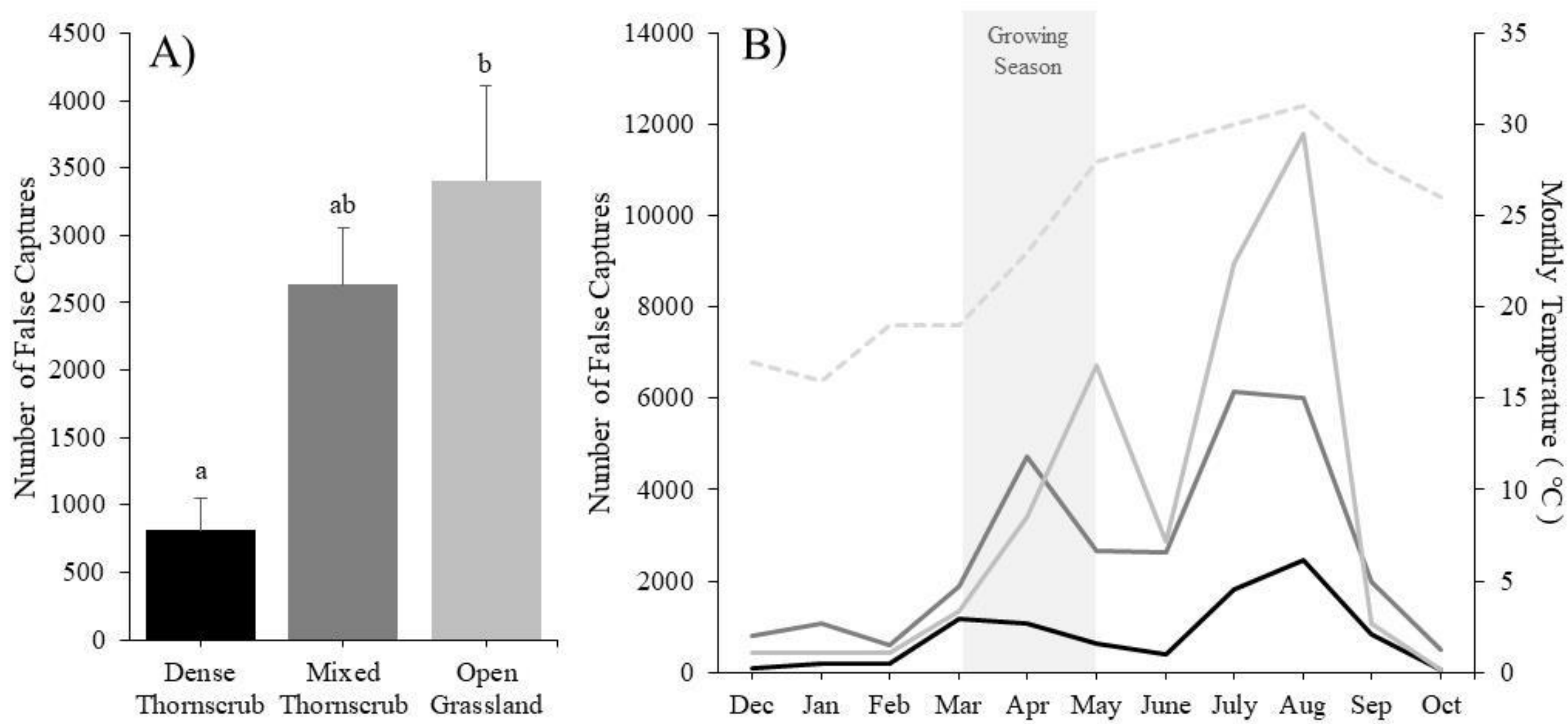


FIG. 10. A) Average number of false captures (± 1 S.E.) within each vegetation class for the control array around State Highway 100, Cameron County, Texas. Categories denoted by different letters were significantly different at p <0.05 as tested with ANOVA followed by Tukey post hoc test. B) Average number of false captures and average temperature by month for the entire study period December 2018 to October 2019. Colors denoting vegetation class are the same in both graphs. Temperature is indicated by the grey dashed line and the south Texas spring growing season is represented with a grey field.

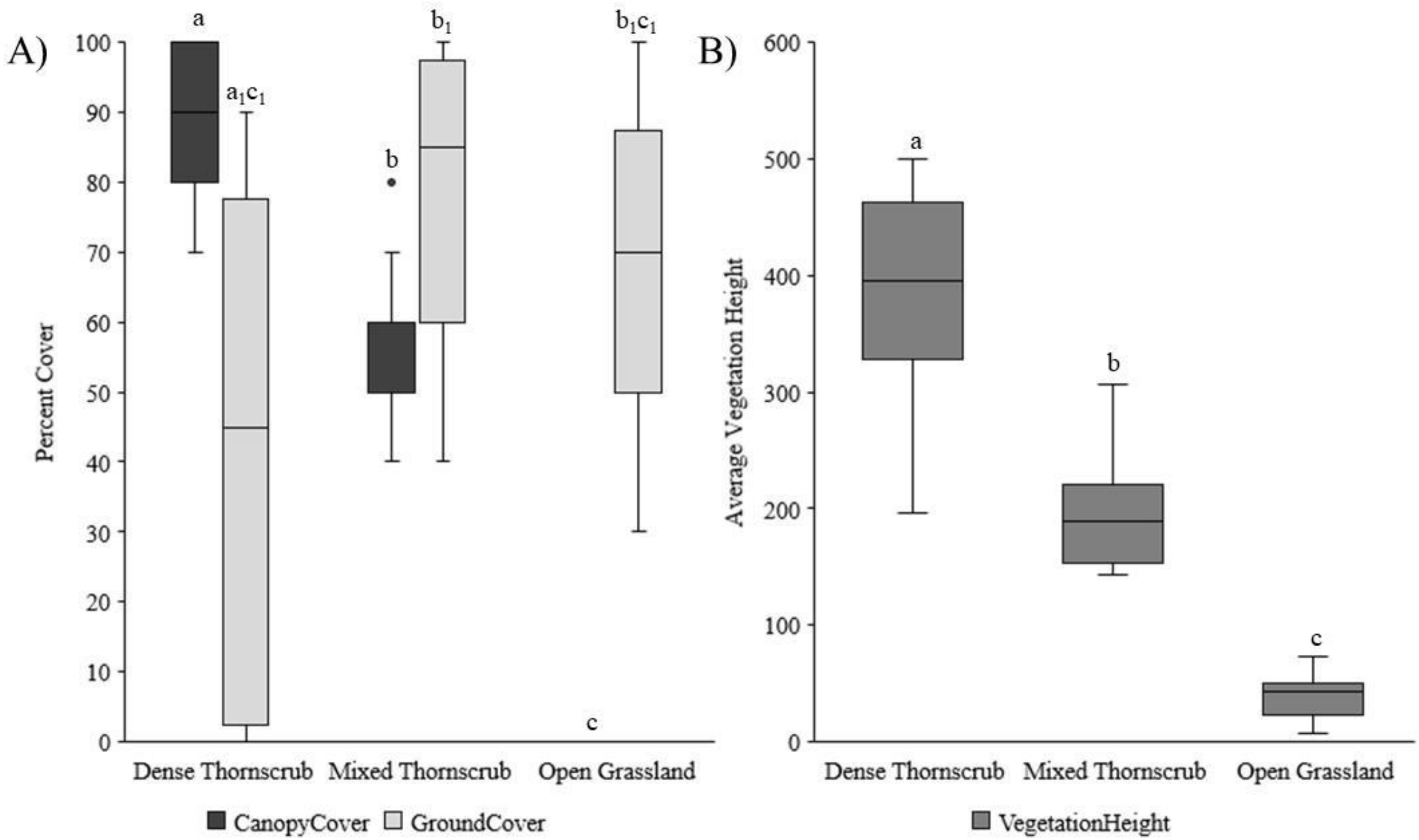


FIG. 11. Box plots of the characteristics of vegetation within each vegetation class of the control array around the State Highway 100 mitigation corridor, Cameron County, Texas. Each box is the first and third quartiles, the median, and whiskers are the full range. A) Median canopy cover and ground cover within each vegetation class, B) the range of canopy height minimum and maximum within each vegetation class. Letters indicate significance between categories according to Dunn's test with Bonferroni correction at $p < 0.05$.

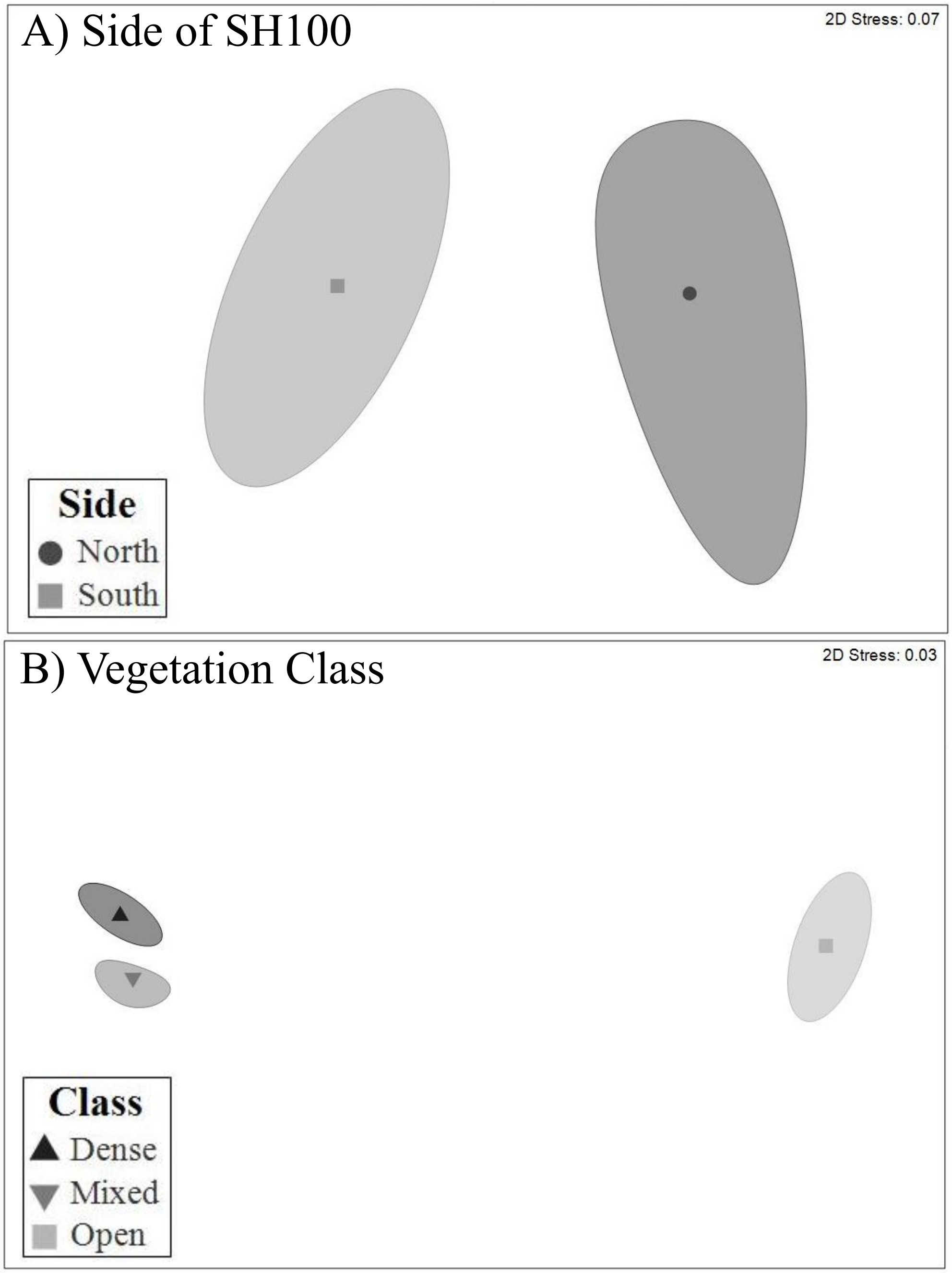


FIG. 12. Non-metric multidimensional scaling (nMDS) plots for the wildlife communities captured in the control array around the State Highway 100 mitigation corridor, Cameron County, Texas. Plots based on bootstrap averages showing clustering within A) North and south sides of SH100 and B) Three vegetation classes of dense thornscrub, mixed for mixed thornscrub, and open for open grassland. Both side of SH100 (pseudo-F = 3.8901; p = 0.0002) and vegetation class (pseudo-F = 7.0793; p = 0.0001) were significantly different in terms of animal communities observed as determined by PERMANOVA. The interaction term of both side of SH100 and vegetation class was also significant (pseudo-F = 1.7112; p = 0.0322).

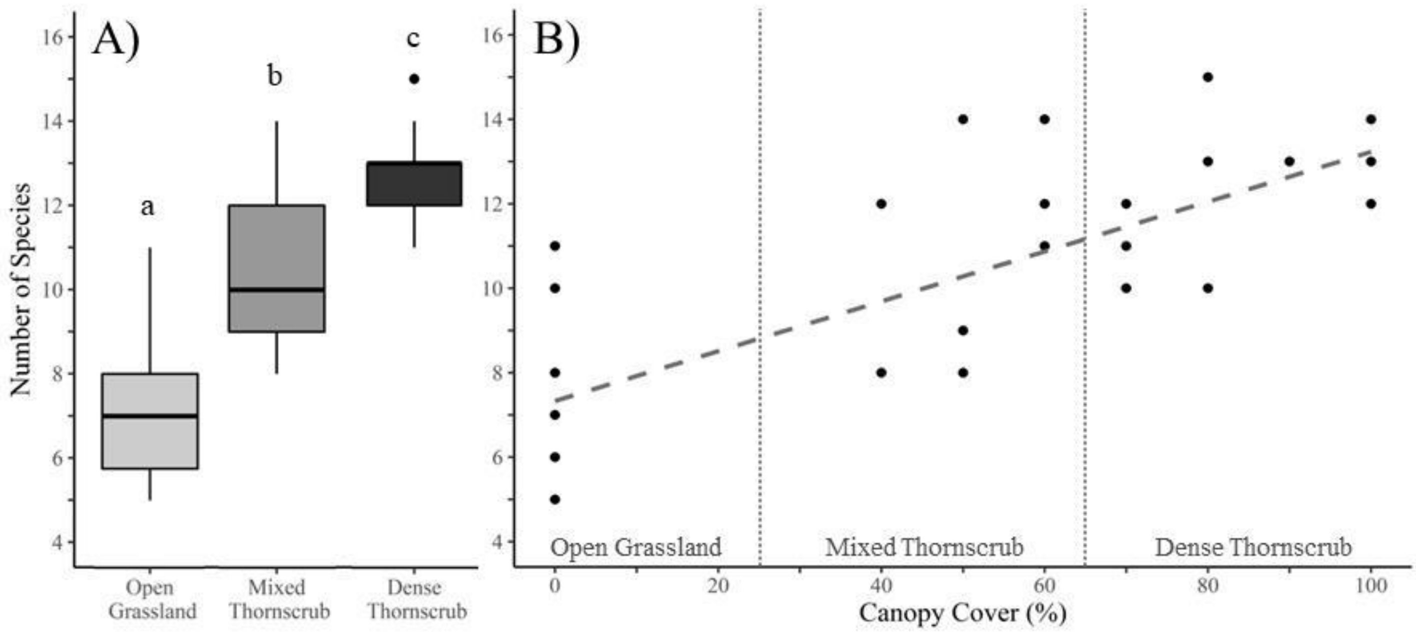


FIG. 13. A) Box plot of species richness (S) within each vegetation class of the control array around State Highway 100, Cameron County, Texas, with outliers represented by a black dot. Categories denoted by different letters were significantly different at $p < 0.05$ as tested with an ANOVA followed by Tukey HSD post hoc test. B) Scatterplot of species richness (S) at each site compared to the corresponding canopy cover at the site, each dot represents a site within the control array (all 36 are displayed, with some overlap). Based on a generalized linear model, canopy cover was found to have a significant positive association with species richness ($p < 0.001$). The trendline is indicated with a dotted grey line and indicates a positive relationship. Vegetation classes are delineated with vertical grey lines.

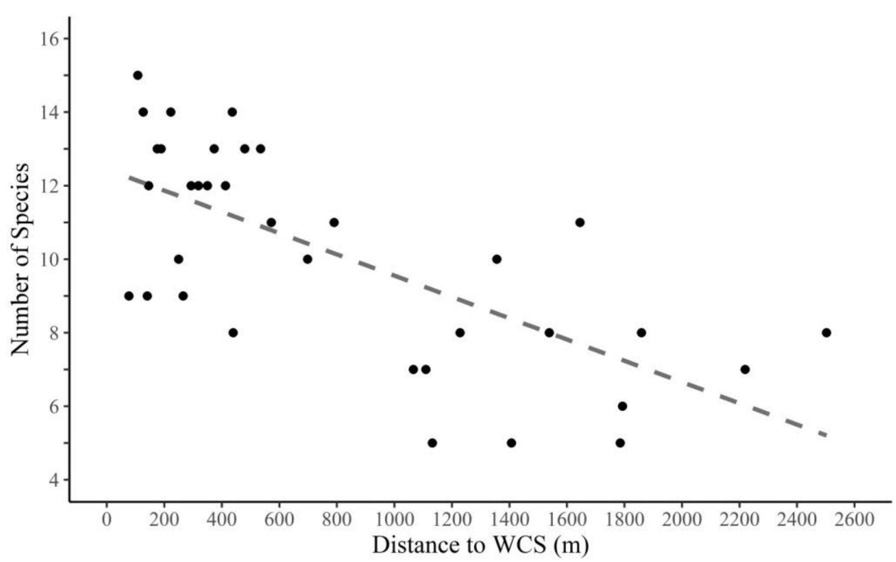
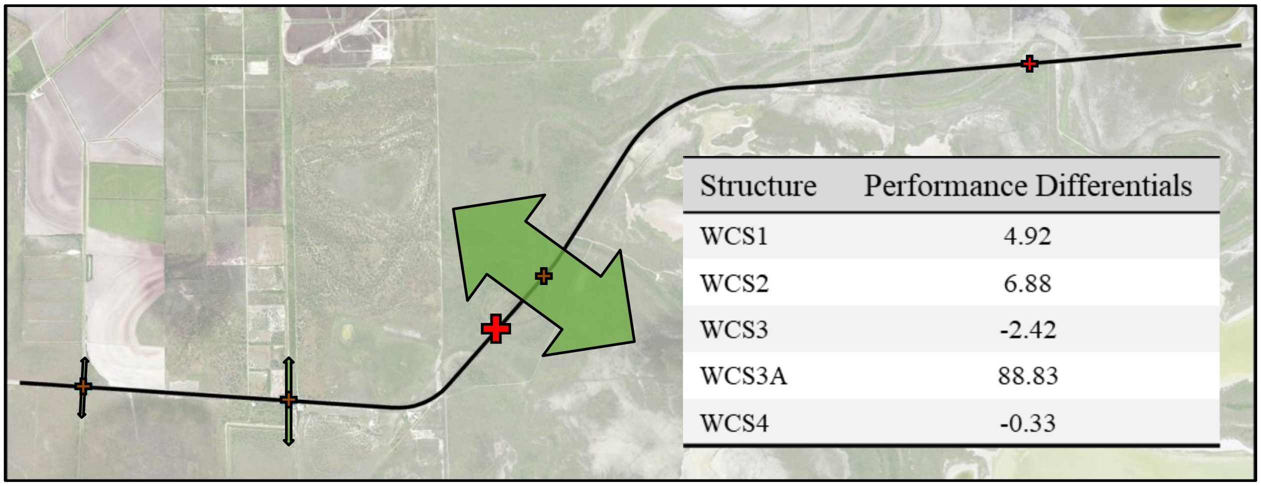


FIG. 14. Species richness (S) within the control array around the State Highway 100 mitigation corridor, Cameron County, Texas, by distance to nearest wildlife crossing structure (WCS), a generalized linear model calculated significant negative association with species richness ($p = 0.0088$). The entire control array is presented, with each dot representing a site. The trendline is represented by a dashed grey line and indicates a negative relationship.

FIG. 15. Number of bobcat occurrences at each site within the control array around the State Highway 100 mitigation corridor, Cameron County, Texas. The three sites within the dotted orange circle are the sites referenced in the Discussion section of Chapter III, indicating the influence of microhabitat on bobcat presence. The dense thornscrub sites had 26 & 34 bobcat occurrences, while bobcats were only observed at the mixed thornscrub site 6 times.



Structure	Performance Differentials
WCS1	4.92
WCS2	6.88
WCS3	-2.42
WCS3A	88.83
WCS4	-0.33

FIG. 16. Visual representation of bobcat movement across the State Highway 100 mitigation corridor, Cameron County, Texas. WCSs are represented with green and red crosses. Green crosses indicate bobcats were observed more often at the structures than expected. The width of the green arrows represent bobcat movement, with wider arrows indicating more movement. Red crosses indicate bobcats were observed less often than expected, and the size of the red cross indicates the degree of the barrier effect. Structures in the table are listed from west to east.

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APPENDICIES

APPENDIX A

Appendix A

Study Design: Preliminary Control Array

Based on the a control array design developed by Andis et al. (2017), an initial proximity-based control array was implemented and monitored from April to November, 2018. A 50-meter grid was overlaid within 1-kilometer by 150-meter placement zones centered on three of the available wildlife crossing structures, WCS3, WCS3A, and WCS4. These three structures were chosen because the only available study area was on the US Fish and Wildlife monitored Laguna Atascosa National Wildlife Refuge. 24 sites were randomly chosen, with eight surrounding each structure, four on either side of SH100. These sites consisted of 24 Bushnell 119874 brand wildlife cameras, with a single camera at each site (Bushnell Corporation, Overland Park, KS, USA). Captures from these cameras were expected to be compared to captures taken within the roadside array using methods developed by Andis et al. (2017). Unfortunately, this array was hobbled with technical difficulties. Half the sites within this array experienced extreme numbers of false captures, with some cameras taking so many that their internal batteries died within three days of being changed. Multiple camera heights, vertical angles, facing, and vegetation clearing techniques were experimented with to attempt to minimize the number of false captures. While these variables appeared to influence the number of false captures taken, it was theorized that the number of false captures were likely most correlated with vegetation type within the camera field of view.

Due to the random placement of this control array, many cameras were placed in areas where the predominant vegetation cover was gulf cordgrass, a fast-growing native species that can reach nearly two meters in height. Not only was camera maintenance difficult at sites predominated by cordgrass, but they were placed within a vegetation type that did not accurately represent vegetation at nearby mitigation structures. The predominant vegetation type surrounding WCS3, WCS3A, and WCS4 is not gulf cordgrass. WCS3 has open coastal prairie and cactus land on the north side of the crossing, with mixed thornscrub on the south side. WCS3A has dense thornscrub immediately surrounding the entrance and exit on both sides of SH100. WCS4 does have open grass and cactus land surrounding the entrance and exit on both sides of SH100; however, this grass is not gulf cordgrass.

Though this array was not effective at accurately observing the wildlife communities around the majority of WCS, it provided an opportunity to understand and experiment with camera placement. Additionally, the variety of camera placements and facings used in this array undermined any statistical or comparative value it may have had. Due to the potential inaccuracy inherent within this control array, it was determined a new control array design had to be created, following a camera placement protocol developed using lessons from the proximity-based control array. The new array also had to account for the variation created by an extremely mosaic landscape, with heterogeneous vegetation types.

APPENDIX B

Appendix B

Example Information for Study Design to be Included in Publication Appendices

Wildlife camera trapping studies should include all following information within their publication appendices to enhance the comparability of their study. This information was also included in the *Tables and Figures* section of this thesis.

APPENDIX B, TABLE 1. The number of cameras within the control array around State Highway 100 mitigation corridor, Cameron County, Texas. and how cameras were distributed between subfactors within each factor. Bushnell 119874 model cameras, Moultrie MCG-13270 model cameras, and Reconyx PC900 HyperFire Professional Covert Camera Traps are referenced by manufacturer name.

	Image Size	Image Format	Multi-Shot	Time Between Pictures	LED Control	Interval	PIR Sensor Level	Night Vision Shutter
BUSHNELL	HD	Full Screen	1 Photo	N/A	Low	0.6	Low	High
MOULTRIE	-	-	1 Photo	N/A	-	None	Low	-
RECONYX	Standard	-	3 Photos	1 Second	High	No Delay	High	1/30th

APPENDIX B, TABLE 2. Protocol used for camera placement requirements in the control array around State Highway 100 mitigation corridor, Cameron County, Texas. These factors are listed in the order they were encountered in the field

1	If canopy cover at the pre-determined site does not align with the vegetation class expected, the site may be placed within 10 meters of the pre-determined site. Starting from the pre-determined placement point, begin moving in a clockwise spiral direction, with each pass 1 meter away from the previous track. The first site reached that matches all criteria is where the site will be placed.
2	The camera cannot face any existing roads, fencing, or water. Do not place cameras in reference to existing trails, however if on a trail is the first point cameras may be placed, avoid facing the camera towards the trail.
3	Place cameras to maximize visibility of both sensor and camera. Terrain must be flat for 2.5 to 3 meters in front of the camera. Additionally, vegetation within 2.5 to 3 meters of the camera must be clearable with a weedwhacker (i.e. not woody or taller than 1.5 meters). Trees and woody vegetation should not be removed aside from minimal clearing of branches from camera sight line.
4	Plant temporary post so camera box is 0.5 meters from ground level at the camera. Post should be placed in a vertical alignment that ensures camera field of view is oriented similarly between all cameras. Ground out to 2.5 to 3 meters should make up ½ of camera field of view. Vegetation/sky then account for the other ½ of the camera field of view.

APPENDIX C

Appendix C

Tables of Statistical Results for Chapters II, III, and IV.

APPENDIX C, TABLE 1. Results of the ANOVA with type III correction determining if there was a significant difference in number of false captures based on side, vegetation, facing, manufacturer, and month included as factors for the control array on State Highway 100, Cameron County, Texas. At the bottom are the results of the generalized linear model testing the influence of average temperature. Significance is indicated with a (*).

Source	Degrees of Freedom	Sum of Squares	*F*	*p*-value
Side	1	7746200	0.2295	0.6322
Vegetation*	2	22887000	3.3901	0.0347
Facing	3	15660000	1.5464	0.2020
Manufacturer*	2	61893000	9.1677	0.0001
Month	1	1053200	0.3120	0.5768
Residuals	386	13030000000		
Source	**Coefficient**	**Standard Error**	***z*-value**	***p*-value**
Temperature*	282	32.3	8.730	< 0.0001

APPENDIX C, TABLE 2. Results of the PERMANOVA and two PERMDISP tests conducted on the wildlife communities captured in the control array around State Highway 100, Cameron County, Texas. Results of the PERMANOVA constitute the upper half of the table, while PERMDISP results occupy the bottom half, with the pairwise results separated. Significance is indicated with (*).

PERMANOVA	Degrees of Freedom	Sum of Squares	Mean Squares	Pseudo-*F*	*p*-value	Unique Terms
Side*	1	5198.8	5198.8	5.6547	0.0001	9937
Vegetation*	2	20069	10034	10.914	0.0001	9933
Side x Vegetation*	2	3827.7	1913.9	2.0817	0.0101	9932
Res	30	27582	919.39			
Total	35	56677				
PERMDISP Results		**Size**	**Average**	**SE**	**F**	***p*-value**
North		18	35.811	2.881	0.0213	0.9036
South		18	47.928	2.35		
Dense Thornscrub		12	26.117	2.1379	10.326	0.0002
Mixed Thornscrub		12	24.451	1.9553		
Open Grassland		12	39.436	3.3472		
Pairwise Comparisons					**t**	***p*-value**
North vs. South					0.14581	0.8922
Dense Thornscrub vs. Mixed Thornscrub					0.5749	0.6025
Dense Thornscrub vs. Open Grassland*					3.3535	0.002
Mixed Thornscrub vs. Open Grassland*					3.8656	0.0006

APPENDIX C, TABLE 3. Results of the SIMPER analysis of wildlife in the control array around State Highway 100, Cameron County, Texas. These results were focused on two factors, vegetation and side of SH100.

Species	Class 1 Average Abundance	Class 2 Average Abundance	Average Dissimilarity	Contribution to Dissimilarity (%)	Cumulative Dissimilarity (%)
Dense Thornscrub and Mixed Thornscrub		Average Dissimilarity: 39.45			
Eastern Cottontail	8.00	7.23	7.09	17.97	17.97
Virginia Opossum	5.38	1.23	6.28	15.91	33.88
Bobcat	4.22	2.27	4.28	10.86	44.74
Northern Bobwhite	3.61	2.82	3.81	9.66	54.39
Dense Thornscrub and Open Grassland		Average Dissimilarity: 66.63			
Eastern Cottontail	8.00	0.58	13.07	19.62	19.62
Virginia Opossum	5.38	0.08	8.98	13.48	33.10
White-tailed Deer	6.16	2.94	7.21	10.82	43.93
Bobcat	4.22	0.39	7.03	10.56	54.48
Nine-banded Armadillo	3.16	0.08	5.54	8.31	62.79
Mixed Thornscrub and Open Grassland		Average Dissimilarity: 63.04			
Eastern Cottontail	7.23	0.58	15.73	24.96	24.96
White-tailed Deer	6.38	2.94	8.53	13.54	38.50
Nine-banded Armadillo	2.76	0.08	6.38	10.13	48.62
Northern Bobwhite	2.82	0.98	4.84	7.68	56.31
North and South (Side of SH100)		Average Dissimilarity: 47.16			
Eastern cottontail	5.21	5.33	5.75	12.20	12.20
White-tailed deer	5.81	4.51	5.15	10.92	23.12
Coyote	1.08	2.68	5.05	10.71	33.84
Nilgai	1.19	2.77	4.68	9.92	43.76

APPENDIX C, TABLE 4. Results of the averaged non-binary general linear models for species richness and all species combined observed at control sites around State Highway 100, Cameron County, Texas. Only coefficients included within the model averaging are provided. A range of McFadden pseudo-R2 scores are also included. These scores cover the range of all models included. The third part of the table are the results of the Student's *t*-test comparing the total number of occurrences within the control array on the north and south sides of SH100. Coefficients with significance *p*-values are indicated with a (*).

Coefficient	Estimate	Standard Error	z Value	*p*-Value
Species Richness		McFadden pseudo-R^2 Range: 0.2525 – 0.2606		
Canopy Cover*	0.0449	0.0087	5.003	< 0.0001
Distance to WCS*	-0.0014	0.0005	2.619	0.0088
Distance to WG	0.0002	0.0006	0.326	0.7445
Side	0.1752	0.4127	0.416	0.6773
All Species Combined		McFadden pseudo-R^2 Range: 0.0651 – 0.0707		
Canopy Cover*	0.0116	0.0029	3.779	0.0002
Distance to WG	-0.00016	0.00021	0.751	0.4528
Distance to WCS	-0.00048	0.00044	1.086	0.2776
Ground Cover	-0.0014	0.0028	0.468	0.6397
Side	0.0205	0.0891	0.224	0.8224
Source		**t**	**Degrees of Freedom**	***p*-value**
Side (North vs. South)		-0.6412	429.14	0.5217

APPENDIX C, TABLE 5. Results of the averaged non-binary general linear models for species observed in the control array around State Highway 100, Cameron County, Texas. Coefficients included within the model averaging for each species are provided. Coefficients with significance *p*-values are indicated with a (*).

Coefficients	Estimate	Std. Error	z Value	*p*-Value
Bobcat	McFadden pseudo-R^2 Score: 0.1826			
Canopy Cover*	0.0287	0.0073	3.916	< 0.0001
Distance to WCS*	-0.0014	0.0053	-2.687	0.0072
Side*	1.552	0.4378	3.545	0.0004
Collared Peccary	McFadden pseudo-R^2 Range: 0.2972 – 0.3068			
Canopy Cover*	0.0564	0.0133	4.092	< 0.0001
Distance to SH100	0.00008	0.0014	0.062	0.9506
Distance to WCS	0.0000089	0.0002	0.057	0.9542
Distance to WG*	0.0086	0.0027	2.914	0.0036
Coyote	McFadden pseudo-R^2 Range: 0.0482 – 0.0753			
Canopy Cover	-0.0093	0.0053	1.686	0.09185
Distance to SH100	0.0070	0.0056	1.216	0.2241
Distance to WCS	0.0003	0.0003	0.947	0.3437
Distance to WG	0.0009	0.0007	1.208	0.2270
Side*	1.2556	0.4254	2.854	0.0043
Eastern Cottontail	McFadden pseudo-R^2 Range: 0.0490 – 0.0643			
Canopy Cover*	0.0328	0.0090	2.483	0.013
Distance to WCS	-0.000087	0.00027	0.316	0.7516
Distance to WG	-0.0015	0.00143	1.029	0.3036
Ground Cover	0.0027	0.0068	0.401	0.6881
Side	-0.0035	0.1009	0.033	0.9735
Nilgai	McFadden pseudo-R^2 Range: 0.0780 – 0.0955			
Distance to SH100	0.0013	0.0030	0.428	0.669
Distance to WG	0.00028	0.00046	0.600	0.549
Ground Cover	0.0013	0.0035	0.380	0.704
Side*	1.422	0.326	4.187	< 0.0001
Side*	1.4373	0.4823	2.872	0.0040

Continuation of Appendix C, Table 5

Coefficients	Estimate	Std. Error	z Value	*p*-Value
Nine-banded Armadillo		McFadden pseudo-R^2 Range: 0.0664 – 0.0822		
Canopy Cover*	0.0396	0.0092	4.167	< 0.0001
Distance to SH100*	0.0227	0.0085	2.534	0.0113
Distance to WCS	-0.000093	0.00030	0.298	0.7654
Distance to WG	-0.00012	0.00046	0.265	0.7909
Ground Cover	0.0018	0.0056	0.311	0.7556
Side	0.0718	0.2760	0.254	0.7994
Northern Bobwhite		McFadden pseudo-R^2 Range: 0.0928 – 0.1089		
Canopy Cover*	0.0246	0.0077	3.085	0.0020
Distance to WCS	-0.00025	0.00045	0.550	0.5820
Distance to WG	-0.00036	0.00077	0.468	0.6400
Ground Cover	-0.0033	0.0067	0.478	0.6328
Side*	1.4373	0.4823	2.872	0.0040
Striped Skunk		McFadden pseudo-R^2 Range: 0.0209 – 0.0285		
Canopy Cover	-0.0160	0.0079	1.949	0.0513
Distance to WCS	-0.00012	0.00032	0.357	0.7214
Distance to WG	-0.00023	0.00059	0.383	0.7017
Side	0.0765	0.2807	0.266	0.7899
Virginia Opossum		McFadden pseudo-R^2 Range: 0.1724 – 0.1849		
Canopy Cover*	0.0529	0.0099	5.139	< 0.0001
Distance to SH100	-0.0066	0.0086	0.759	0.4410
Distance to WG*	-0.0034	0.0012	2.689	0.0072
White-tailed Deer		McFadden pseudo-R^2 Range: 0.0410 – 0.0603		
Canopy Cover*	0.0160	0.0039	3.932	< 0.0001
Distance to SH100	-0.0004	0.0016	0.257	0.7970
Distance to WCS	-0.00017	0.00027	0.623	0.5330
Distance to WG	0.00093	0.00056	1.635	0.1020
Side	-0.1493	0.2530	0.581	0.5610

APPENDIX C, TABLE 6. Total capture range used for calculating expected crossing frequencies using proximity-based occurrences and vegetation-based occurrences calculated from the control array around State Highway 100, Cameron County, Texas. Widths of wildlife crossing structures and wildlife guards are also provided. All measurements are presented in meters.

CONTROL	RANGE	WCS	WIDTH	WG	WIDTH
Dense Thornscrub	24	WCS3	6.0	WG11	4.5
Mixed Thornscrub	24	WCS3A	1.8	WG12	4.5
Open Grassland	24	WCS4	3.0	WG13	9.1
WCS3 Proximity	14			WG14	4.5
WCS3A Proximity	40			WG15	4.5
WCS4 Proximity	18			WG16	4.5
				WG17	4.5

APPENDIX C, TABLE 7. Results of the ANOVA with type III correction comparing the proximity-based performance differentials and vegetation-based performance differentials calculated for the mitigation corridor on State Highway 100, Cameron County, Texas. A Tukey's HSD post-hoc test was used to confirm the results of the ANOVA comparing the proximity-based performance differentials and vegetation-based performance differentials for each structure.

Proximity Vs. Vegetation	Degrees of Freedom	Sum of Squares	Mean Squares	F-value	*p*-value
All	1	235	235	0.042	0.839

APPENDIX C, TABLE 8. Vegetation-based performance differentials for WCS1 and WCS2 in the mitigation corridor on State Highway 100, Cameron County, Texas. If the performance differential is positive it indicates wildlife were seen more frequently at the structure than expected. If the performance differential is negative it indicates wildlife were seen less frequently at the structure than expected. Zero indicates the species was observed the number of times as expected, which may be not at all (indicated by a *).

Species	Black-tailed jackrabbit	Bobcat	Collared peccary	Coyote	Eastern cottontail	Nilgai	Nine-banded armadillo	Northern bobwhite	Northern raccoon	Striped skunk	Texas tortoise	Virginia opossum	White-tailed deer	Total
WCS1	-7.4	4.9	0*	-4	-1	-2.9	0*	-1	69.3	-5.3	-0.2	2.6	-6.5	49
WCS2	-7.4	6.9	0*	-5.6	-1	-2.9	0*	-1	73.2	-5.3	-0.2	272.3	-6.5	323

APPENDIX D

Appendix D

Scientific Names of Species in the Impact Array

Common and scientific names for all species observed crossing wildlife crossing structures or wildlife guards. There were 20 species observed within the roadside array. Species excluded from this list but observed in the array were unidentifiable *rodentia* species and birds that use flight as their predominant transportation.

Common Name Used	Scientific Name
Black-tailed jackrabbit	*Lepus californicus*
Bobcat	*Lynx rufus*
Collared peccary	*Pecari tajacu*
Coyote	*Canis latrans*
Domestic cat	*Felis catus*
Domestic cow	*Bos taurus*
Domestic dog	*Canis lupus familiaris*
Eastern cottontail	*Sylvilagus floridanus*
Greater roadrunner	*Geococcyx californianus*
Long-tailed weasel	*Mustela frenata*
Nine-banded armadillo	*Dasypus novemcinctus*
Northern bobwhite	*Colinus virginianus*
Northern raccoon	*Procyon lotor*
Striped skunk	*Mephitis mephitis*
Texas indigo snake	*Drymarchon melanurus erebennus*
Texas tortoise	*Gopherus berlandieri*
Virginia opossum	*Didelphis virginiana*
Western coachwhip	*Masticophis flagellum*
Western diamondback	*Crotalus atrox*
White-tailed deer	*Odocoileus virginianus*

APPENDIX E

Appendix E

Scientific Names of Species in the Control Array

Common and scientific names for all species observed within the control array. There were 31 species observed within the control array. Species excluded from this list but observed in the array were unidentifiable *rodentia* species and birds that use flight as their predominant transportation.

Common Name Used	Scientific Name
Black-tailed jackrabbit	*Lepus californicus*
Bobcat	*Lynx rufus*
Chachalaca	*Ortalis vetula*
Collared peccary	*Pecari tajacu*
Coyote	*Canis latrans*
Domestic cat	*Felis catus*
Domestic cow	*Bos taurus*
Domestic dog	*Canis lupus familiaris*
Domestic horse	*Equus ferus caballus*
Domestic sheep	*Ovis aries*
Eastern cottontail	*Sylvilagus floridanus*
Feral hog	*Sus scrofa*
Greater roadrunner	*Geococcyx californianus*
Long-tailed weasel	*Mustela frenata*
Mexican ground squirrel	*Spermophilus mexicanus*
Nilgai	*Boselaphus tragocamelus*
Nine-banded armadillo	*Dasypus novemcinctus*
Northern bobwhite	*Colinus virginianus*
Northern raccoon	*Procyon lotor*
Patch-nosed snake	*Salvadora grahamiae*
Striped skunk	*Mephitis mephitis*

Texas horned lizard	*Phrynosoma cornutum*
Texas indigo snake	*Drymarchon melanurus erebennus*
Texas spiny lizard	*Sceloporus olivaceus*
Texas tortoise	*Gopherus berlandieri*
Virginia opossum	*Didelphis virginiana*
Western coachwhip	*Masticophis flagellum*
Western diamondback	*Crotalus atrox*
White-tailed deer	*Odocoileus virginianus*

BIOGRAPHICAL SKETCH

The author, Taylor Miles Hopkins, was born in Anchorage, Alaska, and spent his childhood split between Kenai Peninsula and Columbia Falls, Montana. He earned a Bachelor of Science degree in Resource Conservation from the University of Montana in 2013. He graduated with honors and minors in Climate Change Science & Sociology and Wilderness Studies. While at the University of Montana, Taylor completed an internship with the Montana Transportation Institute studying road ecology on the Highway 93 road mitigation corridor under the guidance of Dr. Marcel Huijser. He also completed an undergraduate thesis with Dr. Andrew Larson, focused on comparing the effects of forest restoration projects to forests under historical fire regimes. The results of which were published in the journal *Forest Science* in 2014. Taylor worked for the United States Forest Service from 2013 to 2017, serving as a forestry technician focused on fire effects monitoring and serving as a wildland fire lookout.

Taylor later enrolled at the University of Texas Rio Grande Valley, where he completed his Master of Science Degree in Biology in 2020. His thesis focused on designing a control-impact study for the road mitigation corridor on State Highway 100 and calculating performance and permeability for the corridor. Outside of academics, Taylor enjoys the outdoors while hiking, fishing, and riding horses, and when it is too stormy outside he enjoys table top roleplaying games. Taylor may be reached at taylor.miles.hopkins@gmail.com.