12-2022

South Texas Wildlife Activity Across a Fragmented Landscape and Road Mitigation Corridor

Caitlin K. Brett

The University of Texas Rio Grande Valley

SOUTH TEXAS WILDLIFE ACTIVITY ACROSS
A FRAGMENTED LANDSCAPE AND
ROAD MITIGATION CORRIDOR

A Thesis

by

CAITLIN K. BRETT

Submitted in Partial Fulfillment of the

Requirements for the Degree of

MASTER OF SCIENCE

Major Subject: Agricultural, Environmental, and Sustainability Sciences

The University of Texas Rio Grande Valley

December 2022

SOUTH TEXAS WILDLIFE ACTIVITY ACROSS

A FRAGMENTED LANDSCAPE AND

ROAD MITIGATION CORRIDOR

A Thesis
by
CAITLIN K. BRETT

COMMITTEE MEMBERS

Dr. Richard J. Kline
Chair of Committee

Dr. John H. Young Jr.
Committee Member

Dr. Sarah Lehnen
Committee Member

December 2022

ABSTRACT

Brett, Caitlin K. South Texas Wildlife Activity Across a Fragmented Landscape and Road Mitigation Corridor. Master of Science (MS), December, 2022, 114 pp., 12 tables, 28 figures, references, 133 titles.

Wildlife crossing structures (WCS) and roadside fencing are commonly installed to mitigate habitat fragmentation, wildlife road mortalities, and other negative effects that roads can have on the surrounding landscape. Eight such WCS were constructed below Farm-to-Market (FM)106 in Cameron County, Texas, across a 16 km corridor transecting the Laguna Atascosa National Wildlife Refuge. These WCS, paired with adjacent roadside fencing, were intended to prevent road mortalities of the endangered ocelot (*Leopardus pardalis*) and to mitigate the barrier effect of FM106 on this and other meso-mammal species. This study will analyze camera trap data from roadside and habitat reference sites to model target species activity throughout the study corridor and identify changes in broader community composition associated with the road and its mitigation structures. This analysis will allow for more accurate estimates of mitigation structure performance while controlling for the influence of land cover characteristics on target species detections.

DEDICATION

I will be forever grateful to my family and friends who supported my decision to move to possibly the remotest corner of the lower forty-eight to study a cat they didn’t know existed in Texas to begin with. Thank you to the delightful community of the Lower Rio Grande Valley for inspiring me to improve my Spanish, for sharing its incredible wealth of cultural and biotic diversity, and for forever ruining my palate for subpar Mexican food. Finally, thank you to my labmates for sharing in and perpetually making light of the struggle that is graduate school, and to my committee members for their tireless support and guidance throughout this process.

ACKNOWLEDGMENTS

I would first like to thank the Texas Department of Transportation and the Laguna Atascosa National Wildlife Refuge for together making this road mitigation project possible and for funding so many associated research opportunities. I am beyond grateful for Dr. Richard Kline, Dr. John Young Jr., and Dr. Sarah Lehnen for their continued advice and support in developing my master's thesis. Finally, a special thank you to everyone who braved the South Texas thornscrub to assist with my field work and/or provided immensely helpful emotional support from the sidelines: T. Miles Hopkins, Adam Sanjar, Anna Mehner, Victoria Hanley, Alexa Campos, Jamie Langbein, Madison Nadler, Molly Picillo, Diego Luna, Marie "Mania" Curie, Dr. Rachael Gwinn, Mom, Dad, Kevin, Andrew, John, and Liz Brett.

TABLE OF CONTENTS

Page

ABSTRACT iii
DEDICATION iv
ACKNOWLEDGMENTS v
TABLE OF CONTENTS vi
LIST OF TABLES viii
LIST OF FIGURES ix
CHAPTER I. INTRODUCTION 1
  Road Ecology 2
    Mitigation Structures 4
    WCS Monitoring 5
    WCS Monitoring Methods 7
  South Texas Road Mitigation 8
    South Texas Ocelots 8
    FM106 Study Area 10
  Methods 12
  Thesis Objectives 17
CHAPTER II. QUANTIFYING WILDLIFE COMMUNITY COMPOSITION ACROSS THE FM106 MITIGATION CORRIDOR 18
  Introduction 18
  Objectives & Hypotheses 19
  Methods 19
  Results 23
  Discussion 26
CHAPTER III. MODELING TARGET SPECIES ACTIVITY ACROSS THE FM106 MITIGATION CORRIDOR 32
  Introduction 32

Objectives & Hypotheses 33
Methods 34
Results 37
Discussion 38
Spatial Analysis 38
Temporal Analysis..40
CHAPTER IV. ESTIMATING THE PERFORMANCE OF THE FM106 MITIGATION STRUCTURES 43
Introduction 43
Objectives & Hypotheses 45
Methods 45
Results 48
Discussion 50
CHAPTER V. CONCLUSIONS 56
REFERENCES 102
BIOGRAPHICAL SKETCH 114

LIST OF TABLES

Page

Table 1: List of Documented South Texas Wildlife Species....61

Table 2: FM106 Mitigation Structure Dimensions....63

Table 3: Detection Event Classification for WCS Interactions....64

Table 4: Species Detections per Vegetation Class....66

Table 5: Species Detections per Distance Bin....67

Table 6: SIMPER Analysis of Vegetation Class....68

Table 7: SIMPER Analysis of Roadside vs Habitat....69

Table 8: Spatial, Temporal, and Road Surface Avoidance per Species....70

Table 9: WCS Performance Scores....72

Table 10: Fencing Performance Scores....73

Table 11: Predation Event Comparison....74

Table 12: FM106 vs SH100 Species Distribution....75

LIST OF FIGURES

Page

Figure 1: South Texas Ocelot Populations and Documented Road Mortalities 76
Figure 2: FM106 Mitigation Corridor...77
Figure 3: FM106 Wildlife Crossing Structures (WCS) 1-8..78
Figure 4: Camera Trap Array for the FM106 Mitigation Corridor...79
Figure 5: FM106 Mitigation Corridor Habitat Site Selection...80
Figure 6: Species Accumulation Curves per Vegetation Class 81
Figure 7: Non-Metric Multidimensional Scaling Plots per Vegetation Class 82
Figure 8: Non-Metric Multidimensional Scaling Plots per Distance Bin.....................................82
Figure 9: DistLM Results for Community Variation..83
Figure 10: DistLM Results for Species Richness 83
Figure 11: Species Richness per Vegetation Class..84
Figure 12: Species Richness per Distance Bin 85
Figure 13: Bobcat Detections over Woody Canopy Cover Density...86
Figure 14: Mammal Community Diel Activity Patterns..87
Figure 15: Bobcat Diel Activity Comparison at Roadside vs Habitat..88
Figure 16: Coyote Diel Activity Comparison at Roadside vs Habitat..89
Figure 17: White-tailed Deer Diel Activity Comparison at Roadside vs Habitat.........................90
Figure 18: Felid Prey Taxa Diel Activity Comparison at Roadside vs Habitat............................91
Figure 19: Striped Skunk Diel Activity Comparison at Roadside vs Habitat...............................92
Figure 20: Bobcat Performance per WCS 93
Figure 21: Coyote Performance per WCS 94
Figure 22: White-tailed Deer Performance per WCS 95
Figure 23: Striped Skunk Performance per WCS...96
Figure 24: Bobcat Performance per FE 97
Figure 25: Coyote Performance per FE 98

Figure 26: White-tailed Deer Performance per FE 99
Figure 27: Striped Skunk Performance per FE 100
Figure 28: WCS Performance Differentials vs % Success Rate 101

CHAPTER I

INTRODUCTION

Wildlife crossing structures (WCS) paired with roadside fencing are commonly used to mitigate the direct and indirect negative effects of roadways on wildlife communities (van der Ree et al. 2015). In Cameron County, Texas, eight such mitigation structures were installed under Farm-to-Market Road (FM)106 to mitigate ocelot (*Leopardus pardalis*) road mortalities and habitat fragmentation. The FM106 mitigation corridor transects the Laguna Atascosa National Wildlife Refuge (LANWR), which hosts one of the two remaining breeding populations of the endangered ocelot in the United States (Haines et al. 2006b, Janečka et al. 2008, Lehnen et al. 2021). Ongoing camera trap monitoring of the FM106 corridor has documented use of WCS by ocelots in addition to dozens of other wildlife species. However, comparing actual crossing rates with expected crossing rates based on adjacent habitat monitoring to determine WCS performance (Clevenger and Waltho 2000, Andis et al. 2017) has not yet been conducted. It is also uncertain to what extent the road-effect zone (Forman and Alexander 1998) impacts the composition and activity of the FM106 wildlife community. To address these questions, camera trap arrays were established to monitor various habitat types at increasing distances from FM106. Vegetation, particularly woody canopy cover, is one relevant and quantifiable factor that can be used to differentiate between habitat types in this study corridor. South Texas ocelots, alongside several other target species, have been shown to closely associate with dense woody cover (Shindle and Tewes 1998, Harveson et al. 2004, Horne et al. 2009, Hopkins 2020) despite its limited availability on the landscape. Understanding how different species associate with vegetation and other land cover characteristics (e.g. proximity to water, distance to secondary roads, and groundcover structure) is important for optimizing future placement of WCS and mitigation fencing. Data collected from the roadside and habitat reference camera arrays will be used to 1) quantify patterns and potential changes in the wildlife community immediately surrounding the FM106 mitigation corridor; 2) model target species detections to determine the importance of the road-effect zone and land cover characteristics in predicting their activity and movement; 3) estimate the performance of each mitigation structure by comparing actual versus expected crossing rates. Due to the low number of anticipated ocelot detections and the overlap in behavior and ecological roles they share with bobcats (*Lynx rufus*), the bobcat will act as a proxy target species for the purposes of this study. Secondary target species will include coyote (*Canis latrans*), striped skunk (*Mephitis mephitis*), and white-tailed deer (*Odocoileus virginianus*). These were chosen to represent a cross-section of native mammal species which could be detected reliably and in sufficient numbers for analysis and to represent a variety of impacts that roads and mitigation structures may have on the wildlife community.

Road Ecology

Anthropogenic, linear infrastructure features (e.g. roads, fencing, and railroads) have long impacted the landscapes they transect (van der Ree et al. 2015, Jakes et al. 2018). The ubiquity and pace of modern construction projects far outmatch that of potential evolutionary adaptation, thereby challenging biotic communities in a way that comparable natural landscape features (e.g., rivers, canyons) do not (Fahrig 2007). Roadways are the most widespread and consistently trafficked of all such infrastructure, spanning over 4 million miles in the U.S. alone (FHA 2020). Furthermore, national road mileage and traffic volume are both expected to increase into the foreseeable future (2020). Rural roads and highways alone make up less than 1% of land in the continental U.S. (Bigelow and Borchers 2017) but may impact a much larger portion of the landscape due to their direct and indirect effects on adjacent ecosystems (Forman 2000).

Roadways have a direct impact on surrounding wildlife communities in two major ways. First, by converting land into road surface, they eliminate habitat, introduce edge conditions, and may impose a structural barrier to wildlife movement between bisected areas (Forman and Alexander 1998, van der Ree et al. 2015). Second, roadways facilitate wildlife-vehicle collisions, often resulting in road mortalities (van der Ree et al. 2015). The full impact of wildlife-vehicle collisions varies depending on the species or population in question and the degree to which individuals avoid or successfully cross the roadway (Jaeger et al. 2005, Fahrig and Rytwinski 2009, van der Ree et al. 2015).

The indirect impacts of roadways on wildlife are often interdependent and negative, especially for medium to large mammal species (Fahrig and Rytwinski 2009). Habitat fragmentation combined with sensory disturbance (i.e. noise, light) and pollution may lead to degraded habitat availability (De Molenaar et al. 2006), altered trophic interactions (Clevenger and Waltho 2000, Ditmer et al. 2021), behavioral or phenological changes (Reijnen et al. 1995, McClure et al. 2013), species introductions (Hansen and Clevenger 2005), long-term erosion of population persistence, and genetic isolation (Jaeger and Fahrig 2004, Jaeger et al. 2005, van der Ree et al. 2015). These effects vary depending on the behavior (i.e. degree of road avoidance) and biology (i.e. movement capability, home range size, reproductive rate) of the species or population in question (Jaeger et al. 2005, Fahrig and Rytwinski 2009). Indirect effects can extend far beyond the road itself so the amount of land experiencing these effects is uncertain. Over 20% of the continental U.S. is located within 150 meters of a road; over 80% is located within a kilometer (Riitters and Wickham 2003). Documenting the extent of the indirect effects of roads on surrounding biotic communities is important in understanding the total amount of disturbance taking place on a national and global scale.

The delineation of these cumulative effects is termed the road-effect zone (Forman and Alexander 1998). This zone may be asymmetrical and varied depending on the taxa studied and the interplay of ecological factors considered, but generally denotes the greatest documented effect-distance across all variables (Forman and Deblinger 2000). Estimates of the road-effect zone have ranged from within 300 m (Reijnen et al. 1995, Forman and Alexander 1998, Forman and Deblinger 2000, Hansen and Clevenger 2005, Benítez-López et al. 2010), to over 5 km (Reijnen et al. 1995, Forman and Alexander 1998, Benítez-López et al. 2010) for various taxa. These effect-distances are commonly based on observed changes in the relative abundance or density of a species, but more recent studies have also considered wildlife movement, activity indices, or diel activity to assess how roads and other forms of human disturbance may alter wildlife behavioral patterns (Barrueto et al. 2014, Andis et al. 2017, Gaynor et al. 2018a, Watabe and Saito 2021).

Mitigation Structures

A common approach to mitigating the effects of roadways on wildlife is the construction of wildlife crossing structures (WCS) paired with roadside exclusion fencing (van der Ree et al. 2015). Whereas fencing alone can only mitigate the direct effects of wildlife-vehicle collisions, fencing paired with WCS can also funnel wildlife from the surrounding landscape towards safe crossings over or under the roadway, potentially restoring some degree of habitat connectivity for species which utilize the structures (van der Ree et al. 2015).

WCS can vary greatly in their design depending on the target species, from expansive vegetated overpasses to minimally modified culvert-tunnels (Clevenger and Huijser 2011). However, WCS placement on the landscape also influences how well these structures are able to meet conservation goals such as habitat connectivity or mortality mitigation (van der Ree et al. 2015). Effective design relies on the predictability and understanding of wildlife responses to mitigation structures themselves whereas effective placement relies on the predictability and understanding of wildlife movement across the landscape. Because of the inherent heterogeneity of most habitat conditions, it is inadvisable to conduct research on the effectiveness of WCS design without first determining the influence of WCS placement (Andis et al. 2017).

WCS Monitoring

If successful, mitigation structures may restore some degree of habitat connectivity while limiting mortality risk for local populations. However, defining this success requires rigorous and often long-term study in order to document behavioral habituation and population-level effects (Clevenger and Waltho 2003, Gagnon et al. 2011, van der Grift et al. 2013, van der Grift et al. 2015). While usage may be the most common metric used in defining WCS success (Denneboom et al. 2021), some argue that measuring WCS performance is necessary to understand their ability to provide connectivity for the local wildlife community (Clevenger and Waltho 2000, van der Grift et al. 2015, van der Ree et al. 2015, Andis et al. 2017). Performance can be calculated by comparing the actual crossing frequency (ACF) at each WCS with an expected crossing frequency (ECF) based on comparable habitat reference sites (Clevenger and Waltho 2000, Andis et al. 2017). By contextualizing WCS use with baseline habitat data, performance accounts for the effect of WCS placement and allows for more accurate assessment of design characteristics (Andis et al. 2017). Determining performance can be accomplished using a Control-Impact study design to compare wildlife activity at the roadway mitigation structures (Impact) to activity at unmitigated sections of the roadside (Control) or in adjacent habitat away from the road (Control) (Benítez-López et al. 2010, van der Ree et al. 2015).

Control monitoring habitat within the estimated road-effect zone concentrates survey effort on species which are willing to approach the road to some degree (Andis et al. 2017). However, if the road-effect zone is unknown and no similar studies have been conducted, then monitoring habitat beyond the potential effect-distance may be necessary to assess the full impact of the road's presence and detect possible avoidance behaviors among the community (van der Ree et al. 2015, Ascensão et al. 2019). In addition to determining WCS performance and the road-effect zone, control reference sites (i.e. locations in surrounding habitat with comparable conditions to those seen at roadside impact sites) may also help quantify land cover characteristics that predict wildlife movement in and around the mitigation corridor (Lewis et al. 2011, Hopkins 2020). Such methods can then inform future placement, design, or modification of mitigation structures (van der Ree et al. 2015).

Regardless of larger study goals, reference sites should be replicated to closely match the habitat conditions present at mitigation structures in order to avoid confounding factors in quantifying the impact of the mitigation corridor (Smith and van der Ree 2015, van der Grift et al. 2015). Some researchers recommend monitoring for a variety of taxa rather than a single target species (Clevenger and Waltho 2003, van der Grift et al. 2015), particularly given the variation in home range size, life span, recruitment rate, and potential habituation that may occur across different taxa. Multi-species monitoring is crucial because both roads and their associated mitigation structures can have disproportionate impacts on the community. Even seemingly positive effects, such as repeat WCS usage, predator release, access to areas for thermoregulation, or foraging opportunities (Fahrig and Rytwinski 2009, Morelli et al. 2014) may actually represent the decoupling of important trophic relationships to the deficit of the overall biotic community (Clevenger and Waltho 2005).

WCS Monitoring Methods

Methods for monitoring road mitigation corridors vary in cost, efficacy, required survey effort, and detection biases (Smith and van der Ree 2015). For mammals in particular, these often include radio-telemetry or GPS tracking, track-pad surveying, road mortality surveys, and camera trapping (van der Ree et al. 2015). While tracking can offer more precise information about the movement of individual animals, it is time and resource-intensive to conduct even for a single species (Kays et al. 2010, van der Ree et al. 2015). Track pads are capable of monitoring multiple species but require frequent visits for data collection and maintenance and are especially limited in their capacity to identify repeat detections of individuals (van der Ree et al. 2015).

Road mortality surveys are commonly used to monitor multiple species, though they are limited by the sample interval and detection probabilities for different taxa (van der Ree et al. 2015). Animals which exhibit a strong degree of road or traffic avoidance will not be well represented in this sampling method, whereas species which are attracted to the road and lack traffic avoidance may be overrepresented (Jaeger et al. 2005, van der Ree et al. 2015). Road mortality surveys detect the direct impact of roads (via wildlife-vehicle collisions) but are a poor indicator of the indirect impacts of roads and corresponding mitigation structures (Ascensão et al. 2019). Especially in the case of small (two-lane) and low traffic roads, these indirect effects likely outweigh mortalities in terms of negative impact on local populations (Jaeger et al. 2005), so additional monitoring is necessary.

Camera trapping offers multiple advantages compared to other techniques as cameras are minimally invasive, capable of monitoring several species simultaneously, low-maintenance, and increasingly cost-effective (Silvy 2020). Photo and video footage allow insight into animal behavior that other observational or interventional surveys could never provide by documenting activity absent a human researcher's presence (Kays et al. 2010). Camera traps with passive infrared (PIR) mechanisms are especially appropriate for medium to large mammals, as their relative size and speed makes them more prone to detection by one or more of the camera's pyroelectric sensors (Meek et al. 2014, Welbourne et al. 2016, Jumeau et al. 2017). However, without the capacity for identifying individuals or repeat detections within a population, abundance or density estimates can be challenging if not untenable to accurately calculate for multiple species across a single camera trap array (Sollmann et al. 2013, Burton et al. 2015, Parsons et al. 2017). Repeated behavior by resident individuals may inadvertently skew detection rates or recorded interaction behaviors. As with any sampling method, camera trapping has drawbacks and the potential for imperfect detection rates; understanding and accounting for these limitations is important when drawing conclusions from this method of data collection (Sollmann et al. 2013, Burton et al. 2015).

South Texas Road Mitigation

South Texas Ocelots

The ocelot is a species of neotropical felid native to every mainland country in the Americas but Canada and Chile (Paviolo 2015). In the U.S., the historic range of the ocelot extended from the southern border throughout east-central Texas to Arkansas and Louisiana, and separately in a small area of southeast Arizona (Murray and Gardner 1997, Haines et al. 2006a, Grigione et al. 2007, Grigione et al. 2009). Citing habitat loss and modification, the U.S. Fish and Wildlife Service (USFWS) added the ocelot to the register of federally endangered species in 1982 (USFWS 1982). Their present-day range is restricted to South Texas, with just two known breeding populations of less than 80 total individuals remaining (Haines et al. 2006a, Janečka et al. 2008, Lombardi et al. 2020b, Lehnen et al. 2021). The larger of these populations is located on private ranchlands in Willacy and Kenedy Counties, while a smaller population exists on and around LANWR in Cameron County, TX (Laack 1991, Haines et al. 2006a, Lehnen et al. 2021) (Figure 1). Little to no movement exists between the remaining groups, and evidence of genetic isolation has been documented between the two U.S. populations and compared to the next closest population in Mexico (Janečka et al. 2011).

South Texas ocelots are shown to be strongly associated with dense (i.e. 95% canopy cover) woody vegetation and may select for patches of dense Tamaulipan thornscrub woodlands where available (Shindle and Tewes 1998, Harveson et al. 2004, Horne et al. 2009). Over 90% of native woodlands in Cameron County have been lost, primarily to agriculture, as of 1983 (Tremblay et al. 2005) and estimates of riparian woodland destruction in the entire Lower Rio Grande Valley are as high as 99% (Jahrsdoerfer and Leslie 1988). In addition to severe habitat loss, anthropogenic fragmentation of remaining habitat appears to reinforce the isolation between populations, despite a limited degree of habitat connectivity existing between the groups (Lehnen et al. 2021). Roadways not only contribute to this fragmentation through their indirect effects, but also account for a significant number of ocelot mortalities; in one 19-year study, 35% of recorded ocelot mortalities were due to wildlife-vehicle collisions (Haines et al. 2005). Their population decline in the region aligns with the more general theory that species of higher trophic levels (e.g. *Carnivora*) with larger home range requirements may be at a higher risk of extinction, especially when combined with naturally low population densities and low reproductive rates (Cardillo et al. 2004).

The ocelot is just one charismatic and well-studied member of the Lower Rio Grande Valley ecosystem, but its population decline may be indicative of risks that threaten other native species (Jahrsdoerfer and Leslie 1988). Likewise, mitigation measures designed for the ocelot may benefit a wider portion of the wildlife community; to date, wildlife observed successfully using at least one WCS have included 20 mammal, 3 bird, and 6 herpetofauna species (Kline et al. 2022) (Table 1).

FM106 Study Area

The study area is located within Cameron County, TX, which is bordered by the Gulf of Mexico to the east and the U.S.-Mexico border (the Rio Grande/Río Bravo) to the south. The area has a semi-arid subtropical climate with an average annual temperature of 23˚C (74˚F) and average annual precipitation of 62.2cm (24.5 inches) (NOAA 2021). It is situated at the intersection of three ecoregions: a) the Laguna Madre Barrier Islands and Coastal Marshes, b) the Lower Rio Grande Alluvial Floodplain, and c) the Lower Rio Grande Valley (EPA 2013). Cameron County's human population has doubled from 210,000 in 1980 to over 421,000 in 2020 (US Census Bureau, 2020). The sprawling urban centers of Brownsville and Harlingen dominate the western half of the county alongside a semi-residential agricultural matrix. The east is defined by a gradient of coastal wetlands, the South Padre barrier island, and the Port of Brownsville industrial complex. Interlaced among these is a patchwork of land belonging to the LANWR (est. 1946) and Lower Rio Grande Valley (est. 1980) National Wildlife Refuge systems, which host a diversity of migratory and resident wildlife (Jahrsdoerfer and Leslie 1988) (Figure 1).

Four major road mitigation corridors in this region were established by the Texas Department of Transportation (TxDOT) to address ocelot mortalities and habitat fragmentation associated with South Texas roadways. These included several underpass wildlife crossing structures and roadside mitigation fencing constructed along sections of Farm-to-Market Road (FM)106, State Highway (SH)100, SH48, and FM1847. This thesis focused on the FM106 mitigation corridor.

The FM106 mitigation corridor spanned a 16 km segment of the roadway and extended 1 km from the road in each direction. FM106 is a two-lane rural highway with a posted speed limit of 97 km/hr (60 mph) and a public right-of-way ranging from 24 to 36 m in total width. Agricultural use (cotton, sorghum crops) comprised 27% of the 1 km buffer study area. The corridor also intersected residential developments near San Roman and Schafer roads, recreational trails available for public access as part of LANWR, and access roads to the Port Isabel Detention Center (a U.S. Immigration and Customs Enforcement facility) and the Port Isabel-Cameron County airport (Figure 2).

The study area contained a variety of vegetation communities: mesquite (*Prosopis glandulosa*) savannah interspersed with understory grasses (native and non-native spp.), prickly pear (*Opuntia spp.*), and sea ox-eye (*Borrichia frutescens*); irregularly inundated salt flats dominated by a handful of low-growing, halophytic species including gulf cordgrass (*Spartina spartinae*), shoregrass (*Distichlis littoralis*), and succulents *Batis maritima* and *Salicornia spp.*; coastal prairie (various spp.); and patches of riparian and upland thornscrub woodlands, largely comprised of Texas ebony (*Ebenopsis ebano*), lime prickly-ash (*Zanthoxylum fagara*), retama (*Parkinsonia aculeata*), huisache (*Vachellia farnesiana*), granjeno (*Celtis pallida*), mesquite, prickly pear, and a variety of other low-growing shrub species (Jahrsdoerfer and Leslie 1988, Richardson 2011).

Because most FM106 wildlife crossing structures (WCS) were built below road grade to serve a dual purpose as road drainages, they were often placed within ditches or along resaca edges and experience episodic flooding with major rain or runoff events. Resacas (e.g. *Resaca de los Cuates*, which bisected the study area) are water bodies comparable to extended oxbow lakes which historically experienced flooding or regular waterflow as distributaries of the Rio Grande/*Río Bravo* (McIntosh et al. 2019). Areas with high moisture availability such as resacas and drainages have been closely associated with dense thornscrub growth, and may provide corridors for wildlife movement in this region (Jahrsdoerfer and Leslie 1988).

All eight WCS were prefabricated concrete box culverts with two basic designs: wing-walled or square-facing with catwalk ledges (Figure 3). Dimensions varied between structures; WCS have an average length of 22.3 m (min. 16.5 m, max. 31.4 m), average height of 1.6 m (min. 1.1 m, max. 2.4 m), and an average width of 1.8 m (min. 1.4 m, max. 2.4 m) for an average openness ratio (defined as width x height/length, (Reed et al. 1979) of 0.2 m across all structures (min. 0.09 m, max. 0.26 m) (Table 2). Each WCS was flanked by a segment of chain-link fencing, with variable lengths ranging from 50 to 550 m.

Methods

Monitoring on FM106 included 36 “habitat array” and 102 “roadside array” Reconyx Hyperfire 2 camera traps, all unbaited. Habitat sites included a single camera placed between 21 m and 1000 m away from the road. Roadside sites, all within 20 m of the road surface, included three cameras facing towards each WCS entrance (48 total), one camera facing directly away from each WCS entrance (16 total), one at the end of each mitigation fence segment (26 total), and one on either side of the road at the six unmitigated right-of-way sites (12 total; Figure 4). All cameras were triggered by a passive infrared (PIR) mechanism which detected changes in radiated heat across portions of the field of view (i.e. when an animal moved in front of the camera), and all sites were monitored by at least one standard PIR camera trap.

In addition to one PIR camera facing each WCS entrance and one PIR camera facing a WCS-adjacent area on either side of the road, WCS sites had two additional methods for collecting animal behavior and felid identification data: "trip" cameras and video cameras (Figure 4). "Trip" cameras were programmed to trigger with an active infrared (AIR) mechanism which connected to an infrared beam emitter and reflector across the WCS entrance and detected when that beam was broken (i.e. when an animal entered or exited the crossing). AIR cameras were located facing each entrance at seven of the WCS; one site (WCS 3) was not monitored with AIR cameras due to the risk of flooding regularly damaging the sensor mechanism. Video cameras, centered to directly face each WCS entrance, were programmed to take a 30-second video with each trigger rather than three individual photos. These additional methods, while often redundant for providing simple detection data, were helpful for determining how wildlife interacted with mitigation structures and for confirming the frequency of actual crossing events.

Unmitigated right-of-way (ROW) sites were selected by surveying unfenced sections of the roadside on foot and identifying areas where wildlife likely approached and crossed the surface of the road (based on scat, prints, or altered vegetation structure). Cameras were secured in plastic utility boxes and positioned to face directly away from the roadway.

Fence-end (FE) cameras were affixed as closely as possible to the second post from the end of each fence segment and angled such that wildlife could be observed moving between the right-of-way and the adjacent habitat area.

Habitat camera monitoring sites were determined using a stratified-random selection process, using three factors to categorize potential sites: side of road (East/West), distance from road (“near” = 20-320 m, “intermediate” = 321-600 m, “far” = 601-1000 m), and density of local canopy cover (open = 0-30%, mixed = 31-69%, dense = 70-100%). Stratifying sites in this way was necessary for delineating potential road-effects across the study area, replicating the variation in conditions observed at mitigation structure sites, and ensuring representation of dense thornscrub habitat despite its limited availability on the landscape. Vegetation classes (open, mixed, dense canopy cover) were initially estimated by referencing the woody cover layer of a classified National Agriculture Imagery Program (NAIP) land cover map. This map (estimated at 87% accuracy) was generated using supervised classification with a random forest model in ArcGIS Pro 2.8 (ESRI 2021, Yamashita unpublished work) to identify major land uses based on 2016 NAIP imagery. These classifications were ground-truthed with canopy cover surveys prior to final camera placement. Potential sites were limited to land managed by LANWR which could be regularly accessed to collect data and maintain camera equipment, and required a minimum of 100 m spacing from each other in order to limit spatial autocorrelation between cameras (Kays et al. 2010, Parsons et al. 2017). Otherwise, sites were numbered by their assigned attribute table ID in ArcGIS Pro and randomly selected until all categories were fulfilled. The final 36 sites included 12 each of open, mixed, and dense vegetation classes overlapped with 12 each 0-300 m, 300-600 m, and 600-1000 m distance bins; 18 sites were located north/east, and 18 sites were located south/west of FM106 (Figure 5).

Prior to habitat camera placement, protocols were established to ensure consistency across the array. These were: a) cameras were faced north (+/- 45 degrees in the case of any obstructions) to limit detection interference or obscured images due to direct sunlight, b) sites had at least 2 m of clearance or maintainable vegetation in front of each camera view, and c) canopy cover density survey results fell within the site's expected vegetation class. If these conditions were not met at the site centerpoint, a spiral track pattern was followed in 1 m increments until a suitable location was found within a 15 m radius of the original coordinates (Hopkins 2020).

The habitat array cameras were set to take a single photo per trigger with no delay between triggers, using the same PIR mechanism as the roadside cameras. Habitat cameras were affixed to powder coated steel fence U-posts (Everbilt model # 901154EB) at 50-55 cm from the ground and angled such that the horizon line in each photo is at least 2/3 from the bottom of the frame of view. This allowed for a wide size range of wildlife to be captured at each site (while minimizing the need for vegetation clearing) based on the placement of the camera's "detection band" within its PIR sensor array (Kays et al. 2010, ReconyxInc. 2018). Herbaceous vegetation was cleared in a 2.5 m arc and a 3 m marker placed ahead of camera facing to ensure comparable detection and species identification across all vegetation classes.

For all habitat, right-of-way, and WCS-adjacent sites, GIS measurements and field vegetation surveys were conducted to quantify environmental variables across both camera arrays. Percentage canopy cover was measured at the centerpoint of each site and at 5 m in each cardinal direction, taken from camera height (50cm off the ground) using a GRS Densitometer (Geographic Resource Solutions, Arcata, CA) and verified with photo documentation. For consistency across roadside sites, survey points were oriented such that the "south" point was represented by the WCS entrance or ROW camera, and the "north" point was 10 m perpendicular to the road. Groundcover surveys were conducted using 0.5 m quadrats to categorize vegetation structure and quantify the percentage of bare ground within 5 m of each camera view. Additional measurements were taken using the classified NAIP imagery (Yamashita unpublished work) to quantify distances from each camera to water sources, woody cover patches (minimum 1000 sq m), and primary (FM106) and secondary (San Roman, Ted Hunt, Buena Vista, Veterans Airport, FM510, or Schafer) roads. The amount of woody cover between each site and primary/secondary roads was also measured in ArcGIS Pro (ESRI 2021) and standardized to calculate the average percent of woody cover along distance-to-road transects.

Camera trap sites were visited every 2-4 weeks to clear vegetative growth, ensure proper functioning of all equipment, exchange SD cards, and collect photo data. Photos were processed using the program ReNamer, which identifies and names each photo based on the date and time it was recorded (Harris et al. 2010, Sanderson and Harris 2013). Photos were sorted according to the species detected in each image and grouped into (typically 30-minute) independent detection events. Photos which captured activity occurring solely beyond 3 meters ahead of the camera (indicated by the survey marker placed at each site) were treated as non-detections in order to standardize detection ranges across sites with variable vegetation structure.

In the case of roadside cameras, detection events were further classified to determine how wildlife interact with mitigation structures. These interaction classifications included crossings, refusals, and non-interactions, and additionally denoted additional possible predator-prey interactions (Table 3). Habitat detection events were only classified by interaction in instances where hunting or scavenging behavior was observed by mesopredator species. In these cases, photos were copied to a second file location for further analysis and comparison with similar events observed at roadside cameras. Whenever applicable, detection or interaction events were weighted based on the number of camera trap nights for which each site was fully operational and standardized to 100 camera trap nights per site.

Thesis Objectives

The underlying goal of this thesis was to determine how well the FM106 mitigation structures accomplished the stated goal of providing habitat connectivity for the surrounding wildlife community, something which cannot be confidently determined with roadside monitoring alone. Combining roadside and habitat camera trap arrays can provide comparative data to assess the performance of WCS and mitigation fencing as well as insights into the South Texas wildlife community which may inform future mitigation structures' design and placement. Chapter I has focused on the existing body of research surrounding road mitigation for wildlife and detailed the study area, methods, and objectives of this study. Chapter II focuses on the community-level analysis of the FM106 corridor, quantifying how factors such as proximity to roads and woody canopy cover may influence species composition and richness. Chapter III uses field data to model the activity of select target species to better predict their movement across the landscape and diel activity patterns as a response to anthropogenic disturbance (i.e. road-effects). Chapter IV combines the results of the previous two chapters to estimate and contextualize performance differentials for all FM106 mitigation structures, done by comparing expected with actual crossing rates for primary and secondary target species. Finally, Chapter V provides conclusions that can be drawn from this work and discusses how they may be applied to future road mitigation projects and research.

CHAPTER II

QUANTIFYING WILDLIFE COMMUNITY COMPOSITION ACROSS THE FM106 MITIGATION CORRIDOR

Introduction

Wildlife crossing structures (WCS) are often designed and placed on the landscape with a singular target species in mind, however, they may affect a much larger portion of the wildlife community. Because both roads and their mitigation structures may have unequal effects on species based on home range requirements, trophic level, body size, behavioral responses, or movement capabilities, it is important to consider these varied effects on community composition and interspecies interactions (Clevenger and Waltho 2003;2005, Jaeger et al. 2005, Fahrig and Rytwinski 2009). Understanding the road-effect zone also requires understanding how land cover characteristics, such as vegetation or proximity to resources, may influence the community at both control (non-roadside habitat) and impact (roadside) sites (van der Ree et al. 2015).

As described in Chapter I, eight WCS were installed beneath Farm-to-Market (FM)106 in Cameron County, TX between 2016 and 2019 to mitigate road mortalities for the local endangered ocelot population and to facilitate habitat connectivity for other species which might experience the road as a barrier or disturbance. This mitigation corridor extended for 16 km along FM106 and up to 1 km into the adjacent Laguna Atascosa National Wildlife Refuge

(LANWR). The study area contains fragments of thornscrub habitat interspersed between expanses of agricultural land, coastal prairie, savannah grasslands, and irregularly inundated salt flats. Despite its limited availability on the landscape, thornscrub (defined here as over 70% woody canopy cover density) is an important habitat type for the ocelot as well as several other meso-mammal species.

Objectives & Hypotheses

The primary objective of this Chapter was to identify and test factors – particularly canopy cover density and distance to the roadway – which might predict changes in the FM106 wildlife community across habitat and roadside sites. The following hypotheses were tested: 1) open, mixed, and dense canopy cover sites will have significantly distinct wildlife communities; 2) roadside, near habitat, intermediate habitat, and far habitat sites will have significantly distinct wildlife communities; 3) species richness will increase with increased canopy cover density and increased distance to the road.

Methods

Ten months of camera trap data were used to quantify the influence of woody canopy density and road proximity on the wildlife community surrounding the FM106 mitigation corridor, as described in Chapter I. Camera traps, capable of capturing a variety of medium to large-bodied species, were deployed at sites up to 1 km into the adjacent habitat and at roadside sites including WCS-adjacent and unmitigated right-of-way (ROW) sites. Camera trap sites were evenly distributed across three vegetation classes (based on % woody canopy cover), distances to FM106, and sides of the road (east/west). This study design represented variation in habitat conditions seen across mitigation sites and established a comparison of control (habitat) versus impact (roadside) sites.

Camera sites were determined with a stratified-random selection process, using canopy cover (open = 0-30%, mixed = 31-69%, dense = 70-100%) and distance to road (roadside = 0-20 m, near habitat = 21-300 m, intermediate habitat = 301-600 m, far habitat = 601-1000 m) and side of road (north/east and south/west) as stratifying factors. Potential habitat sites were designated with a 50 m fishnet grid in ArcGIS Pro (ESRI 2021), numbered, and randomly selected until all strata were evenly represented. Canopy measurements were taken prior to camera deployment to confirm the expected vegetation class for each habitat site (n=36; as described in Chapter I). Roadside sites (n=12) were determined using a stratified-random selection process from within the pool of 28 sites monitoring WCS-adjacent and unmitigated ROW locations. Seven possible roadside sites were eliminated from selection due to camera angle, flooding, or data collection gaps which made them inconsistent with cameras in the habitat array. Only one of each east/west pair of sites was permitted to be selected for inclusion to limit non-independence of samples. Otherwise, sites were randomly selected until all strata were evenly represented. All habitat and roadside sites included in this analysis consisted of a single PIR camera to ensure comparable detection probability of target species across different site types and camera configurations.

Reconyx Hyperfire 2 cameras (n=48) were deployed from July 2021 through May 2022 for a total of 334 possible camera trap nights per site. Sites were visited every 2-4 weeks to collect data, deploy blank SD cards, clear vegetation regrowth up to 3 m around each camera, and ensure continued functioning of all equipment. Any gaps in data collection (typically due to camera malfunction or inclement field conditions) were recorded and all community data was standardized based on the number of available camera trap nights per site.

All data were processed using the software ReNamer, which identifies and names each photo based on the date and time it was recorded (Harris et al. 2010, Sanderson and Harris 2013). Photos were then manually sorted by species observed in each image, and sequences of images were automatically grouped if the same species was observed at the same site in images less than 30 minutes apart from one another. This 30-minute window is the basis for the assumption of independence between detection events (Ridout and Linkie 2009). With the exception of 4 rodent (beaver; *Castor canadensis*, fox squirrel; *Sciurus niger*, Mexican ground squirrel; *Ictidomys mexicanus*, nutria; *Myocastor coypus*) and 4 bird (chachalaca; *Ortalis vetula*, greater roadrunner; *Geococcyx californianus*, northern bobwhite; *Colinus virginianus*, and wild turkey; *Meleagris gallopavo*) species, birds and rodents were generally classified as "unspecified bird" or "unspecified rodent" due to inconsistent photo quality and/or likelihood of identification for many smaller-bodied species.

Species accumulation curves (Primer v.7) were plotted (in original sample order) to determine the minimum number of trap nights necessary to capture the majority of the wildlife community represented in each vegetation class and ensure that all sites were operational for at least this minimum period. This analysis included all taxa, excluding any domestic species (i.e. domestic cat; *Felis catus*, dog; *Canis lupus familiaris*, and cattle; *Bos taurus*) observed in the study area (Table 1), and was performed using untransformed data (Clarke and Gorley 2001).

Hypotheses 1 and 2 were tested using a permutational multivariate analysis of variance (PERMANOVA) and permutational analysis of dispersion (PERMDISP) routines, based off a Bray-Curtis resemblance matrix (Anderson et al. 2008, Anderson 2014). A three-factor

PERMANOVA design (type III sums of squares, 9999 permutations) tested for the significance of canopy cover, distance to road, side of road, and any interaction effects between these factors. A bootstrapped non-metric MDS plot and the PERMDISP routine (999 permutations) were used to visualize and check the validity of any significant results. The community dataset was square root transformed prior to analysis in order to more evenly represent variation across all taxa, as observed using a 3D plot of the distributions. This and subsequent analyses in this chapter incorporated all taxa observed across the study area. The SIMPER routine was also run using square root transformed data to determine how much each species account for differences in the wildlife community across vegetation classes and roadside versus habitat groups (Clarke and Gorley 2001).

PCA was used alongside draftsman plots (Primer v.7) to visualize potentially explanatory environmental factors, identify any correlation between factors, and finalize the list of environmental variables to be incorporated into the DistLM analysis. The factors considered for this analysis included: 1) east/west location (UTM_x), 2) north/south location (UTM_y) , 3) average distance to road (Dist.rd.avg), 4) percent of woody cover between site and nearest roads (%WC.Dist.rd.avg), 5) local canopy cover density (CC.PER15), 6) surrounding canopy cover density (CC.PER300), 7) distance to nearest woody patch (DIST.WC1000SQ), 8) percent bare groundcover (GC.PER.BARE), and 9) distance to nearest water source (DIST.WATER). Distance (m) to nearest woody patch was transformed (v+100) to account for negative values (i.e. when sites were located within a patch, distance to edge was expressed using negative distance). Factors were then normalized and used to develop a Euclidean Distance resemblance matrix in order to compare with community data (Anderson et al. 2008).

The DistLM routine, referencing small sample corrected Akaike Information Criterion (AICc) scores for best model selection, was used to identify significant factors in predicting variation in wildlife community and presence-absence data across sites (Primer v.7). The community dataset was transformed into a second Bray-Curtis resemblance matrix based on presence-absence data. Community matrices were then cross-referenced with the Euclidean Distance matrix developed from environmental site data to run the DistLM routine. Results were visualized using distance-based redundancy analysis or dbRDA plots (Anderson et al. 2008).

Hypothesis 3 was tested with a two-way ANOVA using R package 'car' (Fox et al. 2012) to determine the influence of vegetation class and distance bin on species richness. Data were tested for normality of residuals using a Shapiro-Wilk test (Shapiro and Wilk 1965) and homoscedasticity using a Breusch-Pagan test (Breusch and Pagan 1979). Tukey HSD post-hoc testing was used to compare means across levels for any factors determined to be significant (Tukey 1977, Kim 2015).

Results

The combined roadside-habitat camera trap array was operational for a total of 14,555 trap nights, for an average of 303 nights per site (min. = 231, max. = 329; SE = +/-4.03), or approximately 10 months, for each of the 48 camera sites. In this study period, a total of 264,110 photo detections and 22,151 independent wildlife detection events were recorded. Detections included 4 avian, 19 mammalian (3 domestic), and 8 herpetofauna taxa, as well as a variety of unspecified bird and rodent taxa which could not be consistently identified to species (Table 1). Of these taxa, 3 avian and 15 mammalian (including 2 domestic) species were consistently recorded across the camera trap array (detected a minimum of 10 times after standardizing detections to 100 trap nights per site; Table 4; Table 5).

Species accumulation curves (Figure 6) were plotted for each vegetation class (n=16 trap nights). Within each vegetation class, at least 90% of the wildlife community was observed within 13 weeks for open sites (17 of 18 total species observed), 5 weeks for mixed sites (16 of 17 total species observed), and 3 weeks for dense sites (18 of 20 total species observed). Camera traps used in this and subsequent analyses were operational for a minimum of 231 trap nights (33 weeks) with an average of 303 trap nights per site (SE = +/-4.03), or approximately 43 weeks.

PERMANOVA results indicated that both canopy cover (p=0.0001) and distance to road (p=0.0004) were significant factors in explaining wildlife community variation; side of road and any interaction effects were not significant factors. Pair-wise testing revealed that within canopy cover, all three classes (open, mixed, dense) showed significantly distinct wildlife communities from each other. Pair-wise tests for distance bins (roadside, habitat near, habitat intermediate, and habitat far) indicated that only roadside sites had significantly different communities from the other groups ($p < 0.003$); none of the habitat distance bins were significantly distinct from each other ($p > 0.33$). PERMDISP testing using canopy cover as the group factor showed that this may have been partially due to overdispersion (p=0.001), but bootstrapped nMDS plots showed significant separation across all classes and confirmed the significance of the PERMANOVA results (Figure 7). PERMDISP testing using distance as the group factor indicated that there was no significant overdispersion within distance bins (p=0.89). Bootstrapped nMDS plots illustrated the separation between roadside and habitat groups as well as the overlap between the three habitat distance bins (Figure 8).

SIMPER results showed that in Open sites, over 70% of the similarity was driven by just four taxa: unspecified birds (detected less at open sites), white-tailed deer, nilgai (*Boselaphus tragocamelus*), and coyote (each detected more at open sites). On the other hand, 70% of the similarity within Mixed and Dense sites was driven by six taxa each (white-tailed deer, bobcat, eastern cottontail; *Sylvilagus floridanus*, Virginia opossum; *Didelphis virginianus*, nine-banded armadillo; *Dasypus novemcinctus*, unspecified bird and rodent; Table 6). Between roadside sites (<20 m from FM106) and pooled habitat sites (21-1000 m from FM106), nearly half of the cumulative dissimilarity was driven by northern raccoon (*Procyon lotor*), white-tailed deer, and potential felid prey taxa (i.e. cottontail, unspecified bird, and rodent). Raccoons, along with opossum, bobcat, striped skunk, coyote, nine-banded armadillo, and domestic cats (*Felis catus*) were detected more frequently in roadside sites. White-tailed deer, eastern cottontail, nilgai, javelina (*Pecari tajacu*), feral hog (*Sus scrofa*), roadrunner, and unspecified birds and rodents were detected more frequently in habitat sites (Table 7).

PCA and draftsman plots indicated that some correlation among environmental factors existed, but none were considered strong enough (>0.8) ifto potentially remove from the DistLM analysis. DistLM results for the overall community indicated that north-south location (UTM_y) and local canopy cover density (CC.PER15) were significant in predicting variation across the wildlife community (Figure 9). The best model (based on AICc) included these two factors (R^2 = 0.33), though the best single predictor of variation was canopy cover density (R^2 = 0.25). DistLM results for presence-absence community data indicated that north-south location (UTM_y), local canopy cover density (CC.PER15), and distance to water were significant factors in predicting variation in the community (Figure 10). The best model (based on AICc) included these factors (R^2 = 0.36).

Results of the Shaprio-Wilk test for normality of residuals (p=0.44) and Breusch-Pagan test of homoscedasticity (p=0.22) indicated that untransformed species richness data fulfilled these assumptions required for linear modeling (Poole and O'Farrell 1971). ANOVA results indicated that vegetation class was significant ($p<0.00001$), but distance bin was not. Tukey HSD post-hoc testing showed that Open sites were significantly different from Mixed ($p=0.00001$) and Dense ($p<0.00001$) sites, but Mixed and Dense sites were not significantly different from each other. Species richness increased with increased canopy cover (Figure 11), but was not affected by distance to the road (Figure 12).

Discussion

This study quantified wildlife community data and species richness on the FM106 mitigation corridor by sampling across a variety of habitat conditions and control-impact sites to better understand the effects of this roadway on the surrounding community. These results helped to enhance our understanding of the community and interspecific relationships in the FM106 study area and contextualize more specific future analyses. PERMANOVA results supported Hypothesis 1, that wildlife communities would differ across vegetation classes, but only partially supported Hypothesis 2, that wildlife communities would differ across distance bins. ANOVA results partially supported Hypothesis 3, that species richness would increase with increased canopy cover density and distance to FM106.

Vegetation structure (i.e. woody canopy cover availability) was consistently identified as a significant factor in predicting community variation, and increased cover density was associated with higher levels of richness and overall activity for many target species. This is consistent with previous studies conducted in the region on ocelot and other mesomammals, generally indicating that greater availability and density of thornscrub provides resources to support community diversity (Jahrsdoerfer and Leslie 1988, Shindle and Tewes 1998, Lombardi et al. 2020a). The influence of woody canopy cover density on the South Texas wildlife community is an important factor to consider in any future analyses of the effectiveness of wildlife structures. Failing to account for vegetation structure as a predictor of wildlife activity around WCS may lead to inaccurate conclusions about the effectiveness of mitigation strategies (Clevenger and Waltho 2005).

This analysis considered three different simple metrics for woody canopy cover availability: distance to woody canopy cover (minimum 1000 sq m patch), canopy density measured within 15 m, and canopy density estimated within 300 m. Of these three, density measured within 15 m was consistently identified as a significant factor in modeling community variation and richness. The technique for measuring this canopy density metric was consistent with that used in previous work in South Texas (Hopkins 2020, Lombardi et al. 2020b), but could also be estimated at this scale for future studies using classified NAIP or LiDAR imagery (Hopkins 2020, Lombardi et al. 2020a, Lehnen et al. 2021, Yamashita unpublished work).

Location (north-south) along the FM106 corridor was also a significant factor in community variation, which may have been due to individual species differences and habitat requirements (e.g. increased deer and nilgai activity on the southeast portion of the corridor), but might also attest to the fragmented nature of the study area as a whole. The study area consisted of nearly 30% agricultural land cover and was intersected by two rural residential developments and access roads to the Port Isabel ICE Detention Center and the Port Isabel-Cameron County airport. High degrees of fragmentation have been shown to alter wildlife community composition, movement behavior, and vagility (Fahrig 2007, Tucker et al. 2018), factors which could have contributed to the dissimilarity between communities on the north/west and south/east ends of the FM106 corridor. Lehnen et al. (2021) quantified the bulk of the FM106 study area as low suitability habitat for ocelots, largely due to extensive agricultural land use, and identified highly constrained movement potential between more suitable habitat patches on the east and north/west ends of the corridor. For species which were observed to select for similar dense woody cover (which included bobcat, armadillo, raccoon, opossum, javelina, and others), this same fragmentation may have affected their ability to fully utilize available habitat within the study area.

Distance to the road was a less consistent indicator of overall species richness and community variation, particularly when considering the entire gradient of habitat reference sites; however, it is worth noting that roadside sites (<20 m from FM106) had a significantly distinct wildlife community from habitat reference sites. This community difference was largely driven by increased mesopredator (e.g. raccoon, opossum, bobcat, coyote) presence and decreased bird, small mammal, and ungulate presence at roadside sites (Table 7). This trend was somewhat surprising based on the well-documented negative effects of roads and development pressures on carnivores at the macro scale (Clarke et al. 1998, Kohn et al. 2001, Cardillo et al. 2004, Azedo et al. 2022), but consistent with previous research on more local effects. Carnivores have been documented using roads as travel corridors and territory boundary markers, for hunting or scavenging for roadkill, or as possible areas of release from the threat of competition or predation from other species (Bradley and Fagre 1988, Barrientos and Bolonio 2009, Mata et al. 2017, Silva et al. 2019, Ruiz-Capillas et al. 2021). These potential uses of roads by mesocarnivores may therefore outweigh any perceived or actual risks, especially when more optimal habitat is fragmented or not readily available (Shamoon et al. 2018). Fahrig (2007) contended that when landscape changes outpace the capacity of wildlife to evolve movement responses, certain species may rely on maladaptive cues for decision making in altered habitat, which may ultimately result in non-optimal movement behavior.

Although this study documented fewer rodent detections at roadside versus habitat sites, this may be biased towards the behavior of only larger rodent species (e.g. *Neotoma* spp.) due to the study cameras being setup to optimize larger mammal detections. Concurrent research in this study area, which used rodent-specific baited camera trap booths, has shown that road-adjacent sites may in fact have higher abundances of several rodent species compared to the surrounding habitat (Sanjar unpublished work). Previous studies have also documented road-adjacent areas as highly productive for small mammal hunting (Adams and Geis 1983), and cited this as a driving factor in predator road mortalities (Barrientos and Bolonio 2009, Silva et al. 2019). Hunting or scavenging opportunities could explain the higher average detections of bobcats, coyotes, and other mesocarnivore species at roadside sites. In particular, raccoons and opossums are well-established urban-adapted species which may seek out competitive advantages by foraging in disturbance-prone roadside areas (Prange et al. 2004, Wang et al. 2015, Rodriguez et al. 2021).

Several carnivore species were absent from the dataset in the present study. The American badger (*Taxidea taxus*), gray fox (*Urocyon cinereoargenteus*), and puma (*Puma concolor*) were all observed on this or adjacent mitigation corridor projects during post-construction WCS monitoring; long-tailed weasel (*Mustela frenata*) and ocelot were observed during this study period on FM106 roadside cameras but not in the habitat reference sites (Table 1). While some of these species may have naturally low population densities, their effective absence from this study (and rarity in adjacent monitoring efforts) may be significant. The remaining predator-prey interactions, activity, or spatial distribution of species observed in this study may be influenced by limited intraguild competition, potential mesopredator release, or a lessened landscape of fear effect in the absence of historic apex predators (Soulé et al. 1988, Brown et al. 1999, Crooks and Soulé 1999, Terborgh et al. 2001, Grigione et al. 2009, Prugh et al. 2009, Wang et al. 2015). However, more precise inter-specific research using before-after or large-scale comparative study designs would be necessary to confirm any occurrence of such phenomena, particularly given inconsistent support and definitions of mesopredator release hypotheses in previous literature (Jachowski et al. 2020).

Previous estimates of the road-effect zone have ranged from less than a few hundred meters (Reijnen et al. 1995, Forman and Alexander 1998, Forman and Deblinger 2000, Hansen and Clevenger 2005, Benítez-López et al. 2010), to over 5 kilometers (Reijnen et al. 1995, Forman and Alexander 1998, Benítez-López et al. 2010) for various taxa. Chapter II findings seem to suggest that the most immediate impacts on the wildlife community occur within just 20 m of the road, in the area most subject to direct disturbances such as altered plant communities (due to roadside mowing and possible species introductions), various sources of pollution, and higher levels of human activity. The road-effect was shown to be positive for some species and negative for others, though patterns such as this may represent de-coupling of important trophic interactions to the overall deficit of the biotic community (Clevenger and Waltho 2005). These findings do not rule out larger scale impacts on individual species or effects which extend beyond the 1 km cross-section that was quantified here (Forman and Alexander 1998). However, given that 80% of the continental U.S. is located within a kilometer of a roadway (Riitters and Wickham 2003), and the likelihood of encountering conflicting disturbances beyond this point (e.g. other roadways, resource extraction, urbanization), larger road-effects may not even be feasible to quantify using this type of comparative study.

This study was also limited to an approximation of a control-impact design, whereas an ideal study would also have been able to assess the before-after conditions prior to the construction of FM106 (van der Grift et al. 2013). This stretch of road has been present on the landscape for over 30 years (TxDOT 2022), during which time wildlife may have habituated to incremental increases in traffic and human activity, and shifts in the biotic community composition may have occurred

CHAPTER III

MODELING TARGET SPECIES ACTIVITY ACROSS THE FM106 MITIGATION CORRIDOR

Introduction

While community analysis may be sufficient for detecting general trends across the FM106 mitigation corridor, individual analyses can identify predictors of target species activity that may affect the ultimate performance of road mitigation structures. Bobcats, previously used as a proxy for assessing road mortality risk for ocelots (Schmidt et al. 2020), may also display similar habitat selection patterns (Lombardi et al. 2020a), though some studies suggest that these species exhibit temporal partitioning where their ranges overlap in South Texas (Leonard et al. 2020). Previous research has established that canopy cover density is an important measure of felid habitat in this region (Hopkins 2020, Lombardi et al. 2020b, Lehnen et al. 2021), though the indirect effects of roads on these species are not as well-defined. In addition to or in lieu of abundance data, some recent studies have analyzed wildlife movement, activity indices, or diel activity to assess how anthropogenic disturbance may alter wildlife behavioral patterns (Barrueto et al. 2014, Andis et al. 2017, Gaynor et al. 2018a, Watabe and Saito 2021). Wildlife diel activity was quantified on the adjacent SH100 mitigation corridor project (Yamashita 2020), though this study was limited to sites placed within 150 m of the highway and human activity directly detected at camera trap sites. Analysis of diel activity patterns on FM106 may offer broader insights into the temporal extent of the road-effect zone by comparing target species activity with daily traffic patterns.

This study examined ten months of data from 48 camera trap sites across the FM106 mitigation corridor in Cameron County, TX, as described in Chapter I. Camera traps, capable of capturing a variety of medium to large-bodied species, were deployed at sites up to 1 km into the adjacent habitat and at roadside sites and were evenly distributed across three established levels of canopy cover density (open, mixed, and dense). Bobcats (in the absence of sufficient ocelot data) were treated as the primary target species, and detection data from these sites were modeled to examine the influence of woody canopy cover metrics, distance to road, and other potentially influential factors on their spatial distribution. In particular, this model also tested whether woody cover may impact any observed road-effects (e.g. by buffering sound or light disturbances from traffic activity). Temporal activity patterns were then examined for bobcats, ocelots, secondary target species (coyote, striped skunk, white-tailed deer), and a combined prey taxa group (rodents and lagomorphs) to test how daily road usage may be affecting wildlife activity and interactions on the corridor.

Objectives & Hypotheses

The objective of this chapter was to model target species detection events and diel activity to better understand predictors of their movement and behavior on the FM106 mitigation corridor. This analysis tested the following hypotheses: 4) primary target species (bobcat) detections will increase with increased distance from roads; 5) primary target species (bobcat) detections will increase with the interaction of increased distance and woody cover between site and roads, indicating that woody canopy cover may buffer the road-effect; 6) primary and secondary target species will show significantly distinct diel activity patterns at roadside sites compared to habitat sites; and 7) primary and secondary target species' diel activity will have significantly less overlap with traffic activity at roadside sites versus habitat sites, indicating these species exhibit temporal avoidance of FM106 due to daily traffic activity.

Methods

This analysis incorporated 48 camera trap sites, evenly divided between open, mixed, and dense vegetation classes and across four distance bins (roadside, near habitat, intermediate habitat, and far habitat). Each site was monitored using a single PIR camera trap, and the study used the same site selection, camera placement, and data processing methods as described in Chapters I and II.

Hypotheses 4 and 5 were tested using a negative binomial generalized linear model (NBGLM) using AIC model dredging to determine significant factors in predicting bobcat detections (Barton 2012, Team 2021). The NBGLM was chosen to account for non-normal residuals and overdispersion in the count data, which were identified using a Shapiro-Wilks test for normality of residuals ($p < 0.05$ for untransformed and transformed datasets) and a test for overdispersion (R Studio, 2021). A Moran's I score was calculated using the Spatial Statistics toolbox in ArcGIS Pro (ESRI 2021) in order to test for spatial autocorrelation between camera sites and determine whether this effect should be accounted for in the model. Data used for the NBGLM were standardized to 100 camera trap nights per site but otherwise untransformed.

Draftsman plots were developed to visualize potential multicollinearity between environmental variables and identify whether any should be eliminated from the model based on correlation estimates (Clarke and Gorley 2001). Variables considered included: distance to FM106, distance to nearest secondary road, percent woody cover between site and nearest roads, percent woody canopy cover within 15 m of site, percent woody canopy cover within 300 m of site, percent bare groundcover within 5 m of site, distance to large (>1000 sq m) patch of woody cover, and distance to nearest water source. ArcGIS Pro was used to measure and record distances from camera sites to each relevant land cover type or feature. Vegetation metrics were estimated from the classified NAIP imagery (for the 300 m buffer) or averaged from a five-point survey conducted at each site (for the 5 and 15 m buffers), as described in Chapter I.

The Moran's I test for spatial autocorrelation returned no significant results for the spatial distribution of bobcat detections, therefore location was not considered as an additional factor in this analysis. Possible multicollinearity (|>0.6| correlation) was identified between canopy cover within 15 m and two other factors: distance to woody canopy cover (-0.65) and percent bare groundcover (0.61). Both were initially retained for model selection, but percent bare groundcover was ultimately eliminated from the model due to its dependence on canopy cover density and because canopy cover is an established and more easily quantifiable metric for managing felid habitat in South Texas. The final NBGLM included three metrics of woody canopy cover, five measures of road-distance and vegetation structure interactions, and distance to nearest water source:

*species(x) = glm.nb(x ~ % CC within 15m + % CC within 300m + Distance to Nearest Woody CC + Distance to FM106 * Avg %CC Between Site and Road + Distance to Nearest Secondary Road * Avg %CC Between Site and Road + Distance to Nearest Water)*

Hypothesis 6 was tested using the "compareCkern" function in R package 'activity' (Rowcliffe 2016, Team 2021) to identify if there were significant differences between kernel density distributions of roadside versus habitat activity for target species. Hypothesis 7 was tested using the $\hat{\Delta}_4$ overlap estimator from R package 'overlap' (Meredith 2021) to determine if the activity curve generated from traffic detections overlapped less with roadside activity than with habitat activity (with n=999 bootstrapped 95% confidence intervals) for each target species.

Wildlife detection data for Hypotheses 6 and 7 were generated by converting the times of all photos taken to radians and applying a 5-minute independence rule to group consecutive photos into detection events. Ridout and Linkie (2009) suggest a 30-minute threshold to increase independence of detections but other authors (Shamoon et al. 2018) use a threshold as little as 1-minute to more accurately represent fine-scale species activity, regardless of individual independence; a 5-minute threshold was selected as a compromise between these approaches. All species had a minimum of 100 detection events per site type (i.e. roadside or habitat, vegetation class) for hypothesis testing and $\hat{\Delta}_4$ was selected as the appropriate overlap estimator for these relatively large (n>50) sample sizes, as recommended in Ridout and Linkie (2009). Because ocelots were detected infrequently within this study period, they were not included in these hypothesis tests; however, ocelot detections from the broader FM106 post-construction monitoring period (January 2020 – May 2022, n=42) were used to estimate their diel activity. Traffic activity was estimated by pooling AIR camera trap detections of vehicles from two sites on either end of the FM106 corridor, and then randomly down-sampling to 1 of every 60 detections for ease of running bootstrap estimates for Hypothesis 7. Of over 600,000 original detections, approximately 10,000 were used in the final estimate of traffic activity.

Results

Four factors were included in the most parsimonious and lowest-scoring AIC model: percent woody canopy cover within 15 m ($p=0.002$), distance to nearest patch of woody cover ($p=0.013$), distance to FM106 ($p=0.008$), and percent woody cover between site and nearest roads ($p=0.005$). Bobcat detections increased with increased canopy cover density (by factor of 1.88, SE = +/-0.61; Figure 13), decreased distance to woody cover (by factor of -0.004, SE = +/-0.002), decreased distance to FM106 (by factor of -0.003, SE = +/-0.001), and decreased woody cover between site and nearest roads (by factor of -2.78, SE = +/-1.0). Only one model had a competing score of $\Delta AIC \leq 2.0$; this model included the interaction of distance to primary road and average % canopy cover between site and road as a factor, but this factor was not considered significant and was discarded in favor of the more parsimonious first model.

Diel activity patterns were quantified for several mammal species on the FM106 mitigation corridor (Figure 14). Diel activity analyses showed significant differences between roadside versus habitat activity for bobcat ($p=0.04$; Figure 15), coyote ($p=0.01$; Figure 16), white-tailed deer ($p<0.001$; Figure 17), and the combined prey taxa group ($p<0.001$; Figure 18). In each of these cases, patterns shifted towards more nocturnal activity at roadside sites, demonstrated by significantly less roadside activity overlap with traffic activity compared to habitat activity overlap with traffic activity (using 95% bootstrapped confidence intervals). Striped skunk was the only target species tested which did not demonstrate a shift in activity between roadside and habitat sites ($p=0.35$; Figure 19). Bobcat activity patterns did not significantly differ across open, mixed, or dense vegetation classes.

Discussion

Results of this study found that while increased woody cover was associated with increased bobcat detections, as anticipated based on previous research (Hopkins 2020), bobcat activity did not decrease at roadside (<20 m) locations. Rather, increased proximity to FM106 and decreased woody cover between site and road each appeared to have a slight positive effect on bobcat detections. These results therefore did not support Hypotheses 4 and 5. However, temporal avoidance of road traffic was evident for bobcats as well as secondary target species, indicated by shifts towards more nocturnal activity at sites adjacent to the road. This distinction was statistically significant and supported Hypotheses 6 and 7 for bobcats, coyotes, potential prey species (i.e. lagomorphs and rodents), and white-tailed deer, but not for striped skunks. While the road-effects of FM106 may be subtle when considering the spatial distribution of target species on the landscape, it becomes clearer when considering their behavioral responses over time (Table 8).

Spatial Analysis

Analysis of the spatial distribution of bobcats across the FM106 mitigation corridor confirmed our understanding of thornscrub availability and dense woody canopy cover in general as important predictors of felid habitat. Both the density of and proximity to woody cover were positively correlated with bobcat detections at camera trap sites. Bobcat and ocelot have both been found in previous research to closely associate with dense thornscrub in South Texas (Laack 1991, Harveson et al. 2004, Lombardi et al. 2020a), though bobcats may be less selective than ocelots when it comes to the level of canopy density (Horne et al. 2009). Bobcats were distributed across the entire study area with little to no meaningful spatial clustering; this is distinct from the findings of a similar study on the nearby State Highway (SH)100 mitigation corridor, which noted that 83% of all bobcat detections occurred on the south side of this roadway (Hopkins 2020). This could indicate that FM106 presently poses less of a barrier effect for species movement, or that quality habitat is more evenly distributed across this study area.

Previous work by Lombardi et al. (2020a) considered distance to road as a negative factor in modeling felid abundance and activity but, in that case, distance from the nearest road was conflated with greater woody cover availability and possibly less fragmented thornscrub habitat. Research from the adjacent SH100 mitigation corridor (also in Cameron County, TX) considered road distance as well, but was limited to monitoring sites within 150 m of the roadway (Hopkins 2020, Yamashita 2020). By contrast, this study sampled equally from sites across open, mixed, and dense vegetation structure, and evenly represented these groups at four levels of distance up to 1 km from the roadway. Results from FM106 indicated that bobcats may in fact increase their use of sites with increased proximity to the road; this was consistent with the findings of Chapter II, that average relative abundance of mesocarnivores tended to be higher at roadside versus habitat sites (within versus beyond 20 m from roadway).

Bobcat detections also increased with a decrease in the precent of canopy cover between site and road, which does not support Hypothesis 5, that woody cover may be buffering any disturbances or negative road-effects. Bobcats seem to experience a slight positive effect from the road, and may prefer sites with direct (or edge condition) access to the road for its use as a hunting ground, travel corridor, or territory boundary as observed in previous research (Bradley and Fagre 1988). Concurrent research on the FM106 corridor focusing on the distribution and effects of anthropogenic noise may help to further clarify how this particular disturbance affects bobcats and other species across the study area.

Temporal Analysis

Diel activity analysis revealed that several species, including bobcats, coyotes, and white-tailed deer shifted towards significantly more nocturnal behavior at road-adjacent sites compared to habitat reference sites. These findings are consistent with previous studies documenting shifts in wildlife activity to avoid anthropogenic disturbances (Gaynor et al. 2018b, Shamoon et al. 2018, Kautz et al. 2021, Watabe and Saito 2021).

This shift towards more nocturnal activity was most evident for white-tailed deer (Δ overlap estimates = 0.28) and was observed in non-target ungulate species nilgai and feral hogs (though both of these additional species lacked sufficient detections at roadside sites). These shifts may be an indicator that ungulates tend to be more cautious of traffic activity and road-avoidant as a whole, though more specific research would be needed to investigate this pattern. Deer, nilgai, and feral hogs are also subject to seasonal hunting on LANWR property, which, combined with other human activity in the area, may influence their avoidance of anthropogenic activity in uncertain ways (Gaynor et al. 2018b). Road avoidance could have negative long-term implications for ungulate habituation to WCS usage, as it has been suggested that the likelihood of habituation will increase with increased encounters with mitigation structures (Gagnon et al. 2011).

Smaller but still significant shifts towards more nocturnal roadside activity were observed for bobcat ($\Delta = 0.07$) and coyote ($\Delta = 0.14$), as well as non-target mesocarnivores raccoon ($\Delta = 0.04$) and opossum ($\Delta = 0.04$). Despite this apparent temporal avoidance of traffic activity, each of these species was shown to have higher relative detections at roadside compared to habitat sites (Table 8). As has been documented in several studies, road-adjacent areas likely offer resource opportunities for these species such as hunting, scavenging, or established travel corridors (Bradley and Fagre 1988, Barrientos and Bolonio 2009, Mata et al. 2017, Silva et al. 2019, Ruiz-Capillas et al. 2021). It is difficult to say whether these species exhibit a similar attractance-avoidance pattern at the road surface or of vehicles themselves (Jaeger et al. 2005), and this likely varies per species (Jacobson et al. 2016). Of all the mammal road mortalities detected on FM106 for the January 2021 – January 2022 monitoring period, a significant portion was constituted by coyotes (n=6, 15.4%), raccoons (n=5, 12.8%), and opossums (n=6, 15.4%); no bobcat mortalities were detected during this same time, and no other mammal species had more than four mortalities detected (Kline et al. 2022). While these numbers are likely tied to overall abundance and density of these species within the study area, they may also indicate a lack of road surface or vehicular avoidance.

Temporal shifts for these four mesocarnivores may also mirror similar shifts in diel activity between road and habitat sites observed for small mammal prey species (i.e. large-bodied rodents and lagomorph spp; $\Delta = 0.13$, Figure 18). Shamoon et al. (2018) suggest that prey species may be affected by both human and predator spatio-temporal patterns, indicating cascading trophic effects as a result of anthropogenic disturbances.

For each of these species, temporal shifts may be an effective strategy to avoid traffic-related disturbances while still accessing local resources. FM106 is a high-speed rural highway, but experiences relatively light, daytime traffic compared to other major South Texas roadways (TxDOT 2022). Behavioral responses observed in this study may become less apparent or less effective for certain species if, for example, traffic activity expanded to affect a broader portion of the day-night cycle or increased in its intensity (Jacobson et al. 2016). These findings may also be influenced by the rarity or absence of native carnivore species (i.e. puma, jaguar; *Panthera onca*, jaguarundi; *Puma yagouaroundi*, gray fox, badger) on the landscape, as their top-down effects or competitive presence may change spatial or temporal interaction of mammals within the community (Brown et al. 1999, Crooks and Soulé 1999, Prugh et al. 2009, Kautz et al. 2021, Rodriguez et al. 2021).

While bobcats were the proxy target species due to sufficient detections for statistical analysis across roadside and habitat arrays, ocelot activity was also referenced using detections from the full post-construction monitoring period on FM106 (December 2019 – May 2022). This dataset showed 42 ocelot detection events across 5 WCS and 1 fence-end site. All of these detections occurred in dense or mixed habitat, with 31 (74%) occurring in dense habitat and 11 (26%) occurring in mixed habitat. Even among the 11 mixed habitat detections, 8 of these were at edge conditions where the individual was observed moving to or from patches of dense thornscrub. All detections occurred at or adjacent to resacas or ephemerally-flooded drainage ditches. Ocelot activity roughly matched that documented in previous studies (Laack 1991), though it should be noted that this dataset may be biased by the relatively few (n=4) individuals identified in this study, and the fact that over 70% of detection events came from just one individual male. Nonetheless, this generally aligns with previous research on South Texas felid activity which found bobcats to exhibit more diurnal and crepuscular behavior than the more nocturnal ocelot (Leonard et al. 2020). Further research would be required to say whether ocelots respond to road proximity similarly to bobcats in this study (slight positive association with some temporal avoidance of traffic) or if their more nocturnal diel pattern and preference for extremely dense cover alters any response they may have to traffic activity on FM106.

CHAPTER IV

ESTIMATING THE PERFORMANCE OF THE FM106 MITIGATION STRUCTURES

Introduction

One major conservation goal of the FM106 mitigation corridor has been to lessen the barrier effect of the roadway by facilitating habitat connectivity for the surrounding wildlife community. Meeting this goal assumes that mitigation fencing flanking each of the eight WCS sites will funnel wildlife movement from adjacent habitat towards these safe crossing points, meaning that wildlife activity observed in successful WCS should be equal or greater than activity in comparable surrounding areas for each target species. This comparison of actual versus expected crossing frequency is one way of calculating mitigation structure performance (Andis et al. 2017). Such calculations require monitoring at mitigation structures themselves (to quantify actual crossing rates) and at reference sites absent any road or road mitigation structures (to quantify expected crossing rates).

Calculating performance is complicated by variation in the design and placement of each WCS. Variation in design refers to differences in dimensions, structure, and local factors that may affect an animal's immediate decision to enter and cross at a WCS. Variation in placement refers to the WCS location on the corridor and the surrounding land cover characteristics which may affect whether, and how much, an animal approaches that location. Adjusting expected crossing rates for each WCS based on one or more predictors of wildlife activity (e.g. vegetation structure) is one way to control for the influence of placement and more accurately assess the influence of WCS design. In previous chapters of this thesis, canopy cover was shown to be an important predictor of wildlife community composition as well as target species activity. In previous work, South Texas ocelots and other target species have been shown to closely associate with dense woody cover (Shindle and Tewes 1998, Harveson et al. 2004, Horne et al. 2009, Hopkins 2020) despite its limited availability on the landscape.

WCS designs vary by the dimensions of the structures themselves and of the adjacent mitigation fence lengths. These two factors could be tested to determine if either is correlated with increased WCS performance, with possible implications for the design and construction of future road mitigation corridors. However, other site-specific variables (such as adjacent land use, flooding, or other disturbances) may also influence performance (van der Ree et al. 2015, Brunen et al. 2020), and it is important to recognize the limitations of such predictors when they are not well replicated across the study area. This analysis calls for evaluation of mitigation structures on a per species basis, due to differences in biology and behavior that may affect wildlife responses to fencing and WCS. As per the previous chapters, the primary target species was the bobcat; secondary target species included coyote, striped skunk, and white-tailed deer.

A concern sometimes raised at the idea of funneling wildlife towards concentrated movement corridors such as WCS is that this will lead to an increase in predation events, as carnivores could use these sites to ambush or easily hunt prey species (Little et al. 2002). This "prey-trap hypothesis" has been widely refuted, particularly for larger-bodied mammals (Little et al. 2002, Dickson et al. 2005, Ford and Clevenger 2010, Dupuis-Desormeaux et al. 2015), but Mata et al. (2015) and Little et al. (2002) note a lack of comparative studies determining if predation events occur at similar rates at WCS as they do in adjacent areas. The habitat monitoring aspect of this project offered a much-needed opportunity to compare predator-prey interactions at WCS and roadside sites versus similar behaviors observed across the broader landscape. Predation behavior was tracked across all sites for several mesocarnivore species, including ocelot, bobcat, coyote, domestic cat, long-tailed weasel, raccoon, and opossum.

Objectives & Hypotheses

The objective of this chapter was to evaluate the performance of the FM106 WCS and mitigation fencing, including a comparison of predation events across all study sites. The hypotheses tested were: 8) performance differentials will be positive for all WCS and species, indicating they are used more often than comparable habitat sites and provide reasonable habitat connectivity throughout the study corridor; 9) performance differentials will be negative for all fence-end sites, indicating they are used less often than comparable habitat sites and may be effectively repelling wildlife from the roadway; 10) predation events will not occur more frequently at WCS than at comparable habitat sites, indicating a lack of support for the prey-trap hypothesis.

Methods

This analysis incorporated 36 habitat reference, 15 fence-end (FE), 6 unmitigated right-of-way (ROW), and 8 WCS sites, as described in Chapter I. Camera footage at WCS sites was used to classify how wildlife interacted with the structure into one of the following basic categories: (A) full crossing, (B) entry/exit without crossing, (C) approach/departure without crossing, or (D) non-interaction. For all cameras, any observations of possible predation events were also marked as a separate interaction and noted for further analysis. Habitat reference sites were determined using a stratified random selection process (see Chapter I) in order to closely replicate the variation in vegetation structure seen at roadside mitigation sites. Habitat sites evenly represented open, mixed, and dense vegetation classes and were situated up to 1 km from FM106. Methods for detailed site selection, camera placement, and data processing are otherwise as described in Chapter I.

Habitat reference sites were treated as a control group from which an expected crossing frequency (ECF) was calculated (Andis et al. 2017) based on the average number detections of each species (standardized to 100 trap nights per site) observed per vegetation class. This ECF represented the amount of movement expected at a given mitigation site for each species, based on vegetation class (open, mixed, dense). Vegetation classes were defined by woody canopy cover density due to this factor's role as a predictor of target species presence and activity. ECF was then compared to the rate at which species utilized mitigation sites, referred to as the actual crossing frequency (ACF; Andis 2017). For WCS, the ACF represented the number of events (standardized to 100 trap nights per site) where an individual or group of individuals (both counted as 1, regardless of group size) successfully moved from one side of a crossing structure to the other (classified as an "A" interaction). For fence-end sites, the ACF simply represented the number of events where an individual or group of individuals moved across the area adjacent to a FE site.

The comparison of expected versus actual crossing events was used to calculate a performance differential (PD) for each target species (i) for each mitigation site:

$$PD_i = ACF_i - ECF_i$$

(Andis et al. 2017). This PD was a score by which WCS could be judged on how well they provided habitat connectivity for a species. A positive (+) PD indicated that a species was detected using a mitigation site more often than should be expected compared to its typical movement through the surrounding landscape. A negative (-) PD indicated that a species was detected using a mitigation site less often than should be expected. For WCS, positive PD generally indicated success, and may mean that a species is repeatedly choosing to use this site for safe passage across the road. For FE sites, positive PD may actually be an indicator of failed mitigation, in that species may be circumnavigating fencing and interacting with the roadside at certain sites more often than should be expected. These calculations were used to test Hypotheses 8 and 9.

In order to standardize PD by the relative detection rates and activity levels across species, a percent difference score (Andis et al. 2017) was also calculated for each species (*i*):

$$\%\text{diff}_{i} = PD_{i} \,/\, ECF_{i} * 100$$

This % difference score enabled comparison of mitigation structure performance across species; scores which exceed 100% indicate the magnitude to which mitigation structure use exceeded the expected rate of use (e.g. a score of 200% means the PD doubled the expected crossing rate).

The success or crossing rate, another metric quantifying mitigation structure use, was also calculated for each target species and WCS in order to compare this commonly used metric with the results of the PD calculations. Successful crossing rate was quantified by dividing full crossing (A) interactions over the total observed interactions (A+B+C+D). Wildlife interactions with mitigation structures were classified as described in Chapter II (Table 3). Performance differentials for eight species across all WCS sites were plotted over corresponding successful crossing rates to determine how well correlated these two different metrics (PD versus crossing rate) are for WCS performance and usage. A Shapiro-Wilk test for normality of residuals

(Shapiro and Wilk 1965) and a Breusch-Pagan test of homoscedasticity (Breusch and Pagan 1979) were conducted on raw and transformed datasets to determine if and which transformations may be required to meet assumptions of a linear regression model (Poole and O'Farrell 1971).

In addition to standard detection data, any interactions where possible hunting or predation was observed at any camera site were noted for further behavioral quantification. Predation behavior was defined for these purposes as any terrestrial carnivorous species seen pursuing, capturing, and/or consuming an identifiable vertebrate prey species; this definition was modified from traditional carnivore ethograms (MacNulty et al. 2007) to suit the limitations of camera trap documentation and limit the assumptions of suspected hunting behavior. In a further effort to limit assumptions about whether or not hunting behavior actually occurred prior to the detection event, suspected scavenging behavior (e.g. carnivore observed consuming or carrying part of a large carcass acquired out of camera view) was not included. Predation events were then weighted by survey effort and the total number of the respective species observed at a given site type in order to compare their rate of occurrence at mitigation sites (WCS and FE) versus unmitigated reference sites (habitat and ROW) and test Hypothesis 10.

Results

Results of WCS performance differential calculations showed generally positive scores for most WCS and target species, with the exception of white-tailed deer (which had negative PD across all sites) and WCS 3 (which had negative PD for all target species). Bobcat had positive PD for all sites (overall avg. percent difference of 1914%) except for WCS 3 (Table 9, Figure 20). Coyote had mixed results, with particularly low (negative) PD at WCS sites 2 and 3 but had an overall positive percent difference of 22% (Table 9, Figure 21). White-tailed deer had negative PD for all sites (Figure 22), and only one crossing event was recorded for white-tailed deer, at WCS 2. Striped skunk had positive PD for all sites (avg. percent difference of 1121%) except for WCS 3 (Figure 23). Non-target species nilgai, javelina, and eastern cottontail also had negative PD for the majority of WCS sites, whereas armadillo, raccoon, and opossum had overwhelmingly positive PD and percent difference scores for most sites (Table 9).

Results of FE performance differential calculations were more varied between sites and target species. Bobcats showed particularly high PD and percent difference scores at FE 2, 8, and 3, whereas coyotes had particularly high scores at FE 1S and 3 (Table 10; Figure 24 and 25). White-tailed deer recorded PD within the expected range of detections for all FE sites (i.e. percent difference scores never exceeded 50%; Table 10, Figure 26). Striped skunks had particularly high scores at FE sites 4 and 5 (Table 10; Figure 27).

Performance differentials (log10 transformed) combined across eight species and all WCS sites were significantly correlated with successful crossing rates ($p = 1.756e-10$, $R^2 = 0.48$; Figure 28).

Predation events were considered for several mesocarnivore species, but none were observed for coyote, long-tailed weasel, or ocelot during this study period. Raccoons were ultimately omitted from review due to the overwhelming number of events recorded which involved fishing, scavenging, or consuming aquatic invertebrates, none of which fit the definition of predation used for this study and were likely influenced more by the presence of water than by mitigation structures themselves. Predation events were reviewed and recorded for bobcat (n=14), domestic cat (n=3), and Virginia opossum (n=3), for a total of 20 confirmed predation events recorded during this study period. Across these three species, the highest rate of predation events (0.33%) occurred at habitat reference sites. WCS sites saw the highest raw number of predation events (n=9), but after weighting for survey effort and relevant species detections, this accounted for just 0.07% of the activity recorded at these sites. Fence-end and ROW sites had similar rates of predation events occurring (0.16% and 0.14%, respectively; Table 11).

Discussion

Performance differentials were positively correlated with increased successful crossing rates at WCS ($p = 1.756e\text{-}10$, $R^2 = 0.48$), though this trend may vary depending on which taxa were considered (Figure 28). In particular, coyotes provided a good example of how performance differentials may be a more accurate method for assessing WCS. In this case, there appeared to be significant variation in the success rates for coyotes at different WCS (ranging from <15% to >80%). This metric alone might give the impression that some of these crossing structures are more effective than others based on a higher rate of crossing success for this species. Examining the performance differentials for coyotes however showed relatively little variation across WCS (all fall within +/-100% of the ECF); a result of effectively accounting for the variation in habitat conditions surrounding each site. In this case, coyotes typically utilized WCS at the rate that should be expected based on surrounding landscape characteristics and did not appear influenced by variation in design characteristics. Without accounting for this effect of WCS placement on the landscape, as is done with PD calculations, one might draw misleading conclusions about the varying success of WCS designs (van der Ree et al. 2015, Andis et al. 2017, Denneboom et al. 2021).

Hypothesis 8, which predicted positive PD for all target species and WCS sites, was only partially supported. Positive PD were recorded for seven of the eight WCS for bobcat, coyote, and skunk, but were negative across all WCS for white-tailed deer. Positive PD (or percent difference scores exceeding 100%) for FE sites may have indicated that certain species were using these sites more frequently than expected which, unlike positive scores for WCS, may imply that a mitigation structure is failing to effectively redirect wildlife away from the roadside (or towards a WCS). These results provided mixed support for Hypothesis 9, which predicted negative performance differentials for all FE sites.

Performance differentials and percent difference scores are important for determining how well WCS may be providing habitat connectivity for the surrounding wildlife community (Clevenger and Waltho 2005, Andis et al. 2017, Denneboom et al. 2021) but can also highlight possible failures of combined WCS and mitigation fencing configurations. For example, bobcat and coyote PDs were relatively high at most WCS with the exception of one site: WCS 3. Combined with fence-end data which showed that both species have relatively high rates of movement around corresponding site FE 3, this may indicate that bobcats and coyotes were refusing or unable to utilize this crossing structure and were instead crossing into the roadside at the ends of the adjacent fence segments. A similar phenomenon seemed to occur at WCS2, where bobcats had a relatively low WCS PD paired with relatively high FE activity (coyotes also had a negative PD for WCS2, but did not show the same heightened activity at FE 2).

WCS 3 was placed at the intersection of FM106 and a drainage ditch which borders agricultural fields to the east, LANWR property to the northwest, and semi-residential development to the southwest. Throughout almost the entirety of this study period, some level of (often flowing) water was present, and this site was subject to sporadic but often intense flooding during rain events. Previous research has documented flooding as a deterrent to wildlife crossings, particularly for carnivore species (Craveiro et al. 2019, Rivera-Roy 2020), though Craveiro et al. (2019) noted the importance of riparian systems and some water presence as an attractant for carnivore use. While the design of this crossing anticipated some level of inundation and attempted to address this by constructing concrete catwalks within the structure, these catwalks have been partially or entirely buried with sediment infill since their installation. This sediment infill reduced the effective openness ratio of WCS 3 which, combined with periodic flooding, may have been the reason for this structure's poor overall performance for most species (Craveiro et al. 2019, Brunen et al. 2020, Denneboom et al. 2021). As sediment infill is likely to increase with regular flood events, continued monitoring or modification of this site may be necessary to ensure its capability to function as both a drainage culvert and a wildlife crossing structure.

WCS 2 and FE 2 were situated between two patches of dense thornscrub but were somewhat unique in that the regrading to elevate the road over the crossing structure created a significant buffer between the road, fence, and adjacent shrub-line. Whereas other mixed or dense sites had vegetation that extended to within just a few meters of each WCS entrance, WCS 2 had a 10-m concrete apron and an additional 5-m buffer with little to no substantial vegetation (i.e. mixed grasses and forbs with no canopy cover) on each side (Figure 3). This factor may have also been conflated with the fact that WCS 2 had the highest vertical clearance (2.4 m, compared to an average 1.6 m) and largest openness ratio (width x height / length = 0.26 m, compared to an average 0.14 m) of any structure on FM106. Distance to vegetation and various structural dimensions have been identified in previous studies as potentially significant factors for WCS use by different taxa (Clevenger and Waltho 2000;2005, Denneboom et al. 2021), however the magnitude and direction of these effects appears inconsistent (Clevenger and Waltho 2000). For ocelots in particular, placement of WCS in close proximity to adjacent thornscrub patches may be crucial for mitigating road mortality risk (Schmidt et al. 2020, Blackburn et al. 2021).

Wildlife use of WCS2 stood out for a number of species (relatively low PD for coyotes and bobcats, abnormally high PD for javelina, opossum, and armadillo; Table 10), but it would be inadvisable to make generalizations about any possible predictors due to the lack of replication of these conditions at other sites (van der Ree et al. 2015, Denneboom et al. 2021). A future comparative study across the growing number of WCS in South Texas could indicate whether modifications (e.g. to local vegetation or structure design) could improve performance here for felids.

In general, different types of fencing on the landscape and their varying effects on wildlife movement have been difficult to quantify (Beyer et al. 2016). Placing a pair of cameras at FE sites gave an approximation of activity occurring in these areas and highlighted where that activity exceeded expected rates of movement for target species. However, these were difficult to directly compare with crossing events at WCS sites, which had multiple camera traps monitoring a single possible crossing route. Fence-end pairs were not always situated directly across the road from one another due to staggered fence placement, and even if they were, may have missed road-surface crossings that did not start and end precisely at each camera trap. For this reason, this analysis examined broad patterns in FE detections but did not use classified behavioral interactions as was done for WCS. A longer-term dataset or experimental camera trap study designs would be necessary to understand more specifically how fencing may redirect or constrain wildlife movement (van der Ree et al. 2015).

Jaeger and Fahrig (2004) warned of the possible negative implications of extensive road mitigation fencing, as such structures may further exacerbate any barrier effects of the road for species which are unlikely to be funneled towards and use WCS. Extending mitigation fencing should therefore be done with caution, and only in areas where a) wildlife are at an increased risk of road mortalities (e.g. documented by concentrated road crossings, mortalities, or lack of road/traffic avoidance), b) road mortality is known to contribute to the decline of target species (e.g. South Texas ocelots), and c) have an alternative safe crossing point (i.e. WCS) within reasonable travel distance (Jaeger and Fahrig 2004, van der Ree et al. 2015). One location on FM106 where these conditions are met is at WCS 2. Bobcats (and several other mesomammals) have significantly higher activity at the north fence-end locations than should be expected. Especially considering the bobcat as a proxy for ocelot mortality risk (Schmidt et al. 2020), this site could benefit from extending mitigation fencing farther beyond the adjacent thornscrub tree-line, in an effort to disrupt this current pattern of circumnavigation and encourage increased use of WCS 2. Donaldson and Elliott (2021) suggested that tying fencing into other structures or natural features can effectively deter wildlife from circumnavigating fence ends and entering the roadway. One option to accomplish this on FM106 might be to angle the fence segment ends away from the road, in an attempt to redirect thornscrub-preferential species back towards denser canopy cover; another might be to extend fencing another 500 m to tie into the fencing at FE 1. A before-after study of such a modification could also be useful in investigating target species responses to new impediments to movement within their home ranges (van der Ree et al. 2015).

WCS 2 and 3 may be of particular concern because they could fail to provide the intended habitat connectivity and road mortality mitigation for ocelot individuals dispersing from occupied to unoccupied suitable habitat patches to the west and south of the FM106 corridor, as documented by Lehnen et al. (2021). No ocelots were detected farther west than WCS 3 during this monitoring project, so it is uncertain if or how individuals might respond to WCS 2 or 1.

One ocelot detection was recorded during this study just north of WCS 3 but did not appear to approach the crossing structure (OM347, January 15, 2022; USFWS). Previous monitoring detected another male ocelot (OM331, January 31, 2020; USFWS) approaching but not crossing from the north side of WCS 3, which at the time was flooded above the concrete catwalks. Addressing the flooding and sediment infill at WCS 3 may be one manageable and impactful modification that could be made to the FM106 mitigation structures at this time. Within the study period of this thesis analysis, no ocelots were observed making a full crossing of any WCS, but neither were any ocelots observed at habitat reference sites. In theory, this gives ocelots a neutral (0) performance differential for all sites, but this assessment would obviously benefit from longer term monitoring of individual behavior and movement patterns.

The comparison of predation behavior at mitigation (FE and WCS) versus control (habitat and ROW) sites supported the large body of evidence refuting the existence of any prey-trap occurring at road mitigation structures (Little et al. 2002, Dickson et al. 2005, Ford and Clevenger 2010, Dupuis-Desormeaux et al. 2015). However, predation behavior was observed in just three species for a total of 20 predation events, so conclusions from this small sample size should be interpreted with caution. WCS may be especially important for maintaining populations of small mammal species for which the structure of roadways themselves present a barrier to movement (McGregor et al. 2008), but the felid-focused camera configuration used in this array likely did not capture the breadth of the small mammal community. Concurrent research in this study area has focused on quantifying the rodent community as specific prey sources for the felids in this study and may help to contextualize the results observed here (Sanjar, unpublished).

CHAPTER V

CONCLUSIONS

This study quantified how the FM106 road and road mitigation structures impacted the composition of the surrounding wildlife community, spatial and temporal responses of target species, and the overall performance of wildlife crossing structures and mitigation fencing. Responses to road-effects and mitigation structures were both species-dependent and variable across the wildlife community. Broad patterns across taxa could be extrapolated from this study (e.g. lower WCS performance and greater spatio-temporal road avoidance for ungulates; higher WCS performance and lesser spatio-temporal road avoidance for mesocarnivores), but should be done so carefully. Trends seen here may differ depending on the size and intensity of local road networks, distribution of habitat resources, degrees of habitat fragmentation, presence of competing or apex predator species, or other factors relevant to a different region or study area (Clevenger and Waltho 2003;2005, Jaeger et al. 2005, Barrueto et al. 2014, van der Ree et al. 2015, Denneboom et al. 2021). For most mesocarnivore species studied, WCS appeared to provide effective connection points for wildlife to access resources across both sides of the FM106 roadway while avoiding the risk of vehicle collisions.

Many members of the South Texas wildlife community could not be fully incorporated into statistical analyses, including the ocelot, badger, puma, long-tailed weasel, Texas tortoise (*Gopherus berlandieri*), and several species of snake (e.g. western diamondback; *Crotalus atrox*, Texas indigo; *Drymarchon melanurus erebennus*, Mexican racer; *Coluber constrictor Oaxaca*, and Ruthven's whipsnake; *Coluber schotti ruthveni*). For larger mammalian carnivore species, this was likely due to their low population densities; for the weasel and herpetofauna, this was more likely due to poor camera detection rates considering their movement behavior, smaller size, and temperature regulation (Marcus Rowcliffe et al. 2011, Meek et al. 2012). Different or more specialized methods would be needed to understand how smaller-bodied mammals or ectothermic species respond to the road and mitigation structures on the FM106 corridor.

Canopy cover density was an important predictor of target species activity and community richness across the FM106 corridor, which is consistent with previous work in the region (Jahrsdoerfer and Leslie 1988, Harveson et al. 2004, Hopkins 2020). Of the five species in the adjacent SH100 mitigation study which were detected most often in dense woody cover sites (bobcat, eastern cottontail, javelina, armadillo, Virginia opossum), all five were observed to have similar preferences on the FM106 mitigation corridor (Table 12). Such preferences were less consistent for species (coyote, nilgai, striped skunk, white-tailed deer) which were detected more often in open or mixed habitat (Hopkins 2020). This could indicate that these five species may be more closely associated with thornscrub than other species are with open and mixed habitat or may indicate differences in habitat characteristics between SH100 and FM106 which have not yet been fully quantified.

Bobcats showed consistent detections (>60%) in a single vegetation class (dense woody cover) across both the FM106 and SH100 mitigation corridors (Hopkins 2020); this relatively predictable movement behavior, combined with prior understanding of the ocelot as a species which selects for extremely dense woody cover in this region (Laack 1991, Harveson et al. 2004, Horne et al. 2009) helps to further rationalize the placement of WCS in dense thornscrub corridors, particularly in areas where felid conservation and habitat connectivity is the priority.

WCS performance scores were exceptionally low and road avoidance was especially high for ungulates on the FM106 corridor, with the exception of javelina. It is possible that certain species require a longer habituation period, as observed in previous work (Clevenger and Waltho 2003, Gagnon et al. 2011) or that the felid-focused design of these structures is insufficient for larger-bodied mammals. Male white-tailed deer are reported to range from approximately 0.5 – 1.0 meters tall at the shoulder, though populations found in South Texas are likely on the smaller end of this range considering the area's relatively low altitude and latitude (Smith 1991). In comparison, WCS heights range from 1.1 to 2.4 m, with an average height of 1.6 m (Table 2). It is possible that the majority of deer in this study area could have reasonably fit through the available crossing structures, but that limited clearance, restricted visibility, and perceived risk of entering an enclosed underpass may have prevented their use of WCS. One review of road mitigation studies indicated that ungulate crossing success was broadly correlated with viaducts (i.e. open-span road bridges) rather than underpasses, more open WCS dimensions (i.e. increased width, decreased length), and greater proximity to local vegetation (Denneboom et al. 2021). On the other hand, javelina, which showed greater preference for dense canopy cover, tended to have positive performance scores at all sites where they were present in the adjacent habitat. These results may provide support for the general hypothesis that wildlife prefer passages which mirror their natural habitat conditions (Denneboom et al. 2021), though longer term study would be needed to determine if these structures will ever be effective as ungulate WCS, and which structural characteristics may influence success over time (van der Ree et al. 2015).

It is important to consider possible bias towards repeat individual behavior when relying on camera trap data, particularly when evaluating performance scores for WCS or FE sites. Concurrent research identifying individual bobcat behavior and WCS use may help to clarify how well traditional WCS usage metrics are correlated with the number of discrete individuals using WCS sites (Hanley, unpublished work). However, it is up for debate whether mitigation structures should be considered more or less effective if they serve, for instance, 1 individual 100 times versus 100 individuals 1 time apiece. In either case, the direct risks of wildlife-vehicle collisions are avoided in equal measure (van der Ree et al. 2015). For ocelots in South Texas, these direct mortalities can be extremely detrimental, as even the loss of one individual represents a significant loss of genetic diversity for the entire population (Janečka et al. 2011).

One takeaway from this and similar studies is the importance of using some method of control data to contextualize and assess mitigation structure performance. This study relied on habitat monitoring to calculate WCS performance differentials (Clevenger and Waltho 2000, Andis et al. 2017, Hopkins 2020), while others may use some version of success or crossing rate (%) to weight WCS crossing events by the total number of WCS approaches or expected detections (Denneboom et al. 2021). Results from Chapter IV suggest these two metrics were largely in agreement for several species studied on the FM106 corridor (Figure 28). However, in a recent review of wildlife crossing structure research, a mere 30% of the 270 articles considered used either of these metrics; the rest reportedly used unweighted crossing counts (Denneboom et al. 2021). This review highlights the lack of consistency in road ecology monitoring efforts and may help explain the consequent inconsistency in reportedly significant factors in WCS design or effectiveness across studies (Denneboom et al. 2021).

Of the eight WCS studied on FM106, three of them (WCS 3, 6, and 7) were placed in below-grade drainage ditches which regularly faced the risk of flooding during rain events. Chapter IV results showed that WCS 3 had negative PD for all but one mesocarnivore species, whereas WCS 6 and 7 had overwhelmingly positive PD for mesocarnivore species. These results indicated that while dual-purpose drainage ditch and crossing structures certainly have the capacity to effectively connect wildlife habitat, this may be contingent on the duration and severity of water or sediment inundation at such sites. The negative impact of water presence and levels on WCS use has been well documented in previous work (Craveiro et al. 2019, Brunen et al. 2020, Rivera-Roy 2020). Alternative strategies for increasing drainage capacity or providing routes for dry passage should be considered in instances where – as in the case of WCS 3 – site inundation has become the standard rather than the exception.

Results from Chapter II indicate that habitat monitoring for as little as 1-3 months could be sufficient to quantify the majority of the wildlife community and could provide sufficient control data for comparison and evaluation of WCS performance. A potential next step in this area of research will be to standardize and compare performance scores from multiple road mitigation corridors in order to document performance across a variety of replicated WCS design features. This could include up to 15 WCS between the monitoring projects on FM106, SR48, and SH100, and more could be incorporated if a comparable control array study was implemented on FM1847 where five underpass structures were recently completed. Such research could help to identify consistent predictors of WCS performance across target species and inform future mitigation strategies for habitat connectivity throughout South Texas.

Table 1. List of South Texas wildlife species observed during post-construction camera trap monitoring of the Farm to Market Road (FM)106 or State Highway (SH)100 road mitigation corridors in Cameron County, Texas from 2019-2022. Species were marked if observed between July 2021 – May 2022 on the roadside (0-20 m from the road), habitat (21-1000 m from the road), or both FM106 camera trap arrays.

List of Documented South Texas Wildlife Species				
Latin binomial	**Common Name**	**Seen on project (SH100 and/or FM106)**	**FM106 roadside array (July 2021 - May 2022)**	**FM106 habitat array (July 2021 - May 2022)**
MAMMAL SPECIES:				
Taxidea taxus	American badger	X		
Axis axis	axis deer	X		
Castor canadensis	beaver	X		
Lepus californicus	black-tailed jackrabbit	X	X	X
Lynx rufus	bobcat	X	X	X
Canis latrans	coyote	X	X	X
Sylvilagus floridanus	eastern cottontail	X	X	X
Sus scrofa	feral hog	X	X	X
Sciurus niger	fox squirrel	X	X	
Urocyon cinereoargenteus	gray fox	X		
Pecari tajacu	javelina	X	X	X
Mustela frenata	long-tailed weasel	X	X	
Ictidomys mexicanus	Mexican ground squirrel	X	X	X
Puma concolor	mountain lion	X		
Boselaphus tragocamelus	nilgai	X	X	X
Dasypus novemcinctus	nine-banded armadillo	X	X	X
Procyon lotor	northern raccoon	X	X	X
Myocastor coypus	nutria	X		
Leopardus pardalis	ocelot	X	X	
Mephitis mephitis	striped skunk	X	X	X
Didelphis virginiana	Virginia opossum	X	X	X

Table 1, cont.

Odocoileus virginianus	white-tailed deer	X	X	X
HERPETOFAUNA SPECIES:				
Alligator misissippiensis	American alligator	X		
Trachemys scripta elegans	red-eared slider	X		
Apalone spinifera	spiny softshell turtle	X		
Gopherus berlandieri	Texas tortoise	X	X	X
Coniophanes imperialis	black-striped snake	X		
Micrurus fulvius	coral snake	X		
Nerodia rhombifer	diamondback watersnake	X		
Pantherophis emoryi	great plains ratsnake	X		
Coluber constrictor oaxaca	Mexican racer	X		X
Salvadora hexalepis	patch-nose snake	X		
Thamnophis proximus	ribbonsnake	X		X
Coluber schotti ruthveni	Ruthven's whipsnake	X		X
Drymarchon melanurus erebennus	Texas indigo snake	X		
Coluber flagellum testaceus	western coachwhip	X		
Crotalus atrox	western diamondback	X	X	
Sceloporus variabilis	rose-bellied lizard	X		
Phrynosoma cornutum	Texas horned lizard	X		X
Sceloporus olivaceus	Texas spiny lizard	X		X
Cnemidophorus spp.	whiptail lizard (unspec.)	X		X
Incilius nebulifer	gulf coast toad	X		
Lithobates berlandieri	Rio Grande leopard frog	X		
BIRD SPECIES:				
Ortalis vetula	chachalaca	X	X	X
Geococcyx californianus	greater roadrunner	X	X	X
Colinus virginianus	northern bobwhite	X	X	X
Meleagris gallopavo	wild turkey	X	X	X

note: With the exception of the species listed above, birds and rodents were generally classified as "unspecified bird" or "unspecified rodent" due to inconsistent photo quality and/or likelihood of identification for many species.

Domestic species (cat, dog, cattle, horse) were also observed during CT monitoring but omitted from this list.

Table 2. Individual and summary measurements of each wildlife crossing structure (WCS) monitored along the 16 km FM106 mitigation corridor in Cameron County, Texas. Openness ratio is calculated by multiplying the height and width and dividing by the length of each WCS.

FM106 Mitigation Structure Dimensions (in meters)					
	Measurements (m)			**Openness ratio (m) (Width x Height / Length)**	**Total adjacent fence length (m)**
	Length	Width	Height		
WCS 1	16.8	1.5	1.5	0.14	1115
WCS 2	16.5	1.8	2.4	0.26	736
WCS 3	29.1	2.4	1.1	0.09	722
WCS 4	18.4	1.5	1.4	0.12	118
WCS 5	18.4	1.7	1.4	0.13	157
WCS 6	31.1	2.1	1.7	0.12	107
WCS 7	31.4	2.3	1.4	0.10	270
WCS 8	16.5	1.5	1.5	0.13	387
Average:	**22.3**	**1.8**	**1.6**	**0.14**	**415**
Max:	**31.4**	**2.4**	**2.4**	**0.26**	**1115**
Min:	**16.5**	**1.5**	**1.1**	**0.09**	**107**

Table 3. Wildlife detection event classification table for interactions occurring at WCS on the FM106 mitigation corridor in Cameron County, Texas. Detections are grouped into a single interaction event if photos occur within 30 minutes of the next/previous photo of the same species, with the intent to ensure independence between detection events.

Detection Event Classification for WCS Interactions

Class	Note	Behavior	Description
A	E-W	**Full crossing** (east to west)	Individual(s) confirmed to have crossed, with photo documentation on each side of WCS
	W-E	**Full crossing** (west to east)	Individual(s) confirmed to have crossed, with photo documentation on each side of WCS
B	N/A	Entry and exit; **refusal**	Individual(s) observed entering the WCS before exiting on the same side without crossing
	U	Entry or exit; **refusal**	Individuals(s) observed entering or exiting the same side of the WCS without crossing; some portion of the interaction (entry or exit) is uncertain due to lack of photo evidence
C	N/A	Approach and depart; **refusal**	Individual(s) approach or assess the WCS but do not enter, depart on the same side without entering/crossing
	U	Approach or depart; **refusal**	Individual(s) approach or assess the WCS but do not enter; some portion of the interaction (approach or departure) is uncertain due to lack of photo evidence
D	N/A	**Non-interactio**n	Individual(s) observed nearby but do not approach or assess the WCS

Table 3, cont.

E	N/A	**Other interaction**	Individual(s) observed spending significant amount of time exhibiting unintended behavior inside of WCS (e.g. day bedding, play behavior)
	S**	Hunting/seeking out resource	Individual(s) observed spending significant amount of time seeking out and/or consuming food or water at or around WCS site. ***This interaction must be paired with a typical A-E interaction and sorted separately for further analysis of predation events.*

Table 4. Species detections (standardized to 100 CT nights per site, for species with minimum 10 detections) and percent of detections observed in each vegetation class (open, mixed, and dense), recorded between July 2021 and May 2022 on the FM106 mitigation corridor in Cameron County, TX. Open = 0-30%, Mixed = 31-69%, and Dense = 70-100% canopy cover.

Species Detections per Vegetation Class						
	Detections (standardized to 100 CT nights per site) for each vegetation class			**% of detections per species in each vegetation class**		
Species:	OPEN	MIXED	DENSE	**OPEN**	**MIXED**	**DENSE**
black-tailed jackrabbit	10.2	0.6	0.0	**94%**	**6%**	**0%**
bobcat	39.3	86.8	204.2	**12%**	**26%**	**62%**
chachalaca	0.0	0.0	35.4	**0%**	**0%**	**100%**
coyote	63.7	69.5	31.5	**39%**	**42%**	**19%**
domestic cat	0.0	6.6	40.4	**0%**	**14%**	**86%**
domestic cow	41.8	0.0	0.0	**100%**	**0%**	**0%**
eastern cottontail	8.0	553.5	596.9	**1%**	**48%**	**52%**
feral hog	11.6	19.1	66.0	**12%**	**20%**	**68%**
greater roadrunner	2.5	38.9	37.2	**3%**	**50%**	**47%**
javelina	7.9	17.2	26.6	**15%**	**33%**	**52%**
Mexican ground squirrel	0.0	2.5	11.1	**0%**	**18%**	**82%**
nilgai	89.6	56.3	8.9	**58%**	**36%**	**6%**
nine-banded armadillo	5.5	84.9	214.7	**2%**	**28%**	**70%**
northern bobwhite	3.8	3.9	3.9	**33%**	**34%**	**34%**
northern raccoon	54.3	61.8	379.8	**11%**	**12%**	**77%**
ocelot*	0.0	0.0	0.3	**0%**	**0%**	**100%**
striped skunk	21.6	82.2	23.3	**17%**	**65%**	**18%**
Virginia opossum	20.4	174.5	577.2	**3%**	**23%**	**75%**
white-tailed deer	457.3	319.6	176.8	**48%**	**34%**	**19%**

** ocelot detections were included for reference due to its importance as a target species; however, only one detection was recorded within this study period and array.*

Table 5. Species detections (standardized to 100 CT nights per site, for species with minimum 10 detections) and percent of detections observed in each distance bin (DB 0, 1, 2, 3), recorded between July 2021 and May 2022 on the FM106 mitigation corridor in Cameron County, TX. DB0 = 0-20 m, DB1 = 21-320 m, DB2 = 321-620 m, and DB3 = 621-1000 m from the road.

Species Detections per Distance Bin								
	Detections (standardized to 100 CT nights per site) for each distance bin				**% of detections per species in each distance bin**			
Species:	roadside		habitat		roadside		habitat	
	DB0	DB1	DB2	DB3	**DB0**	**DB1**	**DB2**	**DB3**
black-tailed jackrabbit	0.6	0.0	10.2	0.0	**6%**	**0%**	**94%**	**0%**
bobcat	145.2	69.8	67.1	48.2	**44%**	**21%**	**20%**	**15%**
chachalaca	9.4	4.6	18.2	3.2	**27%**	**13%**	**51%**	**9%**
coyote	70.9	28.5	25.8	39.5	**43%**	**17%**	**16%**	**24%**
domestic cat	10.3	34.2	0.0	2.6	**22%**	**73%**	**0%**	**5%**
domestic cow	0.0	0.0	8.3	33.5	**0%**	**0%**	**20%**	**80%**
eastern cottontail	113.4	367.8	406.4	270.8	**10%**	**32%**	**35%**	**23%**
feral hog	14.0	16.2	28.7	37.6	**15%**	**17%**	**30%**	**39%**
greater roadrunner	6.8	21.3	40.9	9.6	**9%**	**27%**	**52%**	**12%**
javelina	16.5	6.8	19.7	8.7	**32%**	**13%**	**38%**	**17%**
Mexican ground squirrel	3.7	8.3	0.0	1.6	**27%**	**61%**	**0%**	**12%**
nilgai	5.1	38.4	49.2	62.1	**3%**	**25%**	**32%**	**40%**
nine-banded armadillo	134.3	39.0	62.0	69.8	**44%**	**13%**	**20%**	**23%**
northern bobwhite	1.6	3.9	3.3	2.8	**14%**	**34%**	**29%**	**24%**
northern raccoon	367.9	18.5	33.5	75.9	**74%**	**4%**	**7%**	**15%**
ocelot*	0.3	0.0	0.0	0.0	**100%**	**0%**	**0%**	**0%**
striped skunk	60.1	46.9	7.3	12.8	**47%**	**37%**	**6%**	**10%**
Virginia opossum	285.7	272.8	119.9	93.6	**37%**	**35%**	**16%**	**12%**
white-tailed deer	170.3	213.1	158.8	411.4	**18%**	**22%**	**17%**	**43%**

** ocelot detections were included for reference due to its importance as a target species; however, only one detection was recorded within this study period and array.*

Table 6. Results of SIMPER analysis comparing open, mixed, and dense vegetation classes on the FM106 mitigation corridor in Cameron County, TX from July 2021 – May 2022. The contributing percentage indicates how much each species drove the similarity within these groups, with a cutoff of 70% cumulative similarity. This analysis used square root transformed data, therefore average abundances should only be interpreted relative to each other.

SIMPER Analysis of Vegetation Class			
Similarity within each canopy cover class (open, mixed, dense), 70% cut-off			
OPEN SITES	**Avg. Abundance**	**Contributing %**	**Cumulative %**
bird	3.08	29.14	29.14
white-tailed deer	4.45	26.98	56.12
nilgai	1.81	12.86	68.98
coyote	1.76	11.55	80.53
MIXED SITES	**Avg. Abundance**	**Contributing %**	**Cumulative %**
eastern cottontail	5.4	19.59	19.59
bird	4.18	15.55	35.14
white-tailed deer	4.02	14.5	49.63
rodent	3.54	12.48	62.11
bobcat	2.08	6.87	68.98
Virginia opossum	2.68	5.94	74.93
DENSE SITES	**Avg. Abundance**	**Contributing %**	**Cumulative %**
bird	5.98	18.13	18.13
eastern cottontail	5.36	16.52	34.65
rodent	5.78	16.15	50.8
Virginia opossum	4.99	9.69	60.48
nine-banded armadillo	3.38	8.16	68.65
white-tailed deer	2.96	7.47	76.12

Table 7. Results of SIMPER analysis comparing distance groups "roadside" (camera sites 0-20 m from road) vs. "habitat" (camera sites 21-1000 m from road) on the FM106 mitigation corridor in Cameron County, TX from July 2021 – May 2022. The contributing percentage indicates how much each species drove the dissimilarity between these groups, with a cutoff of 90% cumulative dissimilarity. This analysis used square root transformed data, therefore average abundances should only be interpreted relative to each other.

SIMPER Analysis of Roadside vs Habitat

Dissimilarity between roadside and pooled habitat sites, 90% cut-off **Taxa:**	**Avg. Abundance ROAD**	**Avg. Abundance HABITAT**	**Contributing %**	**Cumulative %**
northern raccoon	4.5	1.25	10.94	10.94
white-tailed deer	3.06	4.06	9.52	20.46
bird	2.43	5.08	9.36	29.83
rodent	1.21	4.19	9.15	38.97
eastern cottontail	2.2	4.23	8.34	47.31
Virginia opossum	3.53	2.57	7.59	54.9
bobcat	2.97	1.79	7.12	62.02
nilgai	0.34	1.55	5.32	67.34
striped skunk	1.68	0.9	4.87	72.21
coyote	2.28	1.36	4.51	76.72
nine-banded armadillo	2.55	1.58	3.97	80.69
javelina	0.54	0.56	3.1	83.79
feral hog	0.9	1.0	2.98	86.77
greater roadrunner	0.49	0.86	2.4	89.17
domestic cat	0.46	0.21	1.72	90.89

Table 8. Summary of detection (det.) data and diel activity analysis comparing activity patterns at roadside vs. habitat sites on the FM106 mitigation corridor in Cameron County, TX from July 2021 – May 2022. Higher % of average detections in habitat versus roadside sites indicate spatial road avoidance. Higher Δ overlap in habitat versus roadside activity indicate a higher degree of traffic avoidance (* indicates significance). Road crossing and mortality data from year two of FM106 post-construction monitoring (Kline et al, 2022) was also referenced to estimate possible road surface or vehicular avoidance, indicated by lower % of detected crossings or mortalities.

Spatial, Temporal, and Road Surface Avoidance per Species						
	Spatial road avoidance / attractance:			**Traffic / temporal avoidance:**	**Possible road surface / vehicular avoidance:**	
Taxa	total det. events	% avg. det. at Roadside (DB0) sites	% avg. det. at Habitat (DB123) sites	Δ bootstrapped avg. overlap estimates (habitat - road) * = significance	detected road crossings as % of total roadside detections	road mortalities as % of total detected road crossings
Mesomammal species						
bobcat	1082	71%	29%	**0.06 ***	4.2%	0.0%
coyote	543	73%	27%	**0.13 ***	14.7%	13.3%
nine-banded armadillo	996	71%	29%	-0.01	6.5%	11.3%
northern raccoon	1796	91%	9%	**0.05 ***	1.1%	27.4%
striped skunk	438	76%	24%	0.03	3.6%	31.3%
Virginia opossum	2543	67%	33%	**0.04 ***	1.2%	39.1%
Ungulate species						
white-tailed deer	3393	39%	61%	**0.28 ***	5.2%	2.7%
feral hog	321	38%	62%	**0.20 ***	5.8%	52.1%
javelina	177	61%	39%	0.02	0.0%	0.0%
nilgai	494	9%	91%	**0.31 ***	35.0%	14.9%

Table 8, cont.						
Small mammal species						
potential felid prey spp. (pooled rodent and lagomorph)	8683	20%	80%	0.12	N/A	N/A

Table 9. Summary of performance differentials (PD) and percent difference scores (%diff = PD / ECF*100) for each WCS and each species with at least 10 detections per 100 CT nights per site on the FM106 mitigation corridor in Cameron County, TX from July 2021 – May 2022. Percent difference scores below -50% are flagged, as these indicate the degree to which WCS crossings fell below expected crossing rates, and may represent WCS failures for these species.

WCS Performance Scores										
performance metric (PD and % diff) per species		**WCS1**	**WCS2**	**WCS8**	**WCS3**	**WCS4**	**WCS5**	**WCS6**	**WCS7**	**avg. % diff**
			OPEN			**MIXED**			**DENSE**	
bobcat	PD	30	3	40	-6	31	61	49	27	1914%
	%diff	5259%	495%	7208%	-100%	539%	1082%	534%	290%	
coyote	PD	8	-3	5	-3	-1	-1	1	0	22%
	%diff	238%	-83%	143%	-100%	-44%	-21%	51%	-6%	
eastern cottontail	PD	0	0	0	-38	-38	-37	-48	-48	-89%
	%diff	-16%	-100%	-100%	-100%	-98%	-98%	-100%	-100%	
javelina	PD	5	10	0	-1	-1	-1	-2	-2	2597%
	%diff	7093%	14285%	-100%	-100%	-100%	-100%	-100%	-100%	
nilgai	PD	-7	-7	-7	-4	-4	-4	-1	-1	-100%
	%diff	-100%	-100%	-100%	-100%	-100%	-100%	-100%	-100%	
nine-banded armadillo	PD	1	89	0	-3	93	10	31	-6	3000%
	%diff	179%	20495%	-100%	-100%	2959%	326%	293%	-54%	
northern raccoon	PD	18	27	87	29	125	128	153	189	4952%
	%diff	2404%	3576%	11614%	1864%	7923%	8135%	1832%	2270%	
striped skunk	PD	2	3	50	-4	40	52	3	1	1121%
	%diff	275%	309%	5590%	-100%	1011%	1312%	427%	143%	
Virginia opossum	PD	14	68	39	-8	109	12	271	111	1681%
	%diff	1283%	6110%	3480%	-100%	1293%	146%	875%	359%	
white-tailed deer	PD	-33	-33	-33	-21	-21	-21	-12	-12	-100%
	%diff	-100%	-99%	-100%	-100%	-100%	-100%	-100%	-100%	

Table 10. Summary of performance differentials (PD = ACF – ECF) and percent difference scores (%diff =PD/ECF*100) for each fence-end (FE) site and each species with at least 10 detections per 100 CT nights per site on the FM106 mitigation corridor in Cameron County, TX. Percent difference scores exceeding 50% are flagged, as these indicate the degree to which detections at FE sites exceed expected detection rates and may represent ineffective mitigation.

Fencing Performance Scores

performance metric (PD and % diff) per species		FE1S	FE2	FE8	FE3	FE7	FE4/5	FE6	avg. % diff
		OPEN			MIXED		DENSE		
bobcat	PD	0	6	1	3	-2	-4	-8	186%
	%diff	35%	1105%	265%	52%	-29%	-45%	-82%	
coyote	PD	14	-2	-2	3	1	-1	-1	44%
	%diff	411%	-57%	-65%	126%	52%	-76%	-81%	
eastern cottontail	PD	0	7	4	-37	12	-35	-15	425%
	%diff	-100%	2029%	1216%	-96%	31%	-72%	-31%	
javelina	PD	0	0	0	-1	-1	-2	-2	-38%
	%diff	-100%	335%	-100%	-100%	-100%	-100%	-100%	
nilgai	PD	3	-5	-7	-4	-4	-1	-1	-73%
	%diff	41%	-71%	-99%	-89%	-91%	-100%	-100%	
nine-banded armadillo	PD	1	10	0	4	24	19	-4	472%
	%diff	144%	2199%	-83%	129%	767%	178%	-34%	
northern raccoon	PD	0	3	2	12	3	4	-5	223%
	%diff	21%	389%	223%	735%	206%	44%	-60%	
striped skunk	PD	-1	-1	1	-3	-3	16	0	299%
	%diff	-83%	-91%	156%	-82%	-64%	2198%	62%	
Virginia opossum	PD	-1	-1	5	5	15	-6	-26	52%
	%diff	-100%	-80%	415%	54%	174%	-19%	-85%	
white-tailed deer	PD	-16	-27	-13	-14	3	-10	-5	-51%
	%diff	-49%	-82%	-38%	-69%	12%	-87%	-44%	

Table 11. Summary of all confirmed predation events recorded across all CT site types on the FM106 mitigation corridor from July 1, 2021 through May 31, 2022. Predation events were defined by any predator observations where photo documentation confirmed the capture and/or consumption of an identifiable vertebrate prey species at a CT site. Predation events were weighted by the number of total detection events and survey effort (CT nights) recorded for that site type in order to compare mitigation sites (fence-end=FE, wildlife crossing structure=WCS) versus control sites (habitat, unmitigated right-of-way=ROW).

Predation Event Comparison								
	# of hunting behavior observations:			**predation events per total observations (standardized by total CT nights per site type)**			**total pred. events across species:**	**% of pred. events per total detections across species**
site type	bobcat	domestic cat	Virginia opossum	bobcat	domestic cat	Virginia opossum		
habitat	4	1	2	0.78%	1.06%	0.15%	**7**	**0.33%**
ROW	0	1	0	0.00%	3.56%	0.00%	**3**	**0.14%**
FE	3	0	0	0.52%	0.00%	0.00%	**1**	**0.16%**
WCS	7	1	1	0.16%	0.30%	0.01%	**9**	**0.07%**

Table 12. Percent of species detections observed in each vegetation class (open, mixed, dense) on the SH100 (Hopkins 2020) and FM106 (July 2021 – May 2022) habitat camera trap arrays in Cameron County, TX.

FM106 vs SH100 Species Distribution

Species:	**SH100 HABITAT ARRAY**			**FM106 HABITAT ARRAY**			**Potential veg. class preference across habitat arrays**
	% OPEN	% MIXED	% DENSE	% OPEN	% MIXED	% DENSE	
bobcat	2%	30%	68%	12%	26%	62%	**dense**
coyote	63%	8%	29%	39%	42%	19%	*open, mixed?*
eastern cottontail	1%	41%	58%	1%	48%	52%	**dense**
javelina	0%	11%	89%	15%	33%	51%	**dense**
nilgai	35%	45%	21%	58%	36%	6%	*mixed, open?*
nine-banded armadillo	0%	46%	53%	2%	28%	70%	**dense**
striped skunk	63%	10%	27%	17%	65%	18%	*open, mixed?*
Virginia opossum	0%	8%	92%	3%	23%	75%	**dense**
white-tailed deer	13%	43%	43%	48%	34%	19%	*mixed/dense, open?*

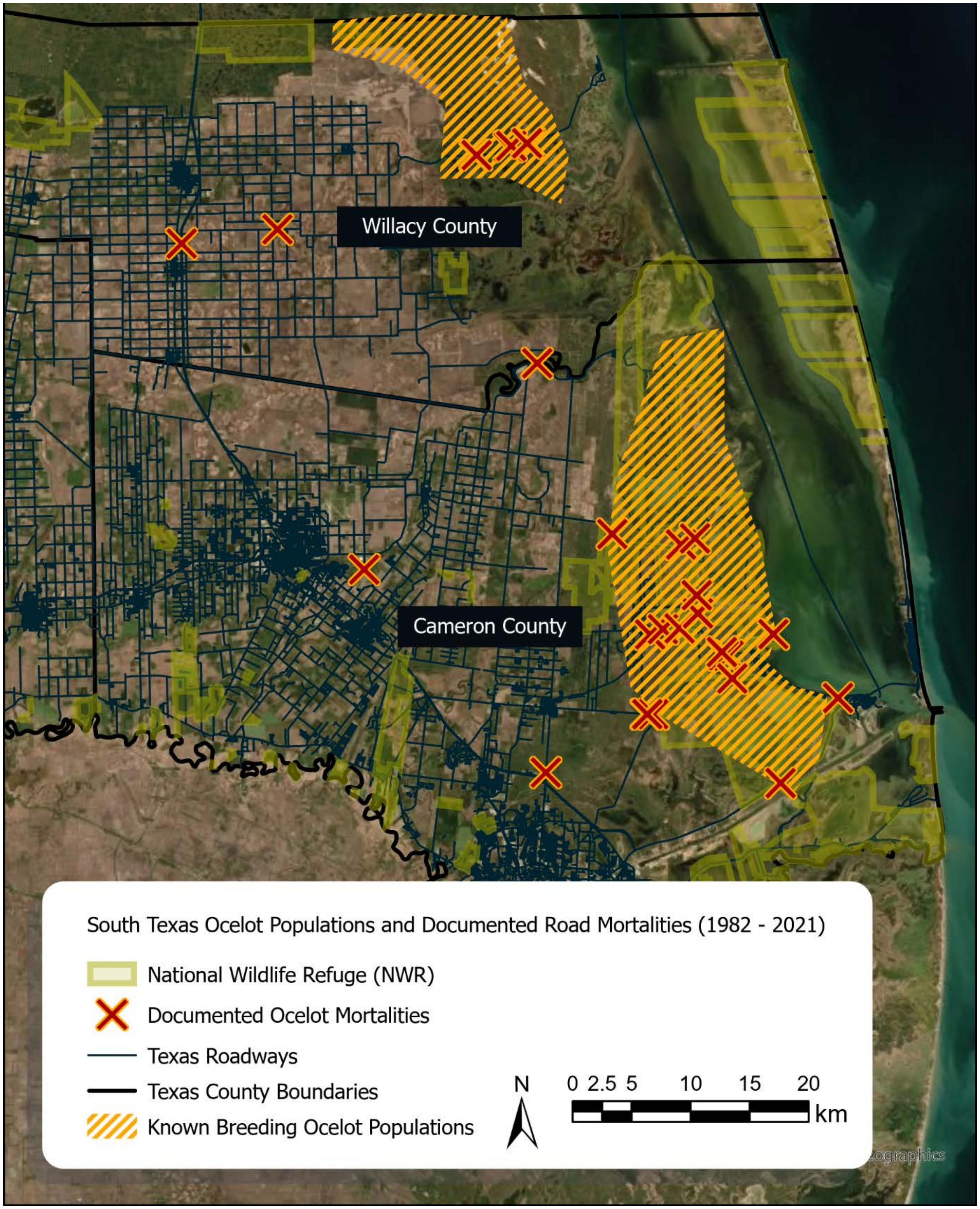

Figure 1. Documented ocelot mortalities between 1982 and 2021 (U.S. Fish and Wildlife; Texas Department of Transportation) and the two known breeding ocelot populations on private ranchlands (Willacy County) and national wildlife refuge lands (Cameron County) in Texas.

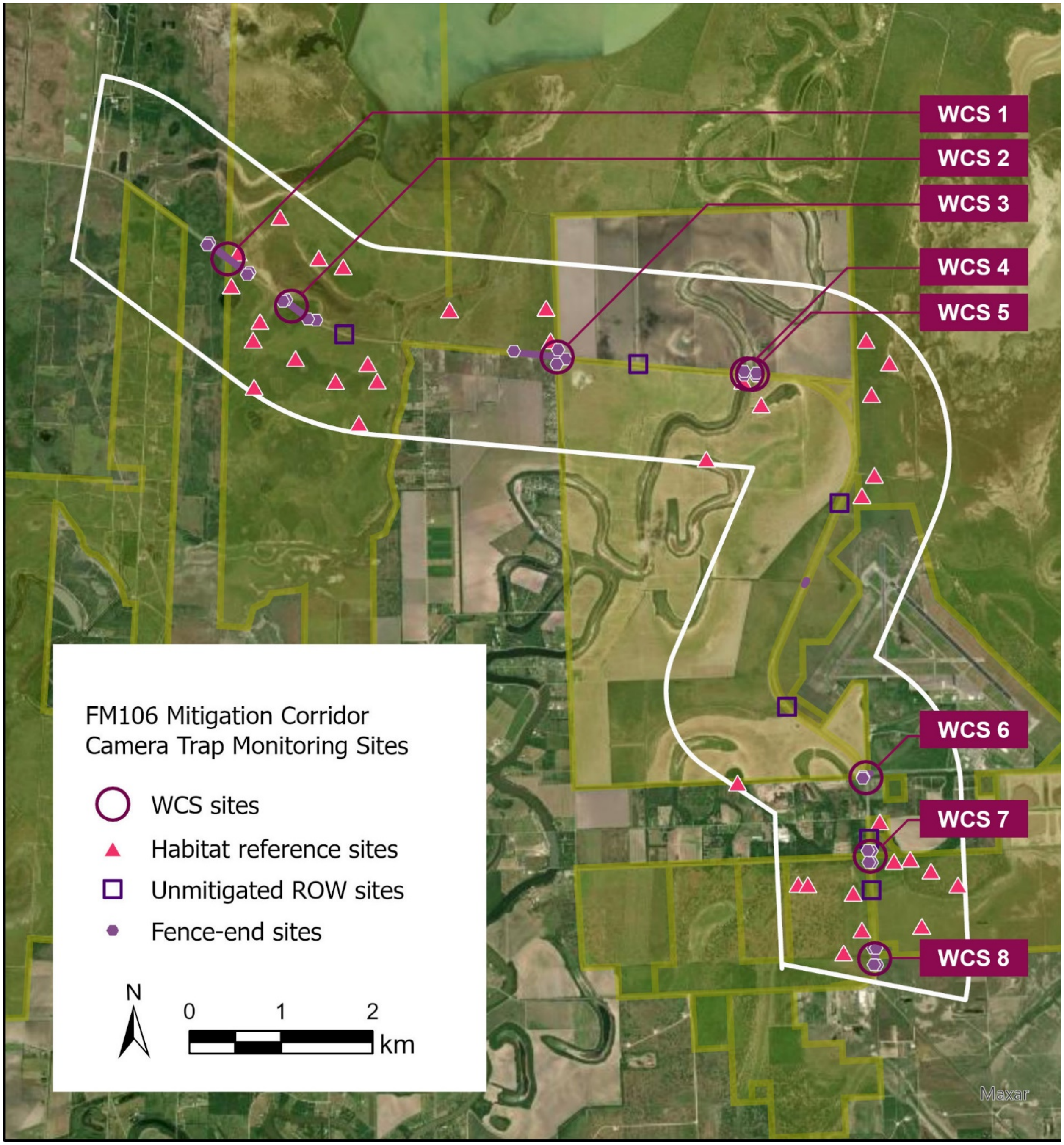

Figure 2. Locations of eight wildlife crossing structures (WCS) installed under a 16 km segment of Farm-to-Market Road (FM)106 in Cameron County, Texas. In addition to WCS sites, camera traps were deployed to monitor 14 mitigation fence-end (FE), 6 unmitigated right-of-way (ROW), and 36 habitat reference sites across the study corridor.

Figure 3. Eight wildlife crossing structures (WCS) were installed under Farm to Market Road (FM)106 in Cameron County, Texas. From top left, they are referred to as WCS 1 - 8. WCS 3 experiences regular sediment infill and flooding and as such is monitored with a fully passive infrared (PIR) camera trap system rather than a partially active infrared (AIR) system.

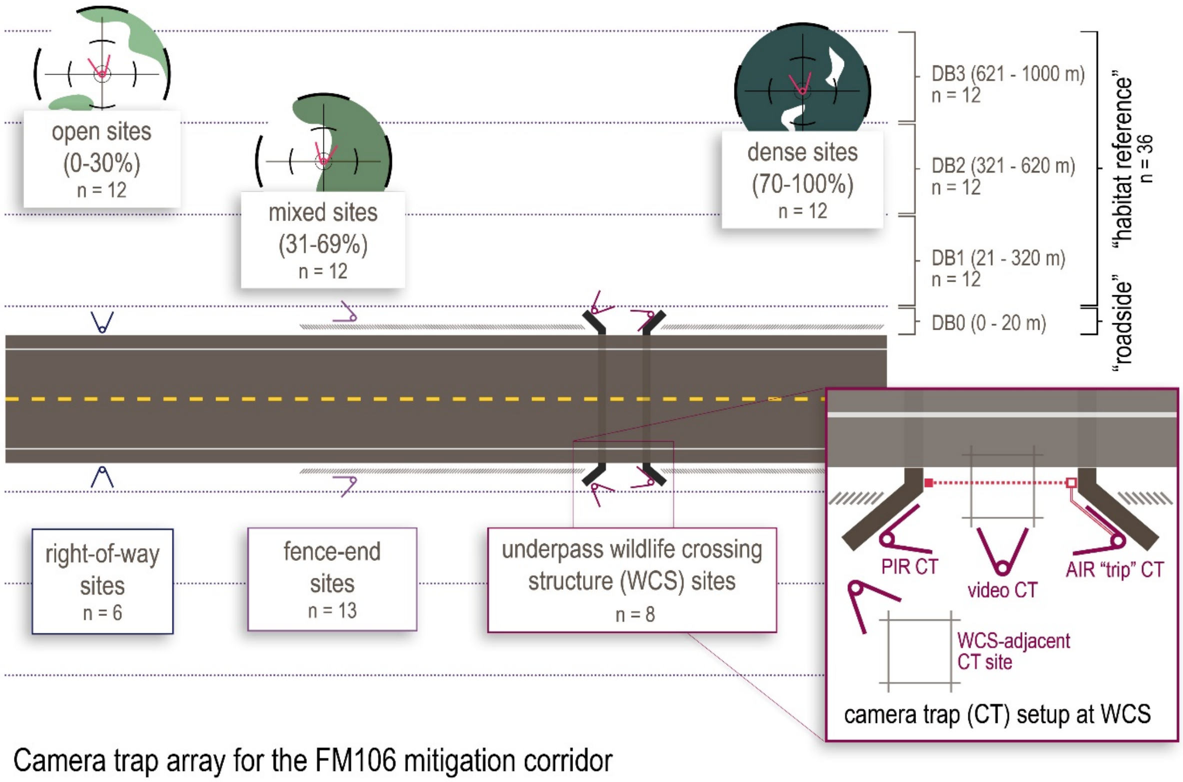

Figure 4. The camera trap (CT) arrays on the FM106 mitigation corridor include 27 roadside sites and 36 habitat reference sites. Roadside sites include WCS underpasses (8 CT each), mitigation fence-ends (2 CT each), and unmitigated right-of-way sites (2 CT each), all located within 20 m of the road surface. Habitat reference sites monitor up to 1000 m from the road surface; sites were randomly selected but evenly distributed across three vegetation classes (open = 0-30%, mixed = 31-69%, and dense = 70-100% woody canopy cover) and three distance bins (DB1 = 20-320 m, DB2 = 321-620 m, DB3 = 621-1000 m) from each side of the road.

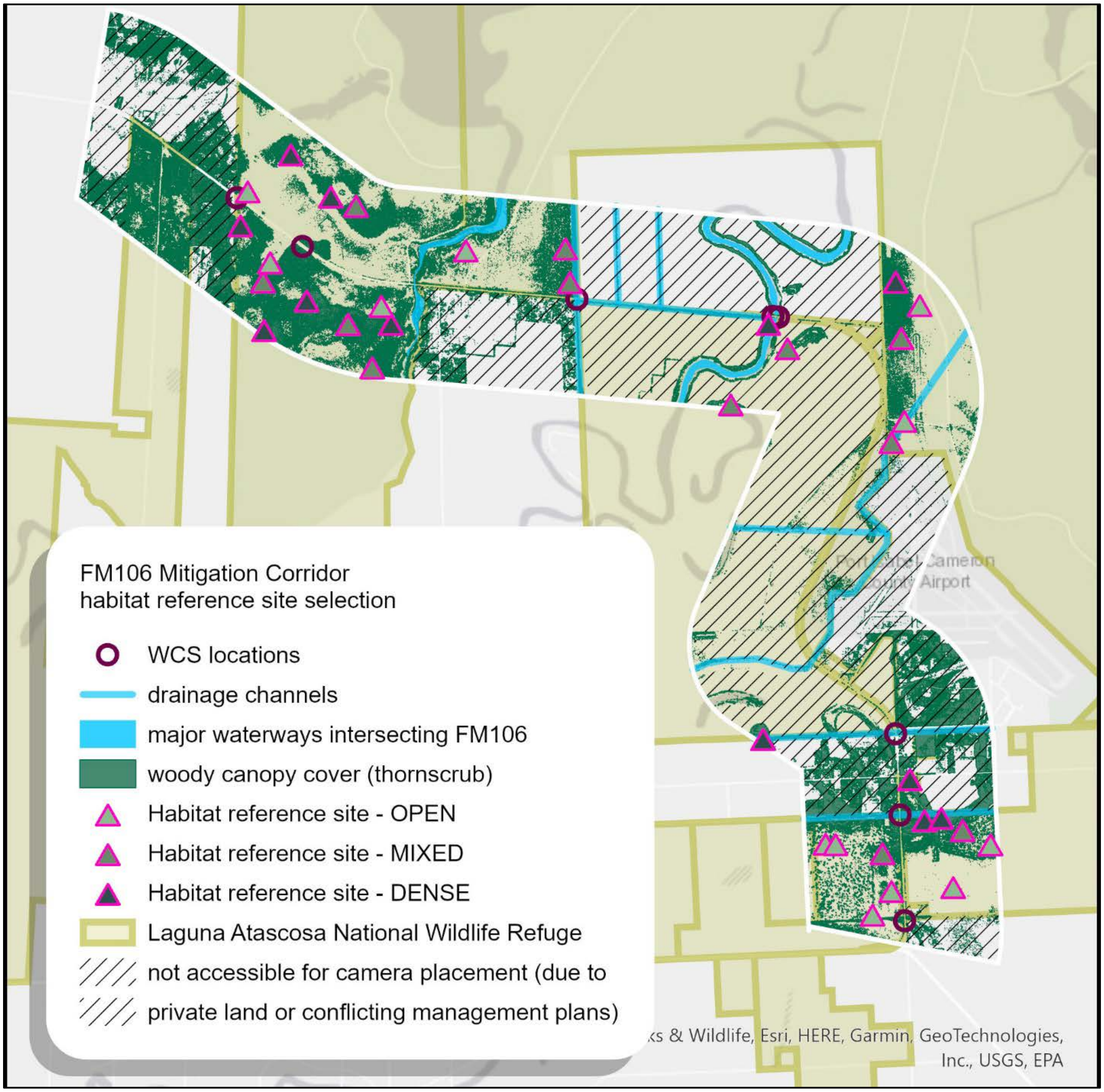

Figure 5. Habitat reference sites for the FM106 mitigation corridor in Cameron County, TX were determined by categorizing sites by estimated canopy cover density. Site selection across the corridor was limited to land managed by LANWR but also restricted by ongoing agricultural use and management plans. Resaca de los Cuates and several drainage ditches intersected the corridor and were closely associated with remnant thornscrub growth as well as WCS placement on the FM106 mitigation corridor in Cameron County, TX.

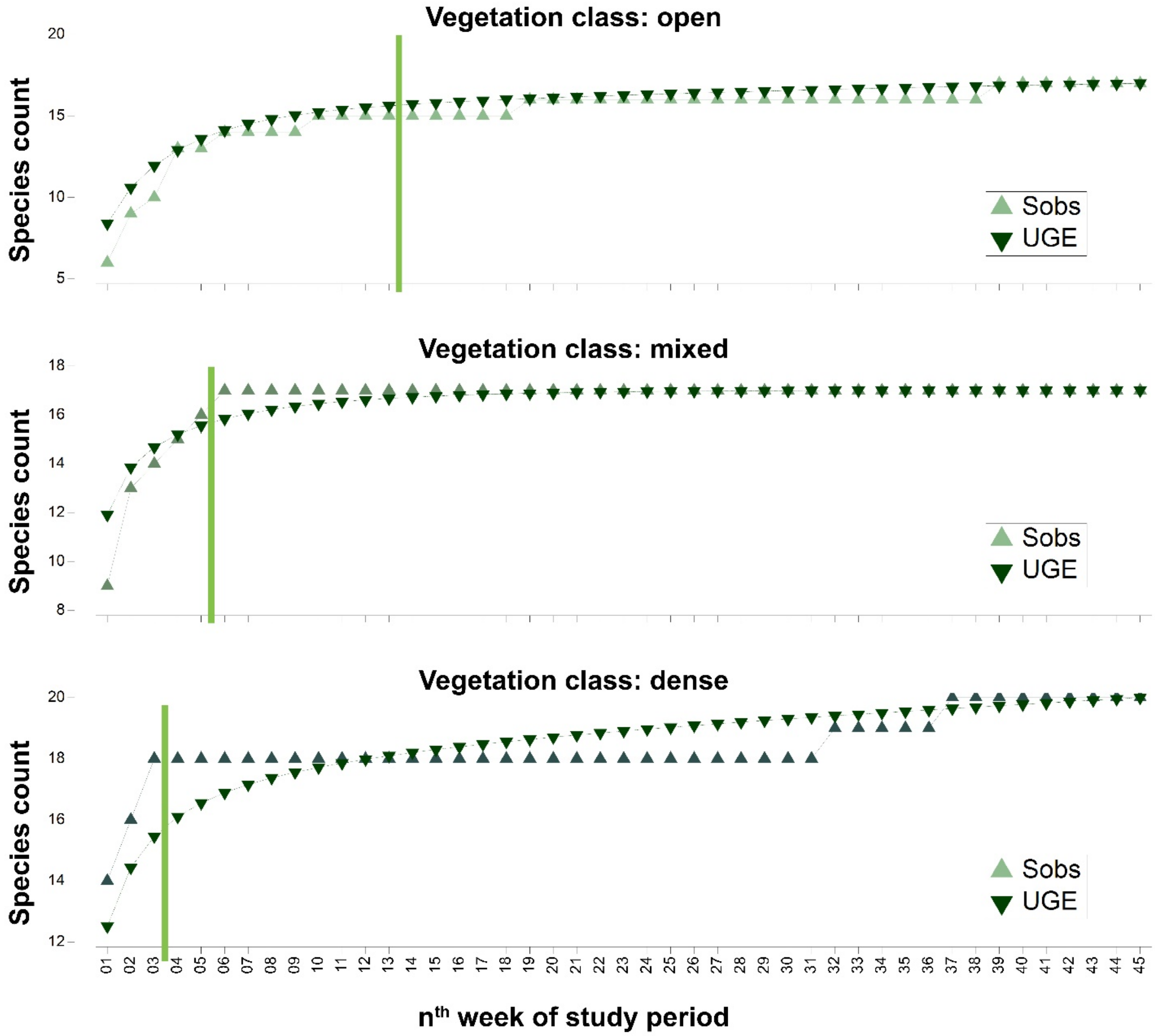

Figure 6. Species accumulation curves were calculated to estimate the number of camera trap nights required to observe the majority of the wildlife community on the FM106 road mitigation corridor in Cameron County, TX. The solid green line represents the time at which at least 90% of the wildlife community had been observed. This occurred at 13 weeks for open sites (17 of 18 total species observed), 5 weeks for mixed sites (16 of 17 total species observed), and 3 weeks for dense sites (18 of 20 total species observed).

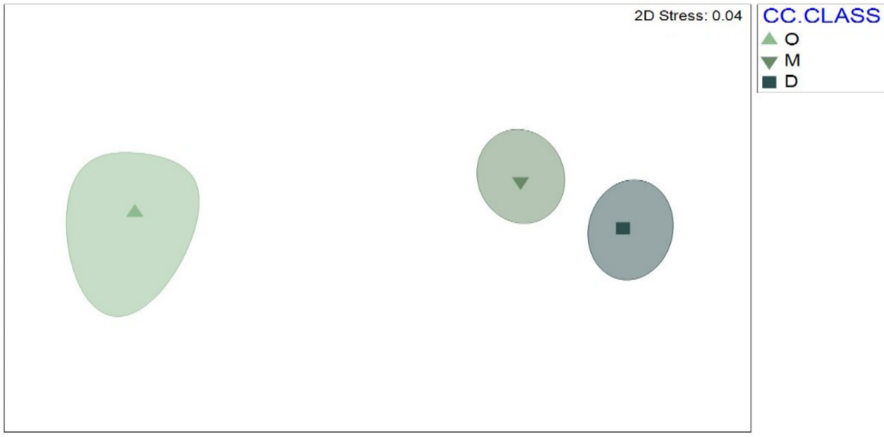

Figure 7. Non-metric MDS plot (stress = 0.04) of open, mixed, and dense camera trap sites sampling the wildlife community of the FM106 road mitigation corridor in Cameron County, TX from July 2021 – May 2022. The lack of overlap between groups indicates significantly distinct communities, which was verified using a PERMANOVA to test significance for open vs. mixed (p=0.0001), open vs. dense (p=0.0001), and mixed vs. dense (0.0007) vegetation classes.

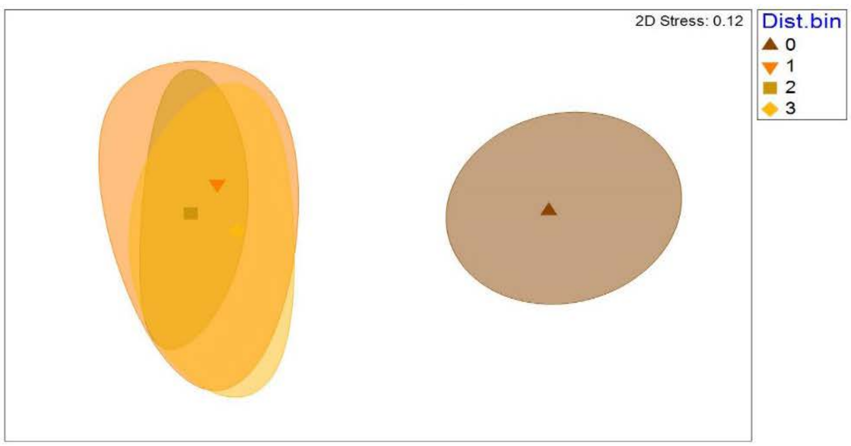

Figure 8. Non-metric MDS plot (stress = 0.12) of open, mixed, and dense camera trap sites sampling the wildlife community of the FM106 road mitigation corridor in Cameron County, TX from July 2021 – May 2022. The overlap between distance bins (DB)1, 2, and 3 indicates there was no significant difference between the wildlife communities sampled from these groups; however, DB0 was observed to have a significantly different community compared to DB1 (p=0.001), DB2 (p=0.0005), and DB3 (p=0.003).

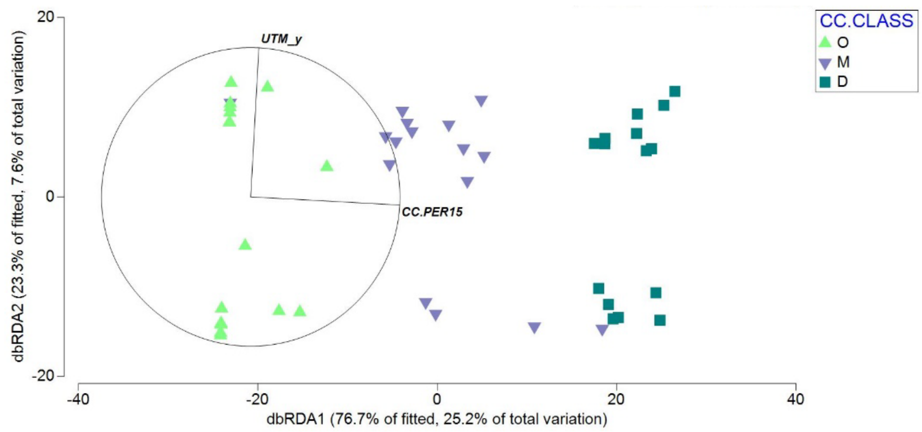

Figure 9. Local canopy cover density (CC.PER15) and north-south location (UTM_y), were identified as significant factors in explaining variation in the wildlife community observed across camera trap sites on the FM106 mitigation corridor in Cameron County, TX. DistLM results indicated that the best model (based on AICc) would include these two factors ($R^2 = 0.33$).

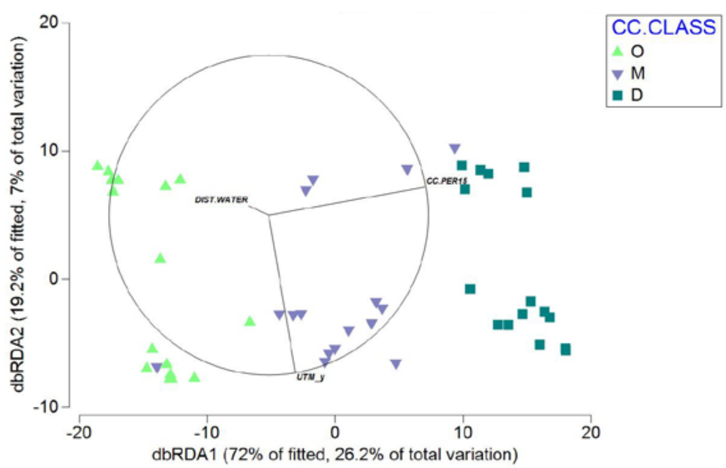

Figure 10. Local canopy cover density (CC.PER15), north-south location (UTM_y), and distance to water source (DIST.WATER) were identified as significant factors in explaining variation in presence-absence community data observed across camera trap sites on the FM106 mitigation corridor in Cameron County, TX. DistLM results indicated that the best model (based on AICc) would include these three factors ($R^2 = 0.36$).

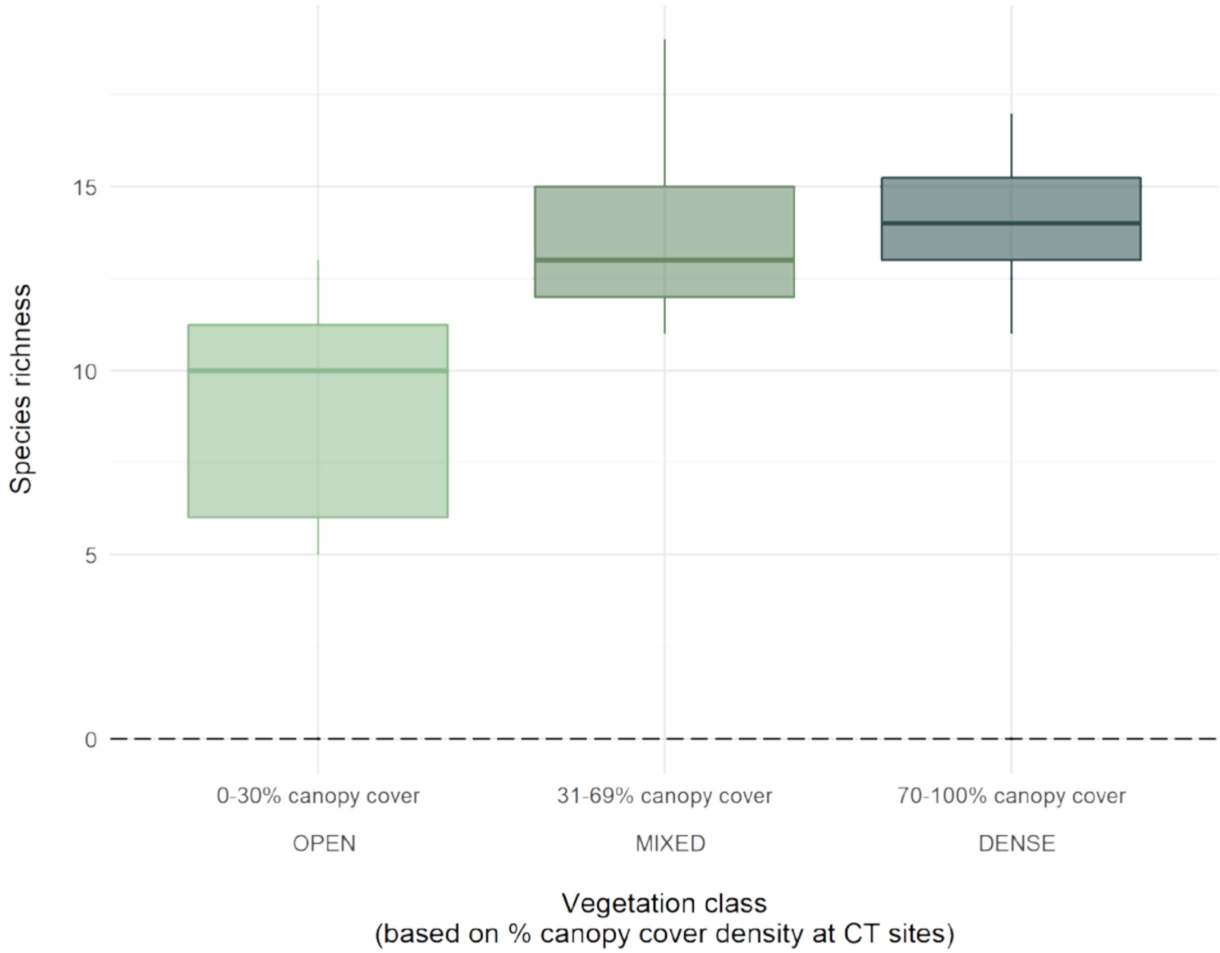

Figure 11. Species richness documented across Open, Mixed, and Dense camera trap sites on the FM106 mitigation corridor in Cameron County, TX from July 2021 – May 2022. ANOVA results and Tukey HSD post-hoc testing indicated significant differences in species richness between Open and Mixed (p=0.00001) and between Open and Dense (p<0.00001), but not between Mixed and Dense sites.

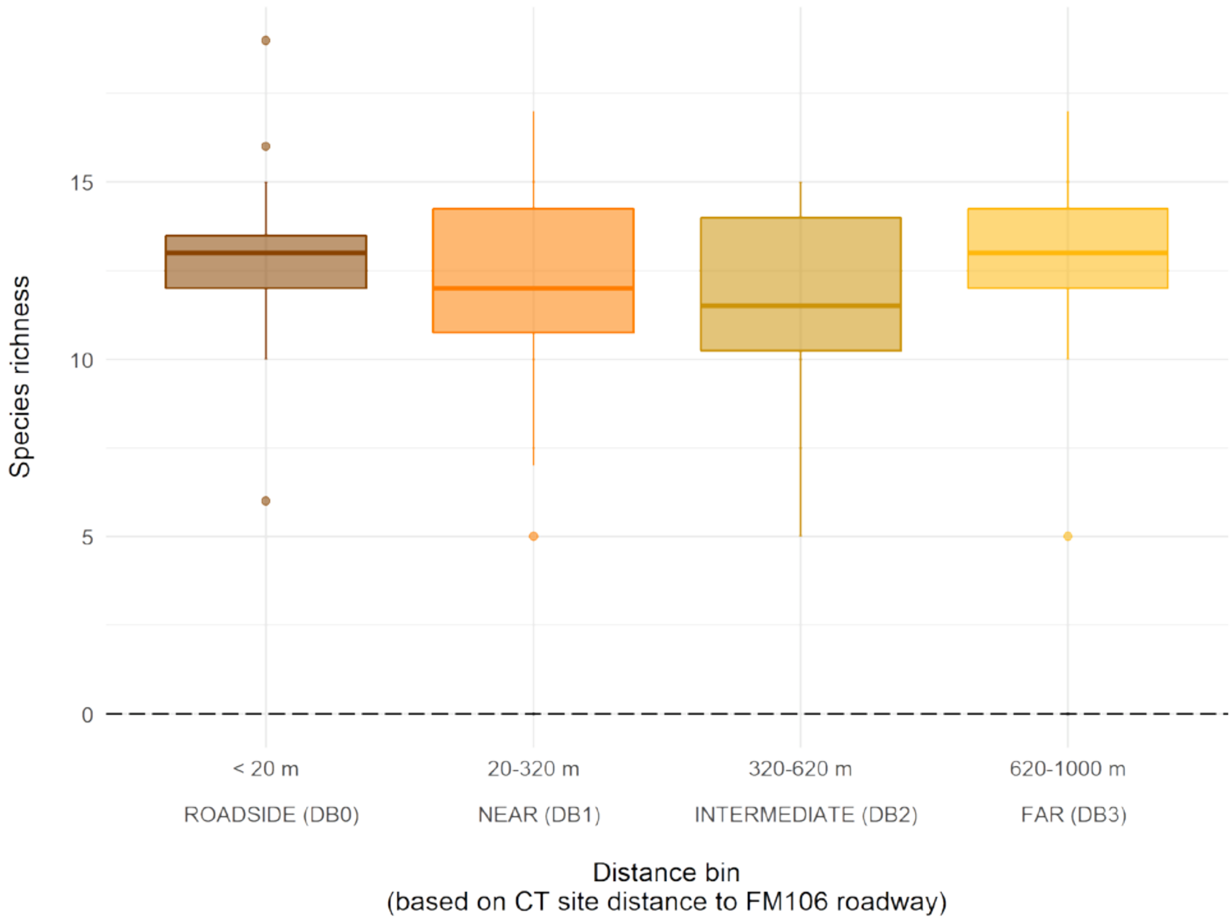

Figure 12. Species richness documented across roadside, near, intermediate, and far camera trap sites on the FM106 mitigation corridor in Cameron County, TX from July 2021 – May 2022. ANOVA results indicated no significant differences in species richness across any distance bins.

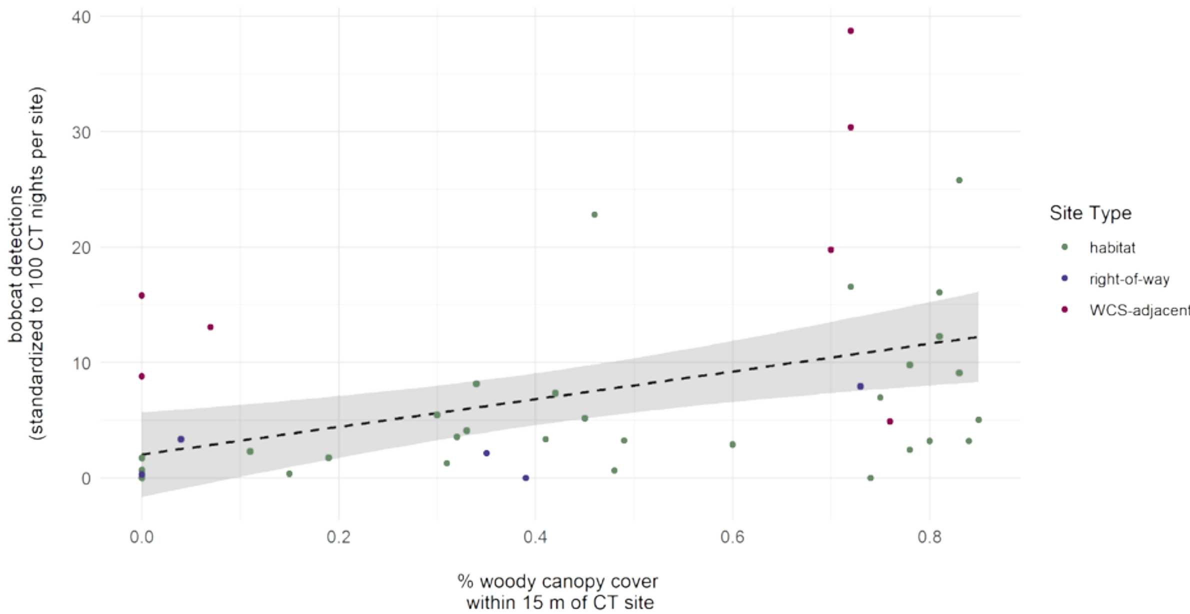

Figure 13. Bobcat detections, standardized to 100 camera trap (CT) nights per site, were correlated with canopy cover density (within 15 m) across habitat and select roadside camera sites from July 2021 – May 2022 on the FM106 mitigation corridor in Cameron County, TX. Raw data visualized here corresponds to the pattern identified using a negative binomial generalized linear model, which indicated that bobcat detections were significantly correlated with increased canopy cover within 15 m as well as decreased distance to woody cover, decreased distance to FM106, and decreased canopy cover between site and road ($R^2 = 0.48$).

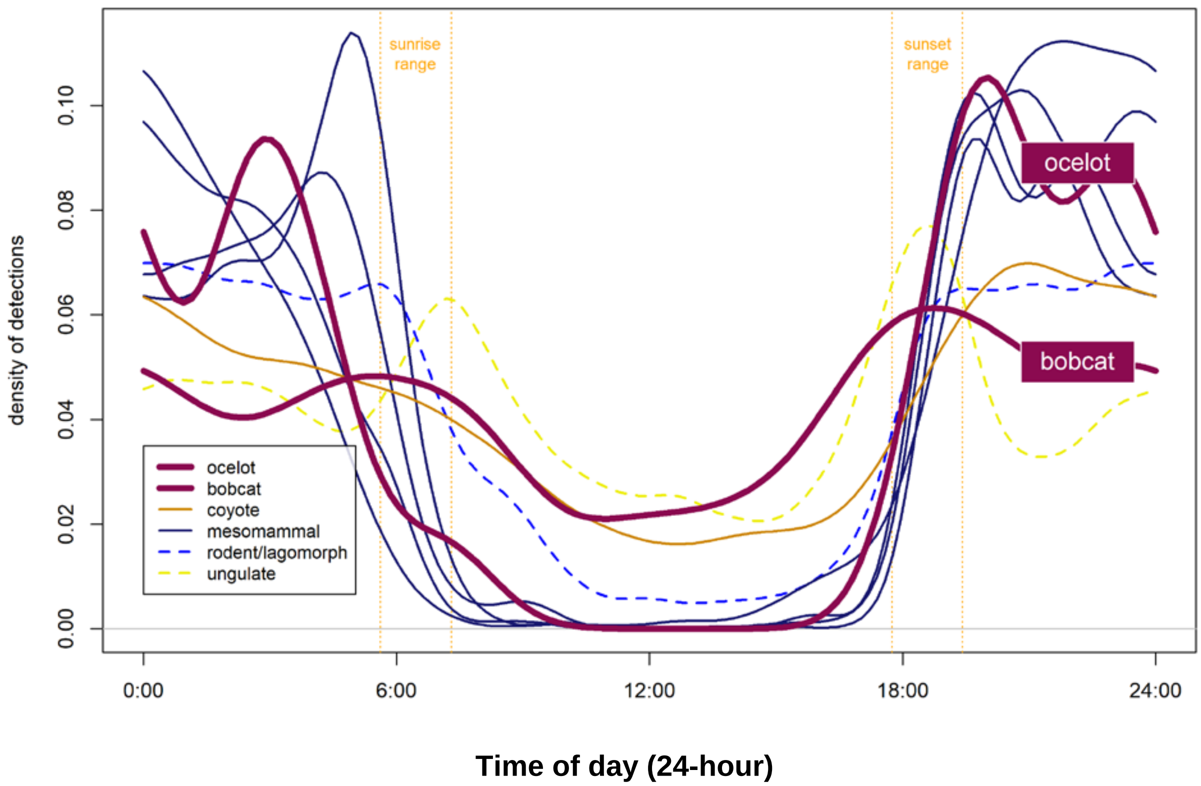

Figure 14. Circular probability distribution plots representing diel activity patterns for groups of species commonly detected on the FM106 mitigation corridor in Cameron County, TX. Ocelot detections (n=42) from over 2 years of post-construction roadside monitoring were used to estimate a diel activity curve; all other species activity curves were developed from habitat and select roadside camera trap detections between July 2021 and May 2022.

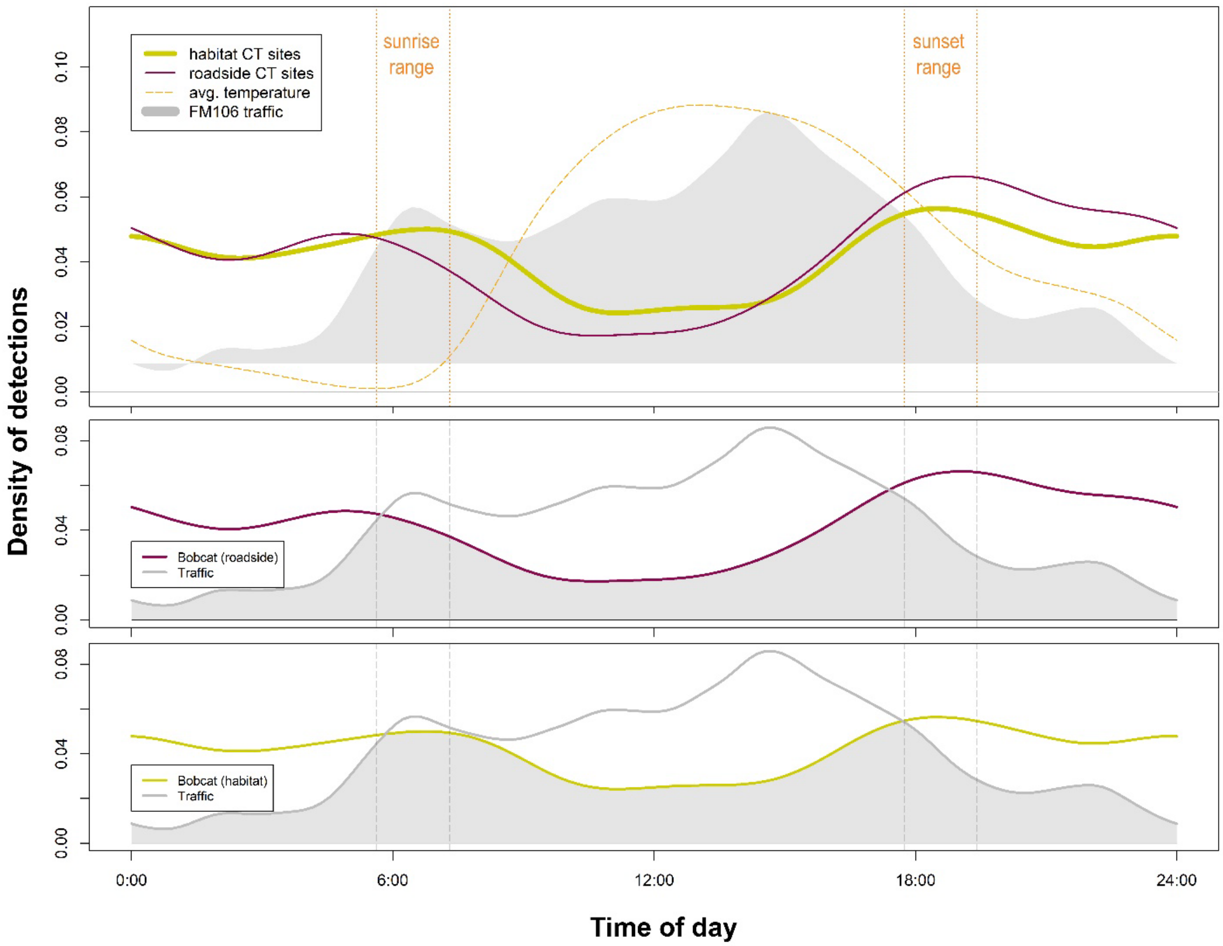

Figure 15. Bobcat (*Lynx rufus*) activity at roadside camera trap (CT) sites was significantly distinct from activity at habitat sites (p = 0.04) on the FM106 mitigation corridor in Cameron County, TX from July 2021 – May 2022. Traffic activity showed less overlap with bobcat activity at roadside sites versus habitat sites, though overlap in the 95% confidence intervals indicates this may not necessarily be a significant directional shift. These results partially support the hypothesis that bobcats may be exhibiting temporal avoidance at times of high human use.

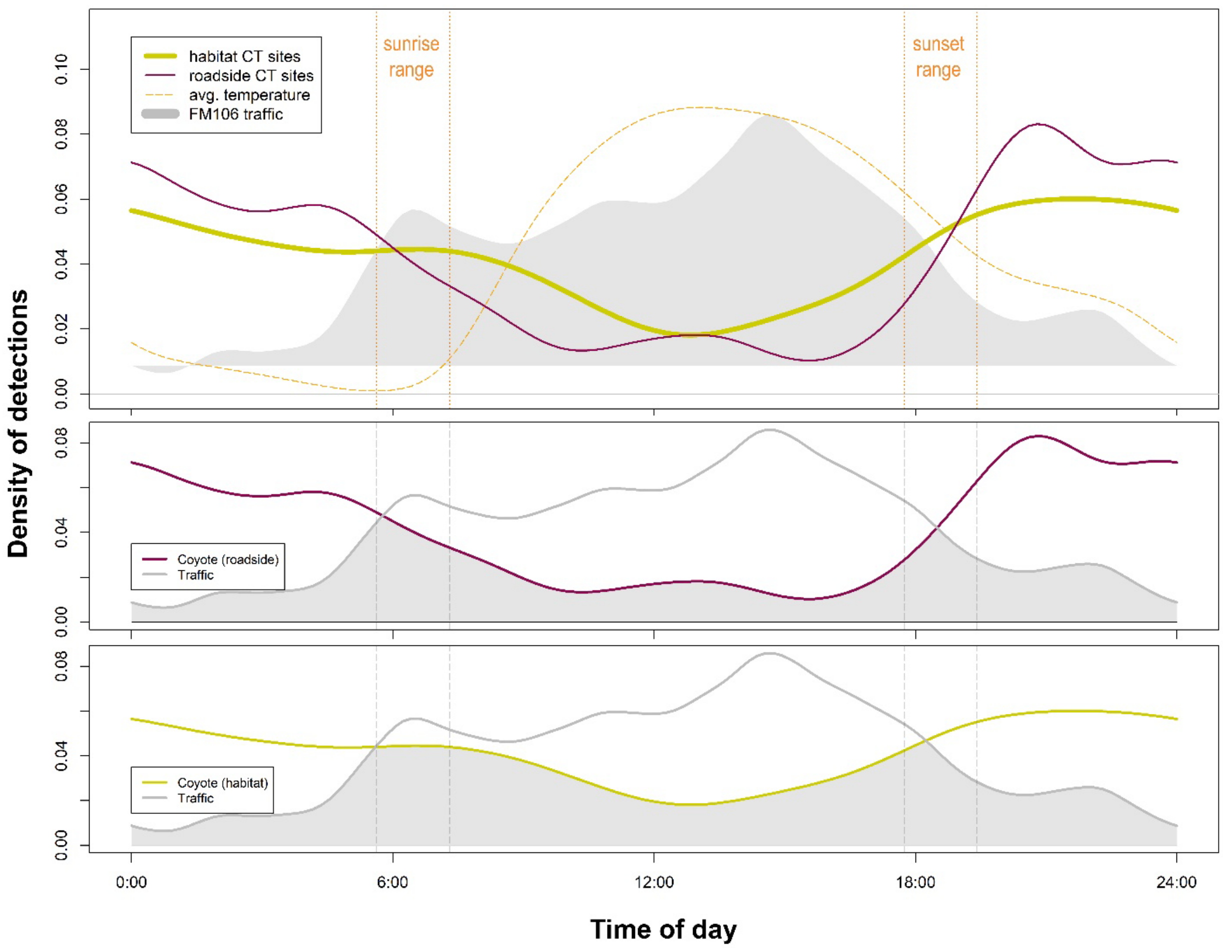

Figure 16. Coyote (*Canis latrans*) activity at roadside (WCS and ROW) sites was significantly distinct from activity at habitat sites (p = 0.01) according to a statistical comparison of kernel density distributions. Activity at roadside sites showed significantly less overlap with daily traffic patterns than coyote activity at habitat sites, indicating that this species may be exhibiting temporal avoidance at times of high human use on the FM106 mitigation corridor in Cameron County, TX.

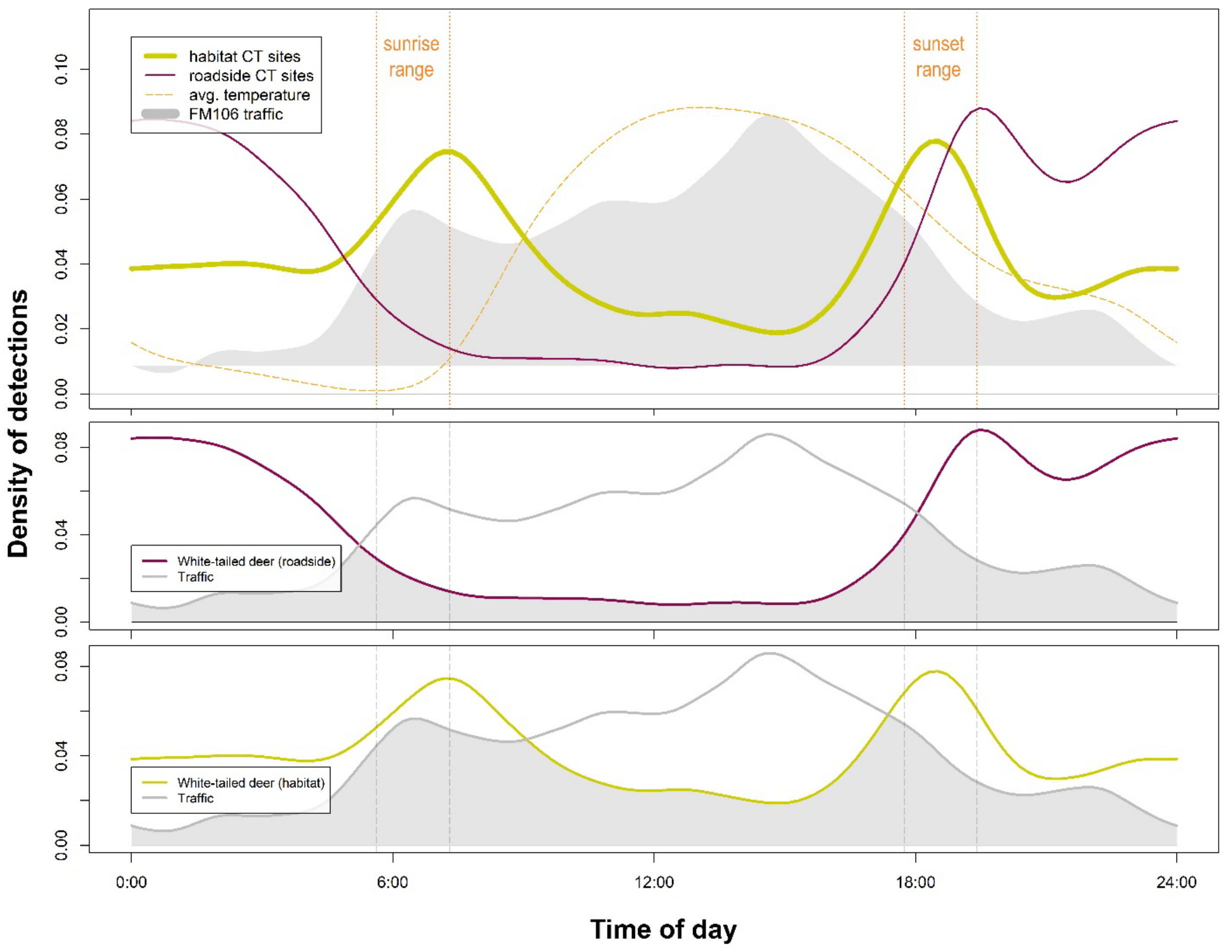

Figure 17. White-tailed deer (*Odocoileus virginianus*) activity at roadside (WCS and ROW) sites was significantly distinct from activity at habitat sites (p = 0.00) according to a statistical comparison of kernel density distributions. Daily traffic patterns showed significantly less overlap with white-tailed deer activity at roadside sites than at habitat sites, indicating that this species may be exhibiting temporal avoidance at times of high human use on the FM106 mitigation corridor in Cameron County, TX.

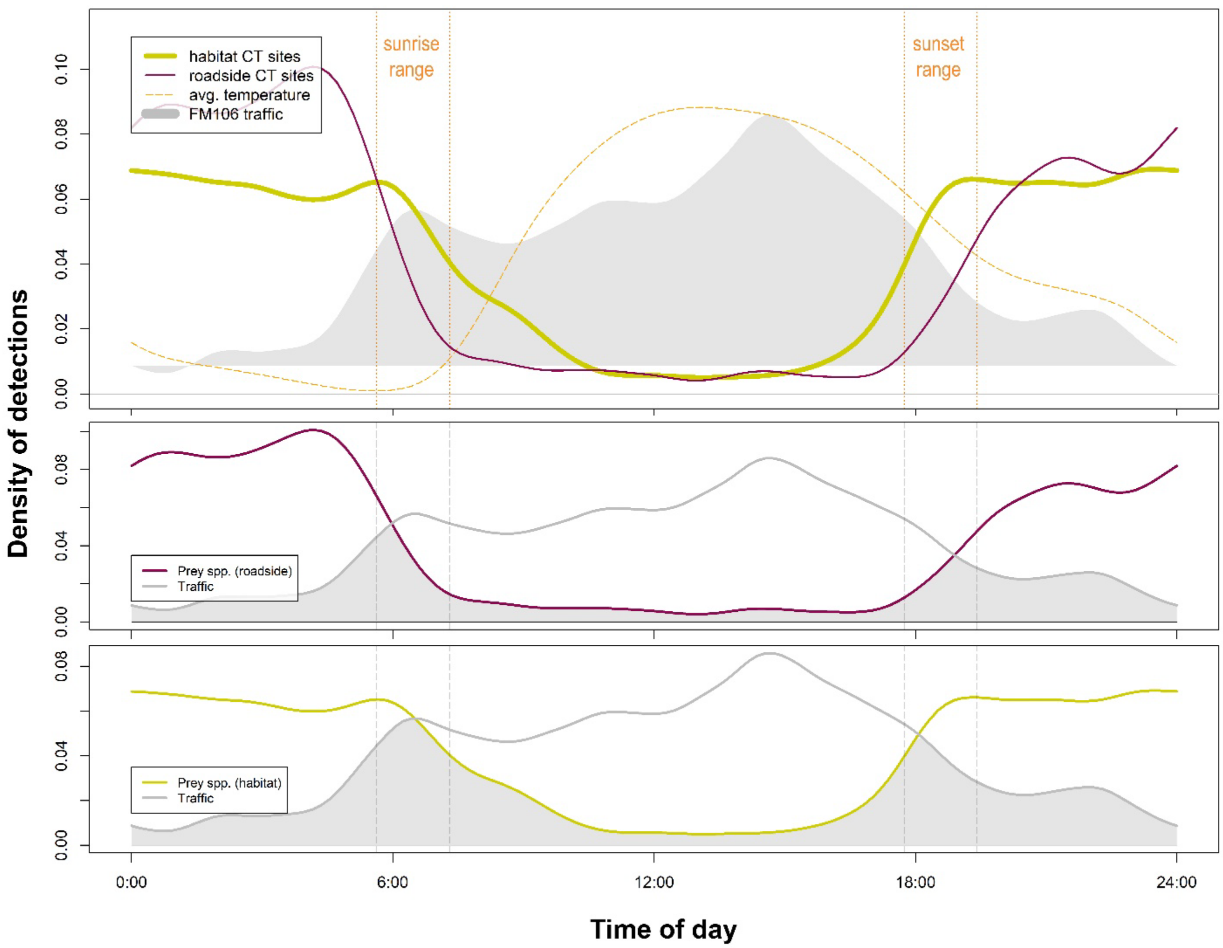

Figure 18. Potential felid prey species (rodent and lagomorph) activity at roadside (WCS and ROW) sites was significantly distinct from activity at habitat sites (p = 0.00) according to a statistical comparison of kernel density distributions. Daily traffic patterns showed significantly less overlap with prey species activity at roadside sites than at habitat sites, indicating that this species may be exhibiting temporal avoidance at times of high human use on the FM106 mitigation corridor in Cameron County, TX.

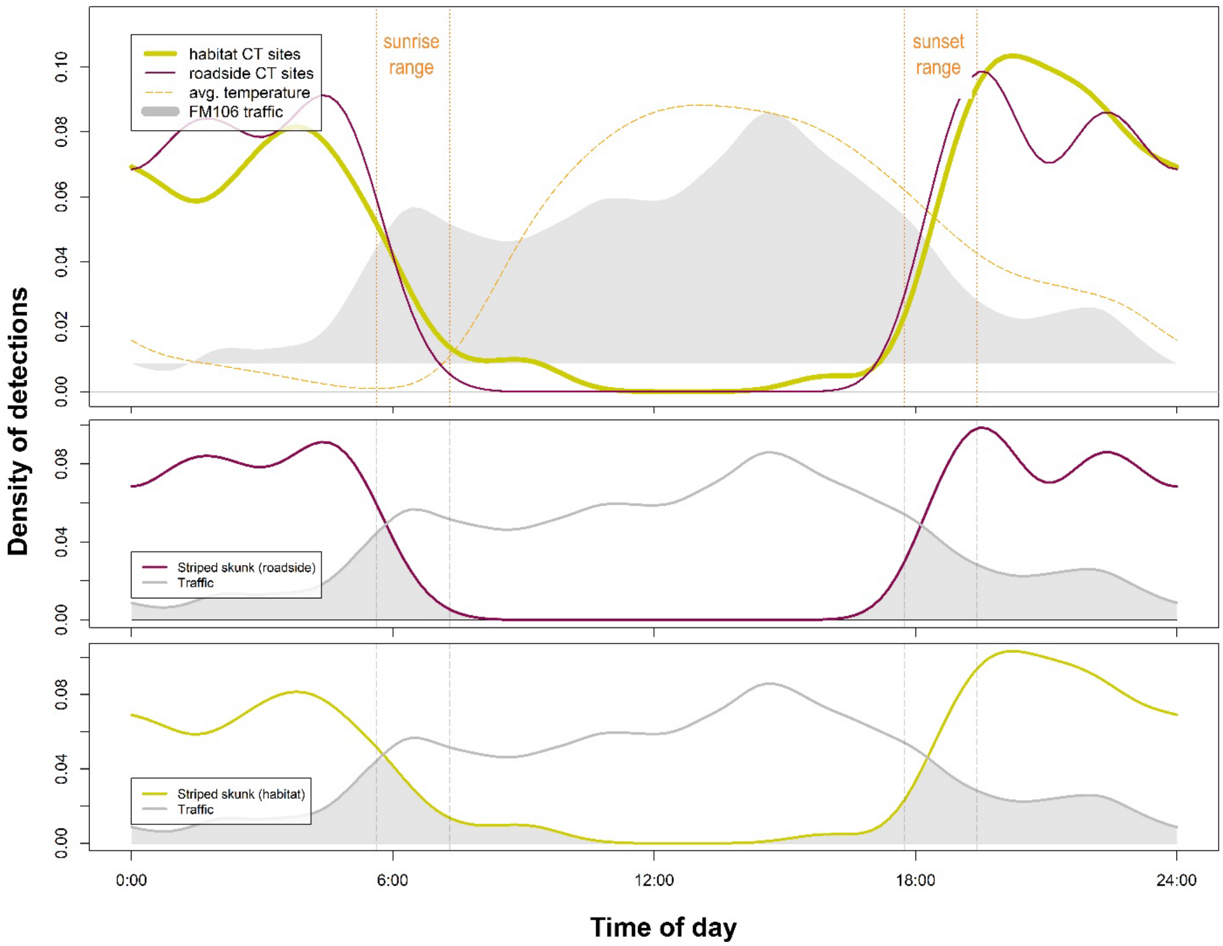

Figure 19. Striped skunk (*Mephitis mephitis*) activity at roadside (WCS and ROW) sites was not significantly distinct from activity at habitat sites (p = 0.35) according to a statistical comparison of kernel density distributions. This species does not appear to be altering its daily activity patterns based on traffic on the FM106 mitigation corridor in Cameron County, TX.

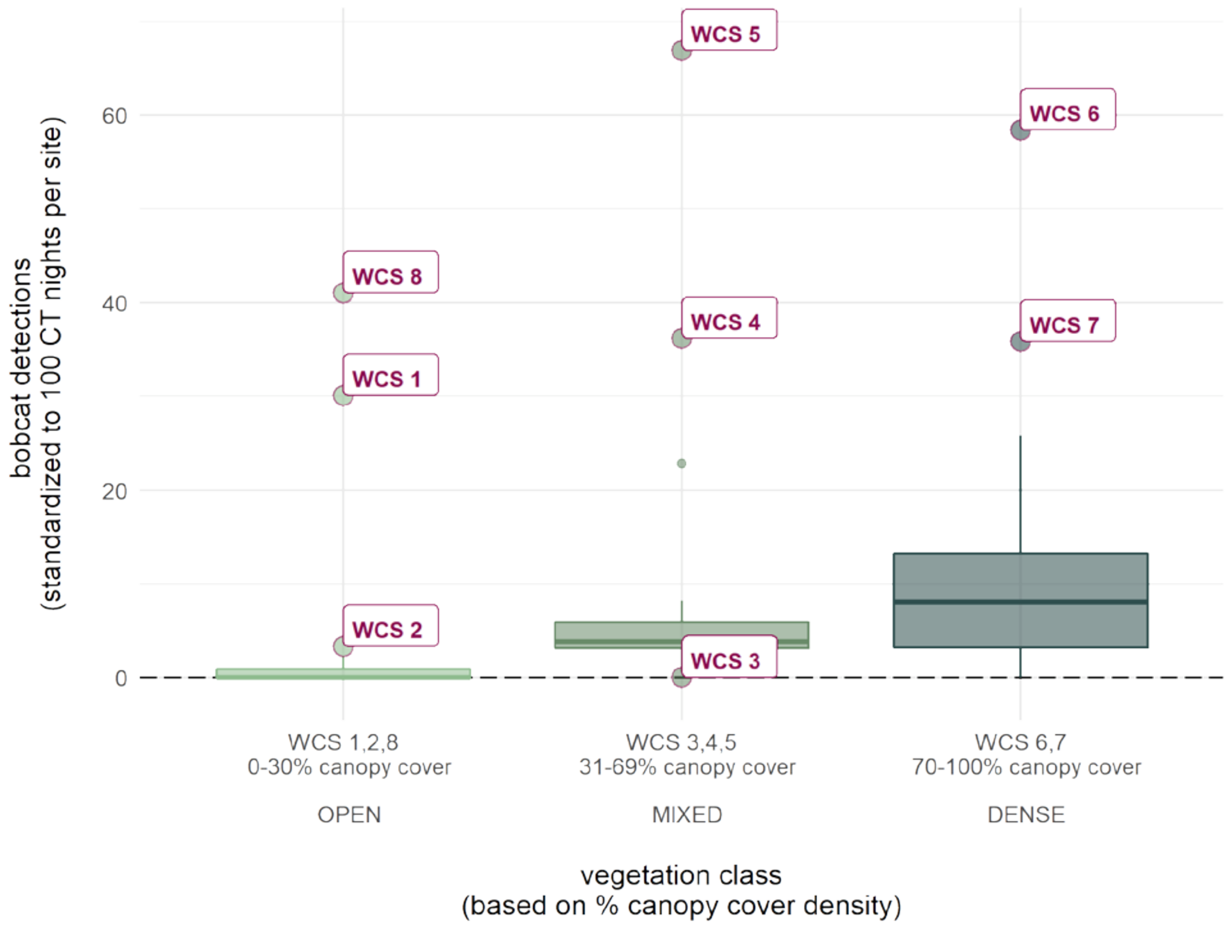

Figure 20. Bobcat crossing events at WCS 1-8 as compared to bobcat detections at habitat reference sites of the same vegetation class (open, mixed, dense). WCS sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Within each vegetation class, performance scores are calculated: ACF - ECF = PD. For bobcats, all WCS sites except for WCS 3 had a positive PD, indicating these sites are used more often than can be expected based on comparable sites in the surrounding habitat, and likely provide habitat connectivity across the FM106 corridor.

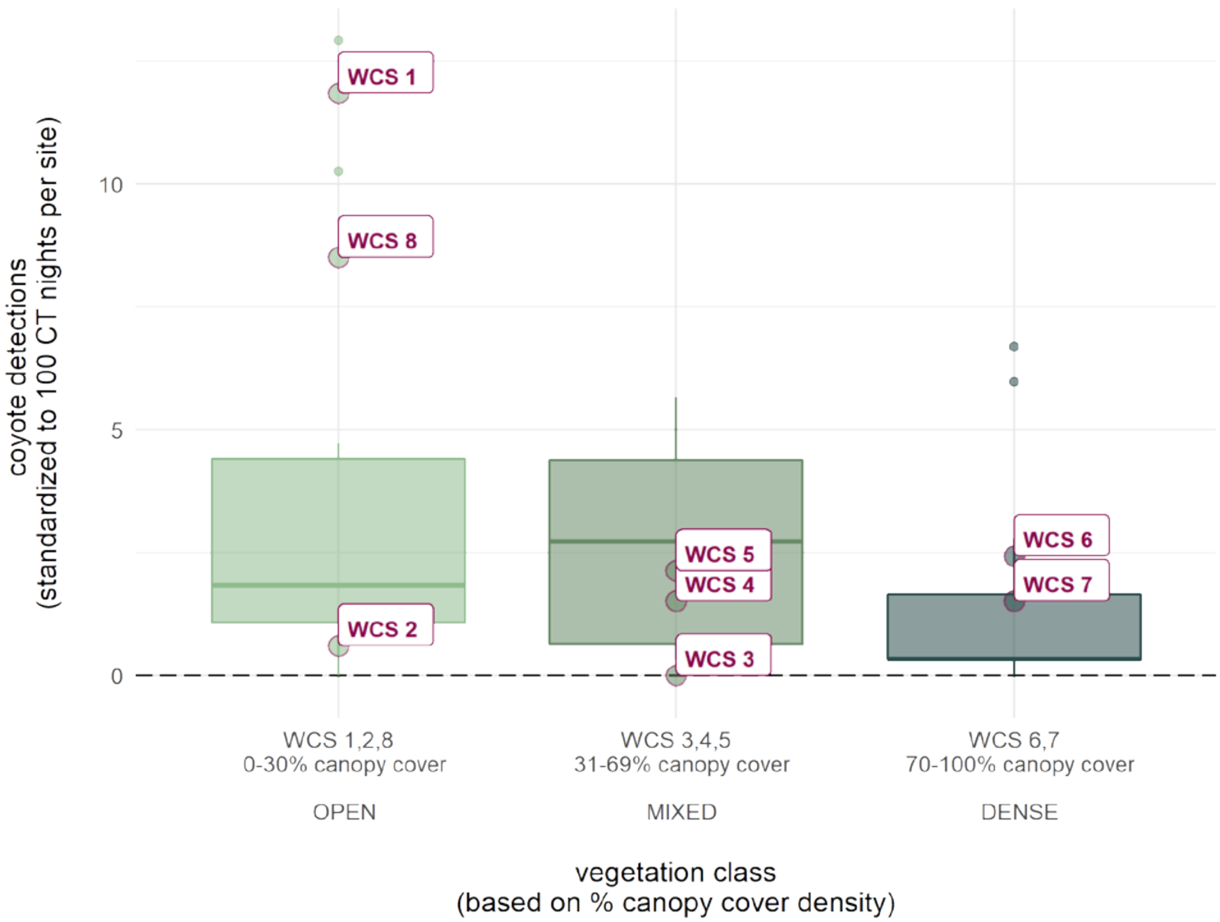

Figure 21. Coyote crossing events at WCS 1-8 as compared to coyote detections at habitat reference sites of the same vegetation class (open, mixed, dense). WCS sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Within each vegetation class, performance scores are calculated: ACF - ECF = PD. Coyotes had particularly high PD at WCS 1, 6, and 8, which may indicate relative success for these sites at providing habitat connectivity. PD were particularly low at WCS 2 and 3, which may indicate relative failure of these sites at providing habitat connectivity.

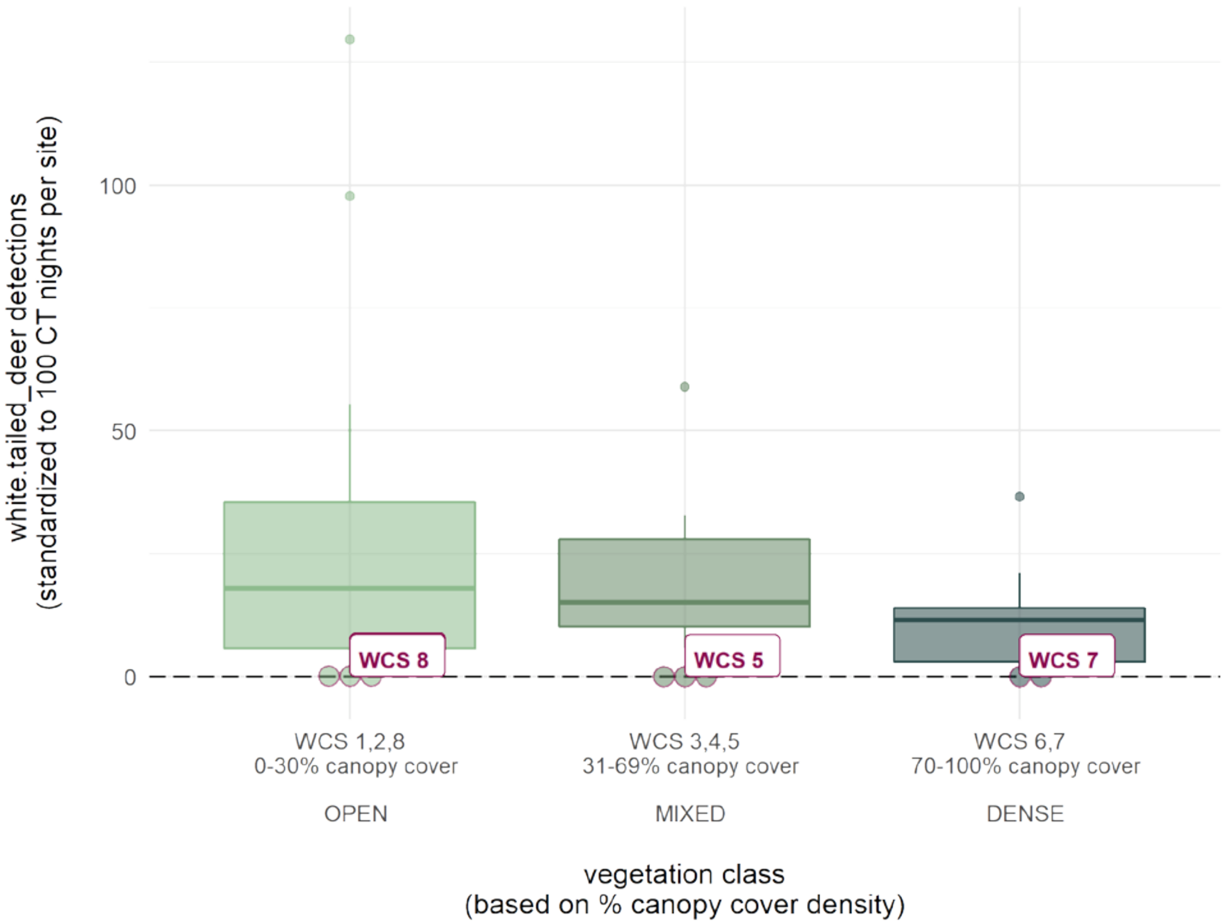

Figure 22. White-tailed deer crossing events at WCS 1-8 as compared to detections at habitat reference sites of the same vegetation class (open, mixed, dense). WCS sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Within each vegetation class, performance scores are calculated: ACF - ECF = PD. White-tailed deer had negative PD at all WCS sites, indicating that structures fail to provide habitat connectivity for this species.

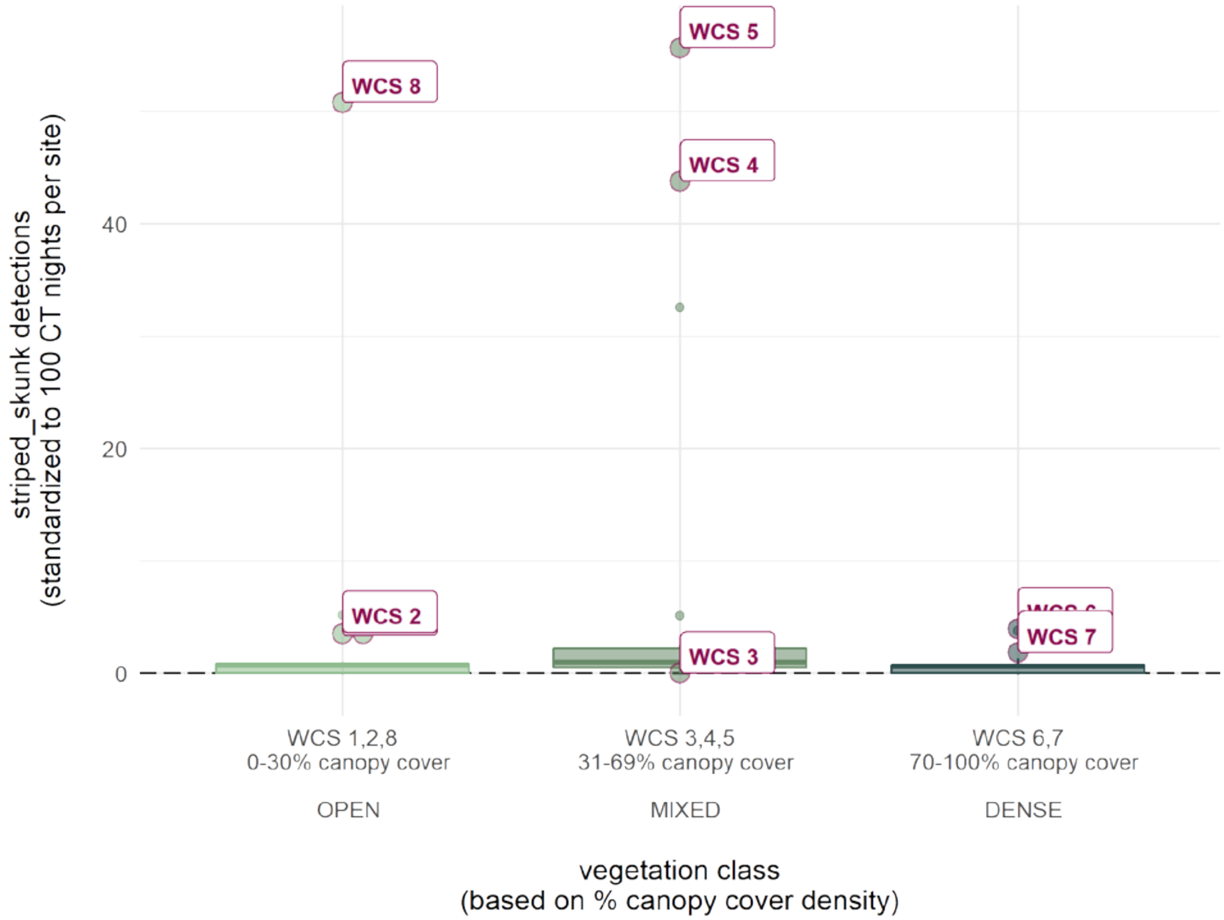

Figure 23. Striped skunk crossing events at WCS 1-8 as compared to detections at habitat reference sites of the same vegetation class (open, mixed, dense). WCS sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Within each vegetation class, performance scores are calculated: ACF - ECF = PD. For striped skunks, all sites except for WCS 3 had a positive PD, indicating these sites are used more often than can be expected based on comparable sites in the surrounding habitat, and likely provide habitat connectivity across the FM106 corridor.

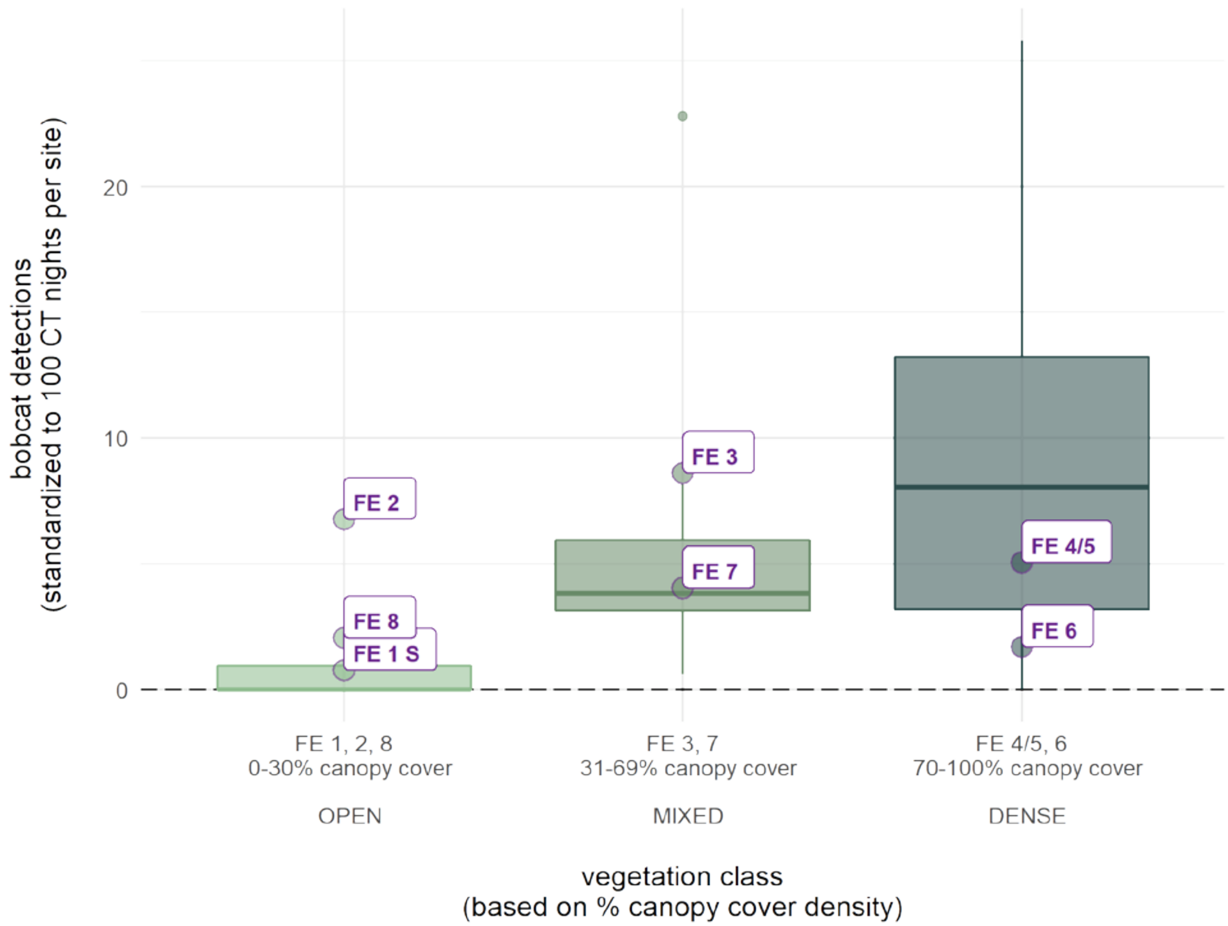

Figure 24. Bobcat crossing events at FE 1-8 as compared to bobcat detections at habitat reference sites of the same vegetation class (open, mixed, dense). FE sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Positive PD means a species is observed at a mitigation site more often than can be expected based on comparable habitat reference sites; for fence-ends, this may indicate that fencing is insufficient at preventing species movement at or across the roadway. Bobcats had particularly high PD for FE sites 2 and 3.

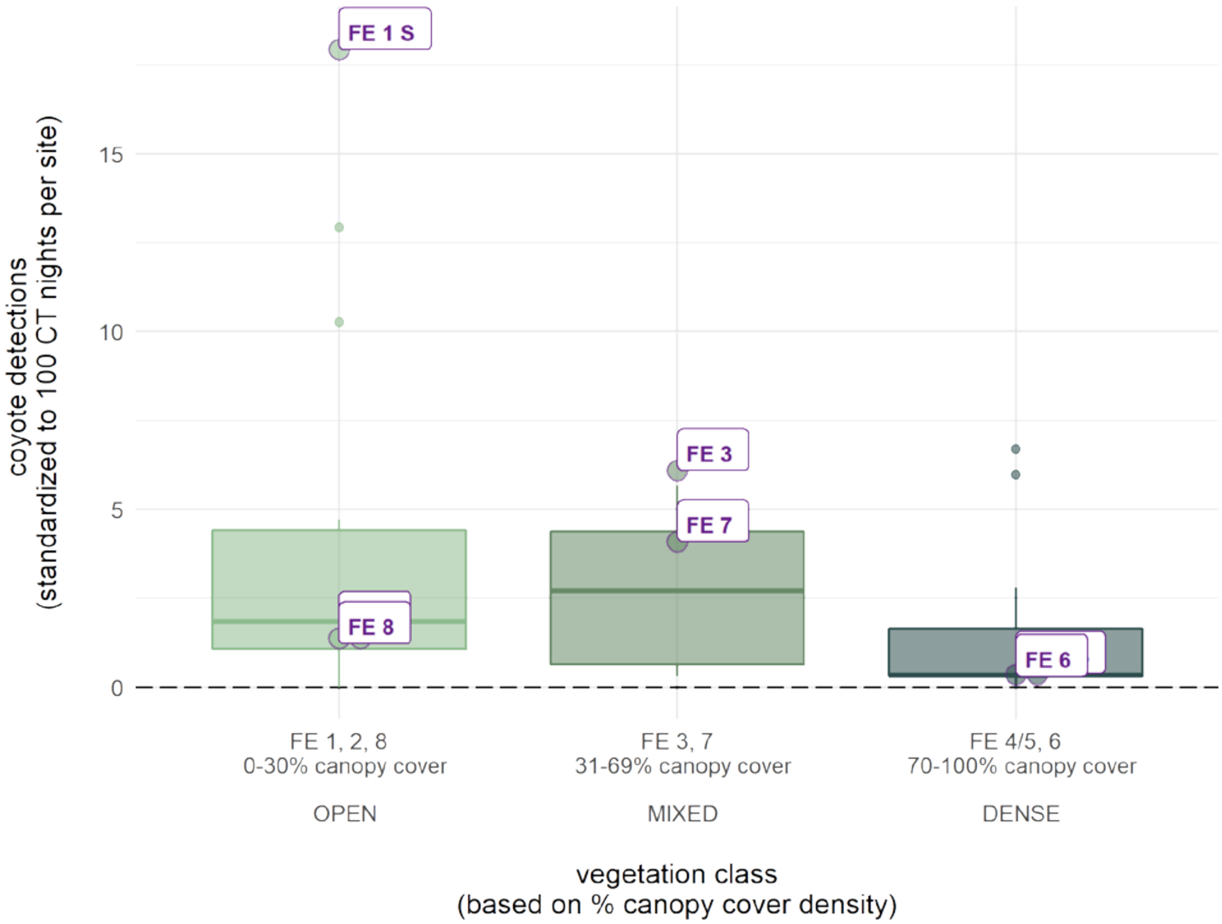

Figure 25. Coyote crossing events at FE 1-8 as compared to coyote detections at habitat reference sites of the same vegetation class (open, mixed, dense). FE sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Positive PD means a species is observed at a mitigation site more often than can be expected based on comparable habitat reference sites; for fence-ends, this may indicate that fencing is insufficient at preventing species movement at or across the roadway. Coyotes had particularly high PD for FE sites 1 and 3.

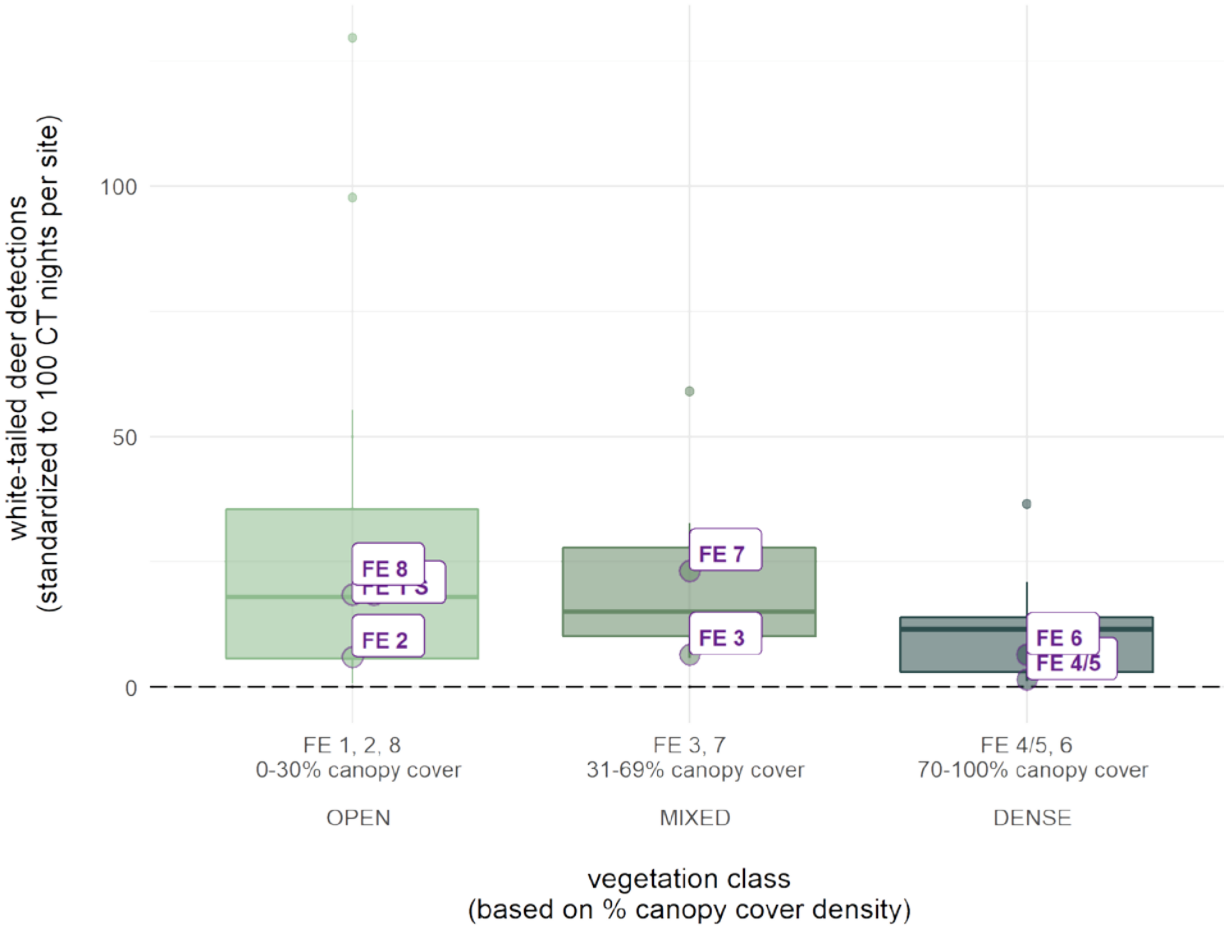

Figure 26. White-tailed deer crossing events at FE 1-8 as compared to detections at habitat reference sites of the same vegetation class (open, mixed, dense). FE sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Positive PD means a species is observed at a mitigation site more often than can be expected based on comparable habitat reference sites; for fence-ends, this may indicate that fencing is insufficient at preventing species movement at or across the roadway, but in this case, white-tailed deer did not have particularly high PD for any FE sites.

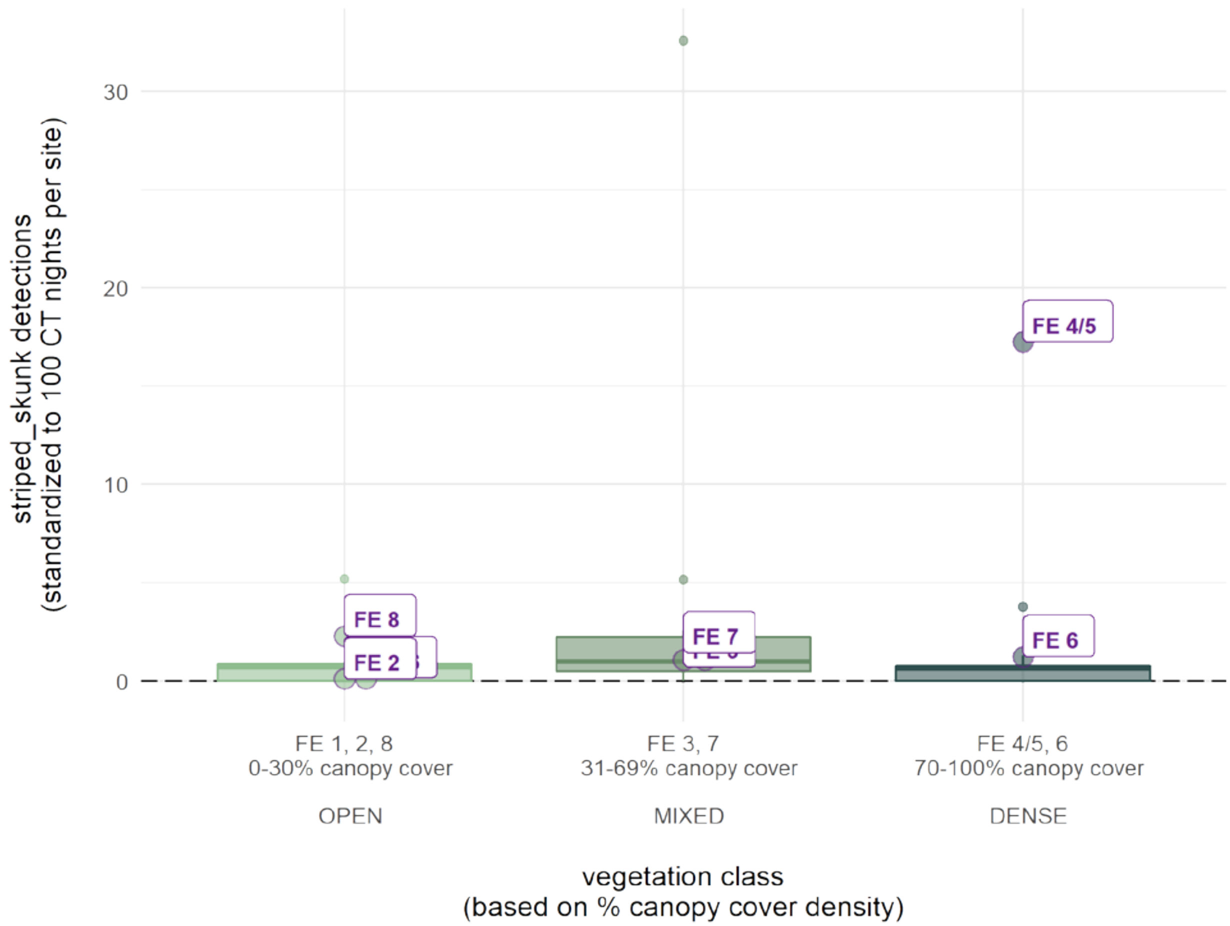

Figure 27. Striped skunk crossing events at FE 1-8 as compared to detections at habitat reference sites of the same vegetation class (open, mixed, dense). FE sites with a higher actual crossing frequency (ACF) than the mean habitat detections (expected crossing frequency or "ECF") are said to have a positive performance differential (PD). Positive PD means a species is observed at a mitigation site more often than can be expected based on comparable habitat reference sites; for fence-ends, this may indicate that fencing is insufficient at preventing species movement at or across the roadway. In this case, striped skunk had particularly high PD for fence-end sites 4 and 5.

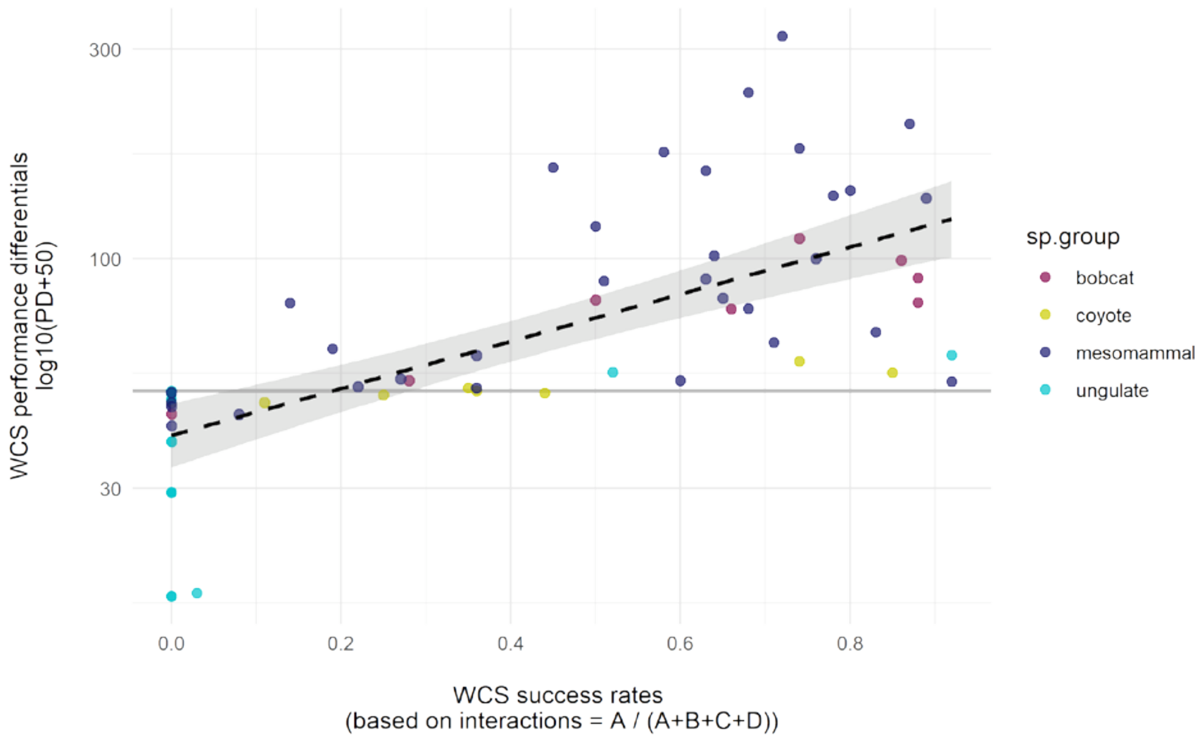

Figure 28. Performance differentials (PD; based on actual crossings versus expected crossing frequencies from habitat reference sites) plotted over WCS success rates (based on % WCS crossings over the total number of WCS interactions) across eight species (bobcat, coyote, javelina, white-tailed deer, northern raccoon, nine-banded armadillo, striped skunk, and Virginia opossum) and eight wildlife crossing structures on the FM106 mitigation corridor in Cameron County, TX. PD scores were significantly correlated with WCS crossing rates ($p < 0.00001$, $R^2 = 0.48$).

REFERENCES

Adams, L. W., and A. D. Geis. 1983. Effects of roads on small mammals. Journal of Applied Ecology:403-415.

Administration, F. H. 2020. Public Road Mileage, Lane-Miles, and VMT 1900 - 2018. *in*.

Anderson, M., R. N. Gorley, and K. R. Clarke. 2008. PERMANOVA+ for PRIMER user manual. Primer-E Ltd, Plymouth, United Kingdom 218.

Anderson, M. J. 2014. Permutational multivariate analysis of variance (PERMANOVA). Wiley statsref: statistics reference online:1-15.

Andis, A., M. P. Huijser, and L. Broberg. 2017. Performance of arch-style road crossing structures from relative movement rates of large mammals. Frontiers in Ecology and Evolution 5:122.

Ascensão, F., A. Kindel, F. Z. Teixeira, R. Barrientos, M. D'Amico, L. Borda-de-Água, and H. M. Pereira. 2019. Beware that the lack of wildlife mortality records can mask a serious impact of linear infrastructures. Global Ecology and Conservation 19:e00661.

Azedo, R., A. Ilhéu, S. Santos, and P. G. Vaz. 2022. Carnivores cross irrigation canals more often through overpasses than through culverts. Basic and Applied Ecology 61:53-67.

Barrientos, R., and L. Bolonio. 2009. The presence of rabbits adjacent to roads increases polecat road mortality. Biodiversity and Conservation 18:405-418.

Barrueto, M., A. T. Ford, and A. P. Clevenger. 2014. Anthropogenic effects on activity patterns of wildlife at crossing structures. Ecosphere 5:1-19.

Barton, K. 2012. MuMIn: Multi-model inference. R package version 1.7. 2. http://CRAN. R-project. org/package= MuMIn.

Benítez-López, A., R. Alkemade, and P. A. Verweij. 2010. The impacts of roads and other infrastructure on mammal and bird populations: a meta-analysis. Biological conservation 143:1307-1316.

Beyer, H. L., E. Gurarie, L. Börger, M. Panzacchi, M. Basille, I. Herfindal, B. Van Moorter, S. R. Lele, and J. Matthiopoulos. 2016. 'You shall not pass!': quantifying barrier permeability and proximity avoidance by animals. Journal of Animal Ecology 85:43-53.

Bigelow, D., and A. Borchers. 2017. Major uses of land in the United States, 2012.

Blackburn, A., C. J. Anderson, A. M. Veals, M. E. Tewes, D. B. Wester, J. H. Young, R. W. DeYoung, and H. L. Perotto-Baldivieso. 2021. Landscape patterns of ocelot–vehicle collision sites. Landscape Ecology 36:497-511.

Bradley, L. C., and D. B. Fagre. 1988. Coyote and bobcat responses to integrated ranch management practices in south Texas. Rangeland Ecology & Management/Journal of Range Management Archives 41:322-327.

Breusch, T. S., and A. R. Pagan. 1979. A simple test for heteroscedasticity and random coefficient variation. Econometrica: Journal of the econometric society:1287-1294.

Brown, J. S., J. W. Laundré, and M. Gurung. 1999. The ecology of fear: optimal foraging, game theory, and trophic interactions. Journal of Mammalogy 80:385-399.

Brunen, B., C. Daguet, and J. A. Jaeger. 2020. What attributes are relevant for drainage culverts to serve as efficient road crossing structures for mammals? Journal of Environmental Management 268:110423.

Burton, A. C., E. Neilson, D. Moreira, A. Ladle, R. Steenweg, J. T. Fisher, E. Bayne, and S. Boutin. 2015. Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. Journal of Applied Ecology 52:675-685.

Cardillo, M., A. Purvis, W. Sechrest, J. L. Gittleman, J. Bielby, G. M. Mace, and C. Moritz. 2004. Human population density and extinction risk in the world's carnivores. PLoS biology 2:e197.

Clarke, G. P., P. C. White, and S. Harris. 1998. Effects of roads on badger Meles meles populations in south-west England. Biological conservation 86:117-124.

Clarke, K., and R. Gorley. 2001. PRIMER user manual: Plymouth routines in multivariate ecological research. Plymouth Marine Laboratory, Plymouth, UK.

Clevenger, A. P., and M. P. Huijser. 2011. Wildlife crossing structure handbook: design and evaluation in North America. United States. Federal Highway Administration. Central Federal Lands Highway ….

Clevenger, A. P., and N. Waltho. 2000. Factors influencing the effectiveness of wildlife underpasses in Banff National Park, Alberta, Canada. Conservation biology 14:47-56.

_____. 2003. Long-term, year-round monitoring of wildlife crossing structures and the importance of temporal and spatial variability in performance studies.

_____. 2005. Performance indices to identify attributes of highway crossing structures facilitating movement of large mammals. Biological conservation 121:453-464.

Craveiro, J., J. Bernardino, A. Mira, and P. G. Vaz. 2019. Impact of culvert flooding on carnivore crossings. Journal of Environmental Management 231:878-885.

Crooks, K. R., and M. E. Soulé. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400:563-566.

De Molenaar, J. G., M. E. Sanders, and D. A. Jonkers. 2006. Road lighting and grassland birds: local influence of road lighting on a black-tailed godwit population. Ecological consequences of artificial night lighting:114-138.

Denneboom, D., A. Bar-Massada, and A. Shwartz. 2021. Factors affecting usage of crossing structures by wildlife–A systematic review and meta-analysis. Science of the Total Environment 777:146061.

Dickson, B. G., J. S. Jenness, and P. Beier. 2005. Influence of vegetation, topography, and roads on cougar movement in southern California. The Journal of Wildlife Management 69:264-276.

Ditmer, M. A., D. C. Stoner, C. D. Francis, J. R. Barber, J. D. Forester, D. M. Choate, K. E. Ironside, K. M. Longshore, K. R. Hersey, and R. T. Larsen. 2021. Artificial nightlight alters the predator–prey dynamics of an apex carnivore. Ecography 44:149-161.

Donaldson, B. M., and K. E. Elliott. 2021. Enhancing existing isolated underpasses with fencing reduces wildlife crashes and connects habitat. Human–Wildlife Interactions 15:20.

Dupuis-Desormeaux, M., Z. Davidson, M. Mwololo, E. Kisio, S. Taylor, and S. E. MacDonald. 2015. Testing the prey-trap hypothesis at two wildlife conservancies in Kenya. PLOS ONE 10:e0139537.

EPA. 2013. Level III and IV ecoregions of the continental United States. *in* U.S. EPA, U.S. Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, Corvallis, Oregon.

ESRI. 2021. Environmental Systems Research Institute, Redlands, CA.

Fahrig, L. 2007. Non-optimal animal movement in human-altered landscapes. Functional ecology 21:1003-1015.

Fahrig, L., and T. Rytwinski. 2009. Effects of roads on animal abundance: an empirical review and synthesis. Ecology and Society 14.

FHA. 2020. Public Road Mileage, Lane-Miles, and VMT 1900 - 2018. *in* Federal Highway Administration.

Ford, A. T., and A. P. Clevenger. 2010. Validity of the prey-trap hypothesis for carnivore-ungulate interactions at wildlife-crossing structures. Conservation biology 24:1679-1685.

Forman, R. T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation biology 14:31-35.

Forman, R. T., and L. E. Alexander. 1998. Roads and their major ecological effects. Annual review of ecology and systematics 29:207-231.

Forman, R. T., and R. D. Deblinger. 2000. The ecological road-effect zone of a Massachusetts (USA) suburban highway. Conservation biology 14:36-46.

Fox, J., S. Weisberg, D. Adler, D. Bates, G. Baud-Bovy, S. Ellison, D. Firth, M. Friendly, G. Gorjanc, and S. Graves. 2012. Package ‘car’. Vienna: R Foundation for Statistical Computing 16.

Gagnon, J. W., N. L. Dodd, K. S. Ogren, and R. E. Schweinsburg. 2011. Factors associated with use of wildlife underpasses and importance of long-term monitoring. The Journal of Wildlife Management 75:1477-1487.

Gaynor, K. M., P. S. Branco, R. A. Long, D. D. Gonçalves, P. K. Granli, and J. H. Poole. 2018a. Effects of human settlement and roads on diel activity patterns of elephants (Loxodonta africana). African Journal of Ecology 56:872-881.

Gaynor, K. M., C. E. Hojnowski, N. H. Carter, and J. S. Brashares. 2018b. The influence of human disturbance on wildlife nocturnality. Science 360:1232-1235.

Grigione, M., K. Menke, C. López-González, R. List, A. Banda, J. Carrera, R. Carrera, A. Giordano, J. Morrison, and M. Sternberg. 2009. Identifying potential conservation areas for felids in the USA and Mexico: integrating reliable knowledge across an international border. Oryx 43:78-86.

Grigione, M., A. Scoville, G. Scoville, and K. Crooks. 2007. Neotropical cats in southeast Arizona and surrounding areas: past and present status of jaguars, ocelots and jaguarundis. Mastozoología Neotropical 14:189-199.

Haines, A. M., J. E. Janecka, M. E. Tewes, L. I. Grassman Jr, and P. Morton. 2006a. The importance of private lands for ocelot Leopardus pardalis conservation in the United States. Oryx 40:90-94.

Haines, A. M., M. E. Tewes, and L. L. Laack. 2005. Survival and sources of mortality in ocelots. The Journal of wildlife management 69:255-263.

Haines, A. M., M. E. Tewes, L. L. Laack, J. S. Horne, and J. H. Young. 2006b. A habitat-based population viability analysis for ocelots (Leopardus pardalis) in the United States. Biological conservation 132:424-436.

Hansen, M. J., and A. P. Clevenger. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological conservation 125:249-259.

Harris, G., R. Thompson, J. L. Childs, and J. G. Sanderson. 2010. Automatic storage and analysis of camera trap data. Bulletin of the Ecological Society of America 91:352-360.

Harveson, P. M., M. E. Tewes, G. L. Anderson, and L. L. Laack. 2004. Habitat use by ocelots in south Texas: implications for restoration. Wildlife Society Bulletin 32:948-954.

Hopkins, T. M. 2020. Influence of Landscape Factors on Wildlife Presence and Road Mitigation Structure Performance. Master of Science, The University of Texas Rio Grande Valley, ProQuest.

Horne, J. S., A. M. Haines, M. E. Tewes, and L. L. Laack. 2009. Habitat partitioning by sympatric ocelots and bobcats: implications for recovery of ocelots in southern Texas. The Southwestern Naturalist:119-126.

Jachowski, D. S., A. Butler, R. Y. Eng, L. Gigliotti, S. Harris, and A. Williams. 2020. Identifying mesopredator release in multi-predator systems: a review of evidence from North America. Mammal Review 50:367-381.

Jacobson, S. L., L. L. Bliss-Ketchum, C. E. de Rivera, and W. P. Smith. 2016. A behavior-based framework for assessing barrier effects to wildlife from vehicle traffic volume. Ecosphere 7:e01345.

Jaeger, J. A., J. Bowman, J. Brennan, L. Fahrig, D. Bert, J. Bouchard, N. Charbonneau, K. Frank, B. Gruber, and K. T. Von Toschanowitz. 2005. Predicting when animal populations are at risk from roads: an interactive model of road avoidance behavior. Ecological Modelling 185:329-348.

Jaeger, J. A., and L. Fahrig. 2004. Effects of road fencing on population persistence. Conservation biology 18:1651-1657.

Jahrsdoerfer, S. E., and D. M. Leslie. 1988. Tamaulipan brushland of the Lower Rio Grande Valley of south Texas: description, human impacts, and management options. US Department of the Interior, Fish and Wildlife Service.

Jakes, A. F., P. F. Jones, L. C. Paige, R. G. Seidler, and M. P. Huijser. 2018. A fence runs through it: A call for greater attention to the influence of fences on wildlife and ecosystems. Biological conservation 227:310-318.

Janečka, J., M. Tewes, L. Laack, A. Caso, J. Grassman, LI, A. Haines, D. Shindle, B. Davis, W. Murphy, and R. Honeycutt. 2011. Reduced genetic diversity and isolation of remnant ocelot populations occupying a severely fragmented landscape in southern Texas. Animal Conservation 14:608-619.

Janečka, J. E., M. Tewes, L. Laack, L. Grassman, A. Haines, and R. Honeycutt. 2008. Small effective population sizes of two remnant ocelot populations (Leopardus pardalis albescens) in the United States. Conservation genetics 9:869.

Jumeau, J., L. Petrod, and Y. Handrich. 2017. A comparison of camera trap and permanent recording video camera efficiency in wildlife underpasses. Ecology and evolution 7:7399-7407.

Kautz, T. M., N. L. Fowler, T. R. Petroelje, D. E. Beyer, N. J. Svoboda, and J. L. Belant. 2021. Large carnivore response to human road use suggests a landscape of coexistence. Global Ecology and Conservation 30:e01772.

Kays, R., S. Tilak, B. Kranstauber, P. A. Jansen, C. Carbone, M. J. Rowcliffe, T. Fountain, J. Eggert, and Z. He. 2010. Monitoring wild animal communities with arrays of motion sensitive camera traps. arXiv preprint arXiv:1009.5718.

Kim, H.-Y. 2015. Statistical notes for clinical researchers: post-hoc multiple comparisons. Restorative dentistry & endodontics 40:172-176.

Kline, R. J., M. Picillo, C. K. Brett, A. R. Mehner, and V. A. Hanley. 2022. Post-construction Monitoring Annual Report for FM 106: January 2021 thru January 2022. University of Texas Rio Grande Valley.

Kohn, B. E., R. Thiel, and J. L. Hansen. 2001. Road density as a factor in habitat selection by wolves and other carnivores in the Great Lakes Region. Carnivore Conservation in the Twenty-first Century 97:110.

Laack, L. L. 1991. Ecology of the ocelot (Felis pardalis) in south Texas. Texas A&I University.

Lehnen, S. E., M. A. Sternberg, H. M. Swarts, and S. E. Sesnie. 2021. Evaluating population connectivity and targeting conservation action for an endangered cat. Ecosphere 12:e03367.

Leonard, J. P., M. E. Tewes, J. V. Lombardi, D. W. Wester, and T. A. Campbell. 2020. Effects of sun angle, lunar illumination, and diurnal temperature on temporal movement rates of sympatric ocelots and bobcats in South Texas. PLOS ONE 15:e0231732.

Lewis, J. S., J. L. Rachlow, J. S. Horne, E. O. Garton, W. L. Wakkinen, J. Hayden, and P. Zager. 2011. Identifying habitat characteristics to predict highway crossing areas for black bears within a human-modified landscape. Landscape and Urban Planning 101:99-107.

Little, S. J., R. G. Harcourt, and A. P. Clevenger. 2002. Do wildlife passages act as prey-traps? Biological conservation 107:135-145.

Lombardi, J. V., D. I. MacKenzie, M. E. Tewes, H. L. Perotto-Baldivieso, J. M. Mata, and T. A. Campbell. 2020a. Co-occurrence of bobcats, coyotes, and ocelots in Texas. Ecology and Evolution 10:4903-4917.

Lombardi, J. V., M. E. Tewes, H. L. Perotto-Baldivieso, J. M. Mata, and T. A. Campbell. 2020b. Spatial structure of woody cover affects habitat use patterns of ocelots in Texas. Mammal Research 65:555-563.

MacNulty, D. R., L. D. Mech, and D. W. Smith. 2007. A proposed ethogram of large-carnivore predatory behavior, exemplified by the wolf. Journal of Mammalogy 88:595-605.

Marcus Rowcliffe, J., C. Carbone, P. A. Jansen, R. Kays, and B. Kranstauber. 2011. Quantifying the sensitivity of camera traps: an adapted distance sampling approach. Methods in Ecology and Evolution 2:464-476.

Mata, C., R. Bencini, B. K. Chambers, and J. E. Malo. 2015. Predator-prey interactions at wildlife crossing structures: between myth and reality. Handbook of Road Ecology. 1st Ed. Wiley, Oxford:190-197.

Mata, C., P. Ruiz-Capillas, and J. E. Malo. 2017. Small-scale alterations in carnivore activity patterns close to motorways. European Journal of Wildlife Research 63:1-12.

McClure, C. J., H. E. Ware, J. Carlisle, G. Kaltenecker, and J. R. Barber. 2013. An experimental investigation into the effects of traffic noise on distributions of birds: avoiding the phantom road. Proceedings of the Royal Society B: Biological Sciences 280:20132290.

McGregor, R. L., D. J. Bender, and L. Fahrig. 2008. Do small mammals avoid roads because of the traffic? Journal of Applied Ecology 45:117-123.

McIntosh, L., A. Fierro-Cabo, and J. A. Benavides. 2019. Macroinvertebrate assemblages from two sampling methods similarly discriminated freshwater wetlands with different ecosystem status in south Texas. Aquatic Ecosystem Health & Management 22:65-76.

Meek, P., G. Ballard, A. Claridge, R. Kays, K. Moseby, T. O’brien, A. O’connell, J. Sanderson, D. Swann, and M. Tobler. 2014. Recommended guiding principles for reporting on camera trapping research. Biodiversity and conservation 23:2321-2343.

Meek, P. D., P. Fleming, and G. Ballard. 2012. An introduction to camera trapping for wildlife surveys in Australia. Invasive Animals Cooperative Research Centre Canberra, Australia.

Meredith, M. 2021. overlap: Estimates of Coefficient of Overlapping for Animal Activity Patterns. *in* M. Ridout, editor.

Morelli, F., M. Beim, L. Jerzak, D. Jones, and P. Tryjanowski. 2014. Can roads, railways and related structures have positive effects on birds?–A review. Transportation Research Part D: Transport and Environment 30:21-31.

Murray, J. L., and G. L. Gardner. 1997. Leopardus pardalis. Mammalian species:1-10.

NOAA. 2021. Climate at a Glance: County Time Series. *in* National Centers for Environmental information.

Parsons, A. W., T. Forrester, W. J. McShea, M. C. Baker-Whatton, J. J. Millspaugh, and R. Kays. 2017. Do occupancy or detection rates from camera traps reflect deer density? Journal of Mammalogy 98:1547-1557.

Paviolo, A., Crawshaw, P., Caso, A., de Oliveira, T., Lopez-Gonzalez, C.A., Kelly, M., DeAngelo, C. & Payan, E. . 2015. Leopardus pardalis.

Poole, M. A., and P. N. O'Farrell. 1971. The assumptions of the linear regression model. Transactions of the Institute of British Geographers:145-158.

Prange, S., S. D. Gehrt, and E. P. Wiggers. 2004. Influences of anthropogenic resources on raccoon (Procyon lotor) movements and spatial distribution. Journal of Mammalogy 85:483-490.

Prugh, L. R., C. J. Stoner, C. W. Epps, W. T. Bean, W. J. Ripple, A. S. Laliberte, and J. S. Brashares. 2009. The rise of the mesopredator. BioScience 59:779-791.

ReconyxInc. 2018. Reconyx Hyperfire2 Outdoor Series Camera User Manual. *in*.

Reed, D., T. N. Woodard, and T. D. Beck. 1979. Regional deer-vehicle accident research.

Reijnen, R., R. Foppen, C. T. Braak, and J. Thissen. 1995. The effects of car traffic on breeding bird populations in woodland. III. Reduction of density in relation to the proximity of main roads. Journal of Applied Ecology:187-202.

Richardson, A. K., Ken. 2011. Plants of Deep South Texas. First edition. Texas A&M University Press, College Station, TX.

Ridout, M. S., and M. Linkie. 2009. Estimating overlap of daily activity patterns from camera trap data. Journal of Agricultural, Biological, and Environmental Statistics 14:322-337.

Riitters, K. H., and J. D. Wickham. 2003. How far to the nearest road? Frontiers in Ecology and the Environment 1:125-129.

Rivera-Roy, A. D. 2020. Wildlife Use of Road Mitigation Structures in Relation to Their Construction, Structural Characteristics, and Environmental Factors along a South Texas Highway. Master of Science, The University of Texas Rio Grande Valley, ProQuest.

Rodriguez, J. T., D. B. Lesmeister, and T. Levi. 2021. Mesocarnivore landscape use along a gradient of urban, rural, and forest cover. PeerJ 9:e11083.

Rowcliffe, M. 2016.

Ruiz-Capillas, P., C. Mata, B. Fernández, C. Fernandes, and J. E. Malo. 2021. Do Roads Alter the Trophic Behavior of the Mesocarnivore Community Living Close to Them? Diversity 13:173.

Sanderson, J., and G. Harris. 2013. Automatic data organization, storage, and analysis of camera trap pictures. Journal of Indonesian Natural History 1:11-19.

Sanjar, A. unpublished work. Personal communication; concurrent research documenting rodent communities on FM106. *in*.

Schmidt, G. M., R. L. Lewison, and H. M. Swarts. 2020. Identifying landscape predictors of ocelot road mortality. Landscape Ecology 35:1651-1666.

Shamoon, H., R. Maor, D. Saltz, and T. Dayan. 2018. Increased mammal nocturnality in agricultural landscapes results in fragmentation due to cascading effects. Biological conservation 226:32-41.

Shapiro, S. S., and M. B. Wilk. 1965. An analysis of variance test for normality (complete samples). Biometrika 52:591-611.

Shindle, D. B., and M. E. Tewes. 1998. Woody species composition of habitats used by ocelots (Leopardus pardalis) in the Tamaulipan Biotic Province. The Southwestern Naturalist:273-279.

Silva, C., M. P. Simões, A. Mira, and S. M. Santos. 2019. Factors influencing predator roadkills: The availability of prey in road verges. Journal of Environmental Management 247:644-650.

Silvy, N. J. 2020. The Wildlife Techniques Manual: Volume 1: Research. Volume 2: Management. Volume 1.JHU Press.

Smith, D. J., and R. van der Ree. 2015. Field methods to evaluate the impacts of roads on wildlife. Handbook of road ecology:82-95.

Smith, W. P. 1991. Odocoileus virginianus. Mammalian species:1-13.

Sollmann, R., A. Mohamed, H. Samejima, and A. Wilting. 2013. Risky business or simple solution–Relative abundance indices from camera-trapping. Biological conservation 159:405-412.

Soulé, M. E., D. T. Bolger, A. C. Alberts, J. Wrights, M. Sorice, and S. Hill. 1988. Reconstructed dynamics of rapid extinctions of chaparral-requiring birds in urban habitat islands. Conservation biology 2:75-92.

Team, R. C. 2021. R Studio. *in* R Foundation for Statistical Computing, Vienna, Austria.

Terborgh, J., L. Lopez, P. Nuñez, M. Rao, G. Shahabuddin, G. Orihuela, M. Riveros, R. Ascanio, G. H. Adler, and T. D. Lambert. 2001. Ecological meltdown in predator-free forest fragments. Science 294:1923-1926.

Tremblay, T. A., W. A. White, and J. A. Raney. 2005. Native woodland loss during the mid 1900s in Cameron County, Texas. The Southwestern Naturalist 50:479-482.

Tucker, M. A., K. Böhning-Gaese, W. F. Fagan, J. M. Fryxell, B. Van Moorter, S. C. Alberts, A. H. Ali, A. M. Allen, N. Attias, and T. Avgar. 2018. Moving in the Anthropocene: Global reductions in terrestrial mammalian movements. Science 359:466-469.

Tukey, J. W. 1977. Exploratory data analysis. Volume 2.Reading, MA.

TxDOT. 2022. Highway Designation File - FARM TO MARKET ROAD NO. 106. *in*.

USFWS. 1982. Endangered and Threatened Wildlife and Plants; Endangered Status for U.S. Population of the Ocelot. Federal Register.

van der Grift, E. A., R. van der Ree, L. Fahrig, S. Findlay, J. Houlahan, J. A. Jaeger, N. Klar, L. F. Madrinan, and L. Olson. 2013. Evaluating the effectiveness of road mitigation measures. Biodiversity and Conservation 22:425-448.

van der Grift, E. A., R. van der Ree, and J. A. Jaeger. 2015. Guidelines for evaluating the effectiveness of road mitigation measures. Handbook of road ecology:129-137.

van der Ree, R., D. J. Smith, and C. Grilo. 2015. Handbook of road ecology. John Wiley & Sons.

Wang, Y., M. L. Allen, and C. C. Wilmers. 2015. Mesopredator spatial and temporal responses to large predators and human development in the Santa Cruz Mountains of California. Biological conservation 190:23-33.

Watabe, R., and M. U. Saito. 2021. Effects of vehicle-passing frequency on forest roads on the activity patterns of carnivores. Landscape and Ecological Engineering 17:225-231.

Welbourne, D. J., A. W. Claridge, D. J. Paull, and A. Lambert. 2016. How do passive infrared triggered camera traps operate and why does it matter? Breaking down common misconceptions. Remote Sensing in Ecology and Conservation 2:77-83.

Yamashita, T. unpublished work. *in*.

Yamashita, T. J. 2020. The influence of human disturbance on wildlife use of a highway in South Texas. The University of Texas Rio Grande Valley, ProQuest.

BIOGRAPHICAL SKETCH

Caitlin K. Brett grew up in the Lake Erie snowbelt town of Leroy, Ohio with her parents, three brothers, and younger sister. In 2011, she left the northeast to attend The Ohio State University where she studied and graduated with a B.S. in Landscape Architecture from the College of Engineering in 2016. She worked with two architecture, planning, and urban design firms in Columbus, Ohio for a combined four years before realizing that anthropocentric design was not what interested her in the study of the environment. Inspired to investigate and better advocate for the role of wildlife – particularly endangered species – in large scale landscape planning and conservation, she moved to New Mexico for a field season surveying for Mexican spotted owls (*Strix occidentalis lucida*) in the Carson and Santa Fe National Forests. She went on to gain experience working with various organic farms in West Virginia and assisting with wildlife biology research in agricultural landscapes at North Dakota State University. In the summer of 2020, she moved to Brownsville, Texas to study the impacts of habitat fragmentation and road mitigation structures on the endangered ocelot (*Leopardus pardalis*) and the associated South Texas wildlife community. In 2022, she graduated with a master’s degree in Agricultural, Environmental, and Sustainability Sciences from the University of Texas Rio Grande Valley. Caitlin can be contacted by email at ckbrett07@gmail.com.