

INFLUENCE OF VEGETATIVE CHARACTERISTICS ON PREDATION AND  
PREDATOR ASSEMBLAGE OF BIRD NESTS

A Thesis

by

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## ABSTRACT

Predation is the primary cause of nesting failure in birds. Thus, understanding the drivers behind nest success is of paramount concern to land managers. In the southern United States, woody vegetation encroachment has altered vegetation structure available for nesting bird species, potentially impacting underlying predator assemblages and predator-prey dynamics in this region. To inform future management of grassland birds, I examined how vegetative characteristics drive predation and predator assemblage of bird nests in semiarid grasslands affected by woody encroachment. In 2015 and 2016, I monitored bird nests within four vegetation types occurring on the San Antonio Viejo Ranch of the East Foundation in South Texas. I also placed infrared cameras at a subset of nests to identify predators. I then collected vegetation measurements at each nest site, including metrics such as percent woody cover, concealment, and distance to the closest anthropogenic or natural edge. I also sampled potential predator activity within the four vegetation types using track plates and data from a concurrent, multi-species monitoring program examining small mammals and herpetofauna.

I monitored 258 nests of 18 bird species and deployed 109 infrared cameras at nest sites. I then divided bird species into four nesting groups to control for variation due to nesting strategy. All nesting strategy groups exhibited similar percentages of success (~25%) in 2015 and 2016 with exception of exposed nests that decreased by ~50% between years. Woody cover at the nest site was a significant driver of nest success of my medium cup nest group, indicating that small increases in woody cover (~10%) could have substantial impacts on birds utilizing this nesting strategy. Snakes were my

primary predator at camera monitored nests (57%) and were also driven by increased woody cover (~15%) at the nest site. Meso- and large mammalian predators were most active in vegetation types dominated by herbaceous cover, small mammals were most active in vegetation types dominated by woody cover, and herpetofauna activity was highly variable. Predator activity did not reflect predator identity at camera monitored nests, indicating activity may not be a valuable metric for quantifying predation risk.

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## INTRODUCTION

Prior to European settlement, 400,000,000 ha of land in the contiguous United States was composed of native tall-, mid-, and shortgrass prairies (Osterheld et al. 1999, Conner et al. 2001). Agricultural industrialization and urban development between 1850 and 1990 resulted in widespread degradation and destruction of these vegetation types, which now totals >98% some regions of North America (Samson and Knopf 1994, Noss et al. 1995, Conner et al. 2001). Grassland conversion resulted in concurrent loss of breeding and foraging areas for some grassland-dependent wildlife (Brennan and Kuvlesky 2005). Population declines are particularly well-documented for birds (Pietz and Granfors 2000, Brennan and Kuvlesky 2005, Stanley 2010) and studies suggest that some species' populations have declined >70% percent in recent decades (Peterjohn and Sauer 1999). Direct loss of habitat is the primary driver of avian population declines. However, fragmentation and degradation of existing native grassland has also had a negative effect on native grassland bird populations through changes in the structure and composition available to birds for nesting (Greenfield et al. 2002) and alterations to predator-prey dynamics (Feeley and Terborgh 2008).

In the southern United States, woody vegetation such as honey mesquite (*Prosopis glandulosa*) has encroached into 38 million ha of what was formerly classified as semiarid savanna (Van Auken 2009), further degrading remnant patches of native grassland (Samson and Knopf 1994). In regions of south Texas, encroachment has become a recent issue (Archer et al. 1995), transforming historical savannas into subtropical thornscrubs consisting of shrub tickets and small trees (McMahan et al.

1984, McLendon 1991). Establishment and encroachment of woody vegetation has been aggressive, and resulted from the synergistic effects of intensive grazing, fire suppression, and increases in atmospheric carbon dioxide levels (Archer et al. 1995). During invasions, woody vegetation quickly crowds out native grasses and forbs, and causes dense thickets (Archer 1995), increased bare ground (McCoy et al. 2001), and diminished quality of vegetation available for nesting birds (Greenfield et al. 2002).

Numerous studies indicate the effects of woody encroachment on bird nesting success and abundance; however, results suggest birds exhibit species-specific responses to changes in vegetation. Keyel et al. (2012) suggested Bobolink (*Dolichonyx oryzivorus*), a ground nesting grassland obligate bird, occupancy and density were negatively correlated to woody cover in fields in central Massachusetts. Alternatively, Winter et al. (2006) suggested density and nest success of Clay-colored Sparrow (*Spizella pallida*), an open cup nesting bird, was positively correlated to woody cover, with nest success rates 1.5 times higher in wooded versus unwooded areas of South Dakota and Minnesota. Additionally, in a grassland bird study in Texas, nest success of Lark Sparrows (*Chondestes grammacus*), another open cup nesting bird, was higher at sites treated more frequently with prescribed burns (Long et al. 2012), suggesting shrub cover may have negative impacts on nest success.

In addition to the direct effects of woody vegetation on nest success, indirect effects on concealment and fragmentation can have substantial effects on birds. Martin and Roper (1988) found that increased woody vegetation increased visual obstruction of nests Hermit Thrush (*Catharus guttatus*) nests in Arizona, thus increasing nest success in

areas with greater cover. In a study of birds nesting in farmlands, nest success responded positively to increased concealment; however, this may have been due changes in parental behavior (Weidinger 2002) because nest defense behaviors are generally negatively correlated with nest concealment (Götmark et al. 1995; Cresswell 1997). Research also indicates concealment has no effect on nesting success of various songbirds (Howlett and Stutchbury 1996, Conkling et al. 2012, Murray and Best 2014).

Fragmentation causes discontinuity of vegetation, decreased grassland patch size, and exposure to edge-dependent predators (Lahti 2001). Batary and Baldi (2003) suggested that proximity to vegetation edges was positively correlated to predation rate of bird nests. Alternatively, a meta-analysis conducted by Lahti (2001) showed no relationship between edge and predation rate. These studies provide evidence that different bird species will respond differently to ecological conditions and the indirect effects of woody vegetation, concealment, and fragmentation on grassland birds needs further study.

Because the nesting phase is arguably the most important stage of the avian life cycle (Donovan and Thompson 2001), understanding the drivers behind nest success and how they are affected by woody vegetation is of paramount importance to land managers. Researchers agree that predation is the primary cause of nesting failure in birds (Martin 1993, Thompson 2007), accounting for up to 80% of seasonal nest failure for some species (Staller et al. 2005, Rahmig et al. 2009, Thompson et al. 2012). Many studies have identified the ecological conditions that influence avian nest success (as detailed above); however, few concurrently examine the ecological conditions that

influence underlying predator assemblages or identify how predator-prey dynamics are influenced by the ecological conditions the species inhabit. Such information will help inform more effective management strategies for birds of conservation concern.

Avian nest predators are diverse and include mammals, snakes, other birds, and invertebrates such as ants. The magnitude of effect that each predator group has on avian nest success is influenced by the nesting strategies of the bird species of interest (Batary and Baldi 2003, Bollinger and Gavin 2004, Renfrew et al. 2005, Borgmann and Conway 2015), predator abundance and distribution (Ribic et al. 2012), predator behavior (DeGregorio et al. 2015), environmental conditions during the breeding season (e.g., precipitation, temperature; Blouin-Demers and Weatherhead 2001, Murray and Smith 2012), and the vegetation characteristics that the predator and prey species inhabit (Klug et al. 2010, Cox et al. 2012), among others.

Mammalian abundance and distribution is influenced by factors such as vegetation and as such, the potential influence of mammalian predation on avian nests varies. Mesomammals such as coyotes (*Canis latrans*) and American badgers (*Taxidea taxus*) are medium-sized, opportunistic, generalist foragers (Schmidly 1994) that rely on both visual and olfactory cues to locate prey (Whelan et al. 1994). Thus, mesomammals will likely have varying effects on nest success, depending on local-scale vegetative characteristics and seasonality and availability of alternate food sources. For example, nest predation by coyotes and badgers accounted for almost 50% of Northern Bobwhite (*Colinus virginianus*; a medium-sized, ground nesting bird) nest predations in south Texas savannas (Rader et al. 2006), but accounted for only 1% of Northern Bobwhite

nest failure in Florida coastal plains (Staller et al. 2005). Similarly, nest predation research on Black-capped Vireos (*Vireo atricapilla*), a small open cup nesting songbird that breeds in early successional shrublands, indicated that coyotes and other meso- and large mammalian predators accounted for a smaller percentage of nest predation events in Oak-Juniper (*Quercus-Juniperus*) woodlands when compared to more arid study sites within their breeding range (Conkling et al. 2012). Conversely, Conner et al. (2010) concluded that mesomammals had little or no effect on nesting success of shrub nesting birds in Georgia. The above research illustrates that the potential impact of meso- and large mammal predation on bird nests can be substantial, but their influence is varied, context-specific, and requires further study (Heske et al. 2001).

In addition to meso- and large mammals, small mammal such as mice, rats, and ground squirrels could also influence nest success to various degrees. Mice such as the deer mouse (*Peromyscus* spp.) prefer increased vegetation structure and grassland patch edges for foraging (Stanley 2010). Similarly, White-footed mice (*P. leucopus*) in a Kansas tallgrass prairie study respond positively to increases in woody vegetation (Matlack et al. 2008). Ribic et al. (2012) found ground squirrels (*Spermophilus* spp.), the primary predator in their study, were more active along wooded edges of agricultural fields in Iowa. Therefore, small mammal nest predators such as mice, rats, and ground squirrels may have a greater influence on avian nest success when birds place their nests close to shrubs or trees (Cocimano et al. 2011). Alternatively, bird nests located in open fields with little to no cover could face less risk of small mammal predation (Cain et al. 2006).



Research indicates that snake predators have a substantial influence on bird nest success. Snakes often aggregate in woody vegetation for cover and thermoregulation (Blouin-Demers and Weatherhead 2001). Woody vegetation in grasslands also provides important resources for snakes (e.g. prey and cover), which can result in increased predation rates of bird nests in close proximity to shrubs, trees, or patch edges (Blouin-Demers and Weatherhead 2001, Bensen et al. 2010). A study conducted in California suggested that snakes accounted for up to 90% of nest predations (Morrison and Bolger 2002). In both rural fields of Missouri and pine forests of Georgia, snake predations are double or triple that of other predators, which is likely influenced by the vegetation found therein (Thompson and Burhans 2003, Conner et al. 2010). Additionally, in tallgrass prairies, grassland birds that have large spatial overlap with snakes exhibit higher predation rates than grassland bird species that do not (Klug et al. 2010). Birds nesting close to forest edges and shrubby patches in grassland ecosystems are also particularly vulnerable to nest predation (DeGregorio et al 2014). Therefore, woody vegetation is likely a critical component when predicting snake predation.

Avian predators also influence nesting success of other birds. As visual predators, it is important for avian predators such as diurnal raptors, Corvids, and Brown-headed Cowbirds (*Molothrus ater*) to have a view of prey (Benson et al. 2010, Conkling et al. 2012). For example, Conner et al. (2010) found higher rates of predation during the nestling phase were the result of increased activity by nestlings and foraging trips to and from the nest by adult birds. The strong visual component to avian predators' life history strategies indicates the need for lower occurrences of visual barriers

obstructing views of potential prey. As a result, avian predators are expected to have a more substantial impact on birds nesting in open areas. The presence of trees or other woody vegetation; however, can also provide perching locations, providing avian predators with foraging vantage points (Chalfoun et al. 2002, Benson et al. 2010), indicating proximity to wooded edges may increase nest predation.

Native and invasive fire ants (*Solenopsis* spp.) can also pose a threat to nesting birds. A study of Northern Bobwhite on south Texas ranchlands indicated native southern fire ants (*S. xyloni*) accounted for 15% of nest depredations (Rader et al. 2006). Imported red fire ants (*S. invicta*) can affect nest mortality similarly with predation rates up to 33% in some shrub nesting birds (Conner et al. 2010). Excluding imported fire ants from Black-capped Vireo and White-eyed Vireo (*Vireo griseus*) nests increased nest success by 6% and 21%, respectively (Campomizzi et al. 2009). Similar to avian predators, fire ants also depredate nests at the nestling phase (Conner et al. 2010).

As described above, the influences of grassland declines and subsequent woody vegetation encroachment on bird nest success is well studied. However, the underlying drivers of nest success (e.g. predation) and how they relate to ecological characteristics in semiarid savannas remain understudied. Additionally, researchers examining nest predation generally focus on a single taxa or functional group, disregarding the diversity of predator assemblages driving nest success. I will examine nest success and nest predators of birds utilizing semiarid savannas. I will also examine activity of predators utilizing these same lands. My research will aid land managers in predicting bird and predator responses prior to implementation of land management decisions.

## OBJECTIVES AND HYPOTHESES

My four research objectives were to: (1) Evaluate the influence of nest-site vegetation characteristics on bird nest success across a continuum of vegetation types in a region historically dominated by semiarid grasslands, (2) identify avian nest predators across these same vegetation types, (3) evaluate the influence of nest-site vegetation characteristics on the probability of predation by mammal, snake, avian, and ant predator types, and (4) examine relationships between predator activity, nest success, and predator assemblage across different vegetation types.

For Objective 1, I assessed the influence of nest site-specific vegetation characteristics on nest success of groups of birds that I categorized based on their nesting strategies. My groups included small cup nesters, medium cup nesters, large cup nesters, and exposed nesters. I describe these groups in detail below. I hypothesized that nest success of both small cup nesters and large cup nesters would increase with decreased woody vegetation cover because bird species included in these groups typically nest in semiarid savannas and could be affected by changes in woody cover (Table 1).

Alternatively, I hypothesized that nest success of medium cup and exposed nesters would increase with increased woody vegetation cover because birds in these groups are generalist nesters and may thrive in areas affected by woody encroachment (Table 1).

Finally, I hypothesized that nest success of all nesting groups was equally influenced by concealment and distance to edge and success would increase with increased concealment and increased distance to edge because increased concealment may obscure

nests from potential predators and increased distance to edge would distance nests from edge dependent predators.

Table 1. Hypothesized avian nest success in relation to woody cover, concealment, and distance to nearest edge for birds that use four different nesting strategies in southern semiarid grasslands.

Nest Type	Nest Success		
	↑ Woody Cover (%)	↑ Concealment (%)	↓ Distance to Edge (m)
Small Cup	↓	↑	↓
Medium Cup	↑	↑	↓
Large Cup	↓	↑	↓
Exposed	↑	↑	↓

For Objective 2, I identified avian nest predators across vegetation types. I hypothesized that meso- and large carnivore species were more prevalent in vegetation types dominated by grasses, rather than woody plants (Table 2). Additionally, I hypothesized that small mammal predators and herpetofauna predators were more prevalent in vegetation types dominated by woody plants (Table 2). Lastly, I hypothesized that avian predators would be more prevalent in vegetation types dominated by grasses (Table 2).

Table 2. Hypothesized avian nest predation frequency by four predator groups in relation to changes in woody plant cover in semiarid grasslands.

Predator	Predation Frequency
	↑ Woody Cover (%)
Meso-/large Mammal	↓
Small Mammal	↑
Herpetofauna	↑
Avian	↓

For Objective 3, I estimated probability of nest predation by predator types. Specifically, I hypothesized that the probability of predation by meso- and large mammalian and avian predators would decrease with increased woody vegetation cover because these predator groups rely on visual cues to find prey, which may be affected by woody cover (Table 3). Additionally, I hypothesized that the probability of predation by snake and small mammalian predators would increase with increased woody vegetation cover because these predators rely on woody vegetation for cover and thermoregulation. Lastly, I hypothesized that all predator groups were equally influenced by nest concealment and distance to edge and predation probability would increase with decreased nest concealment and decreased distance to edge because concealed nests could mask visual and olfactory cues and many of these predators are known to rely on edge for movement and foraging.

Table 3. Hypothesized probability of predation in relation to wood cover, concealment, and distance to nearest edge for four different predator types in southern semiarid grasslands.

Predator	Nest Predation Probability		
	↑ Woody Cover (%)	↑ Concealment (%)	↓ Distance to Edge (m)
Meso-/large Mammal	↓	↓	↑
Small Mammal	↑	↓	↑
Herpetofauna	↑	↓	↑
Avian	↓	↓	↑

For Objective 4, I determined activity of potential predators at the vegetation type scale. I hypothesized that meso- and large carnivore activity as defined by

visits/track station/day would increase in vegetation types dominated by grasses, rather than woody plants (Table 4). Additionally, I hypothesized that small mammal activity (individuals/trap/day) and herpetofauna activity (individuals/trap/day) would increase in vegetation types dominated by woody plants. Lastly, I hypothesized that avian activity (individuals/point/day) would increase in vegetation types dominated by grasses.

Table 4. Hypothesized avian nest predation frequency by four predator groups in relation to changes in woody plant cover in southern semiarid grasslands.

Predator	Predator Activity
	↑ Woody Cover (%)
Meso-/large Mammal	↓
Small Mammal	↑
Herpetofauna	↑
Avian	↓

## STUDY AREA

I conducted my research on the East Foundation's San Antonio Viejo Ranch (hereafter SAV) located in Jim Hogg and Starr counties, approximately 25 km south of Hebbronville, Texas. SAV is the largest of six ranches owned by the East Foundation. SAV occupies 57,035 ha of south Texas rangeland and is located in the South Texas Plains ecoregion (Figure 1). Due to logistical constraints, I utilized approximately 35,000 ha of land for my study, excluding far northern and southern reaches of the ranch (Figure 2). Mean annual temperature in this region is approximately 29° C with annual ranges between 7° C and 36° C. Mean annual rainfall in this region is approximately six cm (National Oceanic and Atmospheric Administration 2014). Mean temperature during the breeding season (March – August) was approximately 25.7° C in 2015 and 28.9° C in 2016 (PRISM Climate Group 2017). Mean rainfall during the breeding season was approximately 6.1 cm in 2015 and 6.1 cm in 2016 (PRISM Climate Group 2017).

Predominant soil types on SAV consist of sandy-loam, clay-loam, clay, and shallow caliche soils (McLendon et al. 2013). Vegetation within my study area is divided into four vegetation types that represent a successional continuum within a semiarid landscape: early seral (predominated by early successional plants), native grassland (dominated by native grasses with no permanent water coverage), shrubland (dominated by woody plants <3 m), and woodland (dominated by woody plants >3 m) (McLendon et al. 2013). Based on the McLendon et al. (2013) classification, early seral vegetation at SAV occurs on <10% of my study area and is characterized by doveweed (*Croton* spp.), sandbur (*Cenchrus* spp.), and horsemint (*Monarda punctata*). Native

grassland occurs on <10% of my study area and includes species such as arrowfeather threeawn (*Aristida purpurascens*), balsamscale grass (*Elyonurus tripsacoides*), Lehmann lovegrass (*Eragrostis lehmanniana*), and seacoast bluestem (*Andropogon littoralis*). Shrubland occurs on <10% of my study area and includes catclaw (*Acacia greggii*), blackbrush (*Acacia rigidula*), and brasil (*Condalia hookeri*). Finally, woodland at SAV occurs on ~70% of my study area and consists primarily of honey mesquite (*Prosopis glandulosa*) and woody species such as amargosa (*Castela texana*) and whitebrush (*Aloysia lycioides*). However, it is important to note that vegetation surveys by McLendon et al. (2013) were conducted under drought conditions, therefore, estimates of herbaceous cover may not reflect current conditions. Personal communications with East Foundation staff indicate the percent of native grassland on SAV is actually much higher.



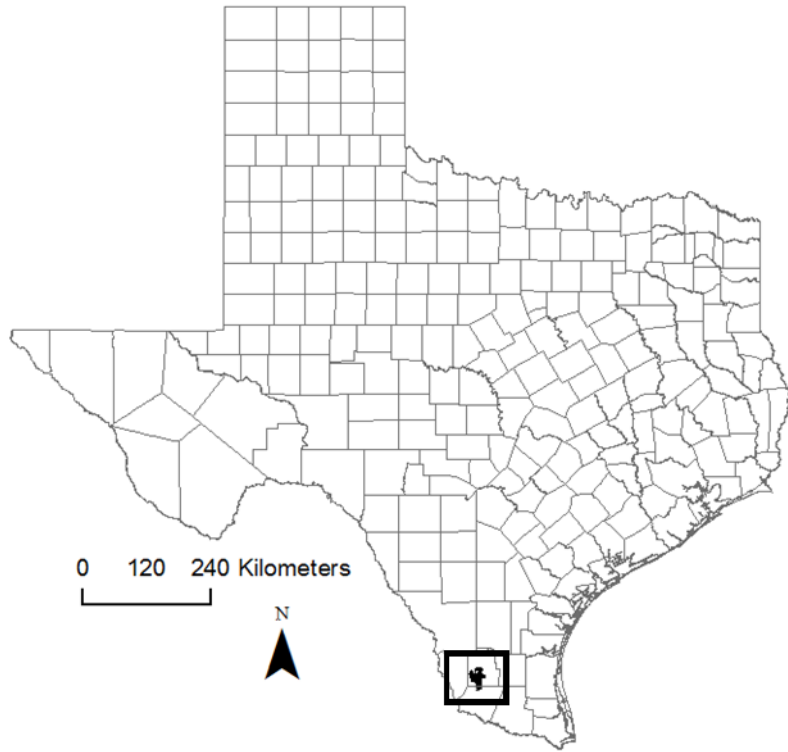


Figure 1. Location of San Antonio Viejo Ranch in Jim Hogg and Starr counties, Texas, USA.

## METHODS

### **Site Selection**

Using a stratified random sampling design (Morrison et al. 2008:146–148), I randomly generated two 600 m square grids (8 total) within each vegetation type to serve as my study sites (Figure 2). These study sites represented the continuum of vegetation conditions that occur on SAV, were representative of nesting densities of birds known to occur on the ranch (Flanders et al. 2006), and were of a size that could be feasibly surveyed by my field technicians and I over the course of the breeding bird season. When logistics allowed me to survey for and monitor additional nests, I deviated outside of my study sites to increase my nest sample sizes and assigned each nest to the most appropriate study site and vegetation category given their locations and the conditions at the nest site.

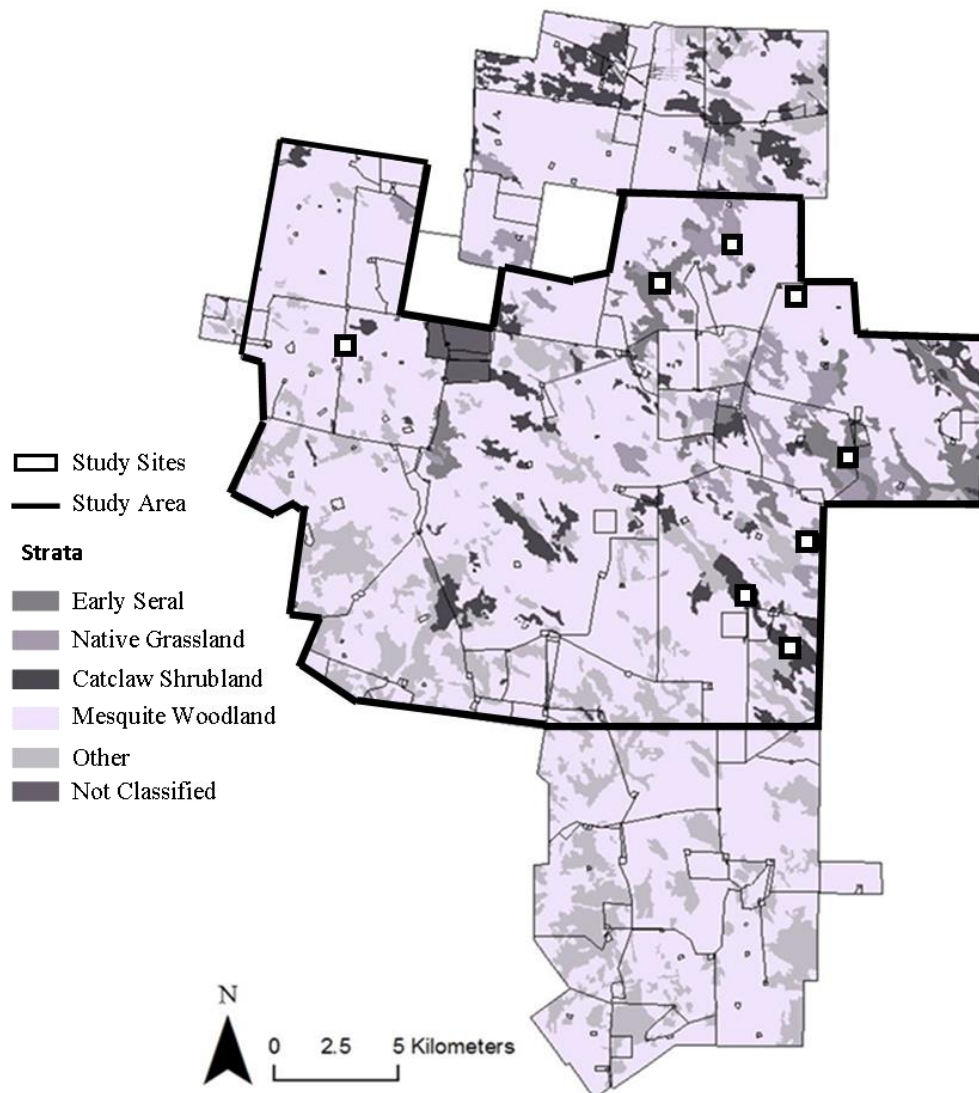


Figure 2. Location of study area and study sites for nest predation study and distribution of vegetation types early seral, native grassland, catclaw shrubland, and mesquite woodland on San Antonio Viejo Ranch, Texas, USA.

### Nest Success

To examine the influence of nest-site vegetation characteristics on bird nest success (Objective 1), I searched for and monitored bird nests between the months of March and August of 2015 and 2016. Hereafter, “success” refers to the probability a nest will fledge at least one young. I searched for and monitored bird nests between the

months of March and August of 2015 and 2016. During this period, I actively searched for nests every 3–5 days within each vegetation type. In grassland vegetation, my technicians and I (hereafter “we”) walked 200 m transects spaced 20 m apart while using a bamboo pole to agitate grass and flush birds (Winter et al. 2003). To cover area between transects, we walked at 10 m spacing to optimize the area agitated without overlapping efforts. Once we flushed a bird from its nest, we identified the location from where the bird flushed and located the nest. If we could not find the nest, we would observe the bird from 15 m away and we used behavioral cues such as nesting material carries or food carries to identify the location of the nest. In shrubland and woodland vegetation types, we used behavioral cues as well as systematic searching to locate bird nests. Once we located a nest, we recorded the number of eggs or nestlings, if present. In all vegetation types, we placed a small flag marker 10 m from the nest in a randomized cardinal direction to aid in relocation of the nest site during subsequent visits. We monitored nests every 2–4 days to determine nest fate. So as not to create a path to the nest location during checks, we did not walk directly to a nest or use the same method of approach when checking nests. If any predators were visible, we did not check the nest. We made no effort to mask our scent as previous studies have shown no changes in predation rates due to observer effects (Kirkpatrick and Conway 2010, Jacobson et al. 2011).

I evaluated vegetative characteristics at the nest site within one to two weeks of successful fledging of young or nesting failure. I used a modified BBIRD protocol (Martin et al. 1997) to quantify percent shrub cover in close proximity to the nest site,

nest concealment, and distance from the nest site to the closest vegetation or anthropogenic edge. I established circular plots around each nest site with 5 m radii to estimate percent shrub cover. I then divided each plot into four quadrats and quantified the percentage of woody perennials <2 m covering the ground to the nearest 10%. I estimated concealment (% visual obstruction) from 1 m away in each cardinal direction and above/below the nest to the nearest 10% (Burnhans and Thompson 1998). Lastly, I estimated closest distance to edge (m) using ArcMap 10.2 and visually in the field. I defined edge as the distinct boundary between two vegetation types, which included the vegetation types described above, roads, fence lines, established cattle paths, or other major anthropogenic structures.

### **Predator Assemblage**

To identify predators of bird nests (Objective 2), I placed infrared video cameras at a subset of nests distributed across the ranch. Since my study focused on all nesting birds, not a specific species, I did not select for an individual species when placing cameras. Due to the limited amount of cameras that could be in operation at one time (~12 units), I selected nests for video monitoring depending on availability of camera units, nesting stage of available nests, distance from other cameras, and logistical constraints (e.g., nests high in the canopy could not be monitored). Video camera systems consisted of an infrared camera (Rainbow, Costa Mesa, California), digital video recorder (Detection Dynamics, Austin, Texas), 12 volt battery, and supplemental 20 watt solar panel (Suntech, San Francisco, California). Video cameras were weatherproof 3.6 mm black and white bullet cameras with 940 nm infrared light emitting

diodes. I connected DVRs to cameras via a 15 m component cable. I used 32 GB memory cards to increase data storage and decrease need for nest visitation to change cards on DVRs. I checked camera systems every 2–4 days to monitor nest status, change memory cards, and repair equipment if necessary. I only placed cameras at nests with contents (eggs or nestlings) to avoid human-caused abandonment by adult birds. I also deployed cameras as far apart as possible to reduce the chances of visitation by individual predators at multiple cameras (Pietz et al. 2012). I determined this distance case by case, based on spatial distribution and availability of nests for study. I mounted cameras approximately 30 cm from the nest or close enough to view contents and activity around the nest without causing unnecessary disturbance to the birds (Pietz et al. 2012). After installation, I covered cables with ground litter to make cameras less conspicuous to predators.

### **Predation Probability**

To assess the influence of nest-site vegetation characteristics (described above) on the probability of predation by mammal, snake, avian, or ant predators (Objective 3), I utilized footage from video cameras deployed at the nest site. I reviewed camera footage until I confirmed an initial predation event. I did not use subsequent predators of specific nests in this study since previous studies indicate that initial predation rates are the most effective measure of predation risk (Pietz and Granfors 2000). If continuous camera footage was interrupted due to failed/damaged equipment, weather, or skewed/blocked camera visuals, I confirmed all nest contents were remaining when footage resumed. If contents were missing, I identified the predator as unknown. I then

grouped predators into one of five categories based on foraging and life history strategy: mammal, snake, bird, ant, or cow/deer.

### **Predator Activity**

*Meso- and Large Mammals* –I measured meso- and large mammal activity (Objective 4) using track plates. I constructed transects using four linear points, spaced 250 m apart. I randomly selected the location and orientation (0 to 359°) of each transect within each vegetation type; however, transect orientation was limited due to shape and size of vegetation types. I did not monitor nests that were located within 250 m to a track plate as most commercial baits are only effective to 200 m and I was thus able to avoid the influence of baiting on nest success (Peterson et al. 2004). I deployed track plates when nest monitoring began and I removed plates once monitoring ended (March through August).

Track stations consisted of one 160 cm x 82 cm aluminum plate. I coated each plate with a 3:1 mixture of isopropyl alcohol and carpenters chalk (Irwin Tools, Martinsville, OH). Once the alcohol evaporated, a thick, uniform layer of chalk remained. I then placed white contact paper sticky side up in the middle of each track plate (Cain 2001) and lured each track plate with a commercial predator attractant that I selected based on potential predators within my study area (Caven's Predator Bait Plus, Minnesota Trapline, Pennock, MN). I opened stations for approximately seven days; however, I closed stations if rain was forecasted. During checks I removed contact paper and bait from the station. I then left the un-lured stations closed for 7–10 days to

discourage predators from becoming acclimated to the bait. After the closure period, I re-lured track plates and replaced chalk and contact paper.

*Small Mammals* – Sampling for small mammals was conducted by a concurrent monitoring project designed by the East Foundation Monitoring program. Moving forward, “we” refers to the East Foundation monitoring crew. We used a stratified random sampling technique to determine the location of each sampling location within each vegetation type. We then ran two trap configurations within each vegetation type between the months of February and April of 2015 and 2016. We used these two configurations to determine the best method for estimating abundance from recaptures (Nichols and Pollock 1983). In 2015, we configured Sherman traps (7.6 x 9.5 x 30.5 cm, model XLK) in a 12 x 12 square grid with 20 m spacing at 50% of sampling locations. For larger rodents such as rats, we also placed extra-large Sherman traps (10.2 x 11.4 x 38 cm, model XLF15) at every other station, resulting in a 6 x 6 configuration with 40 m spacing. At the other half of the sites we used a rectangular configuration consisting of three transects. We constructed transects 300 m in length, with 25 m spacing between Sherman traps (7.6 x 9.5 x 30.5 cm, model XLK) and 25 m spacing between transects, resulting in a 3 x 15 configuration. We also placed extra-large traps (10.2 x 11.4 x 38 cm, model XLF15) every 50 m along the 2 outer transects. In 2016, we used also used two trapping configurations. We used an 11 x 11 configuration with 20 m spacing and placed additional extra-large traps at every other location, resulting in a 6 x 6 configuration. At the other half of sites, we used a 19 x 3 configuration and placed an additional 14 extra-large traps along the outer two lines. Beginning the afternoon prior to



the first trap night, we opened and baited traps, then checked and closed them each morning. We trapped at each site for six consecutive nights before moving traps.

When setting traps, we baited traps with a teaspoon of peanut butter and rolled oats mixture prepared in approximately a 4:1 ratio by weight. Additionally, we provided a 1 x 1 in Nestlet square (Ancare) for bedding material. For each animal captured, we applied a unique mark consisting of a combination of blue, green, purple, and/or red spots on the ventral surface below the neck using Sharpie permanent markers (Root et al. 1999). We then weighed, sexed, and examined all captured animals for reproductive status. We released animals upon completion of marking and inspection at the location of capture. I did not place bird study grids closer than 50 m to small mammal traps to reduce the effects of baiting on nest success. I determined this distance based on the approximate home range of the largest small mammal, the Mexican Ground Squirrel (*S. mexicanus*), which will forage up to 45 m from its burrow (Schmidley 1994).

*Herpetofauna* – We used a stratified random sampling technique to determine the location of each sampling location within each vegetation type. We used trap arrays consisting of pitfall traps, funnel traps, and cover boards with drift fencing to sample the diversity of herpetofauna occurring on the ranch. We first used silt fencing and wooden stakes to construct three arms radiating 15 m from a central point. We placed pitfall traps (five-gallon buckets) at the center of the array and at the distal end of each arm. Additionally, we placed wire-mesh funnel traps on both sides of the drift fence, approximately 7.5 m from the central pitfall trap on all three arms. We provided each pitfall and funnel trap with a plywood cover board for shade; and spaced the cover

boards over pitfall traps approximately three cm from the ground level with sticks or rocks. We also placed a 5 cm x 10 cm sheet of plywood within five m of each of the three distal pitfall traps that acted as cover boards for detecting snakes that are primarily fossorial. To avoid trapping effects on nest success, I did not place bird study grids closer than 50 m to herpetofauna trapping locations to account for disruption of natural predator movement around trapping arrays.

*Avian* – I did not sample for avian predators.

## ANALYSES

### **Nest Success**

I divided the bird species that I monitored into categories based on the species' nesting strategy. Because differences in nesting strategies could potentially mask relationships between nest success and vegetative characteristics, I controlled for these differences using five nesting strategy criteria (hereafter group): nest size, nest shape, nest location (e.g. ground, shrub, or tree placement), incubation time, and nestling time (time nestlings spend in the nest prior to fledging). I evaluated nest size qualitatively by grouping nests with distinctly different sizes based on their circumference (small = 200 mm, medium = 300 mm, large = 750 mm). I evaluated nest shape by grouping nests into two categories: cup shaped nests and minimalist nests with little or no structure (hereafter "exposed"). I then evaluated nest location, incubation, and nestling time using life history traits described in Baicich and Harrison (2005). After considering these five criteria, I arrived at four nesting groups: small cup nests, medium cup nests, large cup nests, and exposed nests (Table 5).

Next, I reported the percentage of successful nests within each group as the number of nests that fledged at least one young as a proportion of all nests within that group. Prior to evaluating the effects of nest-site vegetation characteristics on nest success, I tested the assumptions of normality and homogeneity of variances to determine the appropriate statistical tests to use. First, I assessed normality and homogeneity of variance of shrub cover, concealment, and distance to edge variables for each nesting group and all nesting groups combined using Kolmogorov–Smirnov tests

and visually with scatterplots. I then assessed my achieved power for each nesting strategy group using Gpower 3.1 (Faul et al. 2007). I did not conduct statistical analyses if I could not achieve a desired power of 0.8 (Cohen 1988) given my sample size, medium effect size of 0.3, and  $\alpha = 0.05$ . Because all of my data violated the assumption of normality, I proceeded with statistical evaluation using non-parametric Kruskal-Wallis tests and Wilcoxon Rank Sum tests. I used Kruskal-Wallis tests to test for statistical differences in mean values of shrub cover, concealment, and distance to edge variables associated with each nesting group. I also used Wilcoxon Rank Sum tests to test for differences in mean values of shrub cover, concealment, and distance to edge associated with successful and unsuccessful nests within each group. I then reported mean values and their associated standard errors. In addition to vegetation variables, I also tested if year and presence of nest camera accounted for temporal and human induced variation. I used Wilcoxon Rank Sum tests to compare the effects of year and nest camera presences on the number of successful and unsuccessful nests within each group.

I also developed *a priori* models to evaluate the effects of vegetation variables on nest success of all bird nests combined and for each nesting group. Prior to model evaluation, I checked for statistically significant correlations between variables using variance inflation factors (VIF). I considered VIF values greater than five to be significantly correlated (O'Brien 2007). I used a logistic regression approach (Hosmer and Lemeshow 1989) to evaluate the effects of shrub cover, concealment, distance to edge, nest group, presence of camera, and year on nest success (0 = fail, 1 = success) of

all bird species combined. I also included multiple effects models and interaction models using these variables. If there were significant interactions between hypothesized variables and nesting group, I constructed separate models for each group. Models for each nesting group included shrub cover, concealment, distance to edge, year, presence of camera, multiple effects, and interactions. I evaluated model fit using Akaike's Information Criterion adjusted for small sample sizes ( $AIC_c$ ) and  $AIC_c$  Weights ( $w_i$ ) (Sugiura 1978, Burnham and Anderson 2002:49–97). I then selected my highest ranked model and plotted predicted values and 95% confidence intervals using R 3.3.2 (R Core Team 2017).

Table 5. Common name, scientific name, and associated nesting strategy groups of bird species monitored on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

Common Name	Scientific Name	Nesting Strategy
Bewick's Wren	<i>Thryomanes bewickii</i>	Small Cup
Black-throated Sparrow	<i>Amphispiza bilineata</i>	Small Cup
Cassin's Sparrow	<i>Aimophila cassinii</i>	Small Cup
Lark Sparrow	<i>Chondestes grammacus</i>	Small Cup
Painted Bunting	<i>Passerina ciris</i>	Small Cup
Cardinal	<i>Cardinalis spp.</i>	Medium Cup
Curve-billed Thrasher	<i>Toxostoma curvirostre</i>	Medium Cup
Eastern Meadowlark	<i>Sturnella magna</i>	Medium Cup
Green Jay	<i>Cyanocorax yncas</i>	Medium Cup
Long-billed Thrasher	<i>Toxostoma longirostre</i>	Medium Cup
Northern Cardinal	<i>Cardinalis cardinalis</i>	Medium Cup
Northern Mockingbird	<i>Mimus polyglottos</i>	Medium Cup
Pyrrhuloxia	<i>Cardinalis sinuatus</i>	Medium Cup
Scissor-tailed Flycatcher	<i>Tyrannus forficatus</i>	Medium Cup
Greater Roadrunner	<i>Geococcyx californianus</i>	Large Cup
Yellow-billed Cuckoo	<i>Coccyzus americanus</i>	Large Cup
Common Ground-dove	<i>Columbina passerine</i>	Exposed
Killdeer	<i>Charadrius vociferus</i>	Exposed
Mourning Dove	<i>Zenaida macroura</i>	Exposed

## **Predator Assemblage**

After evaluating camera footage, I grouped predator species by predator type (e.g. mammal, snake, avian, etc.) and I constructed frequency tables of detections of individual species between years and within each vegetation type. I then made comparisons of detection frequencies between predator group, year, and vegetation type.

## **Predation Probability**

To evaluate the effects of nest-site specific vegetation variables on the probability of predation by predator type (e.g. mammals, snakes, etc.), I first divided predators identified on camera footage into categories based on taxonomic class. These predator types included mammals, snakes, birds, ants, and grazing or browsing species such as cow and deer.

Next, I reported the percentage of successful nests and the number of nests depredated by each predator type as a proportion of all camera monitored nests with known fates. Prior to evaluating the effects of nest-site vegetation characteristics on predator type, I tested the assumptions of normality at homogeneity of variances to determine the appropriate statistical tests to use. First, I assessed normality and homogeneity of variance of shrub cover, concealment, and distance to edge variables for each predator type and all camera nests combined using Kolmogorov–Smirnov tests and visually with scatterplots. I then assessed my achieved power for each predator group using Gpower 3.1 (Faul et al. 2007). I did not conduct statistical analyses if I could not achieve a desired power of 0.8 (Cohen 1988) given my sample size, medium effect size of 0.3, and  $\alpha = 0.05$ . Because all of my data violated the assumption of normality, I

proceeded with statistical evaluation using non-parametric Kruskal-Wallis tests. I used Kruskal-Wallis tests to test for statistical differences in mean values of shrub cover, concealment, and distance to edge variables associated with each predator type. I then reported mean values and their associated standard errors.

I developed *a priori* models to evaluate the effect of shrub cover, concealment, and distance to edge on the probability of predation by all predator groups combined. I also included year as a variable to account for potential temporal variation and predator type to account for variation due to different foraging strategies. I also constructed separate models to evaluate the effects of shrub cover, concealment, distance to edge, and year on the probability of predation by each predator type; however, due to low sample sizes, I could only construct models to evaluate the probability of snake predation on bird nests. I used logistic regression (Hosmer and Lemeshow 1989) to evaluate the effects of shrub cover, concealment, and distance to edge, and year on the probability of predation by snakes. I also included multiple effects models and interaction models using these variables. I evaluated model fit using Akaike's Information Criterion adjusted for small sample sizes ( $AIC_c$ ) and  $AIC_c$  Weights ( $w_i$ ) (Sugiura 1978, Burnham and Anderson 2002:49–97). I then selected my highest ranked model and plotted predicted values and 95% confidence intervals using R 3.3.2 (R Core Team 2017).

### **Predator Activity**

I constructed indices of predator activity for meso- and large mammal, small mammal, and herpetofauna sampling efforts using captures or detections per trap or

station night. Prior to constructing indices, I selected species within each predator group that are known to depredate or could potentially depredate bird nests based on life history strategies (Schmidley 1994, Werler and Dixon 2000). For meso- and large mammal sampling, I constructed an index of activity for each species as individual detections of each species per track station per day. Because I could not differentiate between individual prints of the same species on track plates, I counted multiple prints of a single species on an individual plate as one detection for that species within each sampling period. For small mammals, I constructed activity indices as trapped individuals of each species per trap night. Because I used data from two trapping configurations (rectangular and square), I multiplied values collected at rectangular configurations by 3.75 to correct for differences in trap numbers. For herpetofauna, I constructed activity indices as trapped individuals per trap night. I then multiplied all activity indices by 100 to standardize to 100 trap nights.

Next, I performed statistical tests to evaluate the effects of vegetation type on predator activity indices calculated above. I assessed normality and homogeneity of variance of each vegetation type for each predator species using Kolmogorov–Smirnov tests and visually with scatterplots. I then assessed my achieved power for each predator group using Gpower 3.1 (Faul et al. 2007). I did not conduct statistical analyses if I could not achieve a desired power of 0.8 (Cohen 1988) given my sample size, medium effect size of 0.3, and  $\alpha = 0.05$ . Since all of my data violated the assumption of normality, I proceeded with statistical evaluation using non-parametric Kruskal-Wallis tests. I used Kruskal-Wallis tests to test for statistical differences in mean values of



activity between each vegetation type for all predator species. I then reported these mean values and associated standard errors and plotted these values using R 3.3.2 (R Core Team 2017). I then compared results with nest predation frequencies calculated previously.

## RESULTS

### Nest Success

I monitored 258 nests of 17 bird species in 2015 and 2016. I observed the lowest percentage of apparent nest success in small and medium cup groups, with 21.3% and 25.3% of nests fledging at least one young, respectively (Table 6). Large cup nests (31.3%) I monitored were ~1.2 times more successful than small and medium cup nests. I observed the highest success in my exposed nest group (45.7%) which was over two times more successful than small cup nests (Table 6). I observed lower nest success in 2015 when compared with 2016 for all nesting groups except for large cup nests. I only observed a significant difference in success between years for my exposed nest group ( $W = 339$ ,  $P = 0.02$ ), with a ~50% decrease in nest success from 2015 to 2016. In 2015, I found nest success for exposed nests was ~ two times higher than the other three nesting groups ( $\chi^2 = 11.6$ ,  $P = 0.01$ ,  $df = 3$ ). I did not observe significant differences in nest success between the four nesting groups in 2016 ( $\chi^2 = 1.7$ ,  $P = 0.63$ ,  $df = 3$ ). I also did not observe differences between nest success for nests monitored with cameras versus those without ( $\chi^2 = 2.55$ ,  $P = 0.28$ ,  $df = 2$ ).

Table 6. Nest fates, frequency ( $n$ ) of fate, and percentage (%) of fate for 258 small cup, medium cup, large cup, and exposed nests monitored within four vegetation types occurring on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

	Small Cup		Medium Cup		Large Cup		Exposed	
	$n$	%	$n$	%	$n$	%	$n$	%
Abandoned	2	4.3	9	6.2	0	0.0	4	8.7
Depredated	34	72.3	95	65.1	11	68.8	19	41.3
Fell	0	0.0	2	1.4	0	0.0	0	0.0
Fledged	10	21.3	37	25.3	5	31.3	21	45.7
Unknown	1	2.1	0	0.0	0	0.0	1	2.2
Weather	0	0.0	3	2.1	0	0.0	1	2.2

I collected vegetation data for 241 nests of 17 different bird species in 2015 and 2016. I observed 1.6 times more shrub cover at small cup nests than at medium cup nests and 2.8 times more shrub cover at small cup nests than exposed nests ( $\chi^2 = 9.0$ ,  $P = 0.03$ ,  $df = 3$ ). Additionally, I detected a significant decrease of 1.8 times in shrub cover between medium cup nest and exposed nests ( $\chi^2 = 19.1$ ,  $P < 0.01$ ,  $df = 2$ ). I observed 1.4 times more concealment at small cup nests than at medium cup nests and 1.3 times more concealment at small cup nests than at exposed nests ( $\chi^2 = 18.9$ ,  $P < 0.01$ ,  $df = 2$ ). I did not detect any significant differences in concealment between medium cup and exposed cup nests ( $\chi^2 = 18.9$ ,  $P = 0.35$ ,  $df = 2$ ). I also did not detect any significant differences between in distance to edge between any nesting group ( $\chi^2 = 1.98$ ,  $P = 0.37$ ,  $df = 2$ ).

I did not detect any significant differences in shrub cover ( $W = 192.5$ ,  $0.75$ ), concealment ( $W = 197.5$ ,  $P = 0.65$ ), or distance to edge ( $W = 166$ ,  $P = 0.72$ ) for successful and unsuccessful small cup nests (Table 7). For medium cup nests, I observed 1.5 times more shrub cover at unsuccessful nests than successful nests ( $W = 2189$ ,  $P = 0.03$ ). However, I did not observe any significant differences between concealment ( $W = 1902.5$ ,  $P = 0.45$ ) or distance to edge ( $W = 1608$ ,  $P = 0.58$ ). For exposed nests, I did not detect any significant differences in shrub cover ( $W = 205$ ,  $P = 0.28$ ), concealment ( $W = 260$ ,  $P = 0.86$ ), or distance to edge ( $W = 206.5$ ,  $P = 0.42$ ) between successful and unsuccessful nests (Table 7). My sample size was not large enough to carry out these tests on large cup nests at my desired power of 0.8, however, shrub cover was highest and edge was lowest for this nesting group.

Table 7. Mean shrub cover (%), concealment (%), and distance to edge (m)  $\pm$  standard error for successful and failed small cup, medium cup, large cup, and exposed nests on San Antonio Viejo ranch, Texas, USA, 2015–2016.

	Shrub Cover		Concealment		Distance to Edge	
	Success	Fail	Success	Fail	Success	Fail
Small Cup	11.9 $\pm$ 2.9	16.3 $\pm$ 2.6	49.6 $\pm$ 5.1	52.2 $\pm$ 3.3	113.6 $\pm$ 29.0	101.4 $\pm$ 17.0
Medium Cup	5.7 $\pm$ 0.9	10.6 $\pm$ 1.1	36.9 $\pm$ 3.5	38.0 $\pm$ 1.9	104.1 $\pm$ 16.1	92.7 $\pm$ 9.1
Large Cup	18.1 $\pm$ 4.4	23.1 $\pm$ 6.9	50.3 $\pm$ 2.3	40.9 $\pm$ 5.7	34.2 $\pm$ 15.5	76.3 $\pm$ 27.2
Exposed	6.1 $\pm$ 1.4	4.6 $\pm$ 1.2	39.2 $\pm$ 3.8	38.6 $\pm$ 4.0	173.2 $\pm$ 37.7	111.7 $\pm$ 25.3

My best-fit model to predict nest success of all nests combined included interactions between shrub cover, distance to edge, and nest strategy group (Table 8). I detected a ~3% increase in predicted probability of nest success with every 10% decrease in shrub cover and an increase of <1% with every 1 m increase in distance to edge. I observed a 3% decrease of predicted probability of nest success with every 10% increase in shrub cover and a decrease of an additional 2% with every 1 m increase in distance to edge when nesting group was considered with these variables (Table 8). Because my top three models with similar support (Table 8) included interactions between nest group and these two vegetation variables, I analyzed nesting groups separately when examining the effects of vegetation variables on nest success below.

My best-fit model for small cup nests was the null model (Table 8). Thus, my explanatory variables did not explain success for this group. For medium cup nests, my best fit model included shrub cover as a variable and I observed substantially greater support for this model than the second ranked model including shrub cover and distance to edge (Table 8). I observed an ~8% decrease in predicted probability of nest success for medium cup nests with every 10% increase in shrub cover, however, my confidence

for predicting nest success with shrub cover >20% was low due to limited observations above this value (Figure 3). My best fit model for exposed nests included year. From 2015 to 2016, I observed a decrease of ~50% in the predicted probability of nest success for this group (Figure 4).

Table 8. Model selection results from regression analysis of nest success and vegetation variables<sup>b</sup> for all bird species combined and three nesting strategy groups monitored in four vegetation types on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

Nest Strategy <sup>a</sup>	Model <sup>b</sup>	$K^c$	LL <sup>d</sup>	AIC <sub>c</sub> <sup>e</sup>	$\Delta AIC_c^f$	$w_i^g$
All Nests						
	Shrub*Edge*Group	16	-138.03	-235.93	0.00	0.97
	Shrub*Concealment*Group	16	-141.76	-228.49	7.44	0.02
	Concealment*Edge*Group	16	-142.94	-226.11	9.82	0.01
	Shrub	2	-150.21	305.34	541.27	0.00
	Shrub*Edge	4	-147.34	306.32	542.25	0.00
	Edge	2	-150.85	306.62	542.55	0.00
	Global	5	-145.55	307.10	543.04	0.00
	Group	4	-147.76	307.16	543.09	0.00
	Camera	3	-150.75	309.50	545.43	0.00
	Year	2	-152.56	310.05	545.98	0.00
	Null	1	-153.94	310.16	546.09	0.00
	Shrub*Concealment	4	-149.30	310.24	546.17	0.00
	Concealment	2	-152.89	310.69	546.63	0.00
	Shrub*Group	8	-143.20	322.98	558.91	0.00
	Edge*Group	8	-144.24	325.06	560.99	0.00
	Concealment*Group	8	-146.49	329.56	565.49	0.00
Small Cup						
	Null	1	-24.08	50.67	0.00	0.60
	Year	2	-23.56	52.83	2.16	0.21
	Camera	2	-23.96	53.63	2.96	0.14
	Shrub + Concealment	3	-23.39	56.77	6.10	0.03
	Shrub + Edge	3	-23.68	57.36	6.69	0.02
	Shrub*Concealment	4	-23.16	62.31	11.64	0.00
	Global	5	-22.83	70.66	19.99	0.00
	Shrub	8	-143.20	446.41	395.74	0.00
	Edge	8	-144.24	448.49	397.82	0.00
	Concealment	8	-146.49	452.99	402.32	0.00

Table 8. Continued

Nest Strategy <sup>a</sup>	Model <sup>b</sup>	$K^c$	LL <sup>d</sup>	AIC <sub>c</sub> <sup>e</sup>	$\Delta\text{AIC}_c^f$	$w_i^g$
Medium Cup						
	Shrub	2	-71.17	147.84	0.00	0.48
	Shrub + Edge	3	-69.57	148.57	0.73	0.33
	Shrub + Concealment	3	-71.12	151.67	3.83	0.07
	Shrub*Edge	4	-69.33	153.33	5.48	0.03
	Null	1	-75.64	153.72	5.87	0.03
	Camera	3	-72.18	153.78	5.94	0.02
	Edge	2	-74.32	154.15	6.30	0.02
	Concealment	2	-75.03	155.57	7.73	0.01
	Year	2	-75.52	156.55	8.71	0.01
	Shrub*Concealment	4	-71.08	156.83	8.99	0.01
	Global	5	-68.88	159.76	11.92	0.00
Exposed						
	Year	2	-28.27	61.47	0.00	0.53
	Edge	2	-29.48	63.88	2.41	0.16
	Null	1	-31.09	64.47	3.00	0.12
	Camera	2	-30.45	65.82	4.35	0.06
	Shrub	2	-30.73	66.38	4.91	0.05
	Concealment + Edge	3	-29.25	66.50	5.03	0.04
	Concealment	2	-31.08	67.09	5.62	0.03
	Global	5	-26.53	69.05	7.58	0.01
	Concealment*Edge	4	-29.25	70.14	8.67	0.01

<sup>a</sup>Nest strategy group as determined by nest size, shape, location, incubation, and nestling time

<sup>b</sup> Explanatory variable abbreviations are as follows: Null = null model, Shrub = percent shrub cover at nest site, Concealment = percent of nest obscured by vegetation, Edge = distance to closest edge in meters, Year = year sampled, Camera = Camera monitored nest, Group = Nesting strategy group, Global = all variables included

<sup>c</sup>Total parameters in model

<sup>d</sup>LogLikelihood

<sup>e</sup>Akaike's Information Criteria corrected for small sample sizes

<sup>f</sup>AICc relative to the best-fit model

<sup>g</sup>Model weight

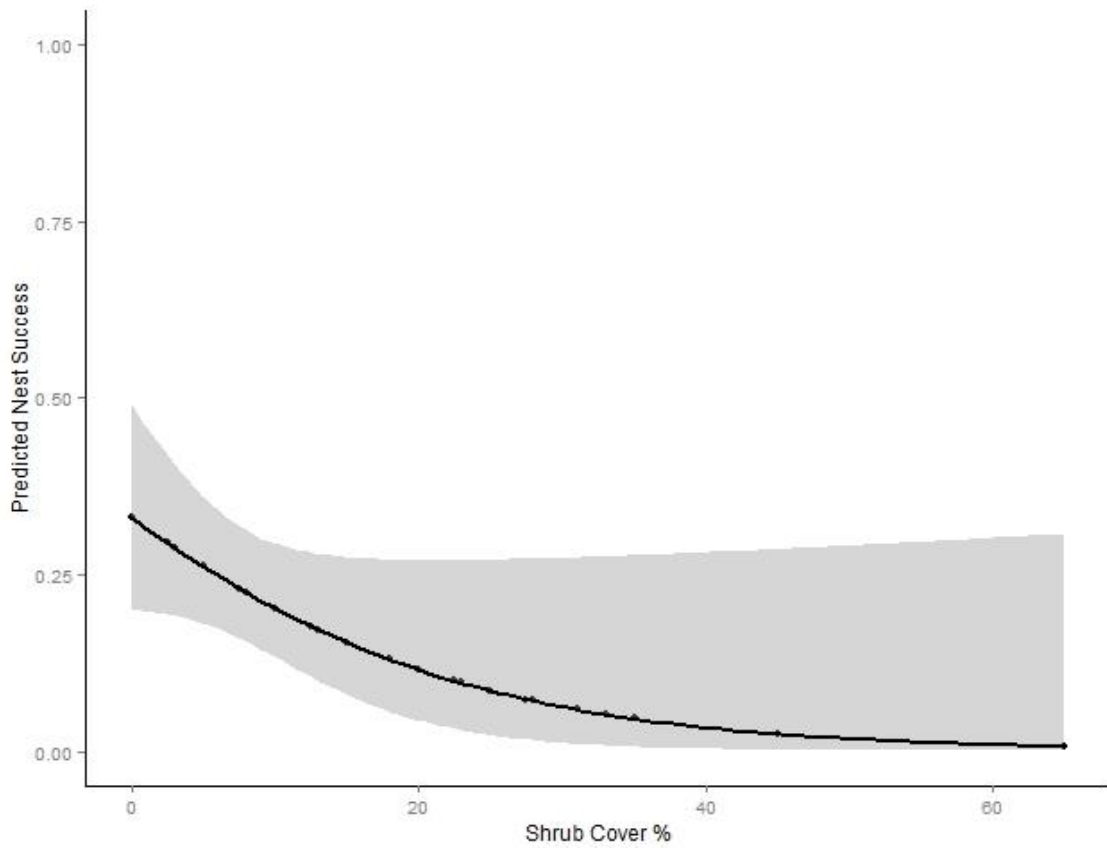


Figure 3. Relationship between predicted probability of nest success and percentage of shrub cover with 95% confidence intervals for medium cup nests located on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

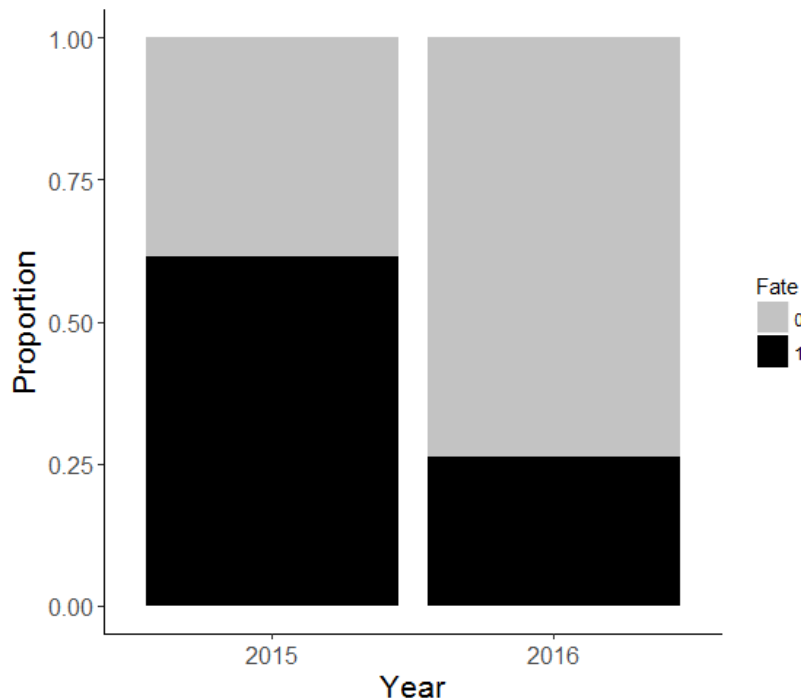


Figure 4. Proportion of successful nests (1) to unsuccessful nests (0) by year for exposed nests located on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

### Predator Assemblage

I deployed infrared cameras at 109 nests and recorded 49 predation events by 12 species (Table 9). Snakes were the most frequent nest predators I detected on camera ( $n = 30$ ). I detected Great Plains rat snakes (*Pantherophis emoryi*) at 33% of predation events and Coachwhips (*Masticophis flagellum*) at 14% of predation events. I detected avian predators ( $n = 9$ ) at 18% of predation events and mesomammals ( $n = 5$ ) at 14% of predation events. I also observed highest species richness in my avian predator group with six bird species detected (Table 9). Within my mesomammals group, I only detected coyotes and badgers. I only observed two nests depredated by ants and two nests depredated by large grazing/browsing species, such as cow and deer (Table 9).



Table 9. Species name, associated predator type, and frequency of detection (*n*) of predators identified from initial predation events recorded on nest cameras deployed on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

Common Name	Scientific Name	Predator Type	2015	2016	Total
			<i>n</i>	<i>n</i>	<i>n</i>
Great Plains Rat Snake	<i>Pantherophis emoryi</i>	Snake	13	3	16
Coachwhip	<i>Masticophis flagellum</i>	Snake	1	6	7
Unknown snake		Snake	4	2	6
Coyote	<i>Canis latrans</i>	Mammal	4	1	5
American Badger	<i>Taxidea taxus</i>	Mammal	1	--	1
Chihuahuan Raven	<i>Corvus cryptoleucus</i>	Avian	2	2	4
Brown-headed Cowbird	<i>Molothrus ater</i>	Avian	--	1	1
Cooper's Hawk	<i>Accipiter cooperii</i>	Avian	--	1	1
Greater Roadrunner	<i>Geococcyx californianus</i>	Avian	1	--	1
Northern Mockingbird	<i>Mimus polyglottos</i>	Avian	1	--	1
Eastern Screech Owl	<i>Megascops asio</i>	Avian	--	1	1
Ant spp.	<i>Solenopsis spp.</i>	Ant	--	2	2
Cow	<i>Bos taurus</i>	Grazing/Browsing	1	--	1
White-tailed Deer	<i>Odocoileus virginianus</i>	Grazing/Browsing	1	--	1
Unknown mammal		Mammal	1	--	1
Total			29	20	49
Unknown			--	1	1
Weather			--	1	1
Predation not recorded			9	--	9

I observed snakes depredating nests most frequently in woodlands (Table 10), accounting for >50% of all predation events in this vegetation type. I observed mesomammals depredating more nests in woodlands (*n* = 2) and grasslands (*n* = 2) than in early seral (*n* = 1) and shrublands (*n* = 1); however, this difference was negligible. Avian, ant, and grazing/browsing predators I observed were variable across all vegetation types (Table 10).

Table 10. Species name, associated predator type, and frequency of detection ( $n$ ) of predators identified from initial predation events recorded on nest cameras deployed in early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

Common Name	Scientific Name	ES	NG	CC	MW	Total
		$n$	$n$	$n$	$n$	$n$
Great Plains Rat Snake	<i>Pantherophis emoryi</i>	4	0	3	9	16
Coachwhip	<i>Masticophis flagellum</i>	3	1	0	3	7
Unknown snake		1	1	1	3	6
Coyote	<i>Canis latrans</i>	1	1	1	2	5
American Badger	<i>Taxidea taxus</i>	0	1	0	0	1
Chihuahuan Raven	<i>Corvus cryptoleucus</i>	1	0	2	1	4
Brown-headed Cowbird	<i>Molothrus ater</i>	0	0	0	1	1
Cooper's Hawk	<i>Accipiter cooperii</i>	1	0	0	0	1
Greater Roadrunner	<i>Geococcyx californianus</i>	0	0	0	1	1
Northern Mockingbird	<i>Mimus polyglottos</i>	1	0	0	0	1
Eastern Screech Owl	<i>Megascops asio</i>	1	0	0	0	1
Ant spp.	<i>Solenopsis</i> spp.	0	1	0	1	2
Cow	<i>Bos taurus x indicus</i>	0	0	0	1	1
White-tailed Deer	<i>Odocoileus virginianus</i>	1	0	0	0	1
Unknown mammal		0	0	0	1	1
<b>Total</b>		<b>14</b>	<b>5</b>	<b>7</b>	<b>23</b>	<b>49</b>
Unknown		0	1	0	0	1
Weather		1	0	0	0	1
Predation not recorded		3	1	0	5	9

### Predation Probability

I observed snakes within my study depredate nests with 2.6 times more shrub cover, 1.2 times more concealment, and were 61.7 m closer to an edge than successful nests (Table 11). I also observed mesomammals ( $n = 5$ ) depredate nests with 1.7 times greater shrub cover, 1.1 times greater concealment, and 61.7 m less distance to edge than successful nests. I observed avian predators selecting similar levels of shrub cover and concealment as snake predators; however, avian predators depredate nests 50.4 m

closer to edges than successful nests (Table 11). I only detected two predation events by ants and these predators depredated nests with low shrub cover, high concealment, and average distance to edge relative to other predators. Lastly, I detected one cow and one deer depredating nests with 1.2 times less concealment, 1.5 times more concealment, and 101.3 m closer to edge than successful nests (Table 11).

Table 11. Mean ( $\bar{x}$ ) shrub cover (%), concealment (%), and distance to edge (m) and standard error (SE) for five predator groups recorded at camera monitored nests on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

Predator	<i>n</i>	Shrub Cover	Concealment	Distance to Edge
		$\bar{x} \pm \text{SE}$	$\bar{x} \pm \text{SE}$	$\bar{x} \pm \text{SE}$
Success	23	6.1 $\pm$ 1.4	41.4 $\pm$ 4.0	128.3 $\pm$ 33.5
Mesomammal	6	11.7 $\pm$ 5.2	44.8 $\pm$ 6.8	66.6 $\pm$ 53.5
Snake	30	15.7 $\pm$ 3.2	48.6 $\pm$ 3.1	96.3 $\pm$ 17.3
Avian	8	15.5 $\pm$ 4.8	40.7 $\pm$ 5.4	77.9 $\pm$ 32.4
Ants	2	11.3 $\pm$ 3.8	57.9 $\pm$ 6.5	89.5 $\pm$ 40.5
Cow/Deer	2	5.0 $\pm$ 5.0	60.0 $\pm$ 11.0	27.0 $\pm$ 25.0

My best-fit model to predict the probability of predation by all predators detected by nest cameras included interactions between shrub, concealment, distance to edge, and predator type (Table 12). I did not detect significant effects of shrub cover, concealment, and distance to edge on the probability of predation by all nests combined. I did observe that predator type altered the impact of these variables on predicted predation of all predators; however, the changes remained negligible (Table 12). Even so, since there were interactions between predator type and these three vegetation variables, I analyzed predator types separately when examining the effects of vegetation variables on the probability of predation reported below.

Because my sample size did not provide my desired power of 0.8 for all predators separately, I only constructed models to evaluate the effects of shrub cover, concealment, distance to edge, and year on the probability of nest predation by snakes. My best fit model for to predict the probability of snake predation included shrub cover (Table 12). I detected an ~10% increase in predicted probability of snake predation with every 10% increase in shrub cover (Figure 5), before reaching asymptote around ~50% shrub cover, similar to the asymptote for my medium cup nest group (Figure 6). However, my confidence for predicting snake predation probability with shrub cover >20% was low due to limited observations above this value.

Table 12. Model selection results from regression analysis of predation probability and vegetation variables<sup>b</sup> for all predators combined and snakes monitored in four vegetation types on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

Predator Type <sup>a</sup>	Model <sup>b</sup>	$K^c$	LL <sup>d</sup>	AIC <sub>c</sub> <sup>e</sup>	$\Delta AIC_c^f$	$w_i^g$
All Predators						
	Concealment*Edge*Type	28	0.00	-68.92	0.00	0.33
	Shrub*Edge*Type	28	0.00	-68.92	0.00	0.33
	Shrub*Concealment*Type	28	0.00	-68.92	0.00	0.33
	Type	7	0.00	28.00	96.92	0.00
	Shrub	2	-40.81	86.54	155.46	0.00
	Shrub + Concealment	3	-39.96	87.93	156.85	0.00
	Shrub + Edge	3	-40.40	88.79	157.71	0.00
	Null	1	-44.72	91.72	160.64	0.00
	Shrub*Edge	4	-40.30	92.24	161.16	0.00
	Concealment	2	-43.94	92.81	161.73	0.00
	Edge	2	-43.98	92.88	161.80	0.00
	Year	2	-44.29	93.51	162.43	0.00
	Global	5	-39.40	94.79	163.72	0.00
	Edge*Type	14	0.00	448.00	516.92	0.00
	Concealment*Type	14	0.00	448.00	516.92	0.00
	Shrub*Type	14	0.00	448.00	516.92	0.00

Table 12. Continued

Predator Type <sup>a</sup>	Model <sup>b</sup>	$K^c$	LL <sup>d</sup>	AIC <sub>c</sub> <sup>e</sup>	$\Delta AIC_c^f$	$w_i^g$
Snakes	Shrub	2	-32.59	72.19	0.00	0.74
	Null	1	-36.27	75.21	3.63	0.12
	Shrub + Concealment	3	-32.36	76.73	5.14	0.06
	Concealment	2	-35.18	76.76	5.17	0.05
	Edge	2	-35.86	78.11	6.53	0.03
	Year	2	-36.01	78.42	6.84	0.02
	Shrub*Concealment	4	-32.30	85.93	14.34	0.00
	Global	5	-31.42	102.84	31.25	0.00

<sup>a</sup>Predator group as determined by life history strategy

<sup>b</sup> Explanatory variable abbreviations are as follows: Null = null model, Shrub = percent shrub cover at nest site, Concealment = percent of nest obscured by vegetation, Edge = distance to closest edge in meters, Year = year sampled, Type = type of predator, Global = all variables included.

<sup>c</sup>Total parameters in model

<sup>d</sup>LogLikelihood

<sup>e</sup>Akaike's Information Criteria corrected for small sample sizes

<sup>f</sup>AICc relative to the best-fit model

<sup>g</sup>Model weight

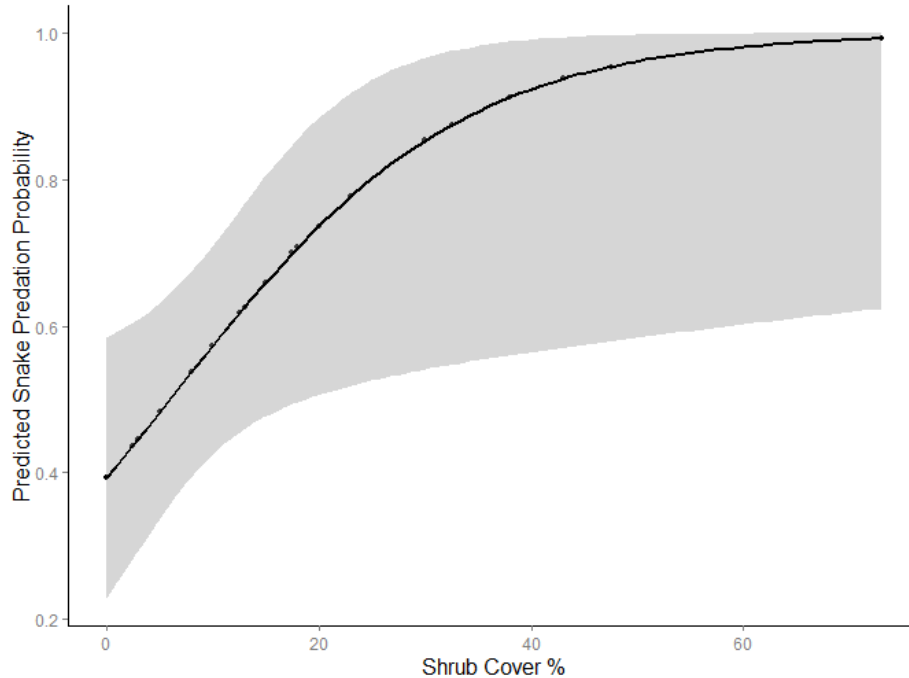


Figure 5. Relationship between predicted probability of predation by snake predators and percentage of shrub cover at the nest site with 95% confidence intervals for camera monitored nests on SAV ranch, Texas, USA, 2015–2016.

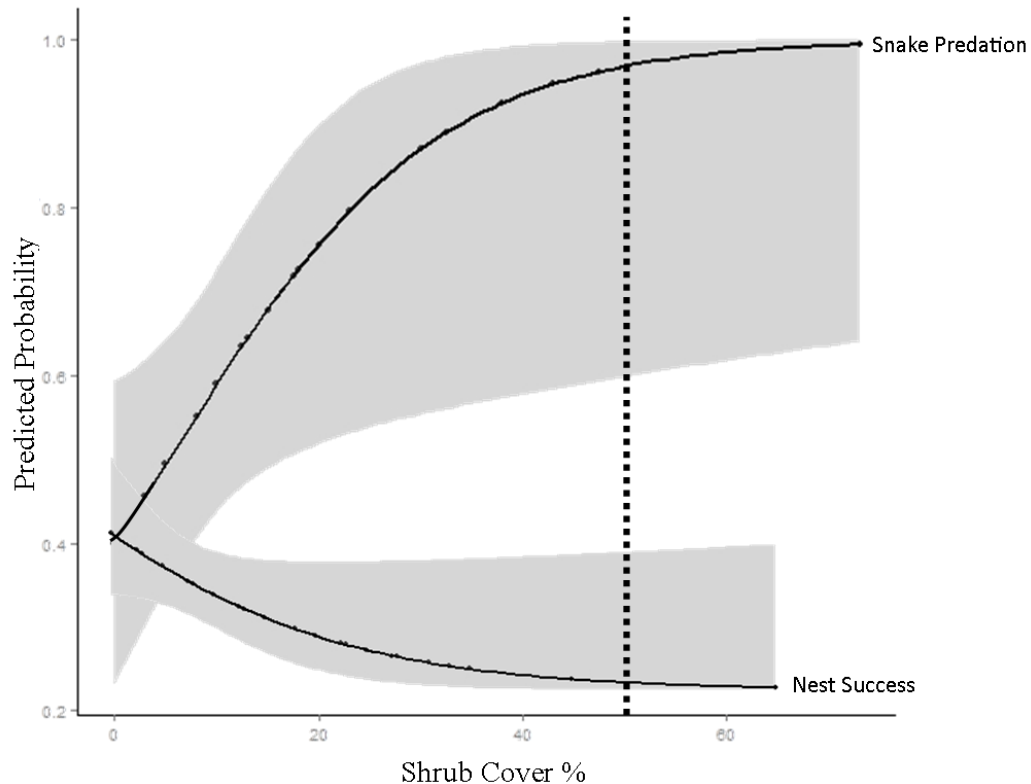


Figure 6. Relationship between predicted probability of snake predation, predicted nest success of medium cup nests, and percentage of shrub cover at the nest site with 95% confidence intervals and threshold line for nests monitored on San Antonio Viejo Ranch, Texas, USA, 2015–2016.

### **Predator Activity**

*Meso- and Large Mammals* – I deployed track plates for a total of 1,454 station days in 2015 and 1,421 station days in 2016. Between 2015 and 2016, I detected three times more coyotes ( $W = 19107$ ,  $P < 0.01$ ) and seven times as many foxes (Tables 13 and 14). I also observed more cow detections (~1.5 times) in 2016 than 2015, but this did not differ significantly between years. All other predator detections remained consistent between years and I only detected one bobcat in 2016. Because I found

statistical differences between years for coyote activity, I analyzed years separately when testing for differences in vegetation type.

In 2015, I detected cows ~five times more frequently in grassland and shrublands than in early seral and woodland vegetation types (Table 13;  $\chi^2 = 22.4$ ,  $P < 0.01$ ,  $df = 3$ ). In 2016, I detected cows ~1.5 times more frequently in shrublands than in the other three vegetation types, however, I did not detect any significant differences at (Table 14;  $\chi^2 = 3.7$ ,  $P = 0.3$ ,  $df = 3$ ). I detected canids in variable numbers across vegetation types, similar to canid detections on camera monitored nests, with the exception of shrublands in 2015 (Table 13). However, in 2016 I detected coyotes three times more frequently in early seral vegetation type than in grasslands, shrublands, or woodlands (Table 14;  $\chi^2 = 16.1$ ,  $P < 0.01$ ,  $df = 3$ ). In 2015, I only detected raccoons in early seral and grasslands, as opposed to 2016 when both detections occurred in woodlands; however, no raccoons were detected on camera (Tables 13 and 14).

Table 13. Meso- and large mammal detections ( $n$ ), mean activity ( $\bar{x}$ ) as defined as detections per 100 track station days, and standard error (SE) within early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types on San Antonio Viejo Ranch, Texas, USA, 2015.

Common Name	Scientific Name	ES		NG		CC		MW	
		$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$
Cow	<i>Bos taurus x indicus</i>	3	1.29±0.80	18	7.02±1.52	17	6.18±1.27	8	2.19±0.80
Coyote	<i>Canis latrans</i>	3	0.86±0.48	4	1.30±0.63	1	0.53±0.53	3	0.72±0.41
Fox spp.	<i>Canidae</i>	3	0.94±0.53	2	0.60±0.42	1	0.27±0.27	1	0.29±0.29
Bobcat	<i>Lynx rufus</i>	0	--	0	--	0	--	0	--
Raccoon	<i>Procyon lotor</i>	1	0.52±0.52	1	0.27±0.27	0	--	0	--
Skunk spp.	<i>Mephitidae</i>	0	--	0	--	0	--	3	0.82±0.48
American Badger	<i>Taxidea taxus</i>	0	--	0	--	0	--	1	0.29±0.29
Unknown Mesomammal		0	--	4	2.02±1.21	1	0.27±0.27	2	0.53±0.38



Table 14. Meso- and large mammal detections (n), mean activity ( $\bar{x}$ ) as defined as detections per 100 track station days, and standard error (SE) within early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types on San Antonio Viejo Ranch, Texas, USA, 2016.

Common Name	Scientific Name	ES		NG		CC		MW	
		<i>n</i>	$\bar{x} \pm SE$	<i>n</i>	$\bar{x} \pm SE$	<i>n</i>	$\bar{x} \pm SE$	<i>n</i>	$\bar{x} \pm SE$
Cow	<i>Bos taurus x indicus</i>	12	3.80±1.08	13	4.40±1.11	21	5.87±1.06	16	4.67±1.00
Coyote	<i>Canis latrans</i>	19	5.75±3.80	6	2.46±1.16	5	1.63±0.74	6	1.56±0.65
Fox spp.	<i>Canidae</i>	0	--	0	--	1	0.30±0.30	0	--
Bobcat	<i>Lynx rufus</i>	0	--	0	--	0	--	1	0.30±0.30
Raccoon	<i>Procyon lotor</i>	0	--	0	--	0	--	2	0.60±0.42
Skunk spp.	<i>Mephitidae</i>	0	--	0	--	2	0.67±0.50	0	--
American Badger	<i>Taxidea taxus</i>	0	--	0	--	0	--	1	0.18±0.18
Unknown Mesomammal		0	--	0	--	1	0.22±0.22	0	--

In early seral, I detected higher activity of cows, coyotes, and foxes in relation to other predators in 2015 and I detected higher activity of coyotes in relation to other predators in 2016 (Figures 6 and 7). I also observed coyotes as the most active mammalian predator at camera monitored nests. In grasslands, cows I observed were seven times more active than other predators in 2015 ( $\chi^2 = 77.8$ ,  $P < 0.01$ ,  $df = 7$ ) and 1.3 times more active than coyotes in 2016 ( $\chi^2 = 106.1$ ,  $P < 0.01$ ,  $df = 7$ ). In shrublands, however, detections of all predators I observed were low in both years, with the exception of cows, which were six times more active in both years (2015:  $\chi^2 = 102.3$ ,  $P < 0.01$ ,  $df = 7$ ; 2016:  $\chi^2 = 102.2$ ,  $P < 0.01$ ,  $df = 7$ ). Lastly, in woodlands I observed cows as the most active predator and cows I observed were two times more active than coyotes in both 2015 (Figure 7;  $\chi^2 = 21.6$ ,  $P < 0.01$ ,  $df = 7$ ) and 2016 (Figure 8;  $\chi^2 = 69.6$ ,  $P < 0.01$ ,  $df = 7$ ). Despite having the lowest activity per predator species, I detected the highest species richness in woodlands in comparison to other vegetation types, similar to predator richness seen on nest cameras.

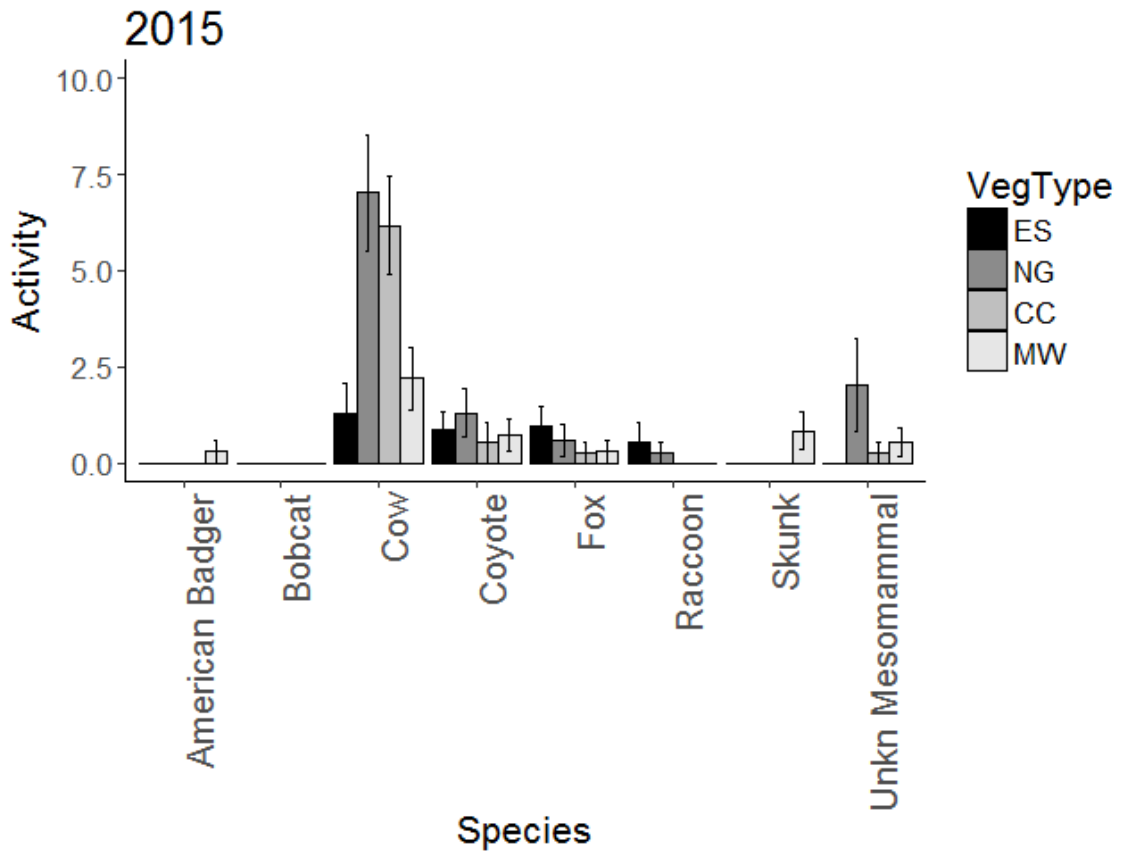


Figure 7. Mean activity (detections per 100 track nights) and standard error bars for meso- and large mammals in early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types on San Antonio Viejo Ranch, Texas, USA, 2015.

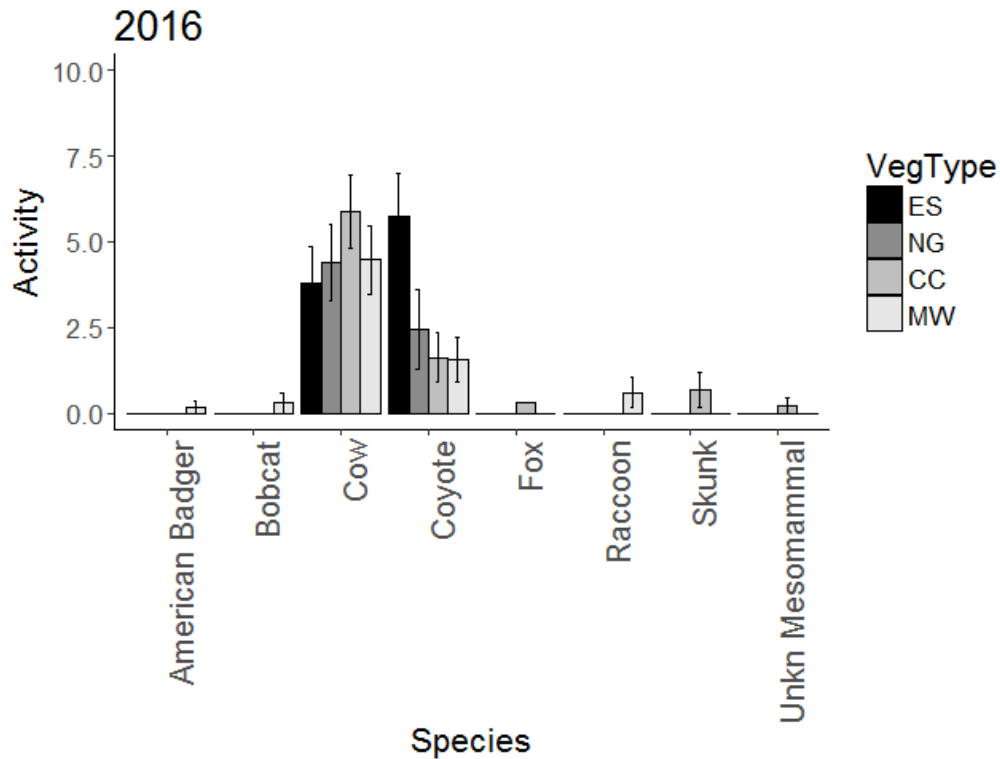


Figure 8. Mean activity (detections per 100 track nights) and error bars for meso- and large mammals in early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types on San Antonio Viejo ranch, Texas, USA, 2016.

*Small Mammals* – We sampled for small mammals for a total of 19,980 trap nights in 2015 and 14,532 nights in 2016. In 2015, I detected grasshopper mice most frequently and Hispid cotton rats most frequently 2016 with 2146 and 782 captures respectively (Tables 15 and 16). I did not observe any difference in Southern plains woodrat activity between years ( $W = 265.5$ ,  $P = 0.11$ ); however, grasshopper mouse activity was seven times higher in 2015 than in 2016 ( $W = 920$ ,  $P < 0.01$ ). Additionally, I observed an increase in Hispid cotton rat activity of seven times in 2016 compared to 2015 ( $W = 265$ ,  $P < 0.05$ ) and an increase in ground squirrel activity of 20 times in 2016 compared to 2015 ( $W = 328$ ,  $P < 0.01$ ).

Table 15. Small mammal detections (n), mean index of activity ( $\bar{x}$ ) as defined as captures per 100 trap nights, and standard error (SE) within early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) types occurring on San Antonio Viejo Ranch, Texas, USA, 2015.

Common Name	Scientific Name	ES		NG		CC		MW	
		<i>n</i>	$\bar{x} \pm SE$	<i>n</i>	$\bar{x} \pm SE$	<i>n</i>	$\bar{x} \pm SE$	<i>n</i>	$\bar{x} \pm SE$
Southern plains woodrat	<i>Neotoma micropus</i>	20	0.91±0.87	34	0.33±0.22	13	0.24±0.23	197	0.93±0.26
Grasshopper mouse	<i>Onychomys leucogaster</i>	282	1.70±0.41	60	2.93±3.38	298	2.52±1.18	1506	6.01±1.53
Hispid cotton rat	<i>Sigmodon hispidus</i>	0	--	0	--	11	0.20±0.20	6	0.03±0.002
Mexican ground squirrel	<i>Ictidomys mexicanus</i>	0	--	3	0.05±0.05	0	--	2	0.01±0.01

Table 16. Small mammal detections ( $n$ ), mean activity ( $\bar{x}$ ) as defined as captures per 100 trap nights, and standard error (SE) within early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) woodland vegetation types occurring on San Antonio Viejo Ranch, Texas, USA, 2016.

Common Name	Scientific Name	ES		NG		CC		MW	
		$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$
Southern plains woodrat	<i>Neotoma micropus</i>	7	0.92±0.87	41	1.89±1.47	12	0.39±0.20	277	1.64±0.58
Grasshopper mouse	<i>Onychomys leucogaster</i>	1	1.71±0.41	1	0.05±0.05	52	4.16±2.67	678	3.84±1.10
Hispid cotton rat	<i>Sigmodon hispidus</i>	0	--	0	--	159	5.15±2.61	622	3.58±1.98
Mexican ground squirrel	<i>Ictidomys mexicanus</i>	0	--	8	0.37±0.26	13	0.46±0.46	101	0.55±0.31

In 2015, I observed substantially higher woodrat activity in early seral and woodland vegetation types by ~three times, however, these differences were not significant ( $\chi^2 = 4.6$ ,  $P = 0.20$ ,  $df = 3$ ). In 2016, I did not detect significant differences in woodrat activity between vegetation types (Figure 9;  $\chi^2 = 1.4$ ,  $P = 0.71$ ,  $df = 3$ ). I also did not detect significant differences between grasshopper mouse activity across vegetation types in either 2015 ( $\chi^2 = 2.8$ ,  $P = 0.43$ ,  $df = 3$ ) or 2016 ( $\chi^2 = 2.1$ ,  $P = 0.56$ ,  $df = 3$ ), but activity was consistently higher in woodlands and decreased as vegetation type changed from shrub dominated to grass dominated. I did not detect significant differences in activity between vegetation types for cotton rats and ground squirrels in 2015 and 2016 at  $\alpha = 0.05$ . I did observe activity that was consistently higher in shrubland and woodland vegetation types than grassland and early seral for cotton rats in both years. I also observed greater activity in woodlands than other vegetation types for ground squirrels (Figures 9 and 10).

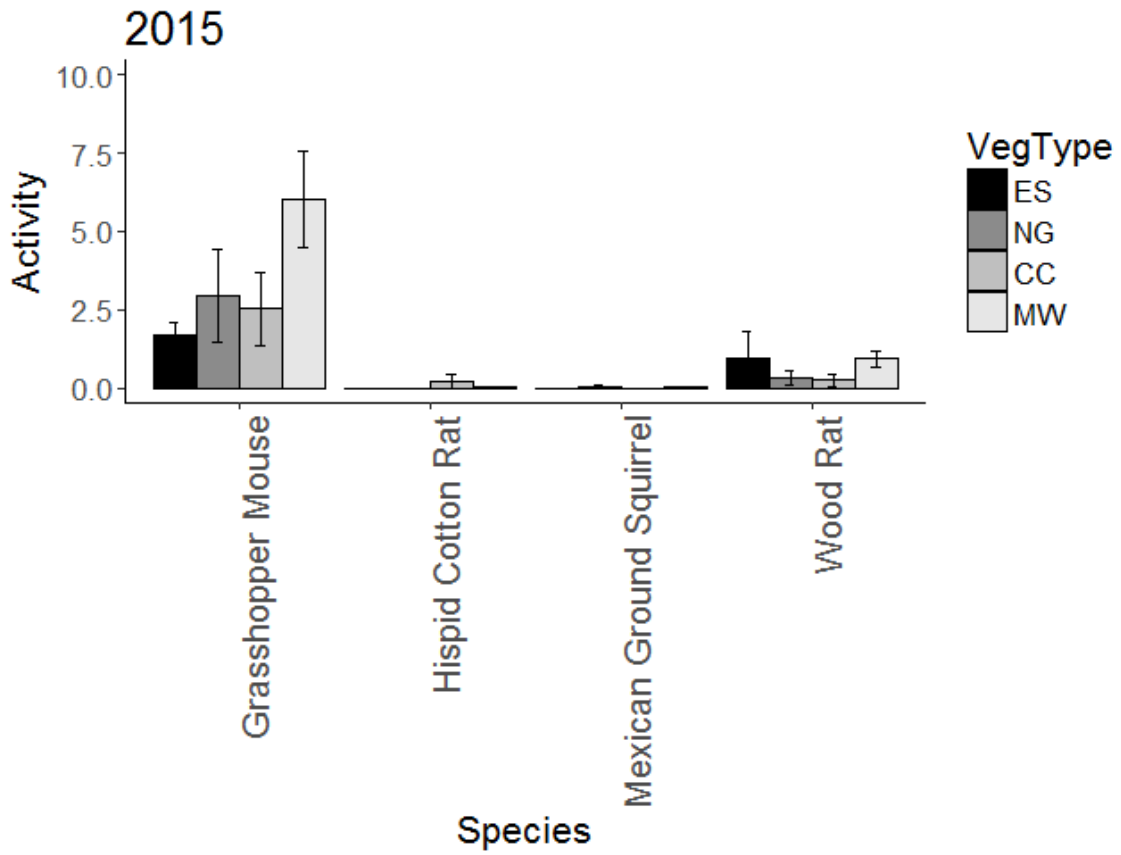


Figure 9. Mean small mammal activity (captures per 100 trap nights) and standard error bars for small mammals in early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types on San Antonio Viejo Ranch, Texas, USA, 2015.



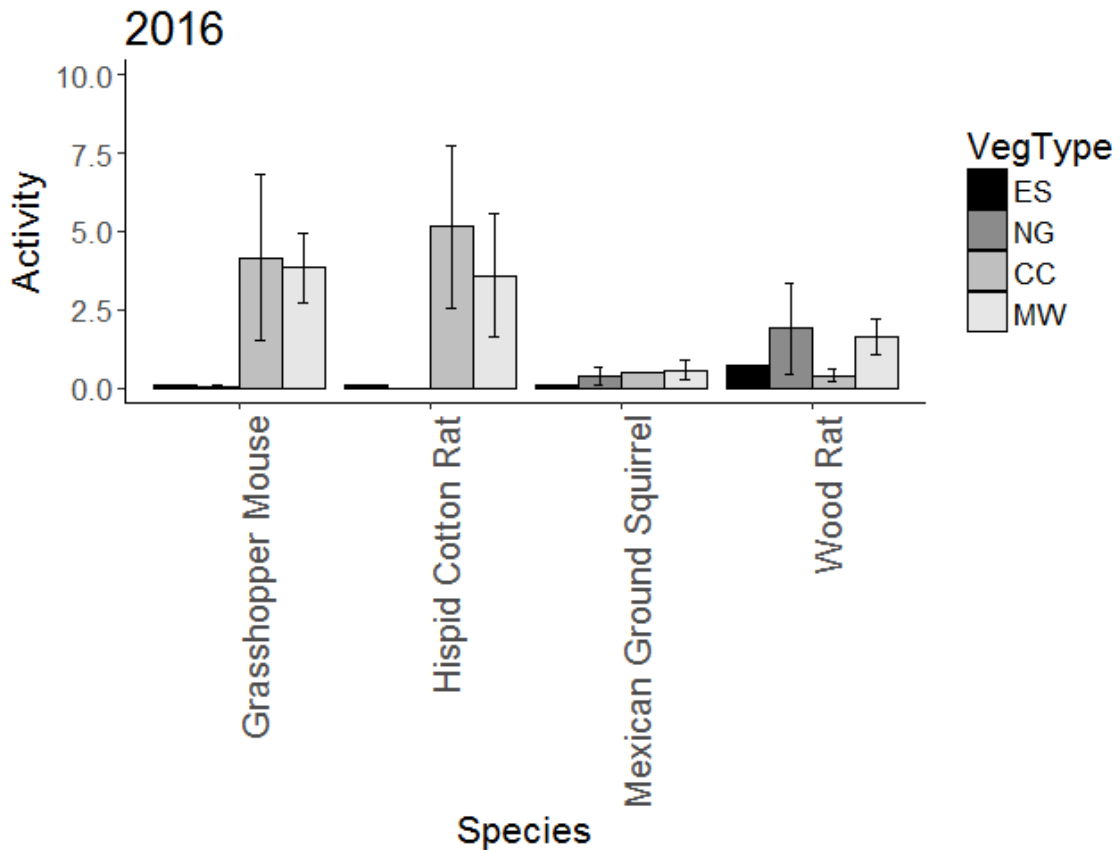


Figure 10. Mean small mammal activity (captures per 100 trap nights) and standard error bars for small mammals in early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types on San Antonio Viejo Ranch, Texas, USA, 2016.

*Herpetofauna* – We sampled herpetofauna for a total of 5,811 days in 2015 and 6,292 days in 2016. In 2015, I detected Texas glossy snakes and whipsnakes more frequently than other snake species, which was not consistent with snake species I detected at camera monitored nests. I detected coralsnakes, rattlesnakes, and the Mexican hognose snake less frequently than other snake species (Table 17). I observed similar trends in 2016, although I detected whipsnakes 1.6 times more frequently, rattlesnakes five times more frequently and did not detect coralsnakes (Table 18).

Table 17. Snake detections ( $n$ ), mean activity ( $\bar{x}$ ) as defined at captures per 100 trap nights, and standard error (SE) within early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types occurring on San Antonio Viejo Ranch, Texas, USA, 2015.

Common Name	Scientific Name	ES		NG		CC		MW	
		$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$
Texas glossy snake	<i>Arizona elegans arenicola</i>	5	0.60±0.36	9	1.1±0.34	9	0.57±0.20	22	0.88±0.27
Texas scarlet snake	<i>Cemophora coccinea lineri</i>	2	0.24±0.03	6	0.72±0.72	0	--	0	--
Rattlesnakes	<i>Crotalus sp.</i>	1	0.12±0.12	1	0.13±0.13	0	--	0	--
Rat snakes	<i>Elaphe sp.</i>	0	--	0	--	1	0.05±0.05	3	0.12±0.09
Mexican Hognose snake	<i>Heterodon kennerlyi</i>	0	--	0	--	0	--	1	0.05±0.05
Texas night snake	<i>Hypsiglena torquata jani</i>	5	0.60±0.36	4	0.48±0.48	1	0.06±0.06	9	0.34±0.19
Kingsnakes	<i>Lampropeltis sp.</i>	0	--	2	0.24±0.24	1	0.06±0.06	1	0.04±0.04
Whipsnakes	<i>Masticophis sp.</i>	5	0.6±0.6	9	1.11±0.34	11	0.66±0.22	11	0.43±0.12
Texas coralsnake	<i>Micrurus tener</i>	0	--	0	--	1	0.05±0.05	1	0.03±0.03
Long-nosed snake	<i>Rhinocheilus lecontei</i>	0	--	0	--	5	0.29±0.18	4	0.16±0.07
Eastern patch-nosed snake	<i>Salvadora grahamiae</i>	0	--	2	0.24±0.24	4	0.25±0.15	5	0.20±0.08

Table 18. Snake detections ( $n$ ), mean activity ( $\bar{x}$ ) as defined as captures per 100 trap nights, and standard error (SE) within early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types occurring on San Antonio Viejo Ranch, Texas, USA, 2016.

Common Name	Scientific Name	ES		NG		CC		MW	
		$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$	$n$	$\bar{x} \pm SE$
Texas glossy snake	<i>Arizona elegans arenicola</i>	3	0.41±0.41	7	0.94±0.12	9	0.11±0.07	30	0.81±0.18
Texas scarlet snake	<i>Cemophora coccinea lineri</i>	0	--	1	0.14±0.14	0	--	0	--
Rattlesnakes	<i>Crotalus atrox</i>	3	0.41±0.41	3	0.40±0.40	1	0.01±0.01	3	0.08±0.06
Rat snakes	<i>Elaphe sp.</i>	0	--	0	--	1	0.01±0.01	0	--
Mexican Hognose snake	<i>Heterodon kennerlyi</i>	0	--	0	--	0	--	2	0.05±0.05
Texas night snake	<i>Hypsiglena torquata jani</i>	1	0.13±0.13	1	0.13±0.13	1	0.01±0.01	3	0.08±0.04
Kingsnakes	<i>Lampropeltis sp.</i>	1	0.13±0.13	0	--	0	--	1	0.03±0.03
Whipsnakes	<i>Masticophis sp.</i>	0	--	0	--	5	0.06±0.02	18	0.50±0.13
Texas coralsnake	<i>Micrurus tener</i>	0	--	0	--	0	--	0	--
Long-nosed snake	<i>Rhinocheilus lecontei</i>	1	0.13±0.13	1	0.13±0.13	1	0.01±0.01	7	0.19±0.08
Eastern patch-nosed snake	<i>Salvadora grahamiae</i>	2	0.27±0.00	0	--	6	0.07±0.02	5	0.13±0.06

Because I did not detect significant differences between years for all snake species, I pooled snakes across year when statistically evaluating differences in activity between vegetation types. For scarlet snakes, I observed significantly higher activity in grassland vegetation types than shrubland and woodland ( $\chi^2 = 10.6$ ,  $P = 0.01$ ,  $df = 3$ ); however, I only detected five scarlet snakes between 2015 and 2016. Snake activity I observed was highest in shrubland vegetation types for patch-nosed snakes, long nosed snakes, rat snakes, coralsnakes, and whipsnakes, which was consistent with snake nest predators I detected on camera. Additionally, I observed greater rattlesnake and kingsnake activity vegetation types dominated by grasses (Figure 11). However, I did not detect significant differences in activity for these species between vegetation types at  $\alpha = 0.05$  (Figure 11).

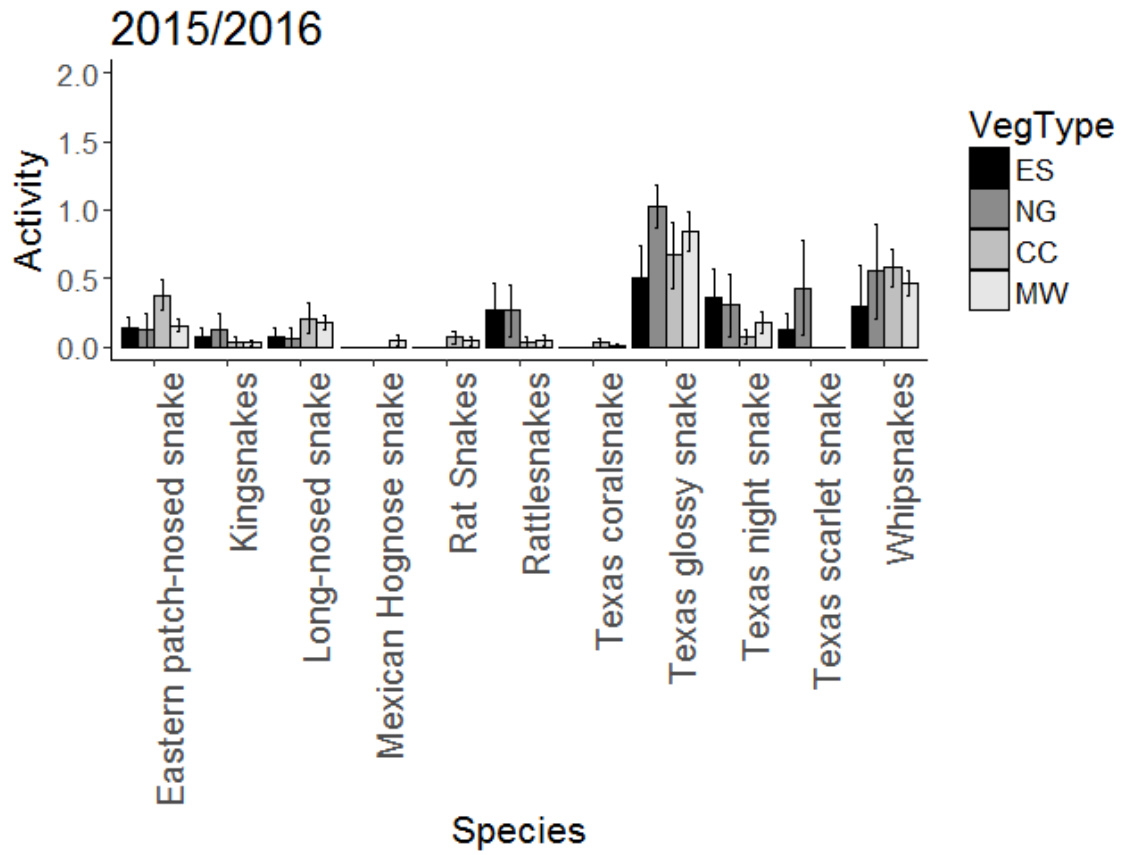


Figure 11. Mean snake activity (captures per 100 trap nights) and error bars for snakes in early seral (ES), native grassland (NG), catclaw shrubland (CC), and mesquite woodland (MW) vegetation types on San Antonio Viejo Ranch, Texas, USA, 2015 and 2016.

## DISCUSSION

Overall, the influence of vegetative characteristics on nest success, predator assemblage, predation probability, and predator activity varied; however, woody vegetation was an important driver of some relationships. Woody vegetation had a significant influence on the success of medium cup nests. At the nest-site scale, woody vegetation was also a significant contributor to increased risk of predation by snakes. Additionally, predator species richness and predator activity (for predominant nest predators) was highest in vegetation types dominated by woody vegetation. Thus, woody vegetation encroachment could substantially decrease nesting success of bird species nesting in semiarid savannas.

Contrary to my hypothesis for my first objective, small increases in percent shrub cover had a negative effect on the success of medium cup nests. As such, shrub encroachment could have substantial negative effects on birds in the region that have this type of nesting ecology. Great plains rat snakes were the predominant predator of medium cup nests in my study and these snakes have been shown to aggregate in areas of woody cover (Klug et al. 2010). Furthermore, predicted snake predation probability and predicted nest success of medium cup nests in my study responded in similar magnitudes (albeit inversely) to increases in shrub cover. These models also indicate a critical threshold of woody vegetation (~ 50% cover), which could be used as a valuable tool when managing these birds.

My hypotheses were not supported when evaluating the effects of concealment and distance to edge on nest success. Avian predation accounted for 50% of events at

small cup nests, therefore, other metrics such as nest height or nest stage may play an important role in driving nest success of this group as previous studies have shown (Conkling et al. 2012). For my exposed nest group, there were significant differences in success between years, indicating that the exposed nature of these nests could make them susceptible to abiotic variables such as rain and temperature that can vary from year to year (Skagen and Adams 2012). Mean rainfall during my study period did not differ between years, however, peak rainfall occurred later in 2016 when compared to 2015 (May as opposed to March), thus, reduced nest success in 2016 may be due to timing of precipitation.

For my second objective, my hypotheses generally were not supported when examining nest predators recorded on camera at the vegetation type scale. I hypothesized that I would detect more mammalian and avian predation events in vegetation types dominated by grass cover, however, these predator detections were variable across vegetation types. Mammal and avian predators may be driven by other factors such as nest stage, as indicated by Conkling et al. (2012). Avian predators were the predominant predator of Black-capped Vireo nests in this south Texas study. Additionally, daily nest survival was lower for nests with nestlings than those with eggs (Conkling et al. 2012). Thus, nests with nestlings as opposed to eggs may be more susceptible to visual predators such as birds and mammals. Snake predators, the predominant predator detected on cameras, occurred predominantly in vegetation types dominated by woody cover, supporting my hypothesis. These observations are consistent with observations in

other studies (Blouin-Demers and Weatherhead 2001, Klug et al. 2010), further supporting the importance of woody cover for snakes.

At the nest site scale (objective three), my hypotheses that increasing woody cover and decreased concealment and distance to edge would increase mammalian predators were generally supported. Coyotes, my predominant mammalian predator, generally depredated nests with low shrub cover and shortest distance to edge relative to other predators. Woody vegetation could increase visual obstruction of nests, thus decreasing predation by these predators. Additionally, coyotes use roads and forest edges as travel routes and for foraging (Heske 1995, Kuehl and Clark 2002), thus mammalian predators are likely to depredate nests closer to edges. I also observed half of coyote predation events at roadrunner nests indicating coyotes may seek roadrunner nests due to the large size of eggs and nestlings to optimize nutritional intake. Additionally, roadrunner nests and adults are large and predators may find nest locations more easily than smaller, conspicuously placed nests.

Nest-site specific vegetation characteristics of nests depredated by avian predators did not generally support my hypotheses. I hypothesized that decreased woody cover, concealment, and distance to edge would result in increased predation rates by avian predators. Woody vegetation cover and concealment was consistent with that of other predators observed at camera monitored nests and was ~10% higher than successful nests. In addition to utilizing visual cues, avian predators may also utilize woody vegetation for perching locations (Bergin et al. 1997), which may explain these observations. Concealment did not differ from other predators, however, avian predators



generally depredated nests closer to an edge relative to other predators. As I hypothesized, avian predators may utilize edges for movement and perching locations to view prey.

As hypothesized, I also observed increased snake predation probability with increases in shrub cover. This explains the decreases in nest success with increased shrub cover in the medium cup nest group as previously discussed. I did not, however, expect for mean shrub cover to be as low as 15% for nests depredated by snakes, although this observation is consistent with other studies examining similar snake species (Klug et al. 2010). Additionally, my hypotheses that decreasing concealment and distance to edge would increase snake predation probability were not supported. Since snake predation events occurred predominantly at night, snakes in my study may rely on non-visual cues for locating prey, therefore, concealment and edge likely does not play an important role.

Although my study was conducted on a working cattle ranch, the influence of Brown-headed Cowbirds on nest success was negligible. Cattle on this ranch graze in large, open pastures and I would often observe them in sparse groups (~5–10 individuals). Thus, the prevalence and effect that cowbirds have on nest success could be linked to the amount of space available for cows to disperse and the location of water sources on this ranch.

I observed only two ant predation events on camera, which was surprising to me due to the prevalence of invasive fire ants in south Texas. I did not sample for fire ant activity on the ranch; however, I would regularly encounter ant hills when searching for

and checking nests. Previous research has indicated despite their prevalence in Texas, ant impacts vary regionally (Campomizzi et al. 2009, Conkling et al. 2012).

For my fourth objective, I hypothesized meso- and large mammal activity was generally higher in vegetation types dominated by grasses than woody plants, however, these differences were not significant for Coyotes, the predominant mammalian nest predator in my study. Coyote activity did not reflect coyote frequency at camera monitored nests. I only recorded five coyote predation events on camera; therefore, activity likely is not an indicator of predation risk. The role of other mammal predators in nest success of birds in this region is negligible.

Small mammal activity was not higher in vegetation types dominated by woody vegetation, contrary to my hypotheses. Small mammal activity has been shown to be driven by smaller scale vegetative features (Thompson and Gese 2013) and food availability which may be an explanation for my results (Reed et al. 2005, Schorr et al. 2007). Despite the high levels of small mammal activity I documented in both 2015 and 2016, I did not observe any small mammal predation events on camera monitored nests. A possible explanation for this was abundant alternative food resources being available during the bird nesting period. Both grasshopper mice and Mexican ground squirrels are known to forage primarily on insects (Edwards 1946, Horner et al. 1964). Ample insects and forbs for small mammals to forage on could be a potential explanation for why small mammal activity did not reflect nest predation in my study.

Lastly, snake activity was variable, depending on the species of snake. Snakes were more active in vegetation types dominated by shrubs, consistent with snake

predation events recorded on camera monitored nests. Activity for rat snakes at sampling locations was low, given that these snakes depredated 33% of all camera nests. Because rat snakes are known to be arboreal (Pierce et al. 2008), the sampling method I used may not adequately sample for snakes with this trait. Thus, activity may be a predictor for nest predation risk, but further study using different methodology may be needed.

## CONCLUSIONS AND MANAGEMENT IMPLICATIONS

My results suggest that woody encroachment in south Texas likely produces negative effects on nesting birds. Even slight increases (~10%) in woody cover resulted in significant increases in nest failure, most likely caused by snake predation. However, woody vegetation clumps (mottes) do offer important locations for breeding, refuge, and foraging for birds and other native wildlife species. Thus, land managers in this region should maintain savanna ecosystems with open grasslands and sparse mesquite mottes. Additionally, these mottes should ideally have ~10% shrub cover and not to exceed 50% to discourage snake predation. Because the majority of predators were rat snakes, which are known to be arboreal, preventing savannas from maturing and transitioning to woodlands should reduce rat snake nest predation and increase nest success. Additionally, impacts of cowbirds and ants on nest success may not be as great as originally believed within this region. The cattle on SAV graze in pastures that can reach up to ~9,800 ha. Thus, the size of pastures and distribution of water sources on this ranch could benefit nesting birds by dispersing cows and cowbirds across the ranch and minimizing their impacts.

Woody vegetation is challenging to manage, however, research has indicated there are several effective means of slowing encroachment or converting shrublands and woodlands back to grassland systems. Reintroducing fire to savanna systems is a valuable management tool shown to reduce woody vegetation in regions of south Texas. Ruthven et al. (2003) saw significant reductions in woody vegetation on rangelands when utilizing winter burns during periods of low or no rainfall. Additionally, Taylor et

al. (2012) indicated high intensity summer burns during periods of low to no rainfall resulted in substantial decreases in woody plants. These high intensity burns can also result in state shifts, transitioning woodlands and shrublands back to grasslands (Taylor et al. 2012). The effectiveness and intensity of prescribed fires are largely dependent on fuel load (Ruthven et al. 2003) and some single treatment fires may not be an effective way of managing woody vegetation (Scifres and Hamilton 1993, Ansley and Jacoby 1998, Ansley and Castellano 2006). Thus, land managers planning to use fire as a land management tool should consider using subsequent fire treatments to maintain grasslands after an initial high intensity treatment is prescribed.

Additionally, herbicide treatments can be an effective alternative to prescribed burning. Two studies conducted in north Texas indicated herbicide treatments were effective at reducing woody vegetation without compromising herbaceous cover (Heaton et al. 2003, Ansley and Castellano 2006). Another study conducted in Oklahoma indicated herbicide treatments reduced woody vegetation temporarily, however, were not an effective long term solution (Stritke et al. 1991). Herbicide treatments are also expensive to implement (Ansley and Castellano 2006), therefore, a combination of prescribed burning and herbicide treatments may be a long-term solution to woody encroachment.

Finally, my study indicates the importance of local-scale research for aiding land owners in managing wildlife populations. Several of my hypotheses were not supported based on my review of similar studies conducted in other areas of the United States. Thus, there likely is not a formula for predicting predation at large scales. Wildlife

biologists should work closely with landowners to help predict the effects of land management on wildlife species. Understanding how vegetation management drives predator assemblage and nesting success at the local scale is paramount to conserving these species.

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