12-2019

A Landscape-Scaled and Community Ecology Approach to Wildlife Corridor Design in South Texas

James A. Stilley
The University of Texas Rio Grande Valley

A LANDSCAPE-SCALED COMMUNITY ECOLOGY APPROACH TO WILDLIFE CORRIDOR DESIGN IN SOUTH TEXAS

A Thesis

by

JAMES A. STILLEY

Submitted to the Graduate College of
The University of Texas Rio Grande Valley
In partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

December 2019

Major Subject: Agricultural, Environmental, and Sustainability

A LANDSCAPE-SCALED AND COMMUNITY ECOLOGY

APPROACH TO WILDLIFE CORRIDOR

DESIGN IN SOUTH TEXAS

A Thesis
by
JAMES A. STILLEY

COMMITTEE MEMBERS

Dr. Christopher Gabler
Chair of Committee

Dr. Owen Temby
Committee Member

Dr. Frank J. Dirrigl Jr.
Committee Member

Mr. W. Boyd Blihovde
Committee Member

December 2019

ABSTRACT

Stilley, James A., A Landscape-Scaled and Community Ecological Approach to Wildlife Corridor Design in South Texas. Master of Science (MS), December, 2019, 75 pp, 17 tables, 23 figures, references, 41 titles

A formidable challenge in landscape ecology is developing a sound resolution to mitigate the impacts of habitat fragmentation and restore connectivity to a degraded landscape. The problem is worldwide, landscapes are becoming primarily anthropogenic and areas set aside for wildlife are small and isolated. Researchers' have developed the concept of the wildlife corridor to remediate this situation but a proper methodology to implement this concept is still in its infancy. This study aims to uncover a quantitative and repeatable wildlife corridor design methodology based on the least cost analysis strategy with both a singular focal taxa approach and a comprehensive community ecology approach. Our study focuses on south Texas as the testing area for our assessment. The study found that neither the focal taxa or community approach were significantly better at protecting the south Texas ecological community but it was successful at creating a methodology for wildlife corridor design.

DEDICATION

The completion of my master thesis would not have been possible without the love and support of my family and friends. I thank my parents and little brother for their support during the course of my thesis. I want especially want to thank my grandmother for her prayers and inspiring me to preserver through this endeavor.

ACKNOWLEDGEMENTS

I am grateful to the professional and technical support provided by Dr. Christopher Gabler. Additionally, I thank Mr. Boyd Blihovde for his support, land manager prospective, and his request for technical assistance in addressing effective land prioritization. I thank all the staff at Laguna Atascosa NWR, from the encompassing South Texas Complex, and from Domain 2 Headquarters in Albuquerque, NM for their support of this project. I thank Dr. Owen Temby for providing a social science prospective and non-specialist review of this study. I thank Dr. Frank J. Dirrigl for reviewing this work from a wildlife ecology prospective and for his helpful recommendations to the methods used in this study. I thank the Friends of Laguna Atascosa NWR for sponsoring this project. I thank Mr. Andy Jones and The Conservation Fund (TCF) for their assistance and comments regarding this project. I thank Mr. Maxwell Pons, TNC, for his technical assistance and review of plant identification. I thank a number of staff members of numerous agencies including TPWD, TXDOT, and NPS for this assistance with this project.

I thank all the private landowners, USFWS, and TCF for allowing access to their respective properties. I want to acknowledge and thank all the members of the Gabler Lab, technicians, fellow graduate students, high scholars, volunteers, Texas Master Naturalists, and USFWS interns whose field and data entry assistance was paramount to the success of this project. I additionally, thank all the staff at UTRGV that provide assistance with different portions of this project including Dr. Rupesh Kariyat, Dr. Saydur Rahman, Ms. Erica Salazar, and numerous others. I additional thank the Graduate College for this opportunity.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Page

CHAPTER I

INTRODUCTION

Landscape connectivity relates how well the spatial characteristics and ecological functions of habitat patches and landscape structure facilitate the movement of wildlife across a landscape (Crooks & Sanjayn 2006). Historically, wildlands were large and continuous enough that connective corridors existed throughout most landscapes without specifically being delineated (Rudnick et al. 2012). In the last two hundred years, the human population has expanded and doubled numerous times (Ezeh et al. 2012) and is projected to reach 9-11 billion people before 2100 (United Nations, 2019). In conjunction with population increase, wilderness areas transformed to an anthropogenic dominated landscape with urban, suburban, and agricultural lands encircling miniscule wilderness parcels (Foley et al. 2011). This rapid human population increases and landscape alteration has created two of the foremost concerns of conservation biology: habitat loss and habitat fragmentation (Rudnick et al. 2012). Habitat loss and fragmentation occur when continuous natural landscapes are modified and transformed into islands of relatively undisturbed land surrounded by a sea of land that is less hospitable for wildlife, such as urban areas and farmland (Crooks & Sanjayn 2006). In response to this global change in landscape composition the scientific community in the field of conservation biology has recently accelerated the development of strategies to preserve and maintain connectivity between remaining habitat fragments (Abrahms et al. 2017). Researchers have proposed the use of wildlife corridors as the method to maintain connectivity, address habitat fragmentation, and restore functionality of biological communities particularily by facilitating connectivity between populations of organisms across a landscape (Abrahms et al. 2017). Although this idea is simple in concept, its implementation has been a formidable challenge to land managers (Abrahms et al. 2017).

There has been extensive research into corridor design, formation, and implemental strategy, yet a universally accepted method does not exist (Rudnick et al. 2012) and may never exist, owing to idiosyncrasies of local environmental conditions and biological communities. In the 1988 management plan developed for Laguna Atascosa, Jahrsdoerfer and Leslie (1988) identified a number of theoretical considerations, including size, width, and structure, that should be addressed in corridor design, however, these considerations were very general and lacked quantifiable distinctions necessary to implement these considerations reliably. Abrahms et al. (2017) stated that mapping the resistance of the landscape toward animal movement and an effective method to delineate corridors is to monitor wildlife use. Two processes, landscape genetics and resource selection, can define landscape resistance. The genetic approach examines the relatedness of wildlife populations and uses their dissimilarity to predict landscape resistance while the resource selection approach examines the natural and life history requirements of a species and assesses the positive and negative relations of landscape features towards that species. In this study we designed our models using the resource selection concept. Crooks and Sanjayan (2006) utilized what Abrahms et al. (2017) later described as the resource selection approach and simplified corridor design to two main components: structural and functional. Structure refers to the geospatial layout of habitats while function relates to the ecology (e.g., habitat use) of a species (Crooks & Sanjayan 2006). They stated that designing these corridors can be done either indirectly by protecting adequate habitat or directly by surveying species movements, and that corridors based on wildlife movement work best if the focal species live in a meta-population fashion (i.e., with multiple populations connected by regular migration of individuals between them). Despite this simple concept, Rudnick et al. (2012) pointed out that a geospatial analysis or theory is needed to incorporate the structural and functional data regardless of method.

Many places have implemented wildlife corridors as part of their management strategy (Crooks & Sanjayan 2006). Rudnick et al. (2012) suggested the most important consideration for choosing a corridor design method is scale. The scales of wildlife corridors vary considerably from wildlife underpasses to regional (also referred to as intermediate or landscape-scale) to continental (corridors that stretch across multiple nations, like the Greater Rocky Mountains Corridor or El Paseo del Jaguar). Additionally, wildlife corridors to date have deficiencies in five areas: species persistence, behavioral ecology, community structure, climate change and the human component (Rudnick et al. 2012). Species persistence is important in determining the theoretical functional life of a wildlife corridor (Rudnick et al. 2012). Knowing the behavioral ecology of a species is advantageous because it will assist in designing a corridor that the target wildlife will use as intended (Rudnick et al. 2012). Community structure is important because wildlife corridors in the past focused on one or a few indicator species, but these corridors were found to have limited utility to the ecosystem, so a community ecology approach should increase the functionality of the corridor for a copious amount of species (Rudnick et al. 2012). Climate change is altering habitats worldwide, thus its inclusion when predicting the future of wildlife population distributions and habitat persistence is essential (Rudnick et al. 2012). Lastly, but perhaps most importantly, is how people will respond to designing, implementing, and living alongside a wildlife corridor (Rudnick et al. 2012).

The determination of sufficient indices and reliably repeatable methodologies to quantify land value or prioritize specific tracts for conservation objectives has been a daunting task (Saura & Pascual-Hortal 2007). Various groups have experimented with different approaches, but a universal methodology has yet to be discovered (Saura & Pascual-Hortal 2007). In addition to prioritizing land for its current conservation value, land prioritization can assess where restoration efforts should be focused to maximize efficient spending when facing budget constraints (Holzmueller et al. 2011). To determine the conservation value of a land parcel or prioritization rank, it is necessary to develop a list of important criteria (Holzmueller et al. 2011) that reflect both biological and socio-ecological attributes (Rohweder et al. 2015). However, Saura and Pascual-Hortal (2007) cautioned researchers and land managers working on these projects that the criteria evaluated and their associated worth have the possibility to allude to false conclusions, so the development of an objective, repeatable, systematic methodology is essential to minimize this problem. In recent years, the U.S. Geological Survey (USGS) Upper Midwest Environmental Sciences Center and the United States Fish and Wildlife Services (USFWS) Morris Wetland Management District, both based in the Midwestern United States, have evaluated ways to rank the conservation priority of management regions based on user-defined criteria (Rohweder et al. 2015). They have developed new software extensions for use in ESRI ArcMap that can make the creation of least cost surface analyses simpler for land managers to implement (Rohweder et al. 2015). Owing largely to their recent creation, these new software extensions, such as LINK (ArcGIS Tools for Conservation Planning), currently lack the testing necessary to assess their utility in other areas (USGS 2005).

Numerous studies have suggested that examining corridors from a community level can meliorate corridor utility for a variety of species essentially improving corridor function (Abrahms et al. 2017). The method we chose to delineate wildlife corridor routes utilized the geospatially termed least cost surface and corridor theory termed resource selection-based resistance surfaces to develop least cost path based corridors. The study demonstrated its utility by its use to develop a corridor in south Texas for a sponsoring land managing agency, the United States Fish and Wildlife Service (USFWS) at Laguna Atascosa NWR. This study developed cost surface models based on empirical, literature, and expert opinion based data on how landscape features influence wildlife use particular for a number of previously identified important local focal taxa species to the Lower Rio Grande Valley (LRGV). The single indicator species for our study region included *Leopardus pardalis albescens* (northern ocelot), *Falco femoralis septentrionalis* (northern aplomado falcon) and *Gopherus berlandieri* (Texas tortoise). Additionally, there was an assessment of the vegetation community which resulted in a model that treated this community as a focal taxa group to see how it compared to single species models.

Ocelots have had a major range reduction in the United States, from once living in four states (Arizona, Louisiana, Arkansas, and Texas) to their current distribution in two populations (80-120 cats total) in southern Texas and a few individuals in Arizona (USFWS 2016). Janecka et al. (2011) examined thirty years of ocelot genetics from Texas and Mexico and found that human-modified landscapes are acting as barriers to ocelot dispersal in South Texas. The barrier to connectivity continues to increase because South Texas is one of the most rapidly growing parts of the United States, and the risk of further habitat loss and fragmentation is substantial (Haines et al. 2006). Landscape-level conservation strategies for wildlife in south Texas will require extensive partnerships and collaboration with private landowners because Texas is unique since Texas is 97% privately owned (Haines et al. 2006). Several other tropical felid species like the Gulf Coast jaguarondi once occupied south Texas but are now considered extirpated from the region (USFWS 2016). The paramount concern of south Texas land managing agencies is that the extirpation of ocelots from the United States is highly likely unless drastic intervention occurs. The northern aplomado falcon's history is even more severe because they were extirpated from South Texas by 1950, presumably from habitat alteration (Jenny et al. 2004). Fortunately, reintroduction efforts in the late 1980s and early 1990s were successful, and current conservation efforts focused on restoring suitable habitat and providing nesting boxes have all contributed to this success (Jenny et al. 2004). On the anthropogenic side of conservation, the USFWS has found that the direct purchasing of land is the most effective means of accomplishing habitat restoration for these endangered species (Jahrsdoerfer & Leslie 1988).

This study included Texas tortoise (*Gopherus berlandieri*) as another single species indicator for wildlife corridor design because this species is both ecologically important to south Texas and other researchers commonly use it as a model species (Webers unpublished). Tortoises are important because all members of the genus *Gopherus* are either state or federally listed as needing protection, and the decline in *Gopherus berlandieri* (Texas tortoise) is known to be a direct result of habitat loss due to conversion form natural lands to agricultural or urban uses (Kazmaier et al. 2001). This species has been extensively studied and many habitat models exist for this species, including some created by the United States Geologic Survey and National Park Service. Studies by Rose and Judd (1975), Bury and Smith (1986), Hellgren et al. (2000), and Kazmaier et al. (2001) have identified the life history, natural history, habitat, and barriers to dispersion for this species, so it was very feasible to utilize the existing information and include this species in our community ecology model. Texas tortoises are a known representative or indicator species of the edges between thornscrub and prairie habitats (Bury & Smith 1986). The particular species included in this study are only found in southern Texas and in northern Mexico (Hellgren et al. 2000), and their range in south Texas only occurs within the Tamaulipan Biotic Province (Kazmaier et al. 2001), which the proposed wildlife corridor is completely contained within this biotic province. Therefore, we utilized the same conceptual framework of the ocelot and aplomado falcon models to develop an additional model specifically for Texas tortoises.

The specific vegetation community examined was the herbaceous composition of forest and grasslands within two land-use histories disturbed and pristine as defined by TPWD TEAM land classification dataset (Elliott 2014) for a gradient of habitat fragments found in the study area. South Texas flora species composition is noted as being primarily herbaceous with 75-80% of all species in the LRGV within this group (Best 2009). Historically the overgrazing of the landscape particularly by sheep caused landscape wide transformations where once higher estimates of herbaceous cover converted into scrubbier mesquite dominated woodlands (Best 2009). Then the introduction of exotic grasses for domestic livestock grazing and soil stabilization has compounded biodiversity loss thus lowering the abundance of native herbaceous species found in the LRGV (Best 2009). These introduced species include *Pennisetum ciliare* (buffelgrass), *Panicum maximum* (Guinea grass), *Dichanthium annulatum* (Kleberg bluestem), *Cynodon dactylon* (Bermuda grass), and *Sorghum halepense* (Johnson grass) (Best 2009). The study assessed four vegetation community parameters: species richness, invasive grass cover, species diversity, and total cover and each were used to develop our cost surface for further analysis.

The objective of this study on a broad scale was to develop a quantitative and repeatable method for assessing the conservation value of land. Second was the development of a method to integrate empirical data (field study derived data) into geospatial models so the ecological community observed in the area of the prospective corridor could be assessed. Lastly, the study assessed the utility of using indicator focal taxa to protect the needs of the ecological community and compared these individual focal taxa models to community cumulative model to evaluate if the community models were more effective than a given individual model at conserving the needs of the focal taxa used to develop the community model. The study evaluated these parameters by using south Texas as a case study. New suitability-based (resource selection based) cost surface models and least cost path-based corridors, to safely traverse these cost surfaces in a way best suited for the focal taxa used to create the cost surface, were developed. The comparison of the similarity of individual focal taxa cost surfaces and corridors to each other and to community cumulative cost surface and corridor models developed by combining all the individual focal taxa models together were evaluated. The first hypothesis was that the single taxa models will differ from each other indicating that these individual focal taxa are not ideal indicator species for the south Texas ecological community. The second hypothesis was multi-taxa-based community cumulative models will be significantly better at integrating the needs of the ecological community in both its cost surface and corridor models. The third hypothesis is that the inclusion of anthropogenic considerations or examining the human dimension side of the landscape will narrow the number of viable corridor options a land managing agency can realistically pursue. Lastly the resulting corridor that satisfies the third hypothesis can be used to assign a prioritization ranking to the individual parcels so a land managing agency can stepwise acquire property for the corridor. A discussion on how to perform this last step is included;

however, due to the sensitivity of the information contained within this analysis, the results were omitted from this publication.

CHAPTER II

METHODS

Study System

The main study took place on over a dozen private ranches and farms and various portions of Laguna Atascosa National Wildlife Refuge (NWR), a fragmented wildlife refuge. All study sites are in the northeastern corner of Cameron County, Texas, an area bordered by the Gulf of Mexico and falls on the international boundary with Mexico. Cameron County is part of the Lower Rio Grande Valley (LRGV) region, a well-known transition zone between temperate and tropical climates, which contributes to its high species diversity (Leslie 2016). The LRGV is home to 19 federally threatened or endangered species and 60 state listed species, with some of the most iconic being *Leopardus pardalis albescens* (northern ocelot), *Puma yagouaroundi cacomitli* (Gulf Coast jaguarundi), and *Falco femoralis septentrionalis* (northern aplomado falcon) (Leslie 2016).

Laguna Atascosa NWR is 36,359 hectares of coastal salt prairie, freshwater wetlands, tidal flats, sand dunes, and thornscrub forest (Leslie 2016). The private ranches and farms have the same habitats found on the wildlife refuge, but with higher levels of human disturbance. The amount of protected native habitat on nearby private lands varies, with some entirely in a conservation easement, some being restored voluntarily by landowners, and others with no formal protection. The natural landscape once present on privately owned farmland or rangeland is highly degraded due to constant anthropogenic use. The LRGV has experienced over a 95%

reduction in native thornscrub habitat and a major reduction in riparian forests (Leslie 2016). This habitat loss and the resulting habitat fragmentation are the primary reasons for the decline of many of the species found in the LRGV (Leslie 2016). The critical habitat for the endangered felids consists of a short canopy (2-4 m) thornscrub forest dominated by a canopy of *Prosopis glandulosa var. glandulosa* (honey mesquite) and other *Acacia* species, with a diverse understory of up to 90 species (Leslie 2016). Another important habitat feature is the Tamaulipan loma shrublands, which are windblown clay sediment deposits that form small, xeric, subtropical, shrubby islands in the middle of the salt prairie and tidal flats (Leslie 2016). The other major native landscape is the salt prairie, which consists of *Borrichia frutescens* (Sea ox-eye daisy), *Spartina spartinae* (Gulf cordgrass) and other species non-woody grasses and forbs. The soils in the study area are over 90% clay and loam soils with 3% considered sandy (Jahrsdoerfer & Leslie 1988). The average annual rainfall is between 38 to 76 cm and is very erratic and seasonal, with temperatures that range from 10ºC in winter to 36ºC in summer (Jahrsdoerfer & Leslie 1988). The topography of the area is very flat with elevations of 0-10 m and slopes of less than one percent (Horne et al. 2009). The individual sites used throughout the study and for the different survey protocols will vary, but will consist of thornscrub (forests) and salt prairie (grassland) habitats with variable amounts of disturbance.

The Ocelot and Aplomado Falcon Models

We began this study by utilizing a simplified computer modeling approach with methods similar to those used by both Holzmueller et al. (2011) and Rohweder et al. (2015) to determine the associated rank of land parcels, with the exception that a standard cost surface analysis following Rouget et al. (2005) was utilized. In this preliminary investigation, the study examined two endangered species, *Leopardus pardalis albescens* (northern ocelot) and *Falco femoralis* *septentrionalis* (northern aplomado falcon) as focal species to assess the biological value and guide the weighting scheme of various attributes across the landscape. The selection of these two species are based on the recommendation of USFWS and non-governmental organizations (NGOs) active in the region since they are both considered important species for the Rio Grande Valley region of southernmost Texas.

This model took into account nine variables from publically available data sources: slope, water, artificial nesting structures, parcel size, soils, road speed, habitat, and the presence of windmills (Table 1). Local experts identified these variables as important during the first landscape prioritization meeting (Stilley et al. 2018), and the model was refined based on a literature review of the biology of the two focal species and their responses to anthropogenic barriers. Not all of the nine variables used in the model are applicable for both focal species, but are listed together for simplicity (Table 2). Despite south Texas lacking significant changes in topography, slope is included for the least cost analysis to run properly and was given the lowest possible weight (1%) however as the study progressed elevation was discovered to be a major abiotic factor influencing vegetation communities so its weight was re-evaluated. The model included the presence of water, and both fresh and brackish sources of water were included. The data source rated ephemeral water sources with a score of 1-9 and permanent water sources with a score of 9-12, so these values were used in the model creation (Table 2). The weight assigned for water was selected by the model creator and are influenced by how the literature stressed the importance of these parameters to a given focal species. Therefore, water weight varied for the different focal taxa; it was considered an important factor for ocelots but is less important for aplomado falcons (Table 2). This general approach was used to assess all the other variables, with the exact values listed in Table 2. For the soils (Table 3) and habitat types (Table 4) the high diversity in types for these factors was too extensive to have listed in the model table so they were separated for the reader's convenience.

Several model variables only applied to one species, namely artificial nest availability for aplomado falcons and road speeds and soils for ocelots. Road mortality is a major source of ocelot mortality in south Texas (Janecka et al. 2011), and the model assumes that roads with speed limits of 40 miles per hour are safer (Forman et al. 2003). This assumption is based on the Forman et al. (2003) 'Road Ecology Science and Solutions' recommendations for most animal species. Soils are the basis of the USFWS' thornscrub restoration strategy, with soil selection origins related to an article by Harveson et al. (2004) (Table 3). Scores for particular habitat types were assigned based on habitat preferences discussed for each focal species in the literature (Table 4). The parcel size classes were based on common land parcel size classes considered in conservation objectives by the USFWS. A binary code for presence of windmills was used in the analysis, but presence of windmills had an unexpectedly strong influence on results, so this document only includes the models that excluded windmills.

To identify specific parcels of land for prioritization, we performed a least cost or resistance surface analysis using ESRI ArcGIS 10.6 software. We organized and preformed multiple geospatial analyses on the factors specified above. The original variables evaluated were downloaded from reputable sources (TPWD, USGS, USFWS, UN, TXDOT, and Cameron County Appraiser Office) and condensed to form a shortlist of classifications (Table 1). These files were clipped to a specific area of interest and all the files were converted to raster format to begin the analysis. The combination strategy is conceptually illustrated in Figure 1 and the Arcmap 10.6 coding view of the model is shown in Figure 2. The weight scheme for each species is based on the biology of that species, with higher weights assigned to those factors found to most limit their distribution according to a literature review and previous Laguna Atascosa NWR wildlife management recommendations (Leslie 2016; USFWS 2016) (Tables 1, 2, 3, & 4).

Texas Tortoise

The development of the geospatial model was similar to the method described for the ocelot and aplomado falcon. The tortoise model utilized six input factors split into three ecologically important factors (lomas, forest canopy and forest edge) and three anthropogenic factors (road speed, protected areas, and urban avoidance) (Table 5). Originally, the model included all of Cameron County but for our analysis presented here, only a small portion on the eastern half of the county excluding the barrier island was included. This model's original larger extent was to satisfy the requirements of a geospatial science class and to assist a partnering agency, the Conservation Fund, with a comparable model for their LRGV wide Texas tortoise model.

Ecological factor data was provided by the USGS with species-specific modifications. To identify lomas, a USGS digital elevation model (DEM) layer with 3 m spatial resolution was acquired, and spatial analyst tools were used to determine areas of high slope. The slope tool was then used to create a layer that clearly showed lomas and had distinct raster values for lomas, which were used to build the cost surface. The USGS forest canopy and forest edge layers required only clipping to the area of interest.

Anthropogenic data came from various sources and required some processing. Multiple layers were combined to create the road speed layer. First, the all roads shapefile from the US Census Bureau Tiger Products website was joined to a table of road speeds from the Texas Department of Transportation (TXDOT). Then two point shapefiles from TXDOT and USFWS containing the locations of wildlife crossing were merged, and this point layer was added to the map with polyline roads using the union function. Next, this file was edited so the wildlife crossings had a road speed value of 0, relevant roads missing values were corrected to true values, and the numerous surface streets lacking speeds outside the focal area were given a value of 5 to expedite the table editing process. This corrected layer was then converted to raster format and the associated scaling value for the expected survivorship of tortoises crossing roads was based on Forman et al. (2003) recommendations. The protected area layer was created from national and state level protection area boundary files from the TPWD Earth Data website. Different protected areas were merged and the union function was used to overlay them on a Cameron county map. The urban avoidance layer was created by the USGS, likely from analyses of satellite imagery, and required only clipping to the focal area.

To create the cost surface, a new model was created with the six layers described, and the reclassify tool was used to prepare the layers for analyses. Tortoise preference classification schemes for each layer were created based on literature about specific Texas tortoise needs, or based on general wildlife parameters if unknown for tortoises, which was the case for assessing the risk of roadways (Table 5). Once the layers were reclassified, the weighted overlay tool was used to assign weights and combine all the input rasters and create a single cost surface output layer.

The final cost surface output layer had eight values associated with the map of Texas tortoises likelihood classes for Cameron county. These eight values were grouped by quartiles and labels as four tiers (optimal (tier 1), sub-optimal (tier 2), marginally suitable (tier 3), and not suitable (tier 4)). This method that used four quartiles to define tiers out of a variable number of potential cost surface layer values was used in all cost surface models. The study assumed that each quartile represented a preference class (optimal, sub-optimal, marginally suitable, and not suitable) for that given taxa and was given a tier (1, 2, 3, or 4) to represent this. This study compared our model to the USGS GAP Analysis habitat map for Texas tortoises, which only shows the binary presence or absence of tortoises, to assess the model's accuracy. For a more rigorous comparison and to decrease analysis bias, multiple classification groups were compared to both the separate presence and absence areas of the USGS GAP Analysis. These groups used for comparison are based on the four tiers previously described and each group was exported as distinct shapefile layer. Each group's shapefile was then compared to the GAP Analysis shapefiles and intersection maps showing the area where the two models overlapped. Then a union was preformed between each cost surface group layer and the USGS GAP Analysis, and again the total area was recorded. Finally, a Jaccard coefficient was calculated and an associated p-value was determined to test for significant differences between classifications.

Vegetation Community Surveys

This study investigated plant community structure in a manner that could additionally be used to classify Lepidoperan communities using a full factorial design with individual study sites representing different treatments for patch size, habitat type, and land-use history. The plant community study examined 2 habitat types (coastal prairie (grassland) and thornscrub forest (forest)), 2 land-use histories (pristine or disturbed), and a gradient of 8 patch sizes (total n = 32); patch sizes were sub-classified into small (30 ha or less) and medium (greater than 30 ha) based on a prior ocelot study by Jackson et al. (2005). It was later classified using a less than 20 ha which was an important cutoff for a Lepidopteran diversity in Skorka et al. (2007) study and the upper cutoff of 50 or 100 ha was based on expert opinion of important cutoffs for vegetation communities.

Lepidopteran surveys typically try to maximize coverage of a habitat by having lengthy (several kilometers) or numerous transects (Kadlec et al. 2012) so our vegetation survey transects systematically varied in length which Lonard and Judd (2002) showed is still valid to use in assessing species richness and diversity of plants in the LRGV. The reason for the transect length variation include; (a) substantially sample a habitat fragment while staying within private property boundaries, (b) maximize sampling effort with a minimum workforce, and (c) employ systematic and repeatable protocols. We used a gradient of transect lengths for small patches (50 m for 1 ha, 100 m for 5 ha, 150 m for 10 ha, and 200 m for 20 ha) and a standard length of 200 m broken into two 100 m transects for patches greater than 20 ha. The paired 100 m transects for patches greater than 20 ha had a minimum spacing of 16 m, which was found to be the minimum distance to observe differences in butterfly composition (Haddad 1999).

We used these same transects for the vegetation community surveys and used a belt transect with both inner and outer plots that are a loosely similar method to Gehlhausen et al. (2000). To do this we placed a 0.25 m^2 quadrate on a single side of a transect every ten meters starting at zero along the transects described earlier. We then classified percent cover to within 1% for all vegetation within this plot and we included bare soil and detritus as a category. Then we took a 1 m^2 quadrate and placed it around the meter mark in four places effectively creating a 4 m^2 plot with the associated meter mark as its centroid. For this outer plot (4 m^2) we recorded the presence of any herbaceous species not present within the 0.25 m^2 plot for all grassland sites. For forested sites we used the out plot to quantify wood vegetation in a similar manner to Gehlhausen et al. (2000) where we recorded only wood species with a dbh greater than 2 cm and any cacti species that were within 2 m of the meter mark.

The habitats being assessed were classified into two general categories: forest or grassland. Land-use history is related to the successional state of the habitat, which was classified as either 'disturbed' or 'pristine' based on the date of the last major land change as provided by the Texas Ecosystem Analytical Mapper (TEAM) Assessment Website (TPWD 2018). The study characterized the landscape utilizing ArcGIS program ArcMap 10.6 to develop a list identifying fragmented forest and grassland habitats that represented a gradient of habitat sizes and a variety of land use histories that occurred within the area of interest for the corridor and on privately owned lands. We then used a series of queries to identify the fragmented habitats that occurred within 300 m of an access road and on lands with cooperative land owners, and then to filter these habitats based on their size class and their designation as either disturbed or pristine forest or grassland habitat. Most Lepidopteron surveys to date have assessed fragmented habitats less than 20 ha, so our study sites included similarly-sized fragments (1, 5, 10, and 20 ha) for direct comparison (Koh & Sodhi 2004). Recent Lepidopteron studies have found that butterfly species diversity can assess the quality of habitats much larger than traditionally suspected (Koh &Sodhi 2004). However, the availability of larger fragment sizes within the study area that met all the other criteria was limited, so the largest forest and grassland sites and the pristine and disturbed sites were different sizes ($n < 32$) (Fig. 3). Habitat distinctions were validated via ground-truthing with a baseline vegetation survey similar to Skorka et al. (2007).

Determining Weighting Schemes

In order to develop weighting schemes for our models we followed the recommendations of Rohweder et al. (2015), which were to hold meetings with local experts to develop weighting scheme values. We held two such meeting called the Landscape Prioritization Meetings at

Laguna Atascosa NWR in June 2018 and in April 2019. The goal of the first meeting was to rank the importance of different biological and anthropogenic factors for mammals (ocelot), birds (aplomado falcon), and general ecology (plant community). To accomplish this task our meeting included USFWS wildlife biologists from the South Texas Refuge Complex (STRC), refuge managers, USFWS realty experts, elected board members of refuge friends' groups, partnering agency (The Conservation Fund and Nature Conservancy), and University faculty. These experts viewed a background presentation into the purpose of the meeting and the importance of developing weighting schemes for model development. Then they divided into four groups: mammals, bird, general ecology, and realty; with the task to develop a list of important biological and anthropogenic factors. At the end of the exercise, each group submitted a ranked list of factors the experts demined important to species conservation. Then once prototype models were developed a second landscape prioritization meeting was held. This meeting's task was to agree upon a specific origin and destination location for least cost path analysis and ultimately for to serve as the bases of the coastal corridor routes. An additional task at this meeting was to develop a ranked list of goals a corridor should aim to achieve so we could assess which corridor route would best meet the goal of our sponsoring land managing agency, USFWS. The study applied the results of both of these meeting to develop the models' weighting scheme and if a weighting scheme was not developed for a given taxa then equal weights were used as in the case for Texas tortoise. The results of the corridor goal survey from the second meeting guided our selection of the most successful models options used to test the third hypothesis.

Statistical Analysis of Empirical Data and Assessing Model Similarity

The only empirical data based model included in this study is the plant community focal taxa model. This model was developed from the assessment of four vegetation community parameters; species richness, invasive grass cover, species diversity, and total cover. The study used a 3-way analysis of variance (ANOVA) to determine which factors; habitat type, land-use history and patch size influenced the data. Then a Fisher's least significant difference (LSD) test with a post hoc Tukey correction were used to assess the grouping of the data. All figures are plotted using 95% confidence intervals so some figures may contain significant differences even if the confidence intervals overlap. Then using regression analysis, the study examined if assessing patch size as a continuous variable would elude to significant trends.

The assessment of models utilized the Jaccard similarity index with GIS corrections (Real 1999) to evaluate the similarity of models. The Jaccard similarity index rely on the concept that the area overlapped by two models (intersect) divided by the total area of the models whose area exclusively belong to either group (union) will result in a value that can be interpreted as the models similarity. This index is unique because it indicates both statistically different or statistically similar where a left tailed (lower value) result indicating significant differences between models and a right tail (higher value) result indicating statistically similar results. An intermediate value is interpreted as inconclusive, thus meaning that random chance could explain a higher or lower Jaccard similarity index value within this intermediate zone. The Jaccard similarity index was used to assess both cost surface model similarity and least cost path with a standard buffer addition similarity. To accurately compare cost surface models' similarity each models tier had to be separated and compared independent of the other tiers otherwise the analysis could not distinguish between tier classes and falsely conclude there is 100% overlap.

The least cost path original output is only a single pixel size in width so using a standard corridor buffer width is necessary so paths have a better chance at overlapping. Our study used 300 m as the buffer width because 300 m is considered the minimum width for a landscape-scaled corridor (Fleury & Brown 1997).

Parcel Based Land Prioritization Scheme

If the land managing agency would like to evaluate the parcels in priority order within a given corridor route the method to do this is as follows. Use the select by attribute function to select all parcels intersecting the desired corridor route. Then export these parcels as a new layer. Then preform a dissolve function on the new applicable parcels layer to create a clipping layer. Use this clipping layer to clip the applicable region of the same corridor's vector formatted cost surface model layer (you may need to convert your cost surface layer from a raster to a vector format using the raster to polygon conversion tool). Now you will merge your newly clipped cost surface layer and your original applicable parcels layer. Now you will need to ensure the spatial analyst extension is selected. Then you will take this newly created merge layer and run two zonal statistic calculations, first run a majority value zonal statistic with your zones being defined by that original applicable parcels layer then you will do the same for the minimal value. Then you will export the results of these two zonal statistics to a spreadsheet application such as excel. Then use a 90% majority value and 10% minority value, or your respective sponsoring land managing agency decided weighting scheme, to create a summarized biological value for each parcel of land. Then ensuring these summarized column table values share a similar column to the attribute table of the original applicable land parcel sheet (ex. parcel id number) preform a join. Now you can use a four tier based symbology scheme where you classify your property values by quartile. The quartiles will serve as your guide to first, second, third, and fourth priority options for land acquisition or conservation easement negotiations. An additional step for future studies would be to survey landowners' conservation attitude and use values from these surveys to amend your priority scheme. Then you can use this information to create a second map with property parcels defined by conservation attitude. Additionally, you can make a summarized output by creating a new priority scheme base 50% on the biological rank (which is still 90% majority value and 10% minimum value of the original cost surface area that falls within a given parcel) and 50% on the conservation attitude dataset.

Corridor Validation Suggestions

To validate the corridor, the investigating party has a variety of options but only two such strategies are discussed here, one based on James and Stuart-Smith (2000) and the other by Haddad (1999). When designing a route for a ground-based species such as an ocelot follow the recommendation of James and Stuart-Smith (2000) where you plot collected telemetry points within the area of the corridor using your original Area of Interest (AOI) clipping layer for corridor range extent. Then take your original raster least cost path for your desired corridor and convert it to a vector format using the raster to polyline tool. Make sure you do not use the corridor with the 300m buffer in this analysis. Then using the assign random points tool to add a desired number of sample points to your vector based least cost path result (try to use the fewest number of sample points possible to help manage the data output for future analyzes). Use the near function to calculate the distance of all the sample points on your least cost path to all telemetry points with a defined distance. I would base this distance on the maximum known distance the animal species chosen to validate the corridor would move in a single day. Then use an array function in excel to find the most minimal distance value for each telemetry point. The true distance between a telemetry point and the least cost path will be the minimum most value of that telemetry point to a sample point on the least cost path. Then place all these minimum values into a separate sheet and run a normality test of your choice. If the points are normally distributed, then your path should be a theoretical corridor already in existence. Future studies can use this approach to validate if wildlife are using an implemented wildlife corridor following completion of the corridor and following successful regrowth of all desired habitats within that corridor route. Alternatively, you can use the method outlined in Haddad (1999) where you mark wildlife (in his case butterflies) on both ends of wildlife corridor with a unique identifier. Then the validation is completed by examining a target species (ex. butterflies), assessing numerous study areas for the presences of marked individuals both within, outside, and at the ends of your corridor for a significant duration (ex. fall migration of butterflies). The validation is successful in mark individuals are observed moving to study areas within the corridor as opposed to outside the corridor in a progression from one end to the other. This same approach theoretically could be used for marked medium to large mammals (whether naturally or artificially) and assessed using well-positioned camera trap stations throughout, adjacent, and outside the corridor. However, these alternative approaches are best suited only after the corridor implementation phase has concluded, which would occur after the successful restoration of desired habitats. On the other hand, the first method by James and Stuart-Smith (2000) could be used with existing data even before a human defined corridor exists.

CHAPTER III

RESULTS

Ocelot and Aplomado Falcon Model

The developed ocelot and aplomado falcon cost surfaces used very similar inputs but resulted in very different model outputs. These cost surfaces were originally part of a much larger comprehensive cost surface model that included a total of 6 input cost surfaces given to our sponsoring land management agency as a technical report. These six layers were combined in a similar manner stated in the method section title "parcel based land prioritization scheme" but none of these analyzes are included due to the sensitive nature of the results. The technical report included a Getis-Ord Gi Spatial Statistical analysis and detected the parcel prioritization has a significantly high clustered distribution ($z = 4.438543$, $p < 0.000009$).

Tortoise Model

The tortoise model was separated into 4 distinct tiers (optimal, sub-optimal, marginally suitable, and not suitable) that predict the likelihood of detecting a Texas tortoise. To validate this model, the Jaccard similarity index was used and Table 7 illustrates how to interpret the Jaccard similarity index assessment in this study since $N > 100$ for all models where a left tailed (lower value) result indicating significant differences between models and a right tail (higher value) result indicating statistically similar results. An intermediate value is interpreted as inconclusive, thus meaning that random chance could explain a higher or lower Jaccard similarity index value within this intermediate zone. The Texas tortoise computer model generated in this study was only statistically similar to the USGS Texas Tortoise Gap analysis when the marginally suitable and not suitable (Tier 3 & 4) categories where combined (J = 0.655, $p < 0.001$) (Table 9). The rest of the generated Texas tortoise model categories were statistically different from the USGS Texas Tortoise Gap analysis (Table 9) and an illustration of this model is in Figure 5.

Vegetation Community

The plant community species richness, invasive grass cover, Shannon diversity, and total cover were assessed for two habitat types (forest and grasslands), two land-use histories (pristine and disturbed), and fragment size. The sizes were assessed as a continuous gradient or in three categorical size classifications including less than or greater than 30 ha, less than 20 ha, 20 to 50 ha, and greater than 50 ha and a less than 20, 20 to 100 ha and a greater than 100 ha. In general, most factors were not significantly different from one another except for a few notable exceptions. For species richness, the assessment of just forest and grassland sites independent of other factors (Fig. 6) and of land-use histories independent of other factors (Fig. 7) both lacked significant differences (Table 10). However, with the addition of fragment size (<30 ha & >30 ha) with habitat type resulted in a significant post hoc comparison using the Tukey LSD test. Forest greater than 30 ha (M = 22.75, SE = 4.5) were found to have higher richness than forest less than 30 ha (M = 8.33, SE = 4.11) and grasslands larger than 30 ha (M = 6.80, SE = 3.77) (Fig. 8). While grasslands less than 30 ha (M=10.58, SE = 4.11) were not significantly different than the other habitat or size class for this comparison. Additionally, species richness when classified by habitat type and the size classes, less than 20 ha, 20 to 50 ha and greater than 50 ha, had significantly higher richness for forest of an intermediate size (20-50 ha) while all other sizes and groups were not different from one another (Fig. 9).

Alternatively, to species richness, invasive grasses cover did show a significant different by habitat type with forest (M = 37.17, SE = 6.26) having higher invasive grass cover than grasslands (M = 8.75, SE = 5.73) (Fig. 10). The significant reactions for invasive grass cover occurred for habitat, size class (<30 ha & >30 ha), the interaction of history and patch size, and the interaction of history, habitat, and patch size but only this last interaction was used to populate the model (Table 11). Additionally when examining the significant ANOVA interactions for habitat type and land-use history and habitat and the size classes based around a 30 ha transition (Table 12). By assessing these groups separately the study found that disturbed forest less than 30 ha (M = 77.8030, SE = 13.08) had significantly higher invasive grass cover while pristine grasslands greater than 30 ha (M = 0.0364, SE = 8.27) had significantly lower invasive grass cover (Fig 11). However, land-use history by itself was not significantly different (Fig. 12). The species diversity of these habitat classifications showed patterns more similar to species richness. When assessing habitat type and land-use history no significant difference where detected. However, once you included the two-size class system based on three size classes (less than 20 ha, 20 to 50 ha, and greater than 50 ha and a less than 20, 20 to 100 ha and a greater than 100 ha) a pattern emerged. Both forest of an intermediate size whether 20 to 50 ha (Fig. 13) or 20 to 100 ha (Fig. 14) were higher than the other disturbed categories. In fact, the intermediate forest within the 20 to 100 ha category had a significant ($p < 0.05$) LSD test value. However, despite having some differences among size classes we failed to find a significantly positive relationship when assessing species diversity by patch size as a continuous variable (Fig. 15).

Total cover was the only plant community metric to have a significant 3-way ANOVA and post hoc comparison using the Turkey LSD test for habitat type and quality independent of fragment size (Table 13). This was seen with disturbed forest (M = 81.4, SE = 13.8) having the higher total herbaceous compared to disturbed grasslands (M = 23.7, SE = 12.6) that had the lowest (Fig. 16). Both pristine forest (M = 62.8, SE = 12.6) and pristine grassland (M = 50.9, SE = 11.5) were similar to both each other and to the previous mentioned habitat type and land-use histories.

Interaction between Individual Focal Taxa Cost Surface Model

In general, the individual models are significantly different to each other thus confirming this study's first hypothesis. Despite this trend of general disagreement between cost surface models, there are a few exceptions particularly for sub-optimal (tier 2) habitats. The cost surface models that had significant similarity included the ocelot sub-optimal (tier 2) habitat and the plant community tier 2 (JSI = 0.69863, $p < 0.001$) and the ocelot tier 2 and the aplomado falcon tier 2 (JSI = 0.64860, $p < 0.001$) (Table 14). Many cost surface models were neither significantly different or significantly similar to the Texas tortoise marginally suitable (tier 3) including plant community tier 2, ocelot tier 3, and aplomado falcon tier 2 and 3 (Table 14). The plant community tier 2 and the aplomado falcon tier 2 were neither significantly similar or significantly different.

Interaction of All Communities Cost Surface Models

The general trend for the comparison of community models to individual focal taxa models is that they are significantly different from each other except for some of the tier 2 and a tier 3. This result rejects our second hypothesis that community-based models will better encompass the needs of multiple individual focal taxa then any given focal taxa model. The areas with the most agreement between cost surface models occurred particularly for the community with equal weights models sub-optimal habitat (tier 2) and the tier 2 levels for the second community model (ocelot preference community) (JSI = 0.74641, $p < 0.001$), plant community focal taxa model (JSI = 0.60537, $p < 0.01$), and the ocelot model (JSI = 0.58252, $p < 0.01$) (Table 15). The only other significant similarity occurred with the ocelot preference weighted community model's marginally suitable (tier 3) and the ocelot focal taxa model tier 3

(JSI = 0.53597, $p < 0.05$) (Table 15). There are areas where the Jaccard similarity index fail to illustrate whether models were significantly similar or significantly different and these occurred in three places in both community models. For the equal weights community model this occurred once in between two models tier 2 and twice between the other community model and a single focal taxa model (Table 15). For the ocelot preference community model, which has the inconclusive areas between the tier 3 of the community with equal weights model and its tier 3, all the other areas of neither difference or similarity occurred between tier 2 of the plant community, ocelot and aplomado faclon (Table 15).

Interaction between All Models' Least Cost Path Corridors and Land Manager Considerations

The comparison of focal taxa corridor models strongly supports our first hypothesis, that a given individual focal taxa model would fail to encompass the needs of other individual focal taxa, since all the focal taxa models are significantly different from each other (Table 16). Additionally, this is the case when comparing all the community corridor models to the individual focal taxa corridor models. Thus, our second hypothesis is rejected since the community models were not significantly better at encompassing the needs of the multiple focal taxa (Table 17). We then illustrated several factors that a land managing agency particularly in our case study area of south Texas would need to consider including forest distribution, restorable areas, current protected areas, parcel sizes of properties within the corridor examination area, other habitats (ex. Wetlands) and anthropogenic barriers like roads and windmills (Fig. 21). When examining all the corridor routes (Fig. 22) only two corridor routes successfully accomplish desired land manager goals. These considerations were to avoid the anthropogenic barriers particular windmills, minimize the amount of land parcels not currently classified as legally protected, and maximized habitat diversity. These two corridor routes were the aplomado falcon focal taxon corridor model and the ocelot preference community corridor model (Fig. 23). The recommended route based on the finding in this thesis is the ocelot preference community model only because its cost surface had fewer significantly different agreement areas (Table 14 & 15) between all this models' tiers than the aplomado falcon model. A land management agency could use the cost surface model as a layer and preform a zonal statistic function using a 90% majority and 10% minimum value to identify the conservation value of each property in the region and give a higher preference for property falling within the least cost path corridor route. Unfortunately, due to the nature of corridor development particularly the plasticity of landowners' willingness to be involved, this study will omit the final figure from this paper but has it available for the sponsoring land managing agency upon request.

CHAPTER IV

DISCUSSION AND CONCLUSION

The plant community in this study was categorized by habitat type (forest or grassland), land-use history (pristine or disturbed) and by size (both continuously and categorically). Then these factors were examined to see if they explained variation in species richness, invasive grass cover, species diversity, and total cover. In general, none of these factors were solely responsible in explaining the variance but when combined together trends emerged. We found unlike Best (2009) claim that both areas of invasive grass and native herbaceous cover will have high (>75%) cover our study area only had high total cover in areas with high invasive grass cover. Our study found the lowest covers for both invasive grasses and total cover in disturbed grassland primarily which may be due to most of our disturbed grasslands consisting of fallow sorghum fields. Both Flanders et al. (2006) and Best (2009) state that invasive grass cover reduces biodiversity for both the vegetative and wildlife community. In our study we found that the factors that explained richness were both habitat type and land use history only partially explained the invasive grass cover which needed a categorical size class in order to detect meaningful patterns. Our study agrees with the findings of Lonard and Judd (2002) since our pristine grasslands which are coastal prairies with saline soils had low to no invasive grass cover particularly for larger grassland sites. This is most likely due to the salinity of the soil restricting invasive grass spread. Our study had a spike in diversity of intermediate size forests and intermediate sized disturbed habitats. All pristine quality habitats were not significantly different from the intermediate disturbed habitat in terms of species diversity but were different from small and large disturbed habitats. This trend could be due to the presence of both interior and edge species in these habitats or due to the lack of evaluation of wetness in the habitats. Therefore, areas that are naturally wetter may have higher diversity by containing wetland species then dryer which lack wetland species and should be examined in future studies. Species richness also had a spike for forest greater than 30 ha while smaller forest had lower richness and all grasslands had similar levels of species diversity to both forest sizes. Our study shows that the 30-ha boundary between small and medium size patches for ocelots (Jackson et al. 2005) can be a real boundary since this boundary was important for forest both in terms of species richness and invasive grass cover. The smaller than 20 ha boundary which is important for arthropods like lepidopterans (Kho & Sodhi 2004) was found to be important in explaining both species richness and species diversity in the herbaceous plant community.

This study found that these four taxa: ocelots, aplomado falcons, Texas tortoise and plant community are not candidate indicator species or indicator groups for the LRGV. This was a surprise since many conservation efforts and local experts thought ocelots (Jahrsdoerfer & Leslie, 1988), aplomado falcons (Stilley et al. 2018), and/or Texas tortoises (Weber unpublished) would be ideal indicator species for the LRGV. It was surprising that the USGS GAP analysis and our Texas tortoise model only agreed in regard to where we predict low likelihood or the absence of tortoises. This result may reflect the inclusion of more anthropogenic considerations (protection status of land, road speed and urban avoidance) in the design of our Texas tortoise model while the USGS only included urban avoidance as an anthropogenic factor in their GAP analysis. We were quite surprised to find that the vegetation community did not predict any of our indicator species assessed in this study. However, the vegetation community assessment was predominantly based on herbaceous community composition within forests and grasslands since it was originally designed to test if this layer could predict Lepidopteran community dynamics. This design bias could explain why none of the indicator species models were related to vegetation community. Although, this plant community could be indirectly related to other wildlife communities not included in this study, such as Lepidopteran and small mammal communities, and if this is the case these results would be similar to Belfrage et al. (2005) findings.

The community-based models were less effective at encompassing the ecological community then hypothesized. This result is interesting because in a review of corridor design by Rudnick et al. (2012) these corridor design experts' general opinion is the inclusion of more ecological community members in the design process would improve corridor models. However, we suspect that the inclusion of additional members of the ecological community especially if these additions include more empirically derived geospatial layers instead of public dataset derived models could result in a more comprehensive community model. Additional further evaluation of the vegetation structure could help develop a better corridor. Vessby et al. (2002) investigation for indicator taxa in Swedish seminatural grasslands suggested that functional group assemblage may be a better predictor of species composition then species diversity or species richness and this could be the case for South Texas as well.

Land Managing Implications

Land managing agencies scientific staff have a variety of ways to go about interpreting the results of the study. The rest of this section will focus specifically on the needs of a land managing agency in south Texas but a land managing agency could use this same general method and test for their region of the world and develop corridors that fit that region's needs. If the land managing agency is restricted to avoiding anthropogenic barriers, then the best corridor route would be either the aplomado falcon corridor or the ocelot preference community corridor (Fig. 23). If these anthropogenic barriers in Figure 21 are not a hindrance to the land managing agency, then the corridor that has the most agreement between models would be the equal weight community model (Table 15). This equal weights model happens to follow a straighter path almost directly north to south and better aligns with recent additions to Laguna Atascosa NWR. However, do not interpret this as the end of corridor investigation in south Texas or as the optimal community corridor. Instead, scientists and researchers should continue to investigate if the inclusion of additional community members (different taxa) into the model, results in the creation of even greater ecological community encompassing models. A few taxa worth examine could include lepidopterans, small mammals, and medium to large mammals. The inclusion of more structural assessments of vegetation communities would be another recommendation worth considering (Vessby et al. 2002). Lastly, future studies should examine using established social science techniques to assess the conservation attitude of landowners and develop ways to derive empirically based geospatial data on this behavior.

REFERENCES

Abrahms, B., S. C. Sawyer, N. R. Jordan, J. W. McNutt, A. M. Wilson, and J. S. Brashares. 2017. Does wildlife resource selection accurately inform corridor conservation? Journal of Applied Ecology 54:412–422.

Best, C. 2009. Chapter 28: Fighting weeds with weeds: Battling invasive grasses in the Rio Grande Delta of Texas. Pages 307-318 *in* Van Devender, T.R., F.J. Espinosa-Garcia, B.L. Harper-Lore, and T. Hubbard, editors. *Invasive plants on the move: Controlling them in North America*. Tucson, Arizona, USA.

Belfrage, K., Björklund, J., and L. Salomonsson. 2005. The effects of farm size and organic farming on diversity of birds, pollinators, and plants in a Swedish landscape. AMBIO: A Journal of the Human Environment 34(8): 582-588.

Bury, B., and E.L. Smith. 1986. Aspects of the ecology and management of the tortoise *Gopherus berlandieri* at Laguna Atascosa, Texas. The Southwestern Naturalist 31(3): 387-394.

Crooks K.R. and M. Sanjayan. 2006. Connectivity conservation: maintaining connections for nature. Connectivity Conservation Eds. Cambridge University Press 1-10.

Elliott, L.F., D.D. Diamond, C.D. True, C.F. Blodgett, D. Pursell, D. German, and A. Treuer-Kuehn. 2014. Ecological mapping systems of Texas: summary report. Texas Parks & Wildlife Department, Austin, Texas, USA.

Ezeh, A. C., J. Bongaarts, and B. Mberu. 2012. Global population trends and policy options. The Lancet 380:142–148.

Foley, J.A., N. Ramankutty, K.A. Brauman, E.S. Cassidy, J.S. Gerber, M. Johnston, N.D. Mueller, C. O'Connell, D.K. Ray, P.C. West, C. Balzer, E.M. Bennett, S.R. Carpenter, J. Hill, C. Monfreda, S. Polasky, J. Rockstrom, J. Sheehan, S. Siebert, D. Tilman, and D.P. Zaks, 2011. Solutions for a cultivated planet. Nature 478(7369), 337-342.

Forman, R.T.T., D. Sperling, J.A. Bissonette, A.P. Clevenger, C.D. Cutshall, V.H. Dale, L. Fahrig, R. France, C.R. Goldman, K. Heanue, J.A. Jones, F.J. Swanson, T. Turrentine, and T.C. Winter. 2003. Road ecology science and solutions. Island Press, Washington D.C., USA.

Flanders, A.A., W.P. Kuvlesky Jr., D.C. Ruthven III, R.E. Zaiglin, R.L. Bingham, T.E. Fulbright, F. Hernandez, and L.A. Brennan. 2006. Effects of invasive exotic grasses on south Texas rangeland breeding birds. The Auk 123(1): 171-182.

Fleury, A.M. and R.D. Brown. 1997. A framework for the design of wildlife conservation corridors with specific application to southwestern Ontario. Landscape and Urban Planning 37(8): 163-186.

Gehlhausen, S.M., Schwartz, M.W., and C.K. Augspurger. 2000. Vegetation and microclimatic edge effects in two mixed-mesophytic forest fragments. Plant Ecology 147: 21-35.

Haddad, N.A. 1999. Corridor and distance effects on inter patch movements: a landscape experiment with butterflies. Ecological Applications 9(2): 612-622.

Haines, A.M., J.E. Janecka, M.E. Tewes, L.I. Grassman, and P. Morton. 2006. The Importance of Private Lands for Ocelot *Leopardus pardalis* Conservation in the United States. Oryx 40(1) 1-5.

Harveson, P.M., M.E. Tewes, G.L. Anderson, and L.L. Laack. 2004. Habitat use by ocelots in south Texas: implications for restoration. Wildlife Society Bulletin 32(3): 948-954.

Hellgren, E.C., R.T. Kazmaier, D.C. Ruthven III, and D.R. Synatzske, 2000. Variation in tortoise life history: demography of *Gopherus berlandieri*. Ecology 81(5): 1297-1310.

Holzmueller, E.J., M.D. Gaskin, JC. Mangun. 2011. A GIS approach to prioritizing habitat for restoration using neotropical migrant songbird criteria. Environmental Management 48(1): 150-157.

Horne, J.S., A.M. Haines, M.E. Tewes, and L.L. Laack. 2009. Habitat partitioning by sympatric ocelots and bobcats: implications for recovery for ocelots in southern Texas. The Southwestern Naturalist 54(2): 119-126.

Jackson, V.L., L.L. Laack, and E.G. Zimmermann. 2005. Landscape metrics associated with habitat use by ocelots in south Texas. Journal of Wildlife Management 69(2): 733-738

Jahrsdoerfer, S.E., and D.M. Leslie Jr. 1988. Tamaulipan brushland of the lower Rio Grande Valley of south Texas: description, human impacts, and management options. U.S. Fish and Wildlife Service, Biological Report 88(36): 1-63.

James A.R.C. and A.K. Stuart-Smith. 2000. Distribution of caribou and wolves in relation to linear corridors. The Journal of Wildlife Management 64(1): 154-159.

Janecka, J.E., M.E. Tewes, L.L. Laack, A. Caso, L.I. Grassman Jr., A.M. Haines, D.B. Shindle, B.W. Davis, W.J. Murphy, and R.L. Honeycutt. 2011. Reduced genetic diversity and isolation of remnant ocelot populations occupying a severely fragmented landscape in southern Texas. Animal Conservation 2011: 1-12

Kadlec, T., R. Tropek, and M. Konvicka. 2012. Timed surveys and transect walks as comparable methods for monitoring butterflies in small plots. Journal of Insect Conservation 16: 275-280.

Kazmaier, R.T., E.C. Hellgren, and D.C. Ruthven. 2001. Habitat selection by the Texas tortoise in a managed thornscrub ecosystem. The Journal of Wildlife Management 65(4): 653-660.

Koh, L.P. and N.S. Sodhi. 2004. Importance of reserves, fragments and parks for butterfly conservation in a tropical urban landscape. Ecological Applications 14(6): 1695-1708.

Leslie, D.M., Jr., 2016, An international borderland of concern-Conservation of biodiversity in the Lower Rio Grande Valley: U.S. Geological Survey Scientific Investigations Report 2016-5078, 120 p., http://dx.doi.org/10.3133/sir20165078.

Lonard, R.I. and F.W. Judd. 2002. Riparian vegetation of the lower Rio Grande. The Southwestern Naturalist 47(3): 420 – 432.

Real, R. 1999. Tables of significant values of Jaccard index of similarity. Miscellania Zoologica 22.1: 29-40.

Rohweder, J., S. Vacek, S. M. Crimmins, and W. E. Thogmartin. 2015. A case study of assigning conservation value to dispersed habitat units for conservation planning. Journal of Conservation Planning 11:13-27.

Rose, F.L. and F.W. Judd. 1975. Activity and home range size of the Texas tortoise, *Gopherus berlandieri*, in south Texas. Herpetologica 31(4): 448-456.

Rudnick, D.A. et al. 2012. The role of landscape connectivity in planning and implementing conservation and restoration priorities. Issues in Ecology 16: 1-20.

Skorka, P., J. Settele, and M. Woyciechowski. 2007. Effects of management cessation on grassland butterflies in southern Poland. Agriculture, Ecosystems and Environment 121: 319-324.

Stilley, J.A., W. Blivohde, and C.A. Gabler. 2018. The First Landscape Prioritization Meeting. Presentation and Discussion.

Stilley, J.A., W. Blihovde, and C.A. Gabler. 2019. The Second Landscape Prioritization Meeting. Presentation and Discussion.

TPWD. 2018. TEAM study area report: study area. TPWD. https://tpwd.texas.gov/gis/team/

Trolle, M. and M. Kery, 2003. Estimation of ocelot density in the pantanal using capture-recapture analysis of camera-trapping data. Journal of Mammalogy 84(2): 607-614.

Vessby, K., B. Soderstrom, A. Glimskar, and B. Svensson. 2002. Species-Richness correlations of six different taxa in Swedish seminatural grasslands. Conservation Biology 16(2): 430-439.

United Nations. 2019. World population prospects 2019. https://population.un.org/wpp/

U.S. Fish and Wildlife Service. 2016. Recovery plan for the ccelot (*Leopardus pardalis*), first revision. U.S. Fish and Wildlife Service, Southwest Region, Albuquerque, New Mexico, USA

USGS 2005. LINK ArcGIS tools for conservation planning. Upper Midwest Environmental Sciences Center. Power Point Presentation. http://www.umesc.usgs.gov/management/dss/bird_conservation_tools_link.html

Webers, T. unpublished. The thornscrub green infrastructure assessment: a tool to help guide conservation and restoration in the Lower Rio Grande Valley. Draft. Pages 1-17. The Conservation Fund and American Forest. Annapolis, Maryland, USA

APPENDIX A

Classification	Source
Slope	USGS
Water	UN
Parcel Acreage	Cameron County
Soil	NRCS - USDA
Habitat	TPWD
Windmills	USFWS
Road Speeds	TXDOT
Nest Structures	USFWS

Table 1: The list of classified criteria used to perform the least cost analysis and the organization that provided the original geospatial data.

Model Design Schemes			Ocelot		Aplomado Falcon	
Classification	Raw Values	Field Value	Scale Value	Weight	Scale Value	Weight
Slope	0 - 0.329579 m	1	1	5%	1	1%
	0.329579 - 1.36507 m	2	1		1	
	1.36507 - 2.838763 m	3	1		1	
	2.838763 - 4.190113 m	4	1		1	
	4.190113-5.943195 m	5	1		1	
	no data	0	1		1	
		no data	no data		no data	
Water	1-9 (ephemeral)	0	1	20%	0	1%
	9-12 (permanent)	1	15		1	
	no data	2	2		2	
		no data	no data		no data	
Nest	1	0	n/a	0%	1	10%
	no data	10	n/a		10	
		no data	n/a		no data	
Parcel Acreage	0-5	5	1	5%	1	28%
	5-10	4	1		1	
	10-50	3	3		3	
	50-400	2	4		4	
	400-11648	1	5		5	
	no data	0	6		6	
		no data	no data		no data	
Soils	See Table 2	0	1	5%	n/a	0%
		1	2		n/a	
		2	3		n/a	
		4	5		n/a	
		no data	no data		n/a	
Habitat	See Table 3	1	1	45%	1	45%
		2	2		2	
		5	5		5	
		10	10		10	
		15	15		15	
		20	20		20	
		25	25		25	
		no data	no data		no data	

Model Design Schemes (Continued)			Ocelot		Aplomado Falcon	
Classification	Raw Values	Field Value	Scale Value	Weight	Scale Value	Weight
Road Speed	Wildlife Crossing	0	0	20%	n/a	0%
	5-40	5	5		n/a	
	40-75	15	15		n/a	
	no data	no data	no data		n/a	

Table 2: These are the values, scaled values and weighting scheme used to create the ocelot no wind and aplomado falcon no wind model. For more specific information in regards to soils and habitats refer to Table 2 and 3 respectively.

Soils					
Optimal		Sub-Optimal		Worst	
Type	Scale Value	Type	Scale Value	Type	Scale Value
LAA	1	LM	2	SE	4
LAB	1	CH	2	BA	4
PO	1	TC	2	HA	4
LC	1	LO	2	DE	4
OM	1			WAA	4
LD	1			LY	4
ON	1			WM	4
LEA	1			LK	4
LEB	1			WAB	4
				RE	4
				HC	4
				RG	4
				BE	4
				BU	4
				HE	4
no data	0			LG	4

Table 3: These are the soils used in the ocelot model. The values indicate how well that soil is suited for thornscrub regrowth based on the recommendations of Harveson et al. (2004).

Habitat	Aplomado Falcon	Ocelot
Texas Saline Coastal Prairie	1	10
Texas Coast Salt Brackish Tidal Marsh	5	10
Laguna Madre Salt and Brackish Tidal Flats	5	10
Tamaulipan Mixed Deciduous Thornscrub	10	1
Tamaulipan savannah Grassland	1	3
Tamaulipan Lomas	1	1
Tamaulipan Floodplains	5	3
Tamaulipan Ramadero	5	1
Tamaulipan Saline Lake	10	10
Barren	5	10
Marsh	2	9
Native Invasive: Mesquite Shrubland	1	5
Native Invasive: Common Reed	5	6
South Texas: Disturbance Grassland	2	9
Non-native Invasive: Saltcedar Shrubland	15	10
Row Crops	10	5
Urban High Intensity	25	25
Urban Low Intensity	20	20
Open Water	20	15
Rio grande Delta Thorn Woodland	10	1
no data	no data	no data

Table 4: These are the scale values used to characterize how experts and the literature would expect the aplomado falcons and ocelots to select using these different habitat types. The basis of these habitat classifications are on the Nature Serve Habitat classification system for Texas and their geospatial-derived equivalents as provided by TPWD TEAM Assessment website.

	Forest Edge	Weight	Type	Cost
Biological		17%	Interior	2
			0-30 m	1
			30-120 m	0
			120-150 m	3
			150-250 m	5
			250-500 m	10
			500-4000 m	15
			No Data	No Data
	Forest Canopy	**Weight**	**Type**	**Cost**
		18%	0-25%	10
			25-75%	0
			75-100%	2
			No Data	No Data
	Lomas	**Weight**	**Type**	**Cost**
		17%	0-40	10
			40-283	1
			No Data	No Data
Anthropogenic	**Protected Areas**	**Weight**	**Type**	**Cost**
		16%	Protected	0
			Not Protected	1
			No Data	No Data
	Road Speed	**Weight**	**Speed**	**Cost**
		16%	0	0
			5	3

			30	3
			35	3
			40	3
			45	5
			50	5
			55	5
			60	5
			65	5
			70	5
			75	5
			No Data	No Data
	Urban Avoidance	**Weight**	**Type**	**Cost**
		16%	Not Urban	0
			Slightly Urban	1
			Moderately Urban	3
			Urban	3
			No Data	No Data
All Categories	**Total Weights**	**100%**		

Table 5: This is a table contains all the model parameters summarized into biological and anthropogenic factors. The table include the 6 input layer names Forest Canopy, Forest Edge, Lomas, Protected Areas, Road Speed, and Urban Avoidance. It has the associated weights (Weight) that come out to 100%, the features within a layer (Type) and an associated cost (Cost) used to calculate the cost surface. The value of a pixel in the final cost surface raster is sum of all six costs since each pixel has a value in all 6 input layers multiplied by their respective weight percentage.

TPWD TEAM 2014 Habitats	Computer Model	Total Cover			Richness Inner & Outer Plot			
Nature Serve Classification	Classification	Category	Value	Scale	Category		Value	Scale
Texas Saline Coastal Prairie		Pristine Grassland			Grassland			
Texas Coast Salt Brackish Tidal Flats								
Laguna Madre Salt and Brackish Tidal Flats								
Marsh	Pristine Grassland		0.3857	2		> 30	0.4783	3
South Texas: Disturbance Grassland		Disturbed Grassland						
Barren								
Row Crops	Disturbed Grassland		0.6528	3		< 30	0.2974	2
Tamaulipan Mixed Deciduous Thornscrub		Pristine Forest			Forest			
Tamaulipan Lomas								
Tamaulipan Floodplains								
Tamaulipan Ramadero								
Rio Grande Delta Thorn Woodland	Pristine Forest		0.2569	1		> 30	0	1
Tamaulipan Savannah Grassland		Disturbed Forest						
Native Invasive: Mesquite Shrubland								
Non-native Invasive: Saltcedar Shrubland	Disturbed Forest		0	1		< 30	0.5548	4
Urban Low Intensity		Urban Low Intensity	0.3264	2				
Urban High Intensity								
Tamaulipan Saline Lake								
Native Invasive: Common Reed								
Open Water	Other	Other	null	No Data	Other		Null	No Data

TPWD TEAM 2014 Habitats	Computer Model	Shannon Diversity Index				Invasive Grass Cover			
Nature Serve Classification	Classification	Category		Value	Scale	Category		Value	Scale
Texas Saline Coastal Prairie		Pristine Grassland	< 20	0.4784	3	Pristine Grassland			
Texas Coast Salt Brackish Tidal Flats			20-100	0.5952	3		> 30	0.0004	2
Laguna Madre Salt and Brackish Tidal Flats									
Marsh	Pristine Grassland		100+	0.3638	3		< 30	0.3467	3
South Texas: Disturbance Grassland		Disturbed Grassland	< 20	0.7012	4	Disturbed Grassland			
Barren			20-100	0.2791	2		> 30	0	1
Row Crops	Disturbed Grassland		100+	0.7313	4		< 30	0.0029	3
Tamaulipan Mixed Deciduous Thornscrub		Pristine Forest				Pristine Forest			
Tamaulipan Lomas			< 20	0.4643	3				
Tamaulipan Floodplains			20-100	0	1		> 30	0.5027	4
Tamaulipan Ramadero									
Rio Grande Delta Thorn Woodland	Pristine Forest		100+	0.9077	4		< 30	0.1848	3
Tamaulipan Savannah Grassland		Disturbed Forest	< 20	0.9779	4	Disturbed Forest			
Native Invasive: Mesquite Shrubland			20-100	0.3164	2		> 30	0.0211	3
Non-native Invasive: Saltcedar Shrubland	Disturbed Forest		100+				< 30	0.7780	4
Urban Low Intensity									
Urban High Intensity									
Tamaulipan Saline Lake									
Native Invasive: Common Reed									
Open Water	Other	Other		Null	No Data	Other		Null	No Data

Table 6: This is a table that contains the values used to create the plant community model. All the numbers in the column value and labels listed under category are derived from means from the ANOVA analyses. These value were used to supplement TPWD (2018) TEAM Assessment land classification.

Weighting Scheme	
Layer	Weight
Total Cover	10
Richness Inner & Outer Plot	20
Shannon Diversity Index	30
Invasive Grass Cover	30
Property Size	10

Table 7: This is the table of weights used to create the plant community model. The weights were derived from expert opinion and land management preferences.

	J value	P value	Symbol
Similar	0.52	0.05	*
	0.56	0.01	**
	0.62	0.001	***
Different	0.25	0.05	*
	0.22	0.01	**
	0.18	0.001	***

Table 8: This table contains the significant Jaccard Similarity for analyses with sample sizes greater than one hundred (N > 100) (Real, 1999). The left tail (lower values) indicate significant different dataset while the right tail (higher values) indicate that models are statistically similar. This table scheme will be used in the next few tables comparing the similarity of the models examined in this study.

Tortoise Model Agreement with USGS GAP Analysis									
	New Tortoise Model	High	High & Medium	High, Medium, & Low	Medium & Low	Low	Absent & Low	Absent	Entire Model
USGS GAP Analysis	Present	0.011 ***	0.116 ***	0.201 **	0.222 *	0.201 **			0.201 **
	Absent						0.655 ***	0.164 ***	

Table 9: This is a table that shows the Jaccard Coefficient calculated based on the total area of features in the output intersection and union layer between the USGS GAP Analysis and the Texas tortoise cost surface model described in this paper. Only the TX tortoise absent and low categories combined were found to be statistically similar to the USGS TX tortoise GAP analysis absence category. All the other model layers were significantly different from the USGS GAP Analysis.

	Species Richness				
	Variable	Sum of Square	Degrees of Freedom	F Statistic	P value
Independent	Intercept	2791	1	34.4383	<0.001****
	History	63	1	0.7813	0.39
	Habitat	223	1	2.7517	0.12
	Size Class	134	1	1.6577	0.22
Interaction	History:Habitat	20	1	0.2505	0.63
	History:Size	32	1	0.3964	0.53
	Habitat:Size	394	1	4.8562	0.046**
	History:Habitat:Size	96	1	1.1787	0.30
	Residuals	1054	13		

Table 10: These are the results of a 3-way ANOVA for the species richness of the vegetation community where we considered ($p < 0.1$) as significant. The model found a significant interaction for habitat and size so means of the LSD post hoc with a tukey correction for the habitat and size category (<30 & >30) were used to populate our geospatial model.

	Invasive Grass Cover				
	Variable	Sum of Square	Degrees of Freedom	F Statistic	P value
Independent	Intercept	10020	1	29.2703	<0.001****
	History	161	1	0.4695	0.51
	Habitat	3837	1	11.2089	0.005***
	Size Class	1846	1	5.3929	0.04**
Interaction	History:Habitat	617	1	1.8022	0.2
	History:Size	1589	1	4.6427	0.05*
	Habitat:Size	24	1	0.0699	0.80
	History:Habitat:Size	5975	1	17.4561	0.001***
	Residuals	4450	13		

Table 11: These are the results of a 3-way ANOVA for the invasive grass cover of the vegetation community where we considered ($p < 0.1$) as significant. The model found significant values for habitat, size class, the interaction of history and size, and the interaction of history, habitat, and size. Therefore we used the means from history, habitat, and size (<20, 20-100, >100) of the LSD post hoc with a tukey correction to populate our geospatial model.

	Shannon Diversity				
	Variable	Sum of Square	Degrees of Freedom	F Statistic	P value
Independent	History	2.58	1	12.5396	0.005***
	Habitat	0.27	1	1.3200	0.28
	Size Class	3.14	2	7.6235	0.0097***
Interaction	History:Habitat	0.88	1	4.2741	0.07*
	History:Size	1.05	2	2.5582	0.12
	Habitat:Size	2.6	2	6.3145	0.02**
	History:Habitat:Size	0.14	1	0.6758	0.43
	Residuals	2.06	10		

Table 12: These are the results of a 2-way ANOVA for the species diversity of the vegetation community using the Shannon Diversity Index where we considered ($p < 0.1$) as significant. The model found significant values for history, size class, the interaction of history and habitat, and the interaction of habitat and size. Therefore, the study used the means from two interactions history and habitat and habitat and size (<20, 20-100, >100) and associated LSD post hoc with a tukey correction to populate our geospatial model.

	Total Cover				
	Variable	Sum of Square	Degrees of Freedom	F Statistic	P value
Independent	Intercept	56880	1	74.9805	<0.001****
	History	87	1	0.1149	0.74
	Habitat	5757	1	7.5896	0.02**
	Size Class	140	1	0.1840	0.67
Interaction	History:Habitat	2499	1	3.2938	0.09*
	History:Size	1028	1	1.3555	0.27
	Habitat:Size	1819	1	2.3984	0.15
	History:Habitat:Size	179	1	0.2364	0.63
	Residuals	9862	13		

Table 13: These are the results of a 3-way ANOVA for the total cover of the vegetation community where we considered ($p < 0.1$) as significant. The model found significant values for habitat and the interaction of history and habitat. Therefore, we used means of history and habitat from the LSD post hoc with a tukey correction to populate our geospatial model.

Similarity of Model Index										
Model		Ocelot			Aplomado falcon			Texas tortoise		
		Tier I	Tier II	Tier III	Tier I	Tier II	Tier III	Tier I	Tier II	Tier III
Plant Community	Tier I	0.0007			0.0015			0.0053		
	Tier II		0.6986			0.4833			0.1062	0.4950
	Tier III			0.1050			0.1326			0.2286
Ocelot	Tier I				0.0450			0.0160		
	Tier II					0.6486***			0.1099	0.0576
	Tier III						0.0213			0.3386
Aplomado falcon	Tier I							0.0008		
	Tier II								0.1412	0.4272
	Tier III									0.2714

Table 14: This is the assessment of similarity between all the computer models created for ocelots, aplomado falcons, Texas tortoises, and the plant community. The assessment was conducted using the Jaccard similarity index and the p-values associated with this table are illustrated in Table 8. Only the second-tier ocelot and aplomado falcon models and the second tier ocelot and second tier plant community models were statistically similar to each other. Four models (Plant community tier 2, ocelot tier 3, and aplomado falcon tier 2 & 3) were neither significantly different or statistically similar to the third tier Texas tortoise models. The second tier aplomado falcon and the second tier plant community models are in this inconclusive zone. All the other models were significantly different from each other. This result mostly confirms our first hypothesis that none of our individual focal taxa are candidate indicator taxa and no single taxa will be able to adequately encompasses the needs of the ecological community.

Similarity of Model Index							
		Community Equal			Ocelot Preference Community		
Model		Tier I	Tier II	Tier III	Tier I	Tier II	Tier III
Ocelot Preference Community	Tier I	0.00742***					
	Tier II		0.74641***				
	Tier III			0.4374			
Plant Community	Tier I	0***			0.00002***		
	Tier II		0.60537**			0.4896	
	Tier III			0.17544***			0.11490***
Ocelot	Tier I	0.00071***			0.07593***		
	Tier II		0.58252**			0.4901	
	Tier III			0.2927			0.53597*
Aplomado falcon	Tier I	0.00077***			0.07066***		
	Tier II		0.4614			0.3656	
	Tier III			0.05119***			0.02278***
Texas Tortoise	Tier I	0***			0.00259***		
	Tier II		0.15033***			0.14371***	
	Tier III			0.15124***			0.22200*

Table 15: This table contains the Jaccard similarity index values determined when comparing the two-community based cost surface models back to the individual focal taxa cost surface models. We were surprised to find that our second hypothesis stating that the community models will better incorporate the individual taxa than the individual taxa models to themselves was only partially correct. Both the equal weighted community model and the ocelot preference community models tier 1 (best suited) area for a given species were significantly different to all individual focal taxa models. However, when examining the next best habitat (tier 2) we see stronger agreement particularly for the equal weights' community model. The ocelot preference model had values that were not significantly different from three of the individual focal taxa models nor were they significantly similar and it only had significant similarity to the equal weight community model at tier 2 and with the ocelot model for tier 3 (barely suitable)

Individual Least Cost Path Comparisons			
Cost Paths	Aplomado Falcons	Texas Tortoises	Plants
Ocelots	0.07836***	0.02386***	0.07084***
Aplomado Falcons		0.07901***	0.08788***
Texas Tortoises			0.04315***
Plants			

Table 16: The table shows that the least cost path with a 300m buffer on both sides of the path were all significantly different from each other. This analysis illustrates that none of these focal taxa would be an ideal indicator species or indicator group (plant community) for this region. This result supports our first hypothesis that individual taxa are not suitable as indicator species.

Least Cost Paths		Community Corridors	
		Equal Weights	Ocelot Preference
Individual Models	Ocelots	0.08112***	0.08386***
	Aplomado Falcons	0.21404**	0.14310***
	Texas Tortoises	0.04456***	0.23139*
	Plants	0.27903	0.08255***

Table 17: The comparison of the community least cost paths to the individual least cost path analyzes illustrate that the models are all significantly different. Therefore, the second hypothesis was incorrect and the community based least cost paths were not effective (significantly different) at encompassing the routes suggested for individual taxa.

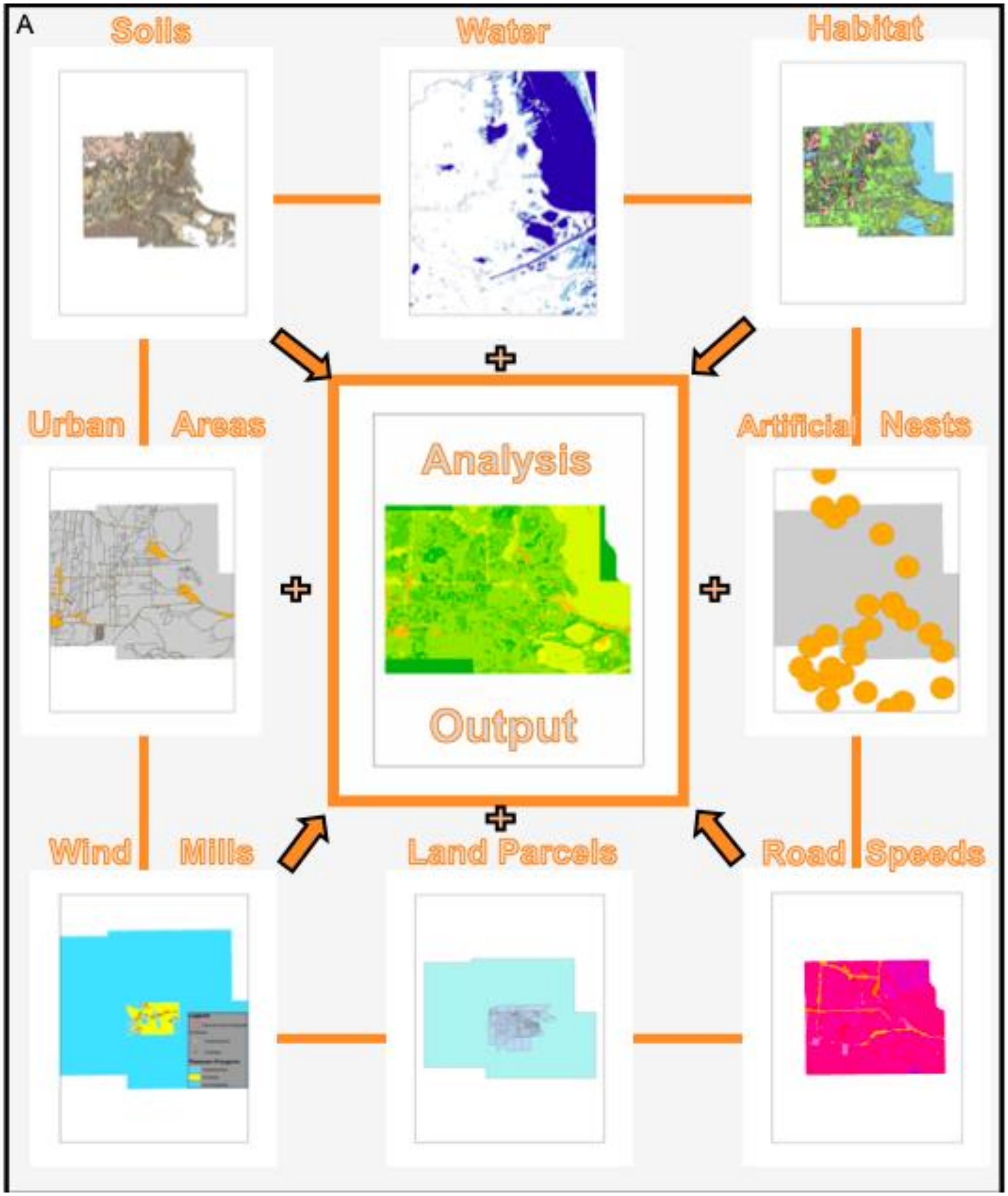


Figure 1: This is a graphic illustration of the nine input geospatial layers used to run the three trials used to develop the maps seen in Figure 1.

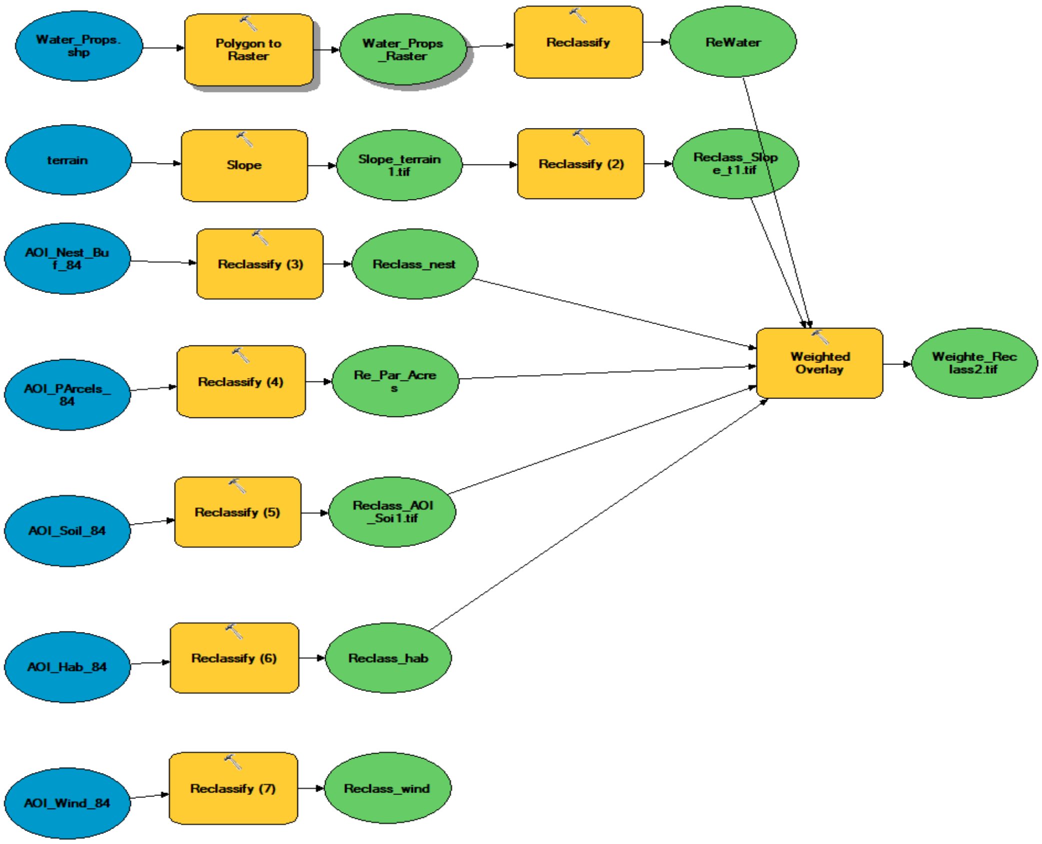


Figure 2: This is how the cost surface model illustrated in Figure 2 was programmed into ArcMap 10.6 to develop the cost surface outputs illustrated in Figure 1.

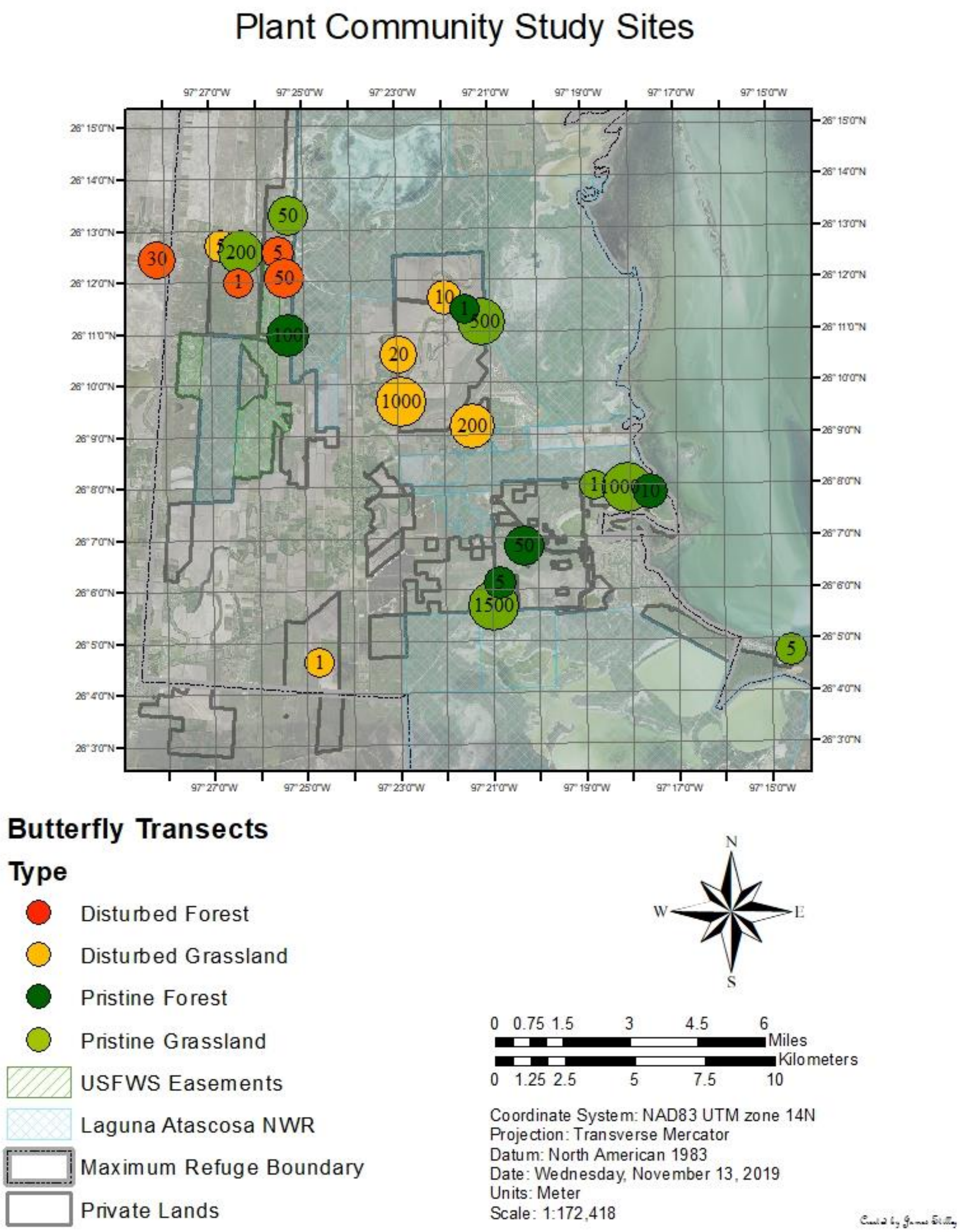


Figure 3: This is map of the plant community study sites and associated habitat fragment type, quality, and size. The number within each circle is the area in hectare of the associated habitat patch that is illustrated in a gradient of sizes.

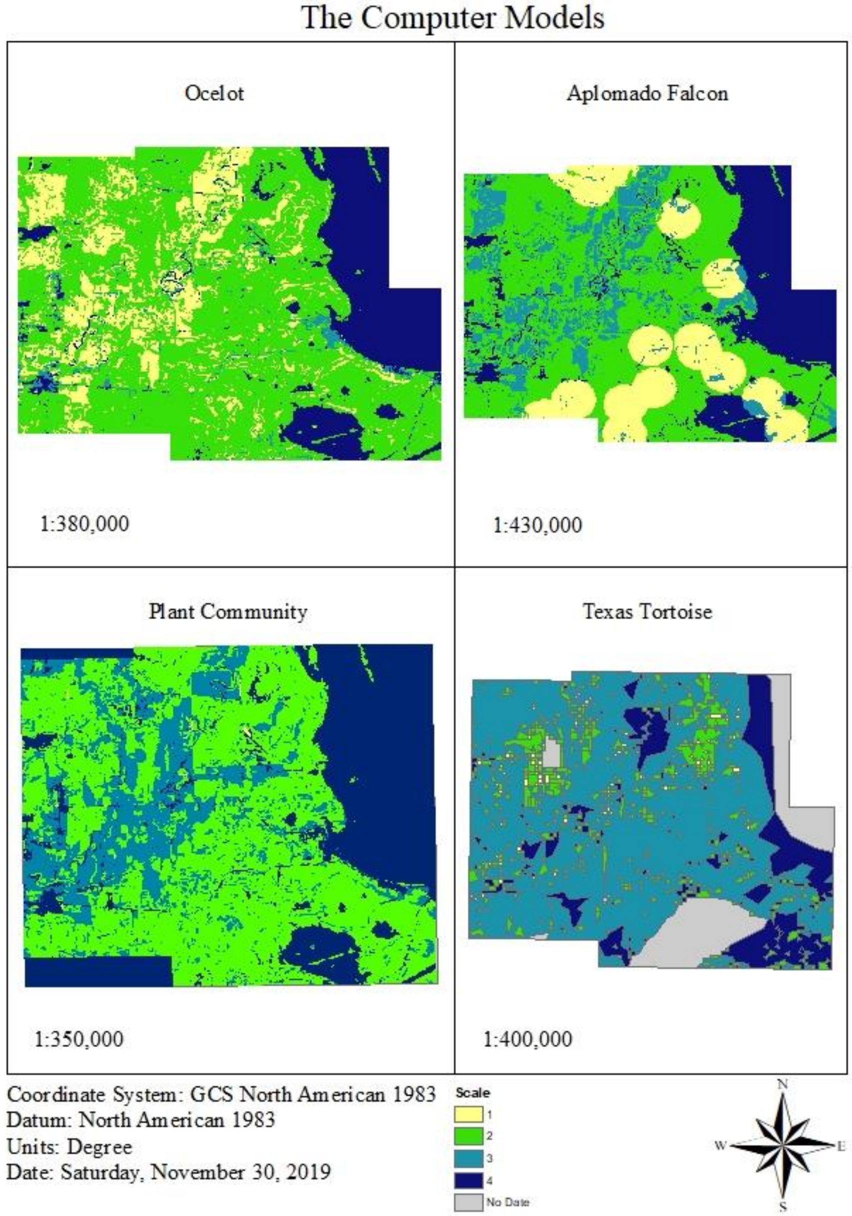


Figure 4: This map illustrates all four focal taxa computer models created to predict the likelihood of occurrence for ocelots, aplomado falcons, Texas tortoises, and plant communities. The three animal models were created from publically available sources with scales and weighting schemes derived from a review of the scientific literature and expert opinion. Only the plant community model is based on empirical data from the study region.

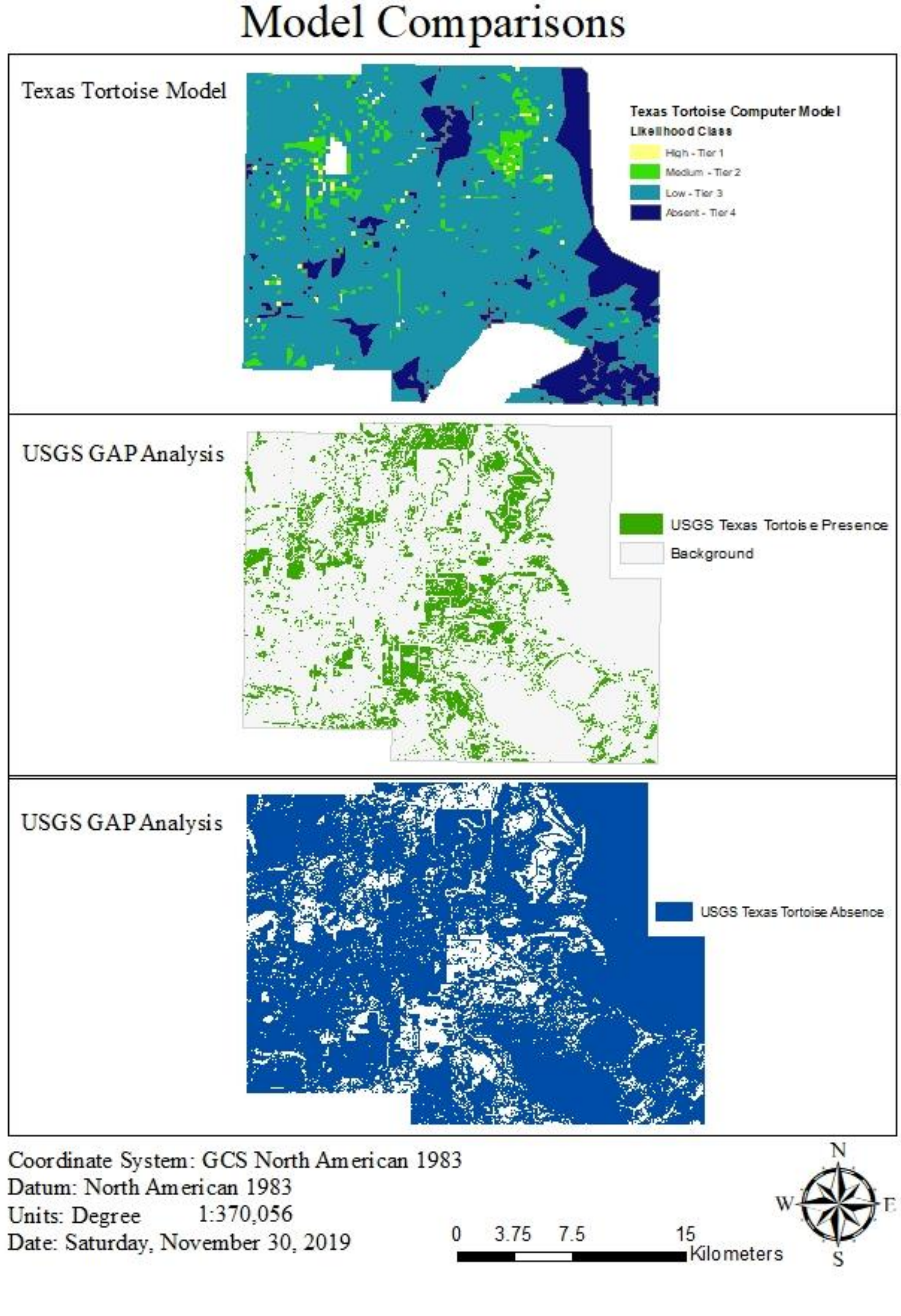


Figure 5: This is a comparison of the Texas tortoise model created in this study compared to the published USGS Texas Tortoise GAP Analysis.

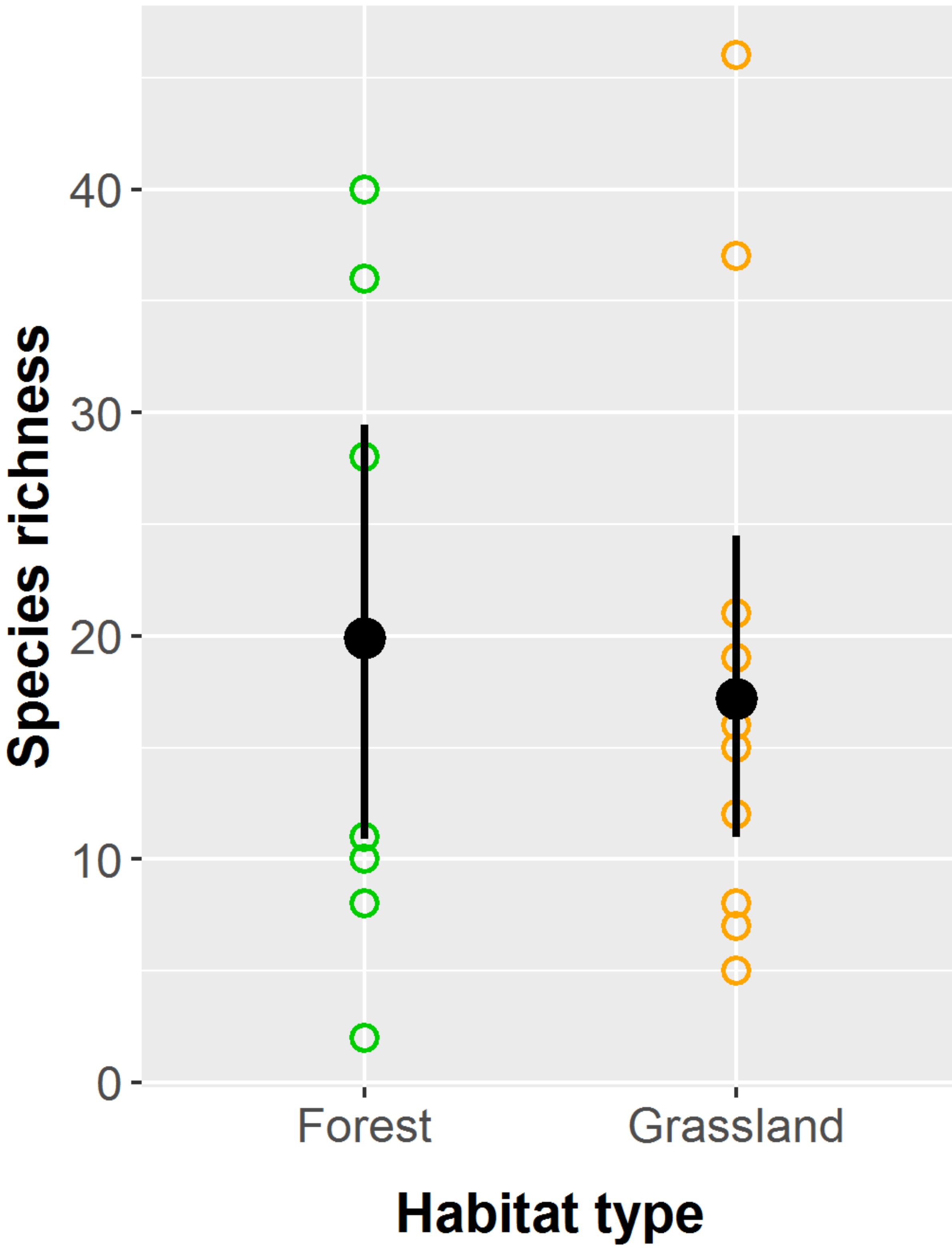


Figure 6: There is no difference in species richness of herbaceous vegetation in forested and grassland sites in the region of the prospective corridor.

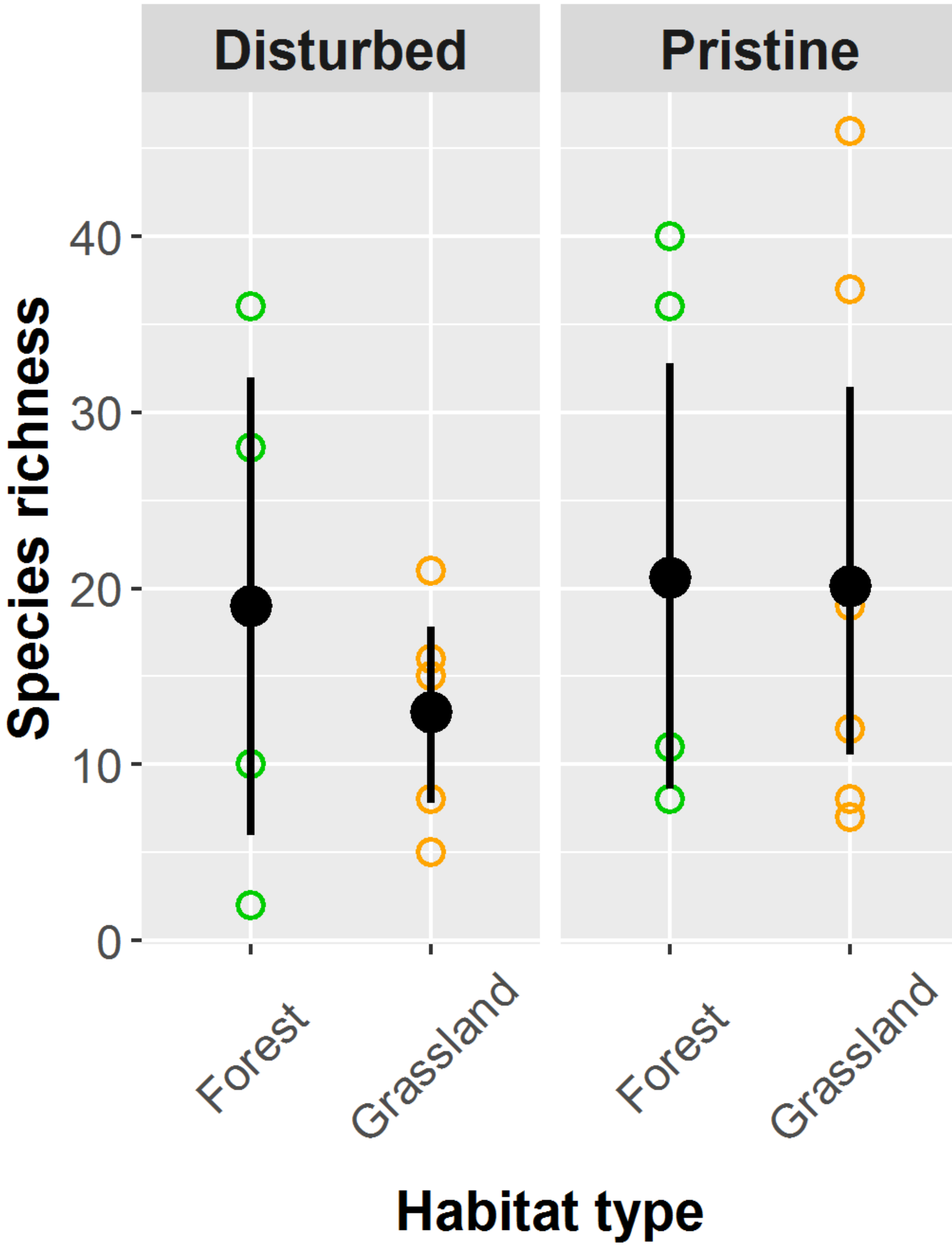


Figure 7: There is no difference in species richness between disturbed and pristine land-use histories for either habitat type (forest of grassland). Nor is there a difference in the species richness of the same land-use history between the two habitat types.

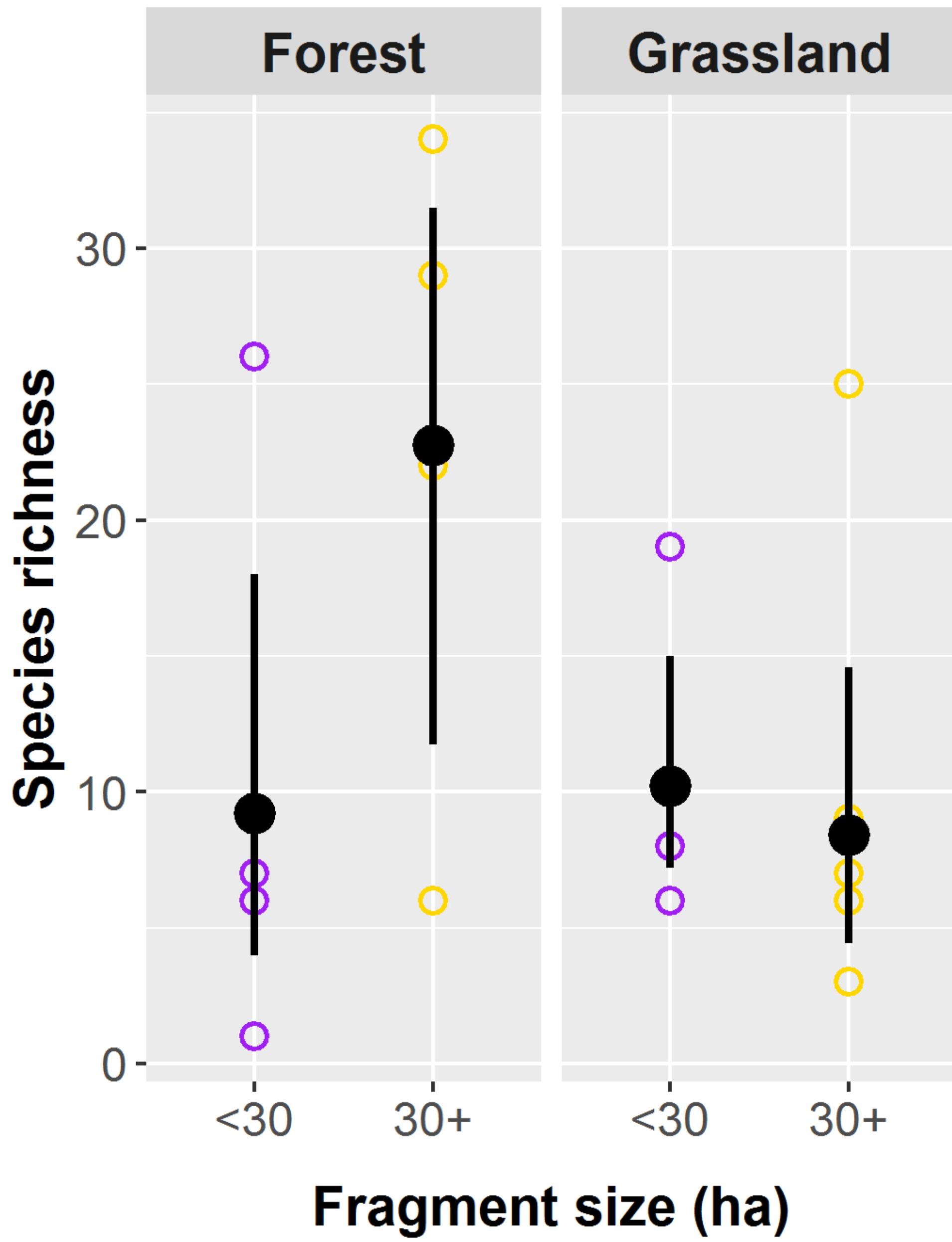


Figure 8: The results of a post hoc comparison using the Tukey HSD test indicate a significant difference exists in the species richness of fragmented habitats split on the south Texas ocelot biology derived fragment habitat boundary of 29 ha (Jackson et al. 2005). Forest larger than 30 ha (M = 22.75, SE = 4.5) have significantly higher species richness then forest less than 30 ha (M = 8.33, SE = 4.11) and grasslands larger than 30 ha (M = 6.80, SE = 3.77). The small forest and large grassland are not significantly different form each other and grassland smaller than 30 ha (M=10.58, SE = 4.11) were not significantly different from any other category.

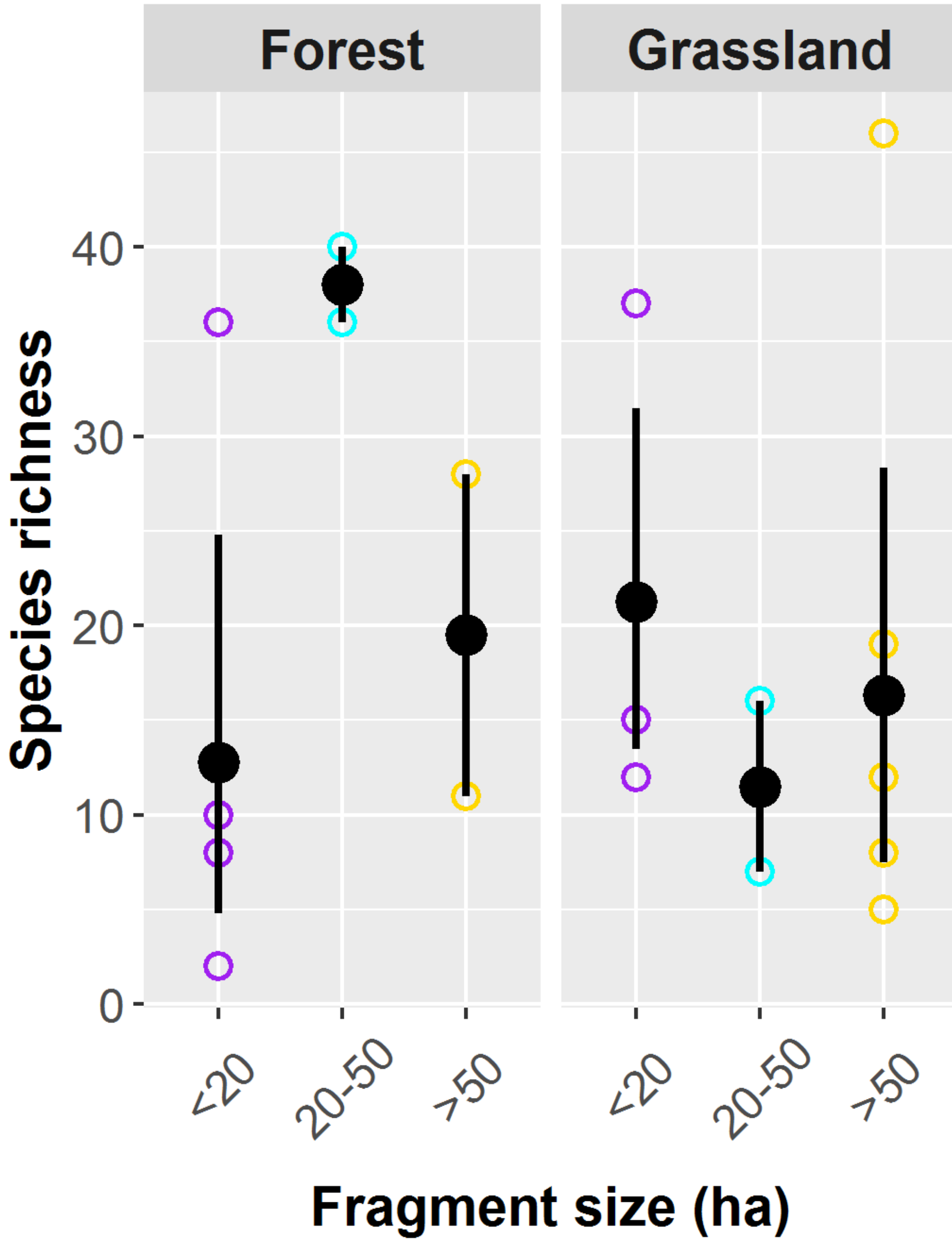


Figure 9: There is a significant increase in species richness in intermediate-sized forest (20-50 ha) compared to all other size classes and habitat types. With the exception of the intermediate sized forest, no significant differences were observed between the remaining size classes for both forest and grasslands.

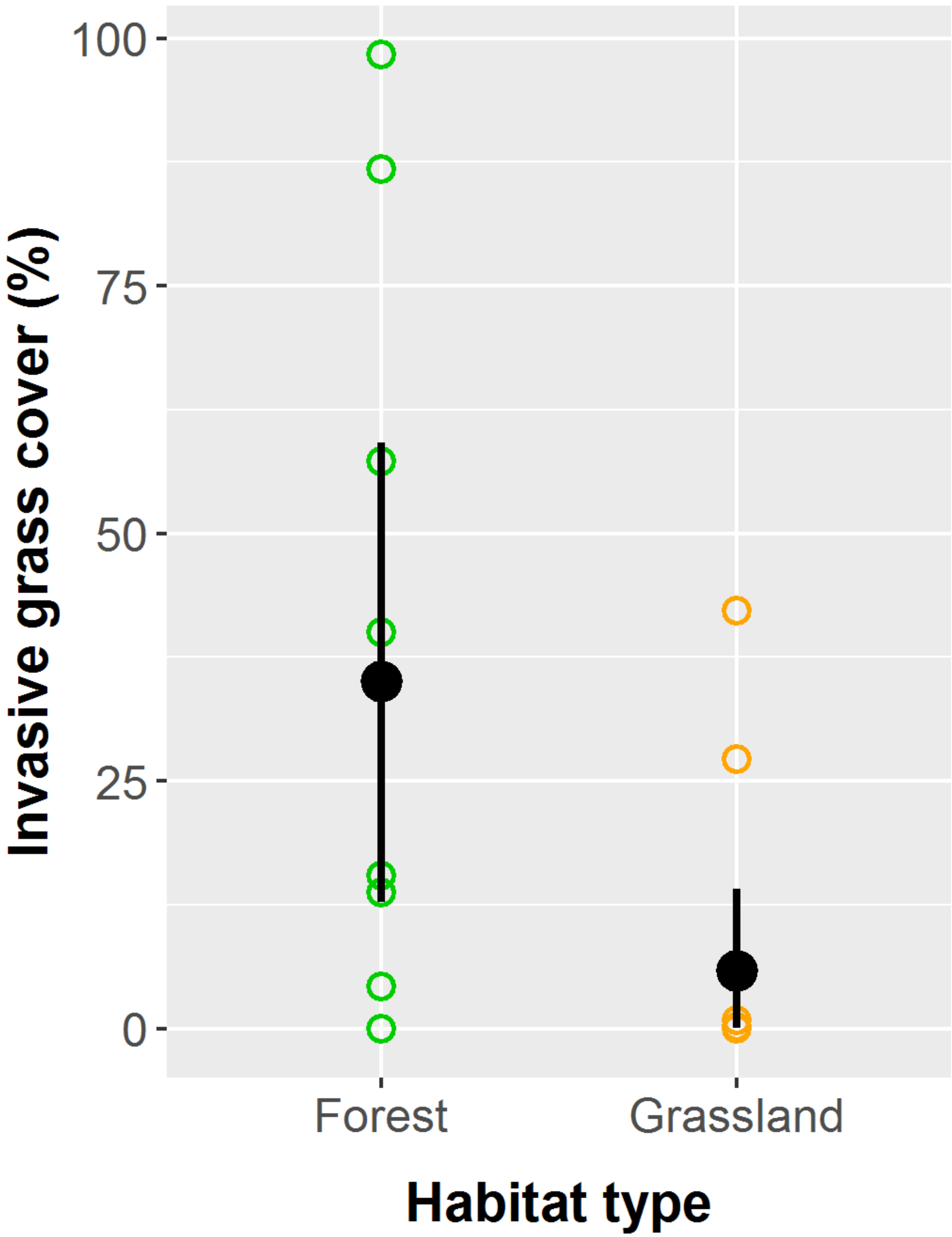


Figure 10: The post hoc comparison using the Tukey HSD test indicate that Forest (M = 37.17, SE = 6.26) have significantly higher invasive grass cover then grasslands (M = 8.75, SE = 5.73).

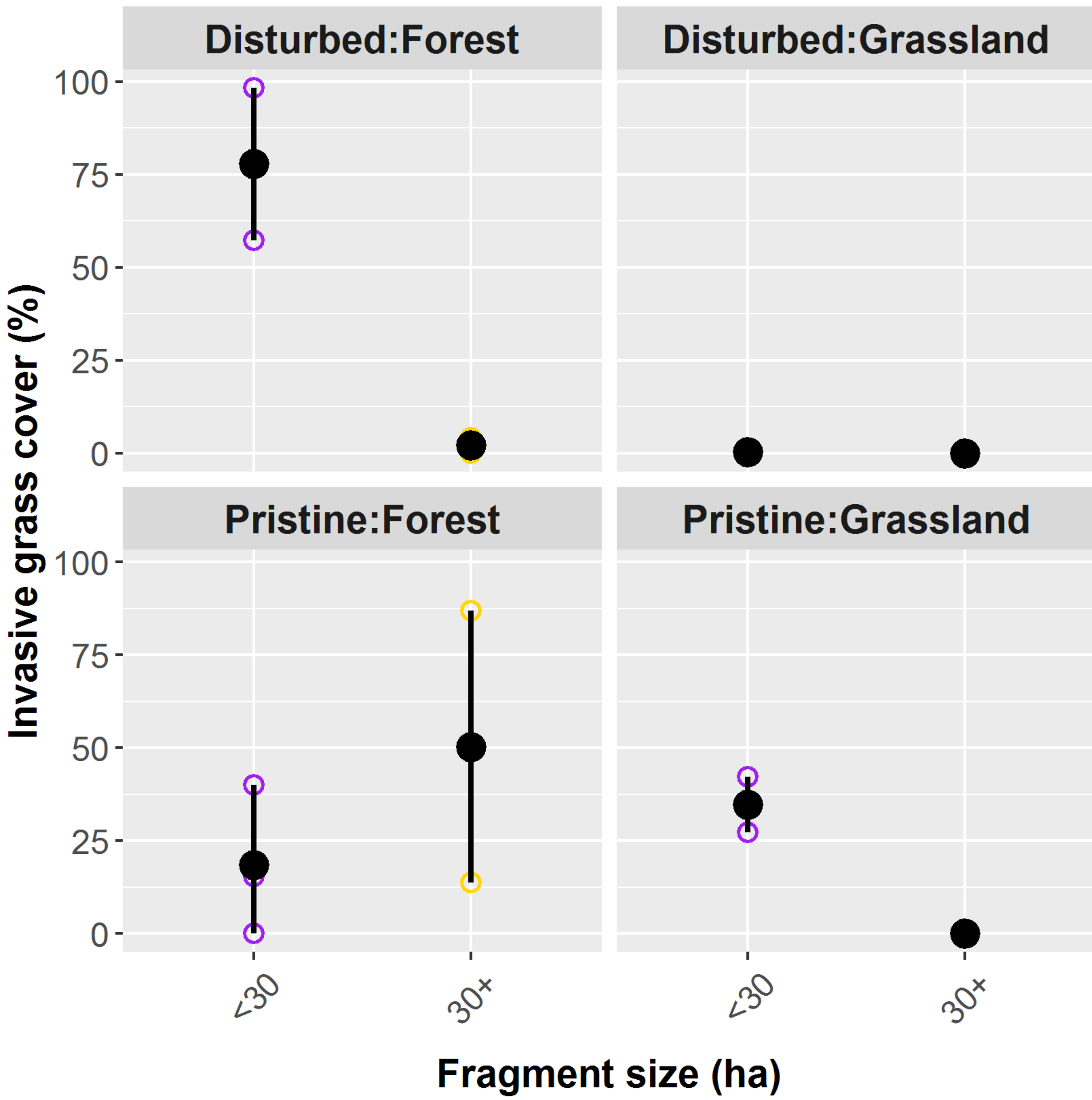


Figure 11: In both disturbed forest less than 30 ha (M = 77.8030, SE = 13.08) and pristine grasslands greater than 30 ha (M = 0.0364, SE = 8.27) a significant difference in invasive grass cover is observed from the rest of the habitat, land history and size classes at the ocelot specific fragment size class transition of 29 ha (Jackson et al. 2005). The rest of the habitat, land history, and size classes are not significantly different from one or more other classes.

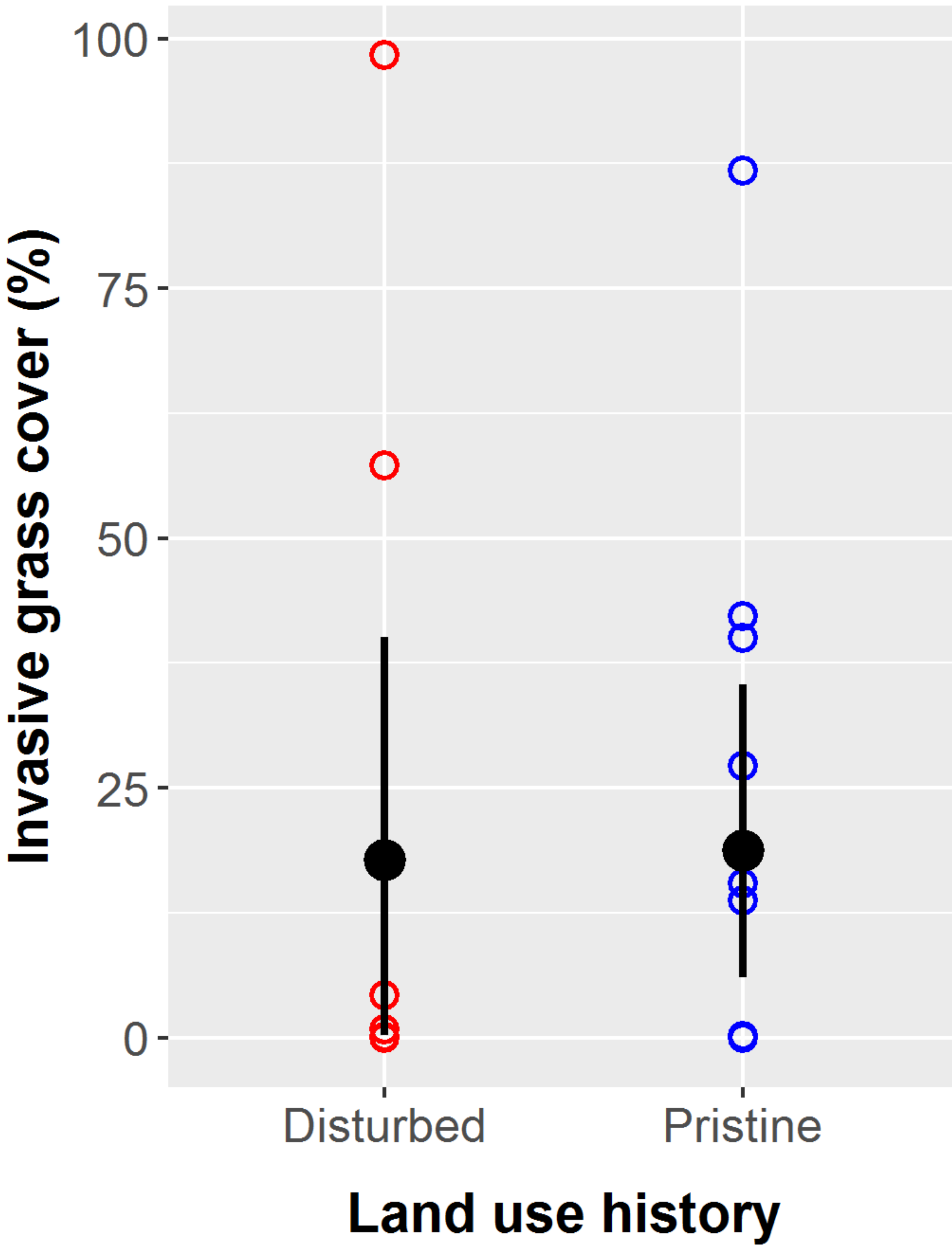


Figure 12: No difference was detected in invasive grass composition between the two land-use history classes (disturbed and pristine).

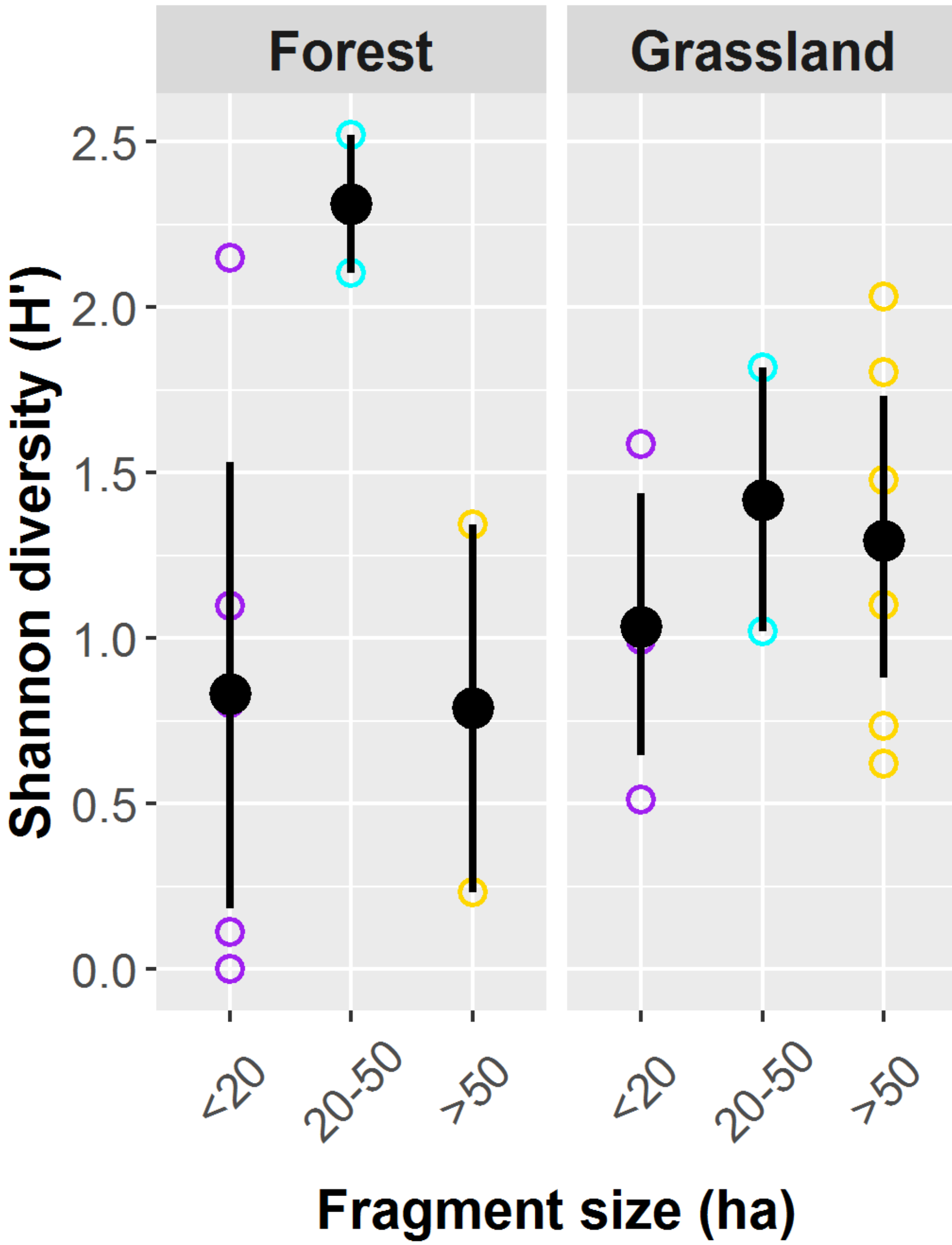


Figure 13: Intermediate sized forest fragments (20-50 ha) have a significantly higher Shannon diversity index than any other forested size class and any of the grassland size classes. It is hypothesized this spike in intermediate forest diversity might be the result of both edge and interior species presence in these forest fragment while the smaller and larger forest fragments primarily are composed of edge species and interior species respectively.

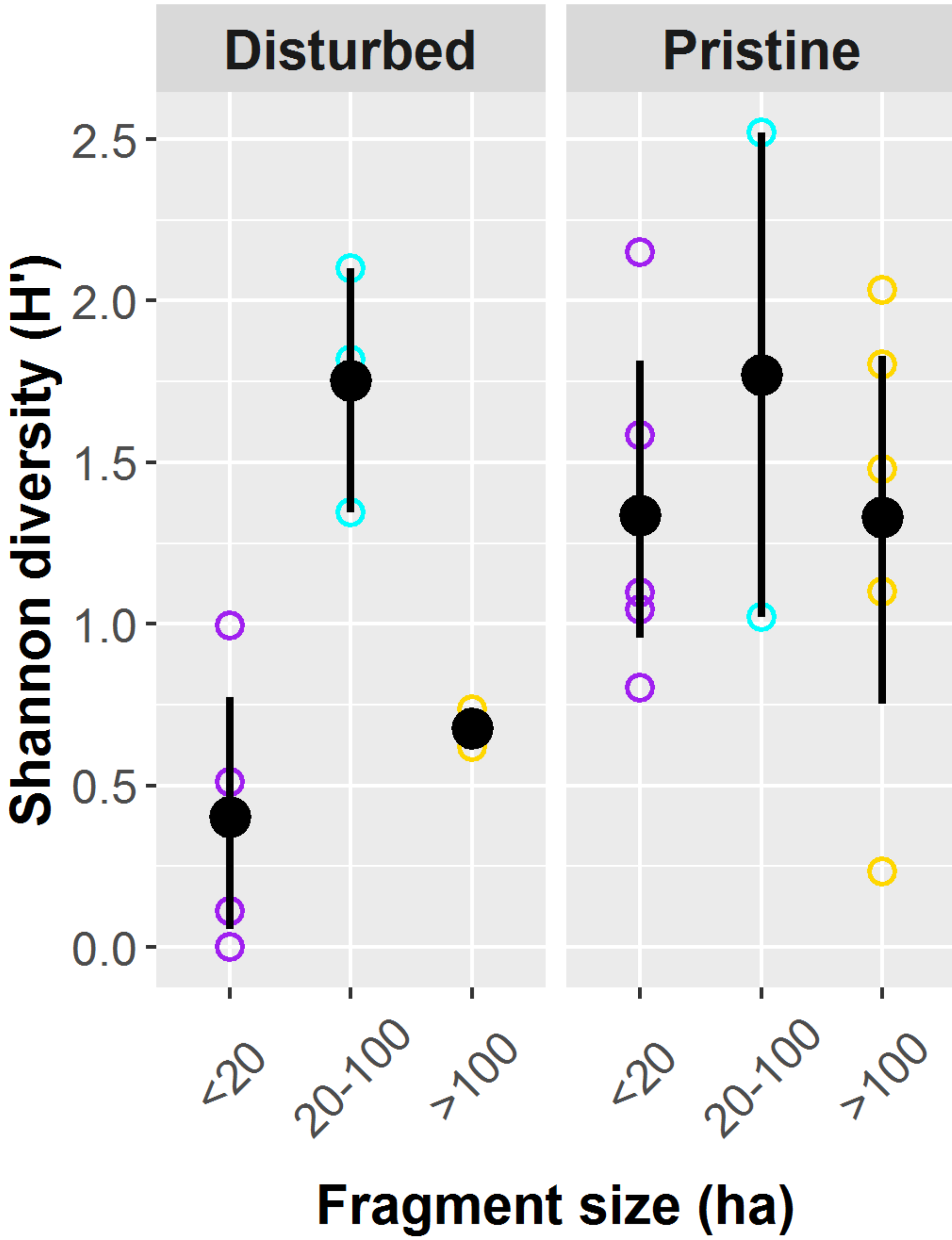


Figure 14: Small disturbed habitat fragments had a significantly lower Shannon diversity index value according to an LSD test ($P < 0.05$) than any size class for pristine habitats and intermediate sized disturbed habitats. However, both the intermediate and large disturb habitats were not different than any pristine habitat size class.

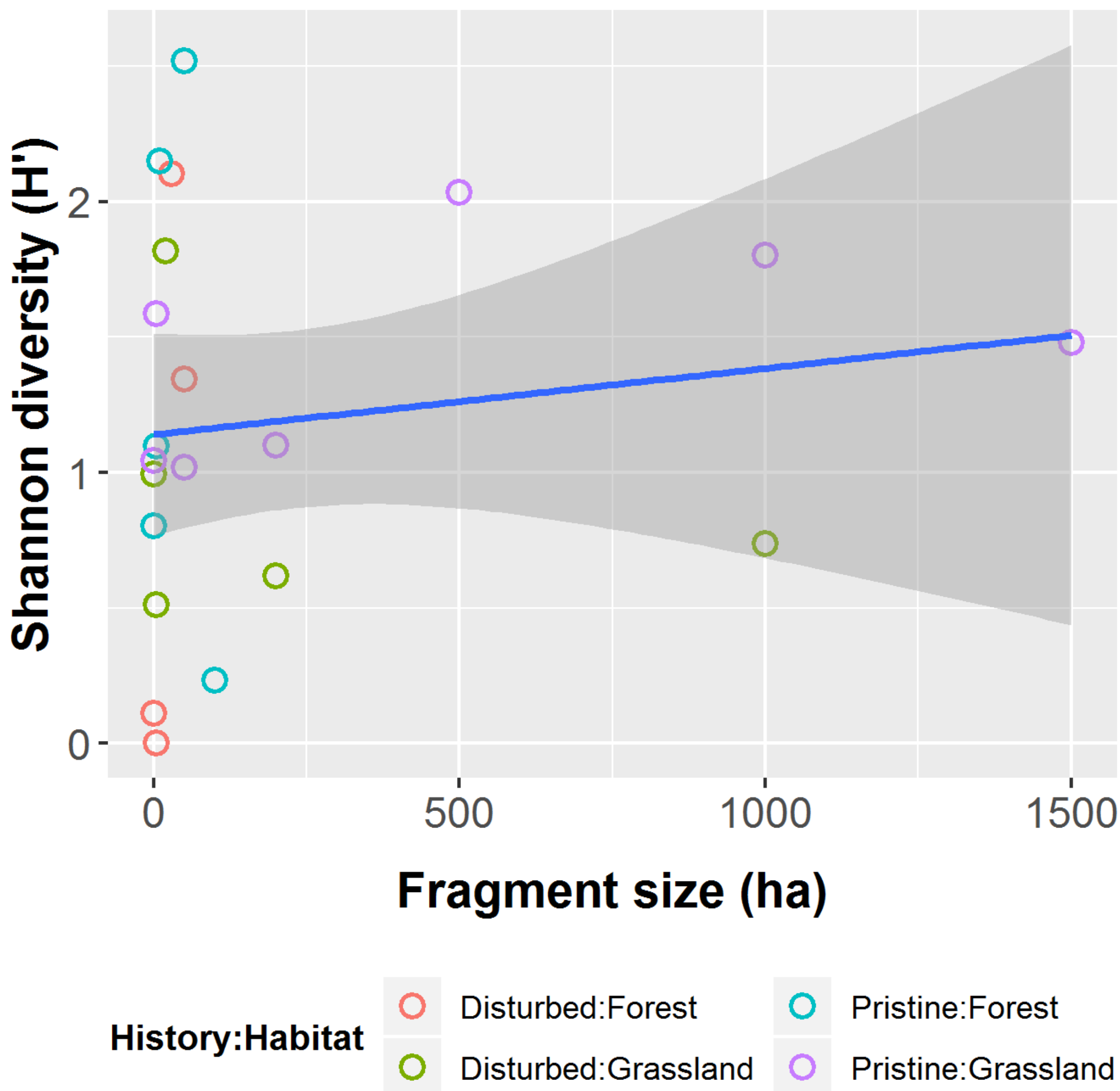


Figure 15: There is no significant relationship between Shannon diversity and fragment size when examined individually. However, if the factors habitat type, history, and size are included in the model then patch size is a significant factor similar to the main regression line.

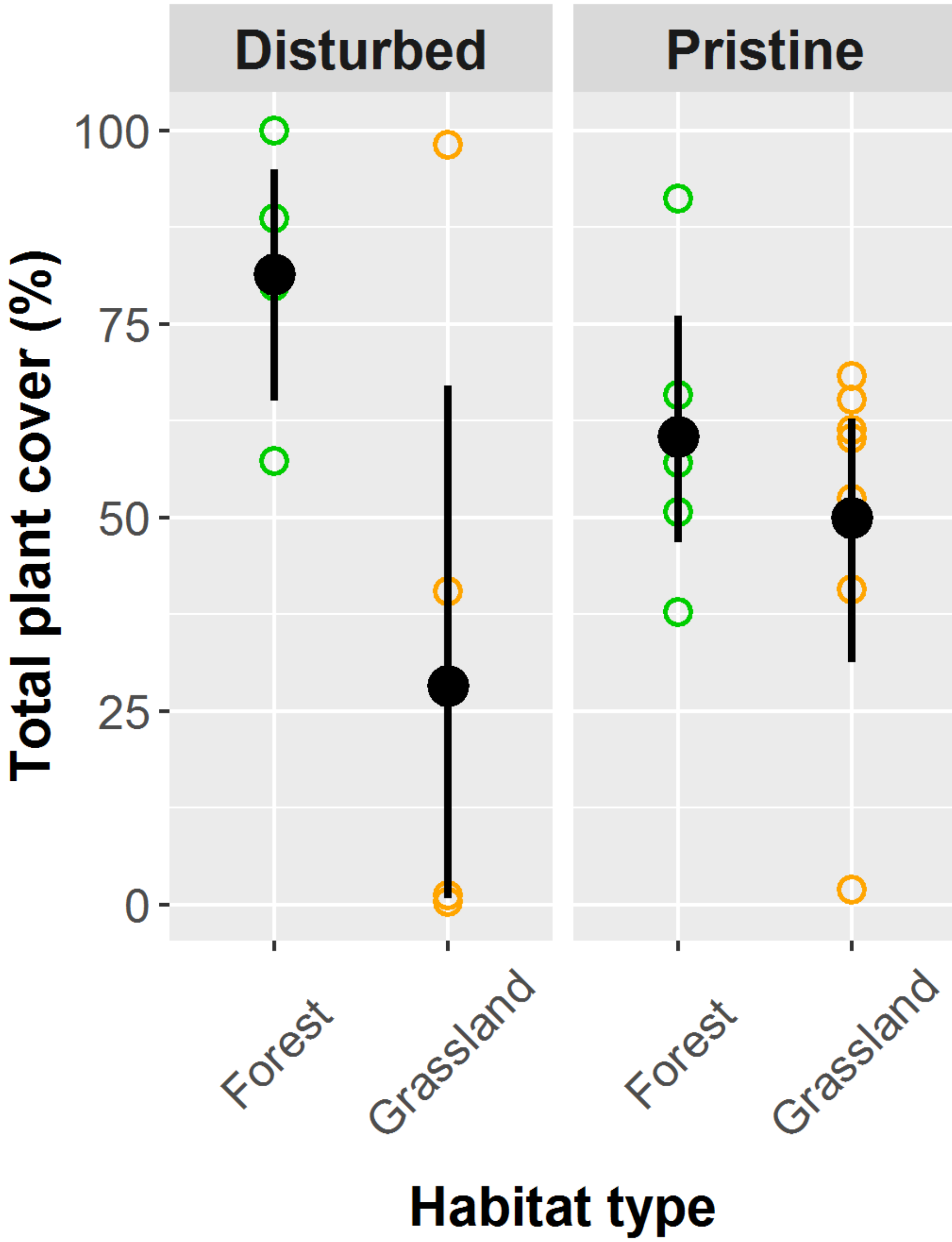
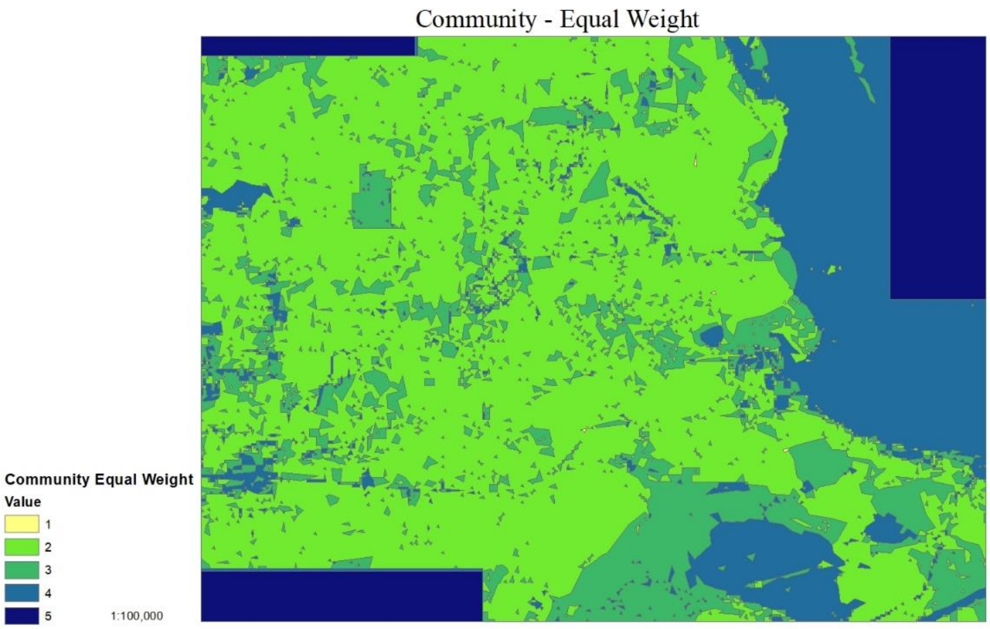


Figure 16: After a post hoc comparison using the Tukey HSD test a significantly higher total herbaceous cover was detected in disturbed forest (M = 81.4, SE = 13.8) compared disturbed grasslands (M = 23.7, SE = 12.6). The two pristine habitat's herbaceous total cover values were not significantly different from each other or from either disturbed habitat category. There values for the post hoc comparison using a Tukey HSD test were pristine forest (M = 62.8, SE = 12.6) and pristine grassland (M = 50.9, SE = 11.5).



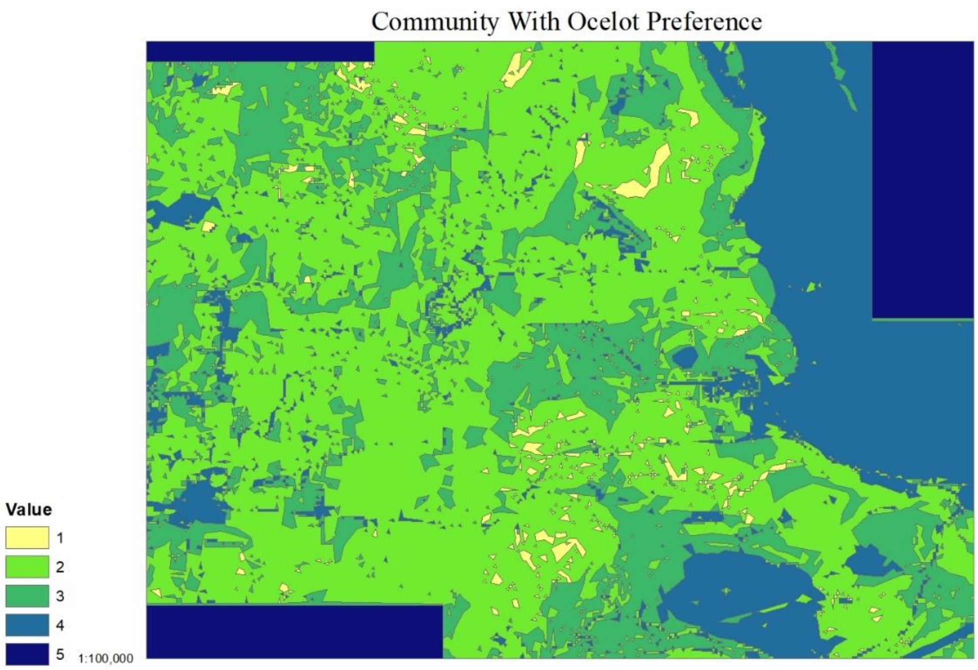


Figure 17: These are the two community models developed by combining the four single focal taxa models together under two weight scheme scenarios. The top map received an equal weighting scheme of 25% for each individual model and the lower map had a weighting scheme of 50% ocelot, 30% aplomado falcon, 10% Texas tortoise, and 10% plant community. The second weighting scheme was developed based on the opinions of experts at the first landscape prioritization meeting.

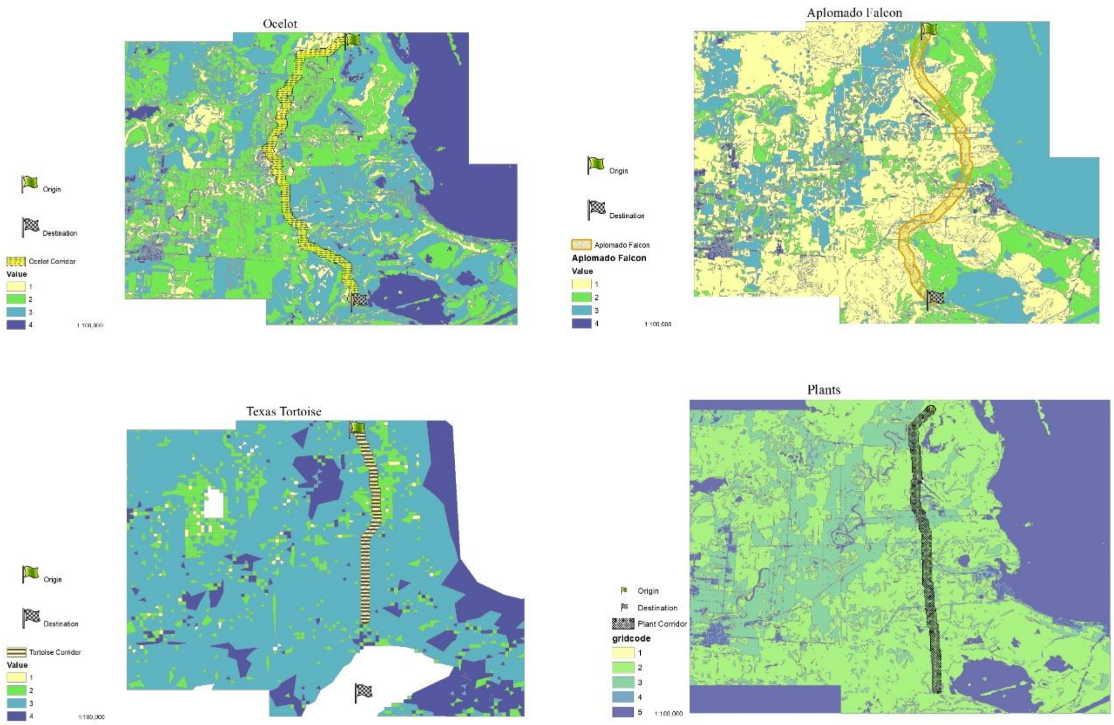


Figure 18: These are the results of the least cost path analysis for each individual focal taxa model. The least cost paths used a standardized origin and destination point that was determined by local experts to be important for ocelots which is the species the land management agency chose as the focal taxon for all corridors to date. Then to make the corridor more realistic a 300-m buffer (based on the minimum size of a landscape-scaled corridor) was placed on each side of the route. The Texas tortoise model is shorter than the other corridors due to the fact the model is undefined in the white space surrounding the destination flag. Therefore, alternative end points were selected surrounding the white space and all the models agreed with the route portrayed in this analysis.

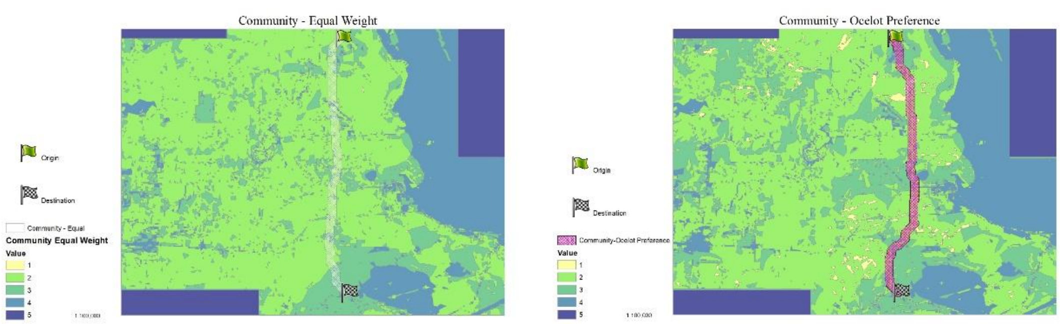


Figure 19: These are the results of a least cost path developed on the community-based corridor model scenarios. The corridor used the same origin and destination points as the individual focal taxa models and the same 300-m buffer on both sides of the path were applied. The difference between the cost surfaces (models) used to create these corridor routes is the equal weights had a 25% weight for each individual taxa model used to create this community cost surface and the ocelot preference use a 50% ocelot, 30% aplomado falcon, 10% TX tortoise, and 10% plant community to develop this model.

All Corridors

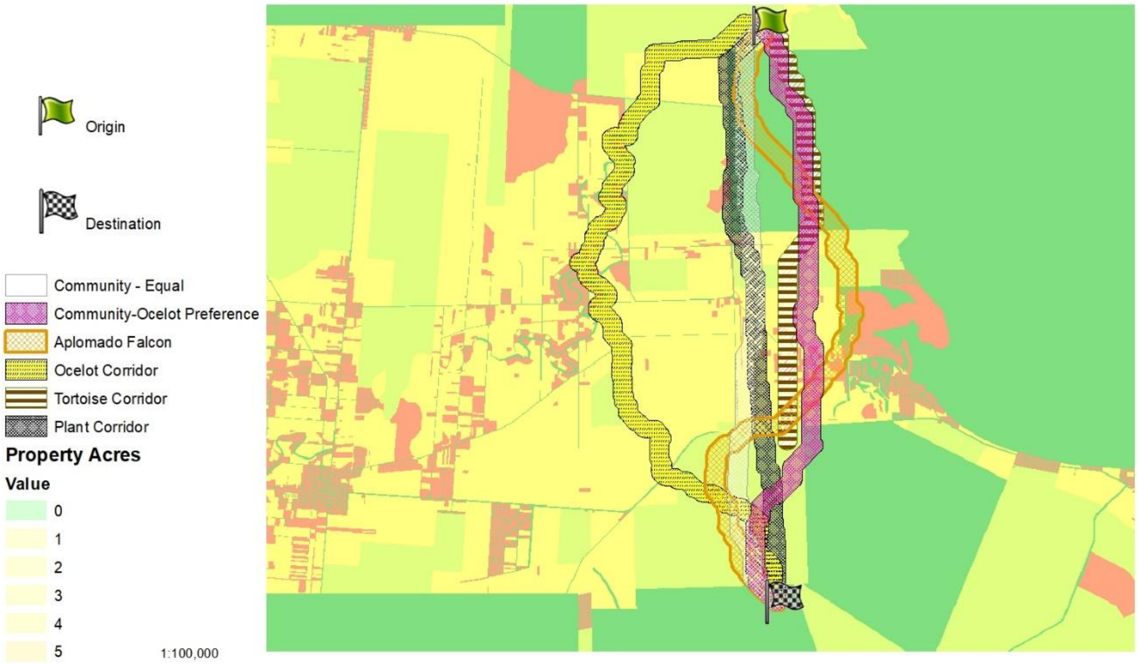


Figure 20: This is a map containing all the least cost path corridors created from each model. The Jaccard similarity index was used to assess how well each model's path agreed with another model's path. The background is a layer that is important to the land managing agency corridor implementation strategy.

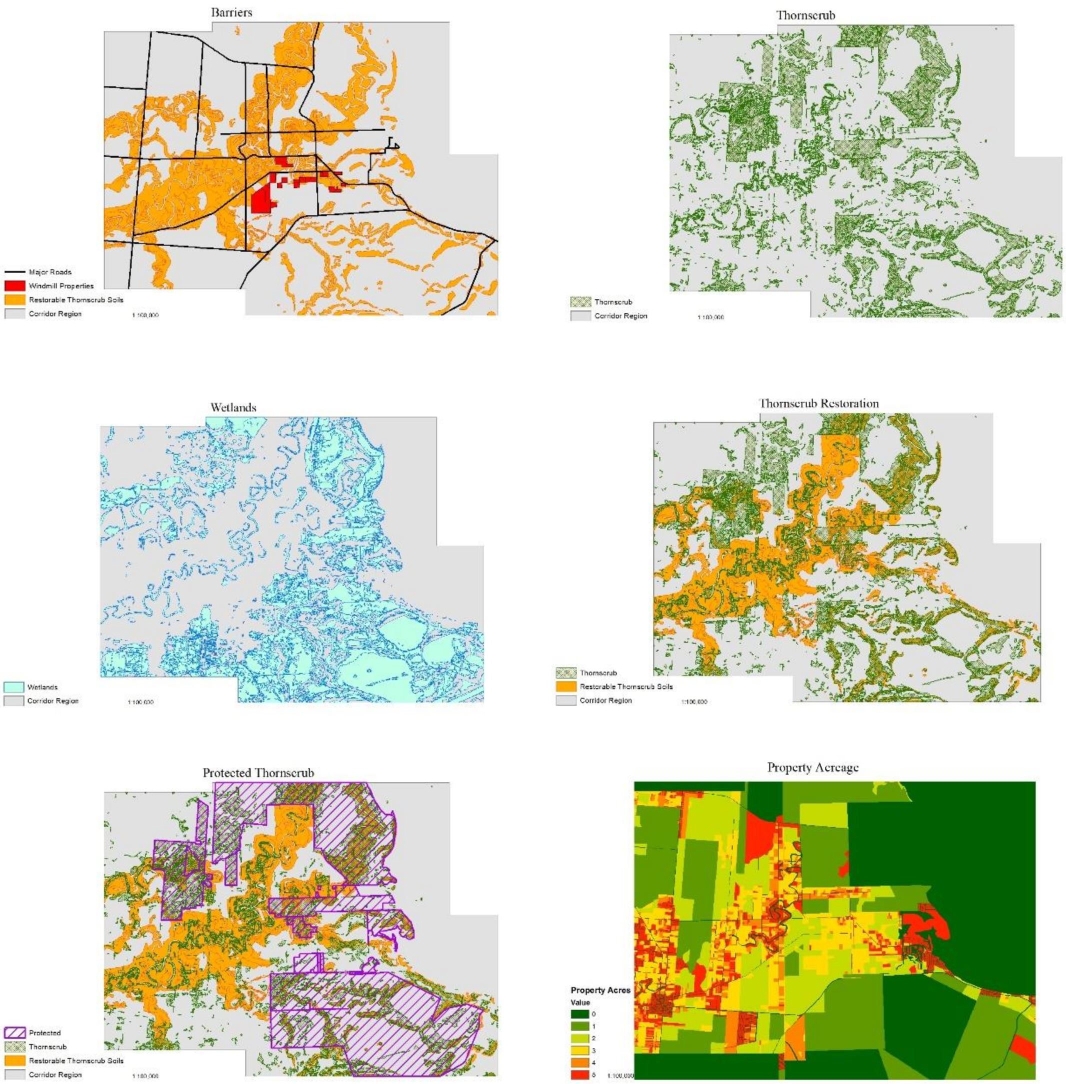


Figure 21: These are layers illustrating various components of the landscape a land managing agency would consider if developing a corridor in this case study region. These layers include barriers to conservation (ex. Roads and windmills), location of forest, location of wetlands, restoration potential (forested areas and areas with soils suitable for forest regrowth), protected area boundaries and parcel allotment distribution in the landscape.

The Mayhem Map

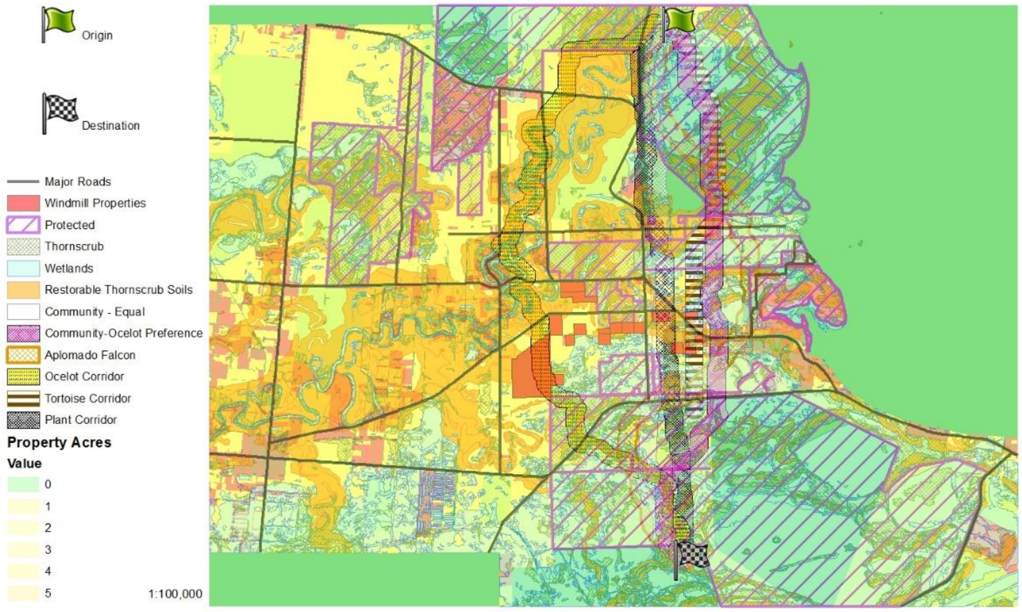
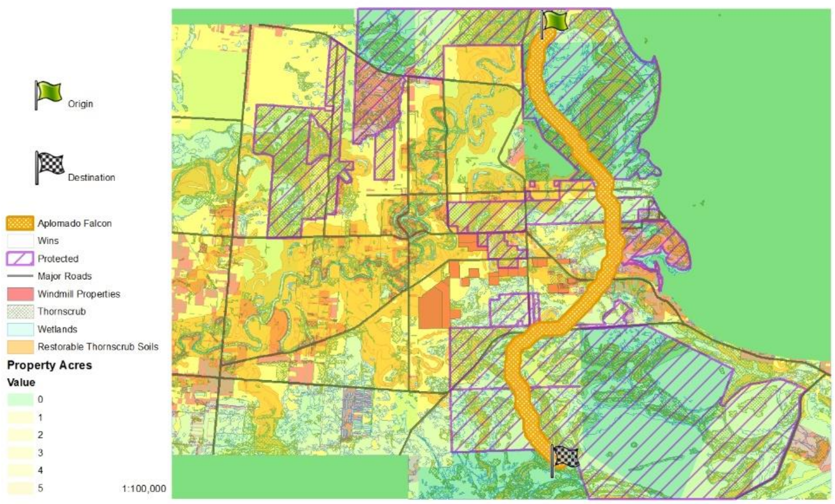


Figure 22: This is an image of all the different factors a land managing agency will consider when assessing whether or not to adopt a corridor. The landscape contains thornscrub supportive soils, current thornscrub forest distribution, wetland areas (includes coastal prairie, freshwater wetlands, brackish water wetlands, and hypersaline wetlands), protected area boundaries, land parcel size by class, roads, and properties containing windmill structures.

Aplomado Falcon Wins



Community - Ocelot Preference is the Grand Winner

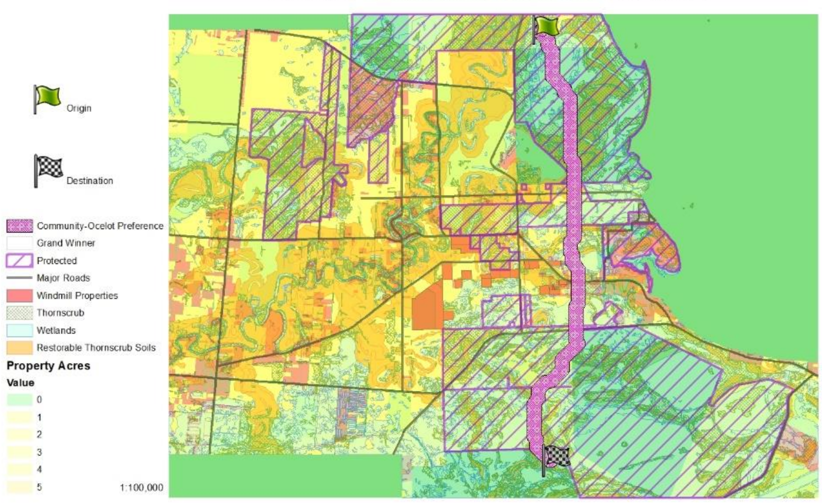


Figure 23: These are the two least cost paths that a land managing agency would consider as the routes that best navigate this landscape. Both models minimize the number of barriers traversed, maximize the amount of protected land already encompassed in the route, minimize the amount of additional areas in need of protection and contain suitable habitat for a variety of species. When comparing these two finalist models, the community with an ocelot preference corridor is preferred over the aplomado falcon corridor because this route has fewer significantly different paths when compared to the other model's corridor routes then the aplomado faclon route that was significantly different to all other corridors.

BIOGRAPHICAL SKETCH

James A. Stilley graduated from the University of Hawaii Hilo with a BS Biology in the Ecology, Evolution and Conservation Tract, minored in Chemistry and Marine Science, and received a certificate in the Marine Options Program in 2014. He graduated with a MS Agricultural, Environmental, and Sustainability Science from the University of Texas Rio Grande Valley in 2019. He is a passionate ecologist engaging in research in both terrestrial and aquatic ecosystems, a scientific scuba diver, and advocate of the conservation of wildlife communities. His permanent address is 5173 N. Banna Ave. Covina CA 91724 and personal email address is jimmystilley@verizon.net.