

ASSESSING THE INFLUENCE OF LANDSCAPE CHARACTERISTICS ON BAT
FATALITIES AT SOUTH TEXAS WIND ENERGY FACILITIES

By

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ABSTRACT

Although wind energy is a viable renewable energy source, strikes by wind turbine blades unintentionally cause bat fatalities. Previous research has suggested siting of wind energy facilities, and turbine placement within facilities, influence the number of bat fatalities; however, there is a knowledge gap regarding the reasons for the variability. This study occurred in Texas, the leading producer of wind energy and home to the greatest diversity and largest colonies of bats in the United States. The objective of my thesis was to assess the influence of landscape characteristics surrounding wind energy facilities and around specific turbines on the number of bat fatalities. I systematically searched 200 wind turbines and collected 1,067 bat carcasses at Hidalgo and Los Vientos Wind Energy Facilities from 2017–2018; I found *Tadarida brasiliensis* (n = 577), *Lasiurus intermedius* (n = 203), *L. ega* (n = 69), *Nycticeius humeralis* (n = 51), *L. xanthinus* (n = 30), *L. cinereus* (n = 18), *L. blossevilli* (n = 2) *Myotis velifer* (n = 2), *Nyctinomops macrotis* (n = 1), *Perimyotis subflavus* (n = 1), and unknown spp. (n = 113). I used Fragstats and ArcGIS Pro to acquire landscape metrics at the two facilities and among the 100 wind turbines at each facility at multiple scales (100 m, 500 m, 1 km, 5 km, and 25 km). Landscape characteristics included landcover types such as barren, crops, herbaceous, developed, shrub/scrub, hay/pasture, forest, wetlands, and open water, proximity to water sources, elevation and degree of slope. Using generalized linear models, zero-inflated and negative binomial models, and AIC model selection, results indicate that landscape characteristics at the broadest scale (5 km and 25 km) examined

were most strongly associated with estimated bat fatality rates. I suggest wind farms should be constructed in areas that consist of uniform and connected habitat throughout the facility, without the presence of fragmented water sources. Managers should also implement mitigation efforts and pre/post-construction assessments to potentially reduce bat fatalities.

I. INTRODUCTION

Chiroptera is the second largest mammalian order globally and one of the most diverse in North America, with 45 recognized bat species occurring in the United States (U.S.) and Canada (Hammerson et al. 2017). Bats are primarily nocturnal, and many emit a wide array of frequencies for echolocation to aid in communication, navigation, and prey capture (Neuweiler 1990; Ammerman et al 2012; Vanderelst 2018). Most bats in the U.S. navigate using echolocation and have evolved body structures, such as fleshy flaps or projections of the ears and nose, that enhance interpretation of echolocated signal (Lawrence and Simmons 1982; Wotton et al. 1995; Neuweiler 2000; Ammerman et al. 2012). Bats are the only mammals capable of sustained flight, with numerous morphological modifications aiding in the process, such as forelimbs modified into airfoils, reduction in their skeleton, and the development of a keeled sternum and clavicle (Ammerman et al. 2012; Vanderelst 2018).

Though some bat species respond to seasonal temperature changes through migration to new roosting sites or by conserving energy using torpor, other species remain residential and active if resources are sufficient (Fleming and Eby 2003; Ammerman et al. 2012; Barclay et al. 2017). Depending on the species of bat, roosting sites, such as naturally occurring caves and trees or anthropogenic structures like bridges and culverts, can be occupied year-round for hibernation and maternity colonies because of adequate temperature, food availability, and/or physiology (Kunz 2005; Dunbar 2010; Ammerman et al. 2012; McGuire 2013). Bats that migrate typically do so between summer and winter areas, with fall transient colonies forming in August, and formation

of over-wintering colonies peaking in September and October (Barbour and Davis 1969; Johnson et al. 2005; Fleming and Eby 2003).

Bats are important bioindicators, susceptible to changes in their population or activity due to their sensitivity to surrounding environments (Vaughan et al. 1996; Jones et al. 2009). Bats also provide essential functions that help support healthy ecosystems, acting as major pollinators, seed dispersers, and insect regulators, which helps to promote biodiversity (Boyles et al. 2011; Ammerman et al. 2012). Insectivorous species of bats in the U.S. are estimated to be valued at an average of \$22.9 billion per year for crop pest management (Boyles et al. 2011).

Despite their environmental and economic importance, bat species face a myriad of threats, including population-level impacts from contaminants (e.g., lead and pesticides) and diseases like white-nose syndrome, destruction of habitat, climate change, and wind turbines (O'Shea et al. 2016; Frick et al. 2017). In fact, bats are considered one of the most threatened groups of vertebrates worldwide (Hammerson et al. 2017) with fatalities at wind turbines creating one of the largest global sources of mass fatality of bat populations (O'Shea et al. 2016). Due to low annual reproductive output (typically 1 offspring a year, some species with as many as 4) and low natural mortality rates, there is a limited capacity for bat populations to rebound after sharp declines (Racey and Entwistle 2000; Barclay and Harder 2003; Podlutzky et al. 2005; Jones et al. 2009; Fleischer 2017). Thus, bat fatalities from wind energy production are a major concern for population stability (Cryan 2011; Barclay et al. 2017; Frick et al. 2017; Sanchez-Navarro et al. 2019). Wind turbines primarily cause bat fatalities through turbine blade strikes, especially impacting fast and high-flying species during the late summer to fall (Kunz et

al. 2007; Rydell et al. 2010; Rollins et al. 2012; Arnett et al. 2016). Despite the lack of consistent bat fatality data, there were an estimated 840,486 to 1,690,696 bat fatalities from 2000 to 2011 in the U.S. and Canada, with roughly 196,190– 395,886 bat fatalities estimated to have occurred in 2012 (Arnett and Baerwald 2013). Other studies have estimated that annual bat fatalities within the U.S. exceed 500,000 per year (Hayes 2013; Smallwood 2013). As the number of wind energy facilities have more than doubled in the past decade, annual bat fatalities have likely increased as a result with one study estimating 1.72 million U.S. bat fatalities in 2019 (Rand et al. 2020; Smallwood 2020).

Fatality risk at wind energy facilities varies by bat species and likely is dependent on a variety of taxa-specific factors such as habitat, foraging and breeding behaviors, morphology, and migration patterns (Denzinger and Schnitzler 2013; Peters 2020). Though species composition and fatality patterns vary across regions, migratory, tree-roosting species, including the hoary bat (*Lasiurus cinereus*), eastern red bat (*L. borealis*), and silver-haired bat (*Lasionycteris noctivagans*), are the most vulnerable (Arnett et al. 2005; Zimmerling and Francis 2016; American Wind Wildlife Institute 2020; Choi et al. 2020). Particularly in the Midwest and Northeast region of North America, this group comprises nearly 80% of fatalities, possibly because they perceive wind turbines as roosting structures (Kerns and Horn 2005; Zimmerling and Francis 2016; Barclay et al. 2017; Thompson et al. 2017). Tree height is typically an important factor for roost selection, with taller trees favored by bat species (Crampton 1998; Kalcounis-rüppell 2005). Recent studies suggest a relationship between increased turbine heights and increased mortality rates (Barclay et al. 2007; Fiedler et al. 2007). However, Brazilian free-tailed bats (*Tadarida brasiliensis*) are under-represented and can constitute

a large portion of fatalities within its range (American Wind Wildlife Institute 2020, Weaver et al. 2020). Furthermore, bat species with high wing loading (mass of bat divided by wing area), such as Brazilian free-tailed bats, are also susceptible to wind turbine fatalities due to this characteristic associated with adaptations of high levels of foraging activity in open spaces (Arnett et al. 2016; Weaver 2020). In addition, the timing of Brazilian free-tailed bat migration coincides with that of many moth species, an important source of food for the species (Rydell 2010; Krauel et al. 2015). Similar migratory patterns and foraging behaviors in various bat species could explain the surge in turbine-related fatalities during fall migration due to insect abundance or the shift in foraging habits, creating high concentrations of bats in some areas (Cryan and Barclay 2009). Additionally, sources of water used by bats for drinking and foraging for aquatic emergent insects may influence distribution of bat species relating to fatalities at wind turbines near water sources (Korine 2016).

Alongside species-specific behavior, wind turbine siting and structure, including height and dimension of rotor-swept areas (the area of the circle created by turbine blades as they sweep or move through the air) on turbines, towers lit with artificial lights, and surrounding habitat, have been suggested to affect the number of bat fatalities (Kerns and Horn 2005; Arnett 2008; Baerwald and Barclay 2009). For example, Brazilian free-tailed bats can fly up to 3,000 m in altitude, however hunting heights can be variable for this species such as 15 meters or ranging from 400-750 meters (Williams et al. 1973; McCracken 1996; McCracken et al. 1999; Best 2003), except when sweeping low over bodies of water (Schmidly 2016). Average turbine heights in the U.S. typically are around 90 meters (Wiser 2022). Wind energy facilities may provide adequate hunting

grounds for bats because of the accumulation of insects near turbines, possibly due to insect attraction to turbine lights or turbine paint color (Ahlén 2003; Cryan et al. 2009; Long et al. 2011; Foo et al 2017; Grof-Tisza et al. 2017, Guest et al. 2022). However, numerous studies (Johnson et al. 2004; Fiedler et al. 2007; Baerwald and Barclay 2011; Bennett and Hale 2014) have not been able to determine significant differences between lit and unlit turbines. Additionally, research indicates the placement of wind turbines in relation to topography may pose significant risks for bat collision (Orloff and Flannery 1992; Hoover 2002; Piorkowski and O’Connell 2010). High-elevation ridge top sites often encourage insect hilltopping behavior, the accumulation of flying insects on topographic features to increase likelihood of mating, which may, in part, lead to higher bat fatalities in these locations as they forage (Arnett et al. 2008; Grof-Tisza et al. 2017, Guest et al. 2022). Studies supporting the insect hilltopping hypothesis are lacking, yet Jansson (2020) used lidar technology to observe swarms of insects at the top of turbines that remained until dark. However, conflicting results from Sanchez-Navarro et al. (2019) indicate lowland areas with lines of crops and pastures are a locality of high bat fatalities due to the abundance of insects near agricultural fields in Spain. Thus, despite the variability in regional topographical trends, topography, elevation, or slope of wind turbines may possibly influence the activity of bats depending on presence of some food sources (insects) (Cruz et al. 2019).

However, there is a knowledge gap regarding landscape relationships with bat fatalities at wind energy facilities because some facilities occur in locations with little landscape variation (Peters 2020). Certain landscape characteristics are likely influencing bat foraging behaviors which, in turn, may influence bat fatalities (Minderman 2012).

These landscape characteristics include proximity to wetlands and water sources, degree of surrounding forest patches and open, non-cultivated habitat, degree of surrounding landscape fragmentation and edge, and configuration of various landcover types within the wind energy facility (Siemers and Schaub 2011; Jantzen and Fenton 2013; Fulton et al. 2014; Ferreira et al. 2015; Starbuck et al. 2015; Bachen et al. 2017; Peters 2020). For instance, some bat species mainly forage and commute along forest edges and linear features, which are commonly created for wind turbine pads and roads, thus indirectly causing increased bat activity and possible fatalities (Walsh and Harris 1996; Kunz et al. 2007; Morris et al. 2010).

Wind energy is a viable source for renewable energy due to its numerous economic and environmental benefits, and by 2020, wind turbines were the source of 8.4% of U.S. electricity generation (U.S. Energy Information Administration 2021). By 2040, the demand for global energy is expected to increase by 25% while simultaneously the need for renewable energy alternatives is increasing due to climate change (International Energy Agency 2018). Renewable energy alone is predicted to provide ~51% of global electricity demand, with a significant portion coming from wind energy (Kiesecker and Naugle 2017). Wind energy is considered one of the most environmentally benign sources of energy because it produces minimal air/water pollution, generates no hazardous waste, and exploits less natural resources, including water, to generate power (Matthew 2006; Slattery 2011). This is unlike other current energy sources, such as fossil fuels, that are responsible for ~75% of CO₂ emissions, ultimately causing greenhouse effects like acid rain, smog, and leaving toxic heavy metals and particulates in the atmosphere (Matthew 2006). Despite wind energy emitting

pollutants during its construction, commissioning, and decommissioning phases, it is one of the “cleanest” sources of energy available today compared to all other sources of energy (Matthew 2006). However, the U.S. Department of Energy (DOE) has predicted approximately 7 million hectares of land are needed to reach onshore wind energy needs, with additional land needed for new transmission lines (U.S. Department of Energy 2015). This amount of land use change and increase in obstacles adds to the stressors on bat populations, creating new concerns among conservationists.

Texas is currently the leading producer of wind energy in the U.S. and provides ~26,000 full time jobs related to the field (Powering Texas 2020). As of 2019, Texas produced three-tenths of the U.S. total wind-powered energy and had an installed wind energy capacity of 28,000 megawatts (MW) (U.S. Energy Information Administration 2021). In 2020, Texas started operation of 168 MW in new capacity, with over 3,000 MW under construction, and new tower heights increased up to 140 m to gain better wind access (Knutson 2020; U.S. Department of Energy 2020). Tower height is important to note as previous research has indicated bat fatalities increase with tower height (Barclay et al. 2007; American Wind Wildlife Institute 2020; Anderson et al. 2022). Anderson et al. (2022) results add to the ongoing pattern of strong relationships between bat mortality and turbine blade heights, especially during the late summer.

With both supply and demand for wind energy increasing in the state, bats in Texas face significant wind energy-related challenges. Texas has the greatest diversity of bat species (68% of all U.S. species) occurring in all major ecoregions (Ammerman et al. 2012), many of which have been documented as fatalities at wind turbine facilities (Weaver et al. 2020). Over a 12-year period ending in 2011, Arnett and Baerwald (2013)

recorded bat fatalities at wind energy facilities throughout the U.S. including species such as Brazilian free-tailed bat, cave myotis (*Myotis velifer*), evening bat (*N. humeralis*), northern yellow bat (*Lasiurus intermedius*), southern yellow bat (*L. ega*), western yellow bat (*L. xanthinus*), western red bat (*L. blossevilli*), and tricolored bat (*Perimyotis subflavus*), all of which migrate through or permanently reside in Texas. Because of the diverse bat species within Texas, various types of habitats and vegetation are used according to species' foraging needs and preferred roosting sites. For example, hoary bats, northern yellow bats, evening bats, and western red bats use forest edges, clearings, and waterways to forage; cave myotis and western yellow bats forage over dense riparian vegetation and dry desert vegetation; and Brazilian free-tailed bats and tricolored bats heavily forage in agricultural areas (Ammerman et al. 2012; Schmidly 2016). Thus, landscape characteristics and resource availability have the potential to influence bat activity and fatalities at wind-energy facilities within Texas due to the large diversity of vegetation throughout the state (Schmidly 2016).

Ultimately, there is a large gap in knowledge of the relationships between landcover and bat fatalities in Texas, primarily due to a lack of publicly available or peer-reviewed studies along with a general lack of consensus among studies, which limits understanding and accurately managing for bats. Investigating how landcover influences bat fatalities in Texas plays an important role in understanding wind energy impacts and aid in guiding future wind energy development and protect bat populations (Weaver 2019). The objective of my study is to examine the influence of landcover on bat fatalities at two spatial scales: wind energy facility and individual wind turbine. I hypothesized that bat fatalities are not equal across the region and that landscape

variables will influence bat fatalities at wind energy facilities. I predicted that bat fatalities will be associated with landcover types considered high quality foraging habitat depending on bat species (i.e., forest, wetlands, and proximity to water sources – habitat that would most likely be used for foraging related to specific dietary preferences). Additionally, I predicted there will be a positive relationship between bat fatalities and amount of surrounding agricultural-related landcover due to the possible high abundance of insects being present near agricultural land, therefore, influencing bat activity in these areas (Sanchez Navarro 2019). Lastly, I predicted relationships between bat fatalities and habitat surrounding the entire facility, rather than at the turbine scale alone (Starbuck et al. 2015; Baerwald 2018). Predictions will heavily vary by bat species and fatality estimates. This research aimed to expand and contribute to the knowledge we have on bat species in Texas and to assist in creating necessary conservation policies and management strategies that can be relevant across regions in the southern U.S.

Study Sites – Los Vientos Wind Energy Facility and Hidalgo Wind Energy Facility

This study was conducted within two wind energy facilities located in southern Texas. Los Vientos III, IV, and V Wind Energy Facilities (combined as one study site - Los Vientos) are owned and operated by Duke Energy Renewables, northeast of the Rio Grande River in Starr County, Texas. Los Vientos consists of 255 wind turbines, extending across ~22,666 hectares of leased land. Hidalgo Wind Energy Facility (also known as Los Mirasoles) is located in Hidalgo County, Texas, which neighbors Starr County. Hidalgo consists of 125 wind turbines, extending over 18,965 hectares of land that is owned and operated by EDP Renewables. Both facilities are within the Texas

Tamaulipan Thornscrub Level IV ecoregion, with dominate vegetation consisting of drought-tolerant, thorn woodland and thorn shrubland (Griffith et al. 2007). These areas are dominated by drought-tolerant, thorn-laden, and small-leaved trees and shrubs. The most prominent tree in this area is honey mesquite (*Prosopis glandulosa*), followed by a dense understory of trees and shrubs such as brasil (*Condalia hookeri*), colima or lime pricklyash (*Zanthoxylum fagara*), Texas persimmon (*Diospyros texana*), lotebush (*Ziziphus obtusifolia*), granjeno (*Celtis ehrenbergiana*), kidneywood (*Eysenhardtia texana*), coyotillo (*Karwinskia humboldtiana*), Texas paloverde (*Parkinsonia texana*), anacahuita (*Cordia boissieri*), and various species of cacti. Xerophytic brush species, such as blackbrush (*Vachellia rigidula*), guajillo (*Senegalia berlandieri*), and cenizo (*Leucophyllum frutescens*), are typical on the rocky, gravelly barren ridges and uplands. Within grasslands common species include cane bluestem (*Bothriochloa barbinodis*), silver bluestem (*B.laguroides*), multi-flowered false rhodesgrass (*Trichloris pluriflora*), sideoats grama (*Bouteloua curtipendula*), pink pappusgrass (*Pappophorum bicolor*), bristlegresses (*Setaria* spp.), lovegrasses (*Eragrostis* spp.), and tobosa (*Pleuraphis mutica*) (Griffith et al. 2007). The bat community within Starr and Hidalgo Counties, Texas is comprised of primarily insectivorous species from families Molossidae, Mormoopidae, and Vespertilionidae (Weaver 2019).

II. METHODS

I used previously collected data from Weaver et al. (2019) examining the impacts of wind energy on bats within Texas at both wind energy facilities. Bat fatality monitoring was completed during March 2017–March 2018 at Los Vientos and between August 2017–August 2018 at Hidalgo. For each site, bat fatalities were estimated by randomly selecting 100 turbines out of the total turbines for fatality searches. Circular plots were established around 8-randomly selected turbines from the 100 selected at each site, ranging from ~60 m to ~100 m in radius. These plots were searched by biologists walking transects placed 5 m apart established across the diameter, searching 2.5 m on both sides of transects. For the other 92 wind turbines, turbine pads and roads 100 m from the turbine were searched. For each individual carcass obtained during searches, the following data were recorded: GPS location, time, species, sex, age, distance from turbine, estimated time of death, visibility class surrounding carcasses (easy or moderate-difficult due to vegetative cover), and completeness of carcass (Weaver 2019). Efforts from carcass searches provided 1,067 bat carcasses were collected under 200 wind turbines at the two wind facilities from 2017–2018 (Table 1); species include Brazilian free-tailed bat ($n = 577$), northern yellow bat ($n = 203$), southern yellow bat ($n = 69$), evening bat ($n = 51$), western yellow bat ($n = 30$), hoary bat ($n = 18$), cave myotis ($n = 2$), western red bat ($n = 2$), big free-tailed bat ($n = 1$), tricolored bat ($n = 1$), and unknown spp. ($n = 113$). GenEst (Generalized Mortality Estimator) software version 1.2.1 was used to estimate fatality rates of the 1,067 bat carcasses collected between both sites (Dalthorp et al. 2018a). This program is a recently developed package of modeling and software tools used for estimating bird and bat fatality rates for various renewable-energy projects

(Dalthorp et al. 2018a; Weaver 2019). During this study, the overall estimated median bat fatality rates were 8,197 (95% *CI*: 6,054–11,883) bats per year at Hidalgo Wind Energy Facility (S. Weaver pers. comm.), and 8,336 (95% *CI*: 5,917–15,082) bats per year at Los Vientos Wind Energy Facility (Weaver et al. 2020).

GIS Analysis

I used ArcGIS Pro to visually organize and analyze previously recorded GPS points of the individual wind turbines at both facilities (Fig. 1; Esri 2019). I imported two layers, one consisting of turbines from Los Vientos and the second consisting of turbines from Hidalgo, into ArcGIS Pro as two separate shapefiles. Once GPS locations of turbines were plotted, I used the 2015 North American Land Change Monitoring System (NALCMS), a landsat-based, 30-m resolution classification system for 19 land cover classes across Canada, United States, and Mexico, as the base layer for this analysis to encompass the study sites (Homer 2020). I combined original NALCMS classes into 8 based on similarities of vegetation category – cropland, wetland, grassland, shrubland, barren land, developed, water, and forest (Table 2). Reducing the number of land cover classes created a more biologically-relevant base layer of data for the wind facilities in southern Texas, excluding specific vegetation categories that do not occur in this area. I calculated the proximity to water using the National Wetland Inventory (NWI) and used the U.S. Geological Survey digital elevation model (DEM) layer for southern Texas elevation and slope. I merged DEM layers into one raster layer and extracted elevation values per cells using Data Management tools.

To determine the structure and cover of vegetation at each wind turbine, I created circular buffers at multiple scales (radii = 100 m, 500 m, 1 km, 5 km, and 25 km) around

each turbine (Fig. 2). I determined 100 m buffer as the minimum scale, because it complimented the 100 m plots established in previous fatality searches. I determined the 25 km buffer as the maximum scale to encompass both wind energy facility study sites. I extracted the percent and square meters of each landcover type within each wind turbine buffer using an ArcGIS Spatial Analyst tool between the NALCMS layer and the corresponding buffers. I used the Near tool to determine Euclidean distance to nearest water source to each turbine. Following this, I generated slope degree and elevation from mosaiced DEM layers for Starr and Hidalgo Counties by using 3D and Spatial Analyst tools through ArcGIS Pro.

Landscape level metrics

I computed landscape level metrics for Los Vientos and Hidalgo using FRAGSTATS, a program that provides a comprehensive set of spatial statistics and descriptive metrics of pattern at the patch, class, and landscape levels (Haines-Young and Chopping 1996). There are many interpretations of “landscape” concept, for the purpose of this thesis, I defined landscape as a pattern and a mosaic of habitat patches (Dunning et al. 1992). Patch level metrics are calculated for individual patches which represent areas of similar characteristics, class level metrics are calculated from all patches of a particular type, whereas landscape level metrics are the combination of all patch and class types in a distinct area (McGarigal 2012; Sertel 2018). There are many landscape metrics such as Largest Patch Index (LPI), Euclidean Nearest Neighbor Distance (ENN), Percentage of Landscape (PLAND), Splitting Index (SPLIT) that can be used in patch, class, and landscape levels (McGarigal 2002). To compare the landscape pattern levels between the two wind energy facilities, I used data with the same spatial resolution at the class level. I

used class level metrics to estimate the spatial structure of the two study areas because information accumulated in class level metrics might not exist at the patch level alone and landscape level could be too broad (Sertel 2018). Class level indices separately quantify the amount and spatial configuration of each patch type and thus provide a means to quantify the extent and context of each patch type in the landscape (McGarigal 2012). Thus, I determined class level landscape metrics are more effective in defining ecological processes in relation to this study (Tischendorf 2001). I clipped the NALCMS layer into two sections and then dissolved each layer with a 5-km buffer to keep wind energy facilities separate to be able to distinguish landscape differences in each facility. I converted clipped sections to a Tagged Image File Format (TIFF) and uploaded each clipped layer into FRAGSTATS. I calculated class metrics such as percentage of landscape (PLAND), total edge (TE), largest patch index (LPI), edge density (ED), Euclidean nearest neighbor distance, area-weighted mean (ENN_AM), splitting index (SPLIT), and number of patches (NP) using FRAGSTATS (Descriptions on Table 3). I used an 8 x 8 m cell neighborhood rule in FRAGSTAT software to calculate all metrics. Once completed, I evaluated the landscape metrics results.

Data analysis

For all statistical analyses, I developed models using counts of estimated bat fatalities as the response variables and amount of present land cover types within each buffer distance, distance to water sources, slope degree, and elevation as independent variables. First, I used the “scale” function in R to scale each variable; essentially this function calculates the mean and standard deviation of the entire vector then subtracts the mean and divides by the standard deviation. I split data by wind energy facility and

scaled all covariates to ensure common scales across models. Next, I calculated variance-inflation factors (VIF) and used $VIF < 3$ to reduce multicollinearity (Zuur et al. 2010). I created all models per facility representing various hypotheses regarding influence of landscape characteristics on bat fatalities. To account for potential model overdispersion, I used a generalized linear regression (GLM) with a negative binomial distribution for data within Hidalgo. I chose to use a zero-inflated regression for Los Vientos due to the frequency with which no carcasses were counted. Additionally, I fitted Los Vientos data to zero-inflated Poisson and zero-inflated negative binomial models. I determined the negative binomial model was the best-fitting model and accounted for overdispersion within the data. I evaluated model selection using Akaike's Information Criterion (AIC) (Bozdogon 1987). I used software R (version 4.04, R Core Team 2020) for all statistical analyses.

Additionally, I created species-specific models (i.e., Brazilian free-tailed bat (TABR) and northern yellow bats (LAIN)) to investigate the possible influence of spatial predictors. I chose to focus on these two bat species because of the large numbers of fatalities compared to the others. If species-specific models failed to converge, I considered the species to have a low sample size ($n < 100$) and removed it from the analysis. I built 16 models for fatality estimates of combined bat species, TABR species, and LAIN species (48 models per facility; 96 total). Once models were built, I used AIC model selection procedure and used the model with the lowest delta value for further estimates. If all three categories had two or more models with ΔAIC_c values < 2 , I model averaged top models to obtain estimates (Mazerolle 2020). Before models were averaged,

I assessed top model coefficients for multicollinearity using $VIF < 3$. (Lukacs, Burnham & Anderson 2010; Cade 2015).

III. RESULTS

The averaged GLMs had few informative predictors for both facility models (Tables 4 & 5). For Hidalgo, after model averaging the top two models with AIC delta <2, the top model for all bat fatality estimates included only one significant predictor, water at 25 km ($\beta = 0.47$; 95% *CI*: 0.27 – 0.68; $P = 0.002$, Table 6), indicating bat fatalities increased alongside the increased proportion of water at the broadest scale alongside bat fatalities. Species-specific models yielded different trend estimates for Brazilian free-tailed bats and northern yellow bats. The top two averaged models indicated that the best fit model for Brazilian free-tailed bat fatality estimates included water at the 5-km scale ($\beta = 0.43$; 95% *CI*: 0.17 – 0.75; $P = 0.001$; Table 6), revealing an increase of Brazilian free-tailed bat fatalities as the proportion of open water increased at this broad scale. The best fit model out of the three averaged models for northern yellow bat fatality estimates included cropland at 25 km ($\beta = 0.52$; 95% *CI*: 0.21 – 0.83; $P = 0.01$, Table 6). This revealed a positive relationship between northern yellow bat fatalities and the proportion of cropland within 25 km.

For Los Vientos, the best fit model for combined bat fatality estimates included a positive relationship with wetland landcover at 25 km ($\beta = 0.19$; 95% *CI*: 0.05 – 0.35; $P = 0.008$; Table 7). The best fit model for the Brazilian free-tailed bat included wetland at 100 m, but conditional averages included wetland at 100 m, forest landcover at 5 km, and wetland at 25 km. I concentrated on full averages from model averaging, therefore, Brazilian free-tailed bat fatalities at this facility had a negative relationship with the proportion of wetland within 100 m of wind turbines ($\beta = -1.25$; 95% *CI*: -2.10 – -0.40; $P = 0.004$; Table 7). The best fit model for northern yellow bat fatality estimates included

barren land at 5 km and shrubland at 5 km for the full average; conditional average included barren land at 5 km, shrubland at 5 km, and water at 5 km. Full average predictors indicate northern yellow bat fatalities have a positive relationship with barren land within 5 km ($\beta = 0.25$; 95% *CI*: 0.12 – 0.37; $P = < 0.0001$; Table 7) and a negative relationship with shrubland within 5 km ($\beta = -0.19$; 95% *CI*: -0.31 – -0.07; $P = 0.002$; Table 7).

Landscape Level Metrics

The FRAGSTATS analysis of landscape within a 5-km buffer around the two wind energy facilities, indicated differences among the two landscapes (Table 8). Landscape metrics were defined as values “within” wind energy facility buffers. The Hidalgo calculated metric values indicated that cropland was the dominant class (PLAND = 48.91). The LPI and ED values for this landcover type were greater than other landcover types within this area, with low ENN_AM and SPLIT values. These values suggested that compact, large, and related spatial units of cropland were concentrated within this landscape class. Forest and water classes were identified as the two smallest habitat types in terms of PLAND, LPI and NP, but had some of the greatest ENN_AM and SPLIT values. These landcover types can be interpreted to have had some of the greatest amounts of fragmentation and average distances between spatial units of the same landcover type. Essentially, these values indicated that the patch-scales of each landcover types (forest and water bodies) were small, located far away, and scattered throughout the landscape (Sertel 2018).

The dominate landcover class within the Los Vientos study area was the shrubland class (PLAND = 56.09). Shrublands, which covers much of this facility, had

low values in ENN_AM and SPLIT, indicating this landcover class was intact and had greater connectivity. The water class had the lowest PLAND, LPI and NP values (PLAND = 0.0036, LPI = 0.0664, NP = 14), but were relatively far apart (ENN_AM = 300.41) and scattering level was high (SPLIT = 2,197,491.32) though not as high as forest and grassland. Grassland had low PLAND (PLAND = 2.68) and LPI (LPI = 0.25) values, with the highest SPLIT and NP values, indicating that the grassland class had an abundance of small patches that were very scattered throughout the landscape, like the structure of forest in this facility. Forest also had a high value for SPLIT and the highest ENN_AM value (ENN_AM = 11,120.28), which indicated a small number of spatial units that were fragmented and scattered throughout Los Vientos.

IV. DISCUSSION

As predicted, bat fatalities were associated with landcover types considered high quality foraging habitat for Brazilian free-tailed and northern yellow bats, specifically foraging habitat related to specific dietary preferences (i.e., water and cropland landcover types). Both wetland and water landcover were predictors for combined bat fatality estimates and Brazilian free-tailed bat fatality estimates at both facilities suggesting the importance of water in this region. One of Hidalgo's best landcover predictors across all three models was proportion of water (25 km and 5 km) with a positive association with estimated bat fatalities. Los Vientos had three different predictors, but wetland (25 km) had a positive association with bat fatalities. Water sources often are associated with frequent bat activity due to the importance of commuting/migration landmarks (Grindal et al. 1999; Serra-Cobo et al. 2000; Schnitzler, Moss, and Denzinger 2003; Johnson et al. 2004), and as a source for drinking and foraging (Grief 2010; De Conno et al. 2018). Additionally, bats lose large amounts of water due to high body surface-to-volume ratio, naked wing membranes and as long-distance flyers, making water balance and food intake necessary (Reichard 2010; Minnaar et al. 2014). Many species also typically choose roosts that are near water and forage in surrounding habitats (Best 2003; Li 2015; McCracken 2018). Within wind energy facilities, bats may be attracted to open water sources, which provide drinking water and an abundance of insects (Grindal et al. 1999, Seidman and Zabel 2001, Ciechanowski 2002, Johnson et al. 2010). In general, water bodies are critical to the foraging habits of bats, but attraction can vary between species. For example, eastern red bats, little brown bats, and cave myotis activity is associated with water sources ranging from rivers to ponds (Nelson and Gilliam 2019; Kunz 1974,

2005). Given the more diverse and abundant insect communities within lentic water bodies, such as ponds, there can be greater levels of bat activity due to the ease of drinking and eating (Mackey and Barclay 1989, Walyesh and Harris 1996). Aquatic habitats such as wetlands, riparian zones, ponds, etc. may also influence bat activity due to abundance of drinking water and insects, therefore, possibly influencing increased bat use in these areas (Grindal et al. 1999, Seidman and Zabel 2001, Ciechanowski 2002, Fukui 2006; Johnson et al. 2010; Peters 2020). Additionally, bats use geographical landmarks and linear features for navigation and orientation (Barclay 1984; Johnson et al. 2004; Limpens et al. 1989; Racey and Entwistle 2003; Timm 1989; Holloway and Barclay 2000; Furmankiewicz Kucharska 2009), such as riparian zones (Serra-Cobo, Sanz-Trullén & Martínez-Rica 1998), in combination as a large-scale cognitive map (Serra-Cobo et al. 2000; Tsoar 2011). Therefore, bats may concentrate in or migrate through areas near prominent landscape features and use these features to forage along their migratory journey (Baerwald and Barclay 2009).

Brazilian free-tailed fatality estimates had an inverse relationship with wetland within 100 m of wind turbines at Los Vientos, possibly indicating proportions of wetland immediately surrounding a wind turbine lead to less Brazilian free-tailed bat fatalities. It is unclear why a negative association with wetland would be observed at a smaller spatial scale and I am not aware of any studies leading to these same results. Indeed, Jain et al. (2007) found the number of bat fatalities were slightly greater at wetlands near turbines in New York, indicating wind turbines located closer to wetlands may lead to greater fatalities. However, these results were not replicated the following year (Jain et al. 2007). There is also the possibility that these small-scale wetland sources are ephemeral and that

bat species would be using other larger more perennial wetland sources (Downs 2006) within the facility and fatalities would be lower at these wetland sources when dry and lack insect aggregations (Korine and Pinshow 2004). Thus, bat species would be scattered throughout the facility and would not be as concentrated around individual turbines. Additionally, a study completed in Poland (Ciechanowski 2015) discovered that species using water as a feeding resource avoided small, isolated water sources at night. Due to the high fragmentation of wetland, according to our FRAGSTATS results (large ENN_AM and SPLIT values equating to great distances between patches, along with low PLAND – less dominate on the landscape) within this facility, this could be another possibility for the negative relationship between fatalities and small-scale wetland sources.

Results supported the prediction that there would be a positive relationship between bat fatalities and amount of surrounding agricultural-related landcover. Proportion of cropland was a positive predictor of northern yellow fatality estimates within Hidalgo, which may suggest the importance of open areas for foraging for this species (Kern 1992). Although northern yellow bats are insectivorous (Schmidly 2016), little is known about its relationship with agricultural pests. Northern yellow bats are a foliage roosting bat and typically prefer to roost under Spanish moss (*Tillandsia usneoides*) and old palm fronds along the Gulf Coast (Barbour and Davis 1969; Mirowsky 1997; Coleman et al. 2012). Because their range follows the coast of the Gulf of Mexico, it is important to highlight that the Rio Grande Valley consists of numerous palms and Spanish moss which may explain the population of northern yellow bats in this area. Few studies have focused on this species specifically, creating uncertainty in

understanding northern yellow bat movements and distribution. In Northeastern Germany, an area with prominent agriculture in the landscape, Roeleke (2016) discovered that common noctule (*Nyctalus noctula*) females flew higher above cropland than males during evening trips (within turbine blade height – up to 100 m) and use foraging areas at least 14 km away from roosts. Despite differences in range between the common noctule and northern yellow, these species have similar foraging behavior; therefore, higher flight and flying longer distances can result in a high collision-risk depending on height of turbines (Altringham 1999; Cryan and Barclay 2009; Roemer 2017, Peters 2020). According to FRAGSTATS, Hidalgo was dominated by intensive cropland with small, disaggregated patches of all other landcover types. This large portion of cropland may attract insectivorous bat species, such as northern yellow bats, due to the abundance of agricultural pests, therefore, resulting in greater bat activity and leading to more wind turbine collisions. In contrast to the results for northern yellow bats, I did not find an association between Brazilian free-tailed bats (also known to forage on crop pests), and agricultural dominance in the landscape. Possible explanations may be that roosts are located closer to water sources or be a result of different habitat use patterns based on other factors such as precipitation, temperature, storm fronts, and wind velocities (Kerns and Horn 2005). It also may be dependent on the types and timing of crops being harvested near this facility.

Northern yellow bat fatalities at Los Vientos had two types of landcover not predicted for, barren land and shrubland, as the best predictors suggesting once again, the importance of open habitat for this species. Barren land can be classified as open, non-cultivated land, in which one study found hoary and eastern red bats had greater fatalities

in facilities adjacent to small, fragmented, open, and non-cultivated habitat vs large, homogenous areas (i.e., cropland and grassland) (Peters 2020). Despite differences, these species may have similar foraging behavior and activity (Altringham 1999). According to the landscape metrics, barren land had very high ENN_AM and SPLIT values which point to small, scattered patches throughout Los Vientos. This heterogenous landscape, fragmented largely with barren and forest patches, has been suggested to lead to greater risks to bats due to bat activity and foraging increasing along edges, especially during migration (Loeb and O’Keefe 2006; Rydell et al 2010). According to the NALCMS layer, barren land is predominately an edge habitat between cropland and other land cover classes, which may be a high collision risk as bats move through or forage in this area. It also possible, due to the barren landcover description being primarily rock, sand, and clay areas, that northern yellow bats may be foraging over this landcover. However, between both facilities, results suggested a relationship between northern yellow bats and both cropland and barren land landcovers, therefore supporting the idea that yellow bats use open habitat as a foraging source. For example, northern yellow bats tend to use clearings to forage, and the somewhat similar species, western yellow bats, tend to forage over dry desert vegetation (Ammerman et al. 2012; Schmidly 2016). The removal or low abundance of vegetation can serve as a concentration zone for prey, therefore, attracting various bat species to these areas (Grindel and Brigham 1998; Hayes and Loeb 2007; Broders 2015). Additionally, though foraging is not well understood in yellow bat species, the western yellow bat has been captured in agricultural areas (Baker 1956; Genoways and Jones 1968), similar to the habitat indicated as an important predictor in our model results.

The inverse relationship with shrubland may suggest that greater proportions of shrubland within 5 km of a facility may lead to less bat fatalities due to its homogeneous structure and lack of bat activity. As mentioned above, disaggregated patch structure of landcover types can lead to greater risks of bat fatalities (Loeb and O’Keefe 2006; Rydell et al 2010), but according to the FRAGSTATS results, Los Vientos was predominately shrubland with uniform and connected structure. Wind energy facilities consisting of predominately shrubland habitat in the western United States have been documented as having considerably lower rates of bat fatalities (Arnett 2013). Furthermore, Rydell et al. (2010) discovered that in Europe, open, flat terrain had relatively low bat fatalities (0-3 per turbine), versus heterogenous agricultural land, where fatality rates increased (2-5 per turbine). This may be related to the reduced bat activity and abundance of foliage and tree-roosting bat species (such as northern yellow bats) within this land cover type, though exceptions to that pattern are present (Jain 2005; Arnett and Baerwald 2013).

My results suggest that these two wind energy facilities differ in landscape structure and bat fatality trends. Los Vientos and Hidalgo estimated mortality rates were similar (each around 8,000 bat fatalities per wind energy site), while Los Vientos had twice as many turbines than Hidalgo. Because of the FRAGSTATS results, I assume Los Vientos consists of predominately shrubland vegetation, whereas Hidalgo largely consists of croplands. The difference in landcover vegetation combined with similar estimated bat fatalities, despite greater turbines at one wind energy facility, may point to a relationship between fatalities and landscape. Additionally, results suggested that landscapes positively associated with bat fatalities are scattered and fragmented throughout both facilities, suggesting that bat fatalities are more closely associated with landscape features

throughout the site rather than at turbine levels alone. This is demonstrated by the single significant predictor variable at 100 m (proportion of wetland), no significant predictors at 500 m and 1000 m, three predictor variables at 5 km, and three predictor variables at the 25 km scales.

Overall, the results of this study suggest that landscape characteristics influence variation in bat fatalities, and proportion of land cover at the fine scale is not as influential on bat fatalities as broader-scale landscape proportions, although the findings herein may not completely explain the reasons for the difference in bat fatalities at these two locations. For example, one prediction was that forested areas would have a positive relationship with bat fatalities, but my data did not support this prediction. This may be due to the low proportion of forested area in this region of Texas. Additionally, we did not measure turbine height, blade length, and turbine density, all factors which have been suggested to affect fatality numbers (Johnson 2003; Arnett et al. 2008; Barclay et al. 2007; Peters 2020). Megawatts and hub height do vary slightly between the turbines at the two wind energy facilities, Los Vientos turbines are Vestas V110-2.0 MW with a hub height of 80 meters while Hidalgo turbines are Vestas V100-2.0 MW with a hub height of 95 meters.

It is critical to note that this study was limited in several aspects. There were no significant associations between bat fatality estimates and distance of water sources, but ground truthing of water sources near wind turbines could lead to a different outcome due to timing of when images were taken. Ground surveys are recommended to locate nearby roosts for future studies. In addition, our combined bat fatality estimates model included many unidentifiable bat species, therefore additional patterns could be missed. Thus,

these results should be used with caution when proposing wind energy siting recommendations due to the variety of bat species in our combined models.

My analysis involved a model-averaging approach which can have several issues, such as leading to bias and poor forecasting in the case of outliers, and potentially distorted model average results due to multicollinearity (Cade 2015; Banner and Higgs 2017; Dormann et al. 2018; Peters 2020). In my thesis, I used a strict cut off ($VIF < 3$) to ensure highly correlated variables were eliminated prior to inclusion in GLM and before models were averaged. Acknowledging these issues, I believe model-averaging was useful for identifying key variables and relationships among bat fatality estimates and landscape structures. Using these metrics, in combination with landcover types, can lead to better understanding of the landscape structures of the two wind energy facilities. Therefore, these types of metrics used in combination with statistical results, may aid in deriving interpretation of fatality influences in future studies. Overall, my findings indicated that landscape structure at a broad scale (i.e., facility wide scale - 5 km and 25 km) can be as or more informative at assessing bat fatality risks at wind energy facilities than at the turbine-scale. These results are beneficial for the management and conservation of bat species within Texas and areas with similar vegetation.

Conclusion

Based on this study, I recommend southern Texas wind turbine siting should focus on large-scale patterns of land cover, rather than at the local scale alone. From my results, we suggest that wind farms should be constructed in areas that consist of uniform and connected habitat throughout the facility. I also recommend that areas with widespread fragmented water sources and wetland landcover should be avoided, if

possible, for wind energy facility siting. Similar models to those herein will be particularly useful in the planning and permitting stages of wind energy development. Specifically, these types of models can aid in wind energy siting and mitigation. Mitigation efforts such as acoustic deterrents, a pre-construction risk assessment with adequate surveys of the landscape and nearby roosts, and a post-construction assessment are suggested to ensure curtailment of bat fatalities. Additionally, I recommend similar research in areas with similar vegetation and climate, such as northern Mexico and the southern region of the United States.

APPENDIX

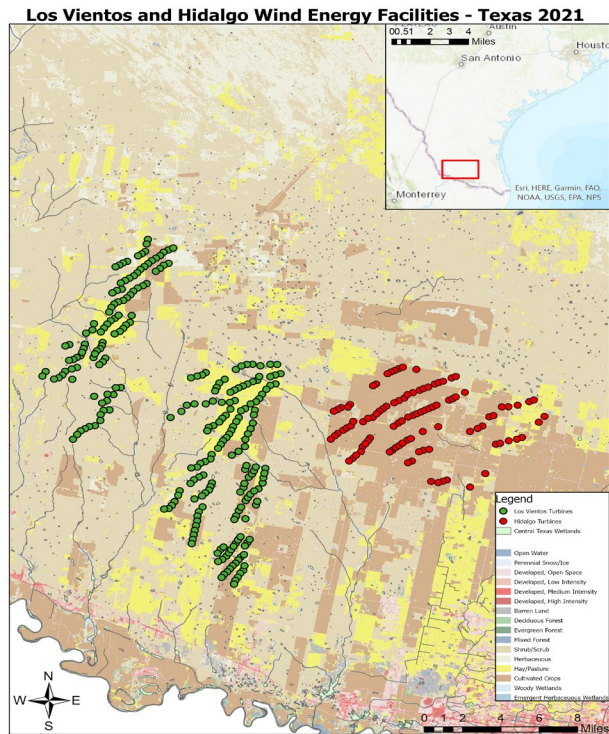


Figure 1: GPS points of individual wind turbine locations within Los Vientos (green; $n = 255$) and Hidalgo (red; $n = 125$) wind energy facilities in Starr and Hidalgo counties, Texas, U.S..

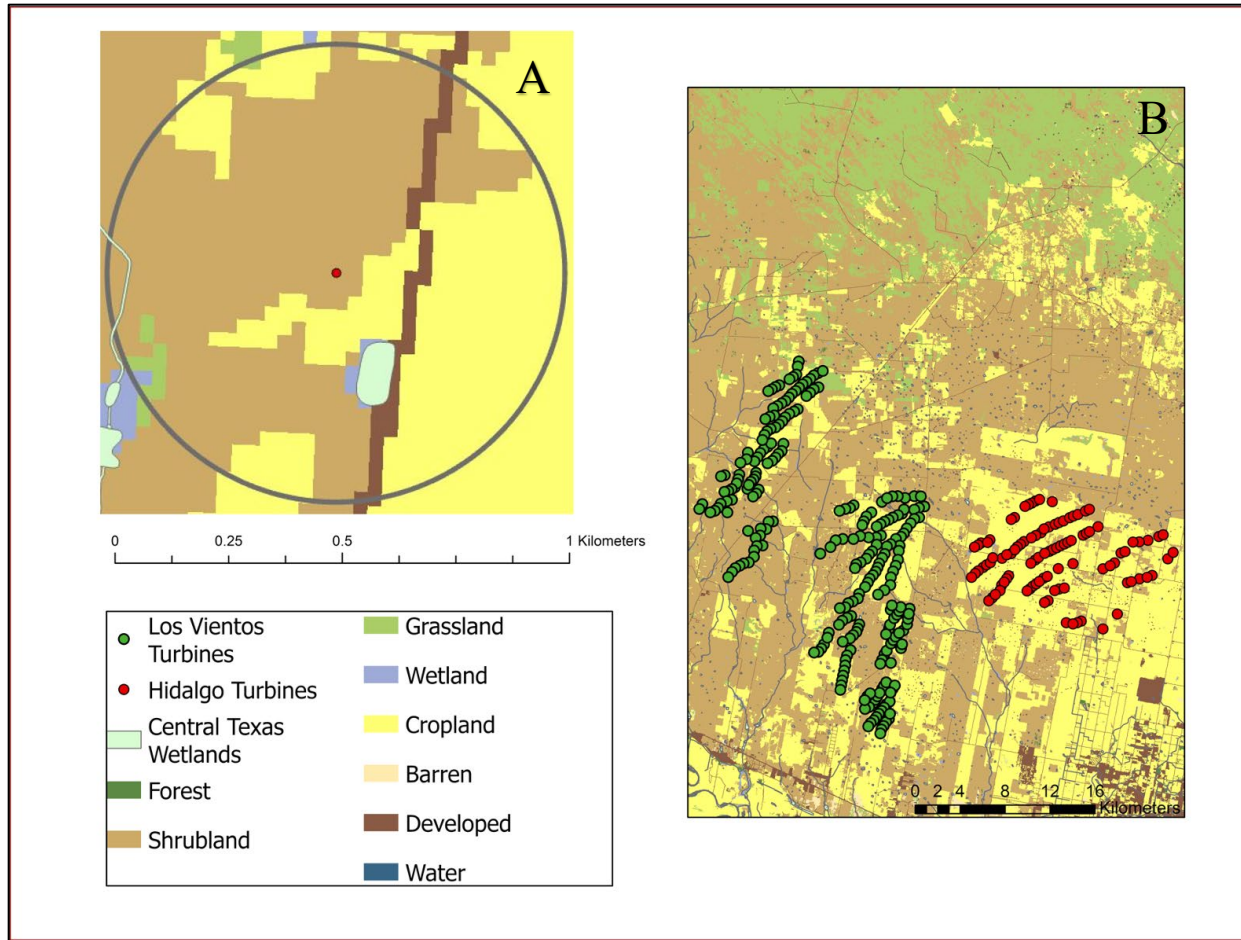


Figure 2: (A) Example GIS maps (ArcGIS; ESRI, Texas, U.S.) of the different landscape types used (only one example landscape buffer is shown). Buffer below is 100 m in radius with colors denoting land-use types in each landscape and light blue denoting NWI

layer. The red dot in the center denotes a wind turbine. (B) Individual GPS points of each turbine from both wind energy facilities: green is Los Vientos, red is Hidalgo. Legend is included.

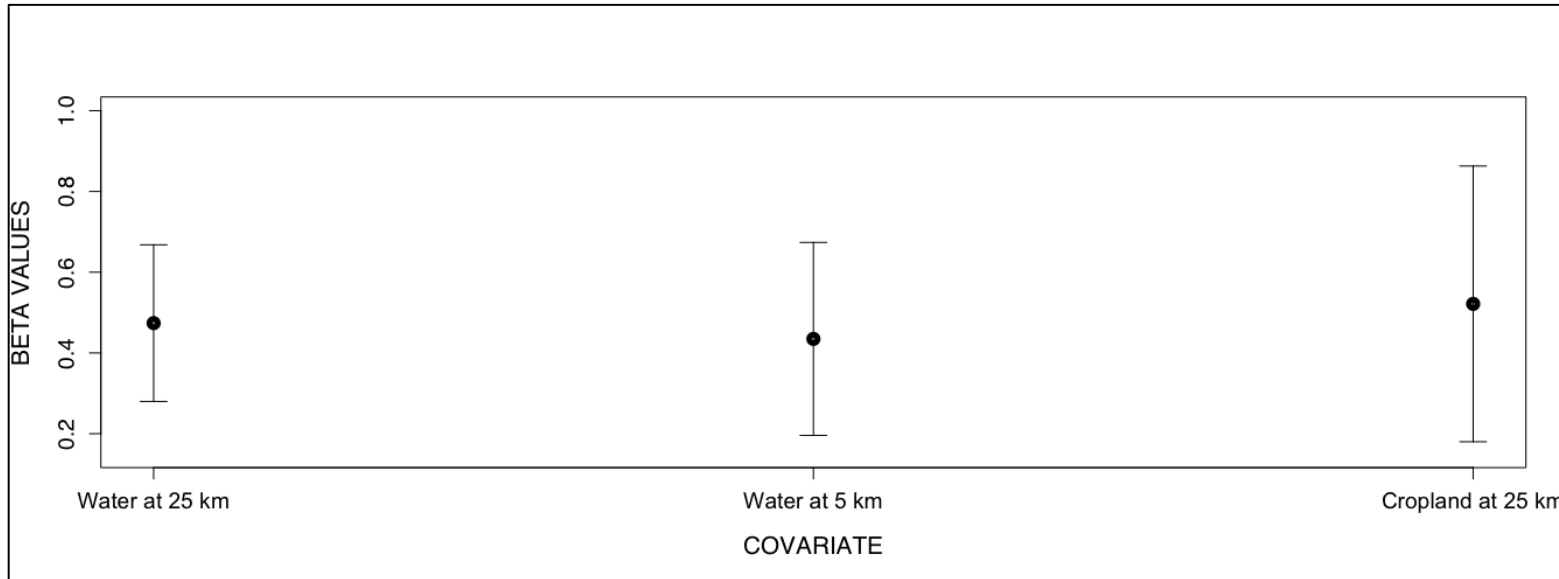


Figure 3: Effect size plot of Hidalgo top predictors from all three models (pooled bat fatalities/Brazilian free-tailed fatalities/northern yellow fatalities); water at 25 km, water at 5 km, and cropland at 25 km.

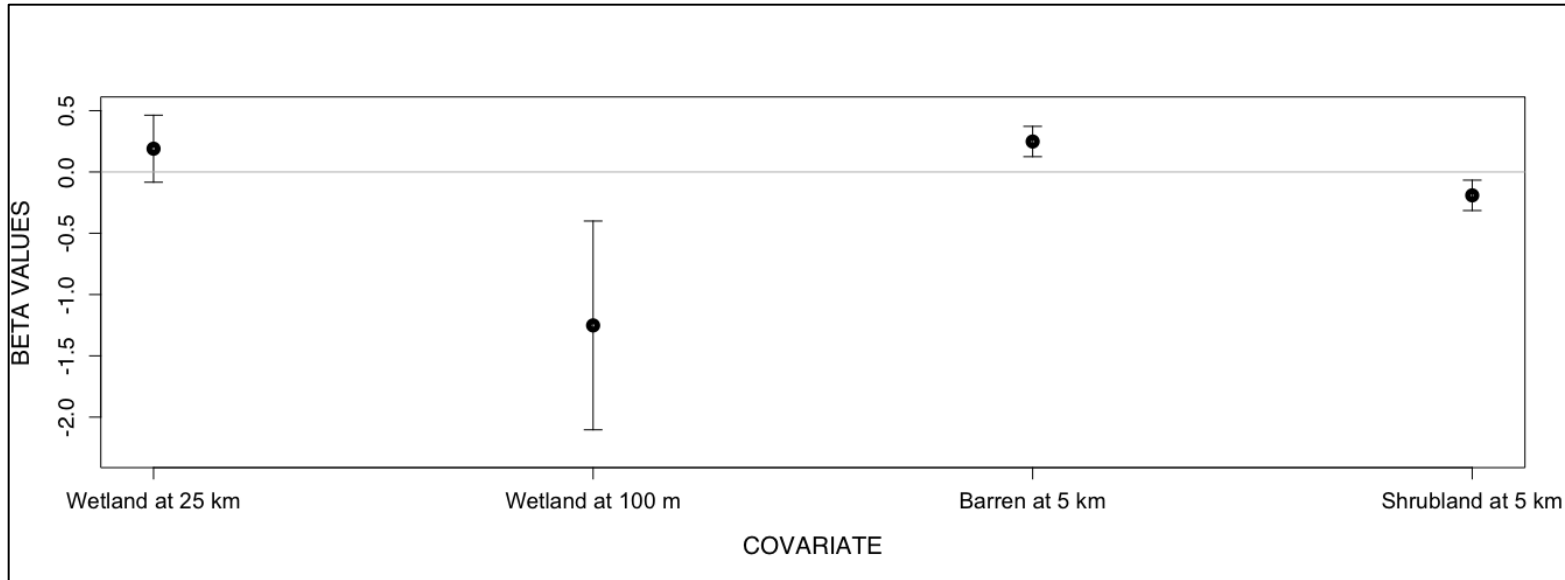


Figure 4: Effect size plot of Los Vientos top predictors from all three models (pooled bat fatalities/Brazilian free-tailed fatalities/northern yellow fatalities); wetland at 25 km, wetland at 100 m, barren land at 5 km, and shrubland at 5 km.

Table 1: Total bat carcasses collected from under 200 wind turbines at Hidalgo and Los Vientos Wind Energy Facilities from 2017–2018; species included *Tadarida brasiliensis* (TABR), *Lasiurus cinereus* (LACI), *L. intermedius* (LAIN), *L. ega* (LAEG), *L. blossevilli* (LABL), *L. xanthinus* (LAXA), *Myotis velifer* (MYVE), *Nycticeius humerali* (NYHU), *Nyctinomops macrotis* (NYMA), *Peromyscus subflavus* (PESU), and unknown spp.

Location	TABR	LACI	LAIN	LAEG	LABL	LAXA	MYVE	NYHU	NYMA	PESU	Unidentified species	Total Fatalities
Hidalgo Facility	418	13	180	62	1	27	2	46	0	1	112	862
Los Vientos Facility	159	5	23	7	1	3	0	5	1	0	1	205

Table 2: North American Land Change Monitoring System legend compared with land cover reclassification used in this study.
 *Areas of open water, generally with less than 25 percent cover of vegetation and/or soil (Homer 2020). **Areas dominated by forest/herbaceous vegetation and has soil periodically covered or saturated with water over extensive periods of time (Homer 2020).

NALCMS classification	Reclassification
Water*	Water
Urban	Developed
Temperate or sub-polar shrubland	Shrubland
Temperate or sub-polar grassland	Grassland
Barren lands	Barren
Temperate or sub-polar needleleaf forest	Forest
Sub-polar taiga needleleaf forest	Forest
Temperatue or sub-polar broadleaf deciduous forest	Forest
Mixed forest	Forest
Cropland	Cropland
Wetland**	Wetland

Table 3: FRAGSTATS class metrics calculated for Los Vientos and Hidalgo Wind Energy Facilities. Index and definitions adapted from McGarigal and Marks (1995).

Metric	Description
Percentage of Landscape (PLAND)	The percentage of the landscape comprised of a particular patch type
Number of Patches (NP)	Number of patches of corresponding patch type (class)
Patch Density (PD)	Number of patches of corresponding patch type (class) per unit area
Largest Patch Index (LPI)	The area (m ²) of the largest patch in the landscape divided by total landscape area (m ²)
Total Edge (TE)	The sum of the lengths (m) of all edge segments in the landscape
Edge Density (ED)	The sum of the lengths (m) of all edge segments in the landscape, divided by the total landscape area (m ²)
Euclidean Nearest Neighbor Distance Area-Weighted Mean (ENN_AM)	Shortest straight-line distance (m) between a focal patch and its nearest neighbor of the same class
Splitting Index (SPLIT)	The number of patches obtained with subdividing the landscape into equal-sized patches based on the effective mesh size

Table 4. Top performing generalized linear models (GLM) with a negative binomial regression at Hidalgo Wind Energy Facility. Models defined as having AICc scores within two points of the top-performing model. The following models were then averaged to determine the best predictors. Combined bat fatality estimates (n = 7983) included two separate models: a water and a shrubland model with combined bat fatality estimates as the response variable. Brazilian free-tailed fatality estimates (n = 4024) included two separate models: water model and buffer model at the 5-km scale with Brazilian free-tailed bat fatality estimates as the response variable. Northern yellow bat fatality estimates (n = 1443) included 4 separate models: a water model, a buffer model at the 25-km scale, a forest model, and a shrubland model with northern yellow bat fatality estimates as the response variable.

*K is the number of parameters in the model, LL is loglikelihood, ΔAIC_c is the information score of the model, AICcWt is the Akiake weight, n is sample size.

Model Description	K	LL	ΔAIC_c	AICcWt
Combined bat fatalities				
Water model: Water @ 5 km + water @ 25 km	4	-526.69	0.00	0.48
Shrubland model: Shrubland @ 100 m + shrubland @ 5 km + shrubland @ 25 km	5	-526.36	1.55	0.22
Brazilian free-tailed bat fatalities				
Water model: Water @ 5 km + water @ 25 km	4	-457.55	0.00	0.35
5 km buffer model: Forest @ 5 km + grassland @ 5 km +	6	-457.92	1.23	0.19

water @ 5 km + cropland @ 5 km				
Northern yellow bat fatalities				
Water model: Water @ 5 km + water @ 25 km	4	-331.27	0.00	0.29
25 km buffer model: Cropland @ 25 km + grassland @ 25 km	4	-331.34	0.13	0.27
Forest model: Forest @ 1 km + forest @ 5 km + forest @ 25 km	5	-331.61	0.90	0.19
Shrubland model: Shrubland @ 100 m + shrubland @ 5 km + shrubland @ 25 km	5	-330.83	1.33	0.15

Table 5. Top performing zero inflated negative binomial regression (ZINB) for bat fatalities at Los Vientos Wind Energy Facility. Models defined as having AICc scores within two points of the top-performing model. The following models were then averaged to determine the best predictors. Combined bat fatality estimates (n = 7983) included two separate models: a buffer model at the 25-km scale and a water model with combined fat fatality estimates as the response variable. Brazilian free-tailed fatality estimates (n = 4024) included two separate models: a forest model and a wetland model with Brazilian free-tailed bat fatality estimates as the response variable. Northern yellow bat fatality estimates (n = 1443) included 3 separate models: a buffer model at the 5-km scale, a forest model, and a barren model with northern yellow bat fatality estimates as the response variable.

*K is the number of parameters in the model, LL is loglikelihood, ΔAIC_c is the information score of the model, AICcWt is the Akiake weight, n is sample size.

Model Description	K	LL	ΔAIC_c	AICcWt
Combined bat fatalities				
25 km buffer model: Water @ 25 km + grassland @ 25 km + wetland @ 25 km	9	-527.48	0.47	0.36
Brazilian free-tailed bat fatalities				
Forest model: Forest @ 5 km + forest @ 25 km	7	-465.44	1.31	0.18
Wetland model: Wetland @ 100 m + Wetland @ 500 m + wetland @ 1 km + wetland @ 5 km	13	-458.93	1.34	0.17

+ wetland @ 25 km				
Northern yellow bat fatalities				
5 km buffer model: Developed @ 5 km + water @ 5 km + barren @ 5 km + wetland @ 5 km + forest @ 5 km + barren @ 5 km	4	-331.27	0.00	0.29
Forest model: Forest @ 5 km + forest @ 25 km	4	-331.34	0.13	0.27
Barren model: Barren @ 500 m + barren @ 5 km + barren @ 25 km	5	-331.61	0.90	0.19

Table 6. Parameter estimates (LCI, 95% lower confidence intervals; UCI, 95% upper confidence intervals) for predictor variables. Derived from model-averaged top performing generalized linear models (GLM) with negative binomial regression for bat fatalities at Hidalgo.

Parameter	Beta	LCI	UCI	P
Combined bat fatalities				
Water @ 25 km	0.47	0.27	0.68	0.0027
Brazilian free-tailed bat fatalities				
Water @ 5 km	0.43	0.17	0.75	0.0011
Northern yellow bat fatalities				
Cropland @ 25 km	0.52	0.21	0.83	0.0104

Table 7. Parameter estimates (LCI, 95% lower confidence intervals; UCI, 95% upper confidence intervals) for predictor variables. Derived from model-averaged top performing generalized linear models (GLM) with negative binomial regression for bat fatalities at Los Vientos.

Parameter	Estimate	LCI	UCI	P
Combined bat fatalities				
Wetland @ 25 km	0.19	0.05	0.35	0.0082
Brazilian free-tailed bat fatalities				
Wetland @ 100 m	-1.25	-2.10	-0.40	0.0040
Northern yellow bat fatalities				
Barren land @ 5 km	0.25	0.12	0.37	< 0.0001
Shrubland @ 5 km	-0.19	-0.31	-0.07	0.0025

Table 8. Landscape metrics results calculated from class metrics: percentage of landscape (PLAND), total edge (TE) largest patch index (LPI), edge density (ED), Euclidean nearest neighbor distance, area-weighted mean (ENN_AM), splitting index (SPLIT), and number of patches (NP) for Hidalgo and Los Vientos Wind Energy Facilities using the 2015 North American Land Change Monitoring System landcover layer.

	Class Name	Landscape Metrics						
		PLAND	LPI	TE	ED	ENN_AM	SPLIT	NP
Hidalgo	Crop	48.91	39.72	1,611,540	23.20	66.48	6.30	208
	Urban	2.40	1.39	865,620	12.46	111.58	5,110.04	537
	Shrubland	35.81	17.91	1,435,800	20.67	65.34	24.57	237
	Grassland	0.32	0.04	115,320	1.66	465.72	2,748,333.70	138
	Wetland	0.89	0.06	248,160	3.57	408.15	1,270,065.92	344
	Barren	0.27	0.01	99,900	1.44	497.21	14,060,842.41	186
	Water	0.06	0.02	15,480	0.22	1,154.74	14,690,561.76	12
	Forest	0.06	0.01	25,980	0.37	1,721.24	353,485,824.96	39
LosVientos	Shrubland	56.09	53.61	4,723,440	30.78	63.25	3.48	490

Cropland	23.95	5.49	3,002,460	19.56	75.50	223.26	864
Grassland	2.68	0.25	1,255,740	8.18	1,069.95	6,320,905,232.04	1,012
Urban	2.31	1.49	1,543,800	10.06	87.85	4,445.06	478
Wetland	1.09	0.04	631,620	4.12	389.85	1,298,089.54	677
Forest	0.17	0.02	151,260	0.99	11,120.28	2,380,532,976.64	187
Barren	0.32	0.03	228,540	1.49	394.74	5,716,793.94	296
Water	0.09	0.07	51,120	0.33	300.41	2,197,491.32	14

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