

University of Nevada, Reno

**Persistence of Wildlife Populations in Low-Diversity Renewable Wind-Energy
Landscapes**

A thesis submitted in partial fulfillment of the requirements for the degree of Master of
Science in Biology

by

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ABSTRACT

The global benefits of increased renewable energy production can come at a cost to local biotic communities and even regional ecosystems. For example, wind energy developments kill birds and bats, and fragment habitat for terrestrial vertebrates within developed project areas. If sensitive species are extirpated, patterns of biotic interactions may be altered at wind farms. In this study, I assessed whether wind energy developments produced other significant ecological costs. Specifically, I determined whether wind farms reduced community diversity and affected the persistence of wildlife populations. I examined this question first by comparing richness, abundance, and diversity of species and communities at wind farm sites, relative to similar reference sites in the San Geronimo Pass of Southern California. Secondly, I used a focused demographic study of side-blotched lizards, *Uta stansburiana*, to identify whether altered wind farm habitats still supported viable populations of common terrestrial vertebrates. I found that wind farms, when compared to study areas without wind turbines, were often noisier and more disturbed, with less diverse communities of plant and predators. On wind farms, the probability of endemic plant presence declined, while presence of non-native plants increased. For populations of *U. stansburiana*, I found no differences in demographic parameters between wind farm and non-wind farm sites, suggesting that wind farms may be viable wildlife habitats for common terrestrial prey species. While some differences in community composition and diversity were evident, I noted that wind farms were able to support many of the same species found in nearby natural areas. And yet, wind energy technology was associated with local declines in diversity, which may affect the sustainability of this technology in local ecosystems.

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THESIS OVERVIEW

Development of wind energy farms causes change to existing land-use patterns at the landscape scale. These wind farms may reduce habitat quality for animals persisting within the turbine-habitat matrix (Lovich and Ennen, 2013). However, the quality of wildlife habitat at wind farms may vary for diverse animal taxa (Madders and Whitfield, 2006). For example, volant species may be more affected by large, noisy turbine structures than terrestrial organisms; whereas guilds such as scavengers might thrive at wind farms by taking advantage of easy prey such as carcasses from turbine collisions (Smallwood et al., 2010). The loss of species sensitive to the presence of wind turbines (and gain of species tolerant to this disturbance) can alter patterns of interspecific interactions, with potentially detrimental changes to ecosystem health (de Groot et al., 2007; Finke and Denno, 2004).

In Chapter 1, I addressed whether community composition and diversity were affected by altered biotic and abiotic environments at four wind energy farm study areas in the San Geronimo Pass of Southern California. I selected target species from (1) predator and (2) plant trophic levels, and for each of these defined communities, I compared composition and diversity at wind farms to five reference sites which ranged from low to high anthropogenic disturbance (from human activities not related to wind energy development). These target communities were selected because they were comprised of species that often act as important drivers of ecosystem-wide patterns of species interactions, affecting diversity of other trophic levels (for example, Gordon, 1998). Altered abundance or diversity of predatory species may reduce the value of habitat patches for prey, which in turn may result in cascade-like changes in abundance

for species across trophic levels (Finke and Denno, 2004; Hawlena et al., 2010).

Alternatively, changes in habitat cover or plant species composition may drastically affect habitat suitability for animals ranging from detritivores to herbivores and predators (de Groot et al., 2007; Gordon, 1998).

I quantified richness (of plants and predators) and abundance (of predators) during field surveys in spring and summer of 2014 and 2015. The two communities were visualized independently using a non-metric multidimensional scaling (NMS) ordination approach to explore differences in community composition between sites. Most dissimilarity between communities could be explained by two axes. The first axis was associated with diversity and richness, and was negatively correlated with disturbance and noise. The second axis showed unique community composition along a gradient of elevation and associated slope; for predator communities, evenness decreased at low elevations. Wind farm sites were clustered along the high noise and disturbance ends of ordination axes, with a trend of lower richness and diversity of plants and predators. For plants, non-native species were more likely to be present at wind farms while endemic species and unique species observations were more common at reference sites. Avian predators such as red-tailed hawk (*Buteo jamaicensis*) and loggerhead shrike (*Lanius ludovicianus*) were less common at wind farm sites. Wind turbines may act as an additional constraint or filter on community structure, reducing abundance of sensitive species. Yet, many of the same species were present at both wind farm sites and reference areas; when physical ground disturbances are minimized, wind farms may provide suitable habitats for more disturbance-tolerant wildlife species.

Changes in habitat quality at wind farms resulted in reduced abundance, richness, and diversity within targeted plant and predator communities (Chapter 1; there was also limited evidence for reduced avian predation pressure on terrestrial vertebrates at wind farm, see Supplement S1). In Chapter 2, I used a reptile model system to ask whether persistence of wildlife populations differed between wind farms and reference habitats, as a possible consequence of observed changes in habitat quality (bottom-up) and predation pressure (top-down). I focused on demographic responses of side-blotched lizard (*Uta stansburiana*) populations, as a species with a highly variable survival rate that is often strongly affected by predation pressure and resource abundance (Ferguson and Fox, 1984; Wilson and Cooke, 2004).

I compared populations of adult side-blotched lizards (*Uta stansburiana*) in 60 x 60 m study plots from reference sites without wind turbines ($n = 5$), and from nearby wind farms ($n = 4$). I marked lizards and recorded individual fates from June 2013 through July 2015. I calculated average body condition and age across populations, and estimated survival rates and population sizes in Program MARK to allow for complex models that accounted for variable probability of capture and survival (as a function of survey conditions, individual quality, and local habitat, climate, and disturbance conditions at study sites). I found that survival rates, population sizes, and individual condition and age were similar between populations with and without wind energy developments. Survival rates varied widely across time and for different age classes; models supported differences in survival as a function of plant cover, but not in response to differences between turbine and non-turbine sites (measured as noise, total anthropogenic disturbance, or turbine presence or absence). Thus, it is possible that

human disturbances associated with wind farms did not affect demographic trends for side-blotched lizards (nor was there evidence of reduced wariness to predators at wind farms, see Supplement S2). It is also possible that other potentially important consequences of disturbance (e.g., high stress levels, reduced reproductive output, or loss of genetic diversity) were present, but not evident in the results of this study.

It is difficult to say with any certainty that lizard populations were not affected by disturbance; however, demographic analyses supported the conclusion that wind farms provided valuable wildlife habitat, and that these habitats were successfully inhabited by terrestrial prey species such as the side-blotched lizard, as well as a number of disturbance tolerant plants and predators. Further research is needed to identify whether other terrestrial guilds (detritivores, herbivores, etc.) are also able to thrive in these altered wildlife habitats. If so, then wind energy production could be a viable alternative energy strategy that outperforms other utility-scale green-energy technologies (for example, utility-scale solar energy or biofuel production) in terms of compatibility with terrestrial wildlife conservation goals.

Land managers can further minimize the consequences of wind energy developments on local communities by placing wind farms in areas of low value to local biodiversity, or at sites with existing human disturbances (Janke, 2010; Kiesecker et al., 2011; McDonald et al., 2009). The next steps towards understanding the value of wind farms as wildlife habitats will be to identify (and minimize) the specific changes in abiotic and biotic selective pressures at wind farms that are most likely to exclude disturbance-intolerant organisms. More precise estimates of target responses could be generated by minimizing the confounding effects of biotic (habitat cover) and abiotic

(elevation and slope) variation, which were important drivers of population and community trends in this study.

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INTRODUCTION

I.1 Overview of renewable energy development

In industrial regions of the world including Europe, North America, and Japan, fossil fuels have defined a way of life for more than a century. Yet researchers predict that as early as 2020, the curtain will begin to fall on an era of cheap fossil fuels (Miller and Sorrell, 2014). Renewable energy development (e.g., biomass, geothermal, hydroelectric, wind, and solar) is highlighted as a way to support growing energy demands, while also addressing global climate change driven by fossil fuel consumption (Panwar et al., 2011). The benefits of renewable energy are clear: by investing in non-finite methods of energy production, we reduce carbon extraction and consumption, increase energy security, reduce greenhouse gas emissions responsible for global climate change, improve public health, and create high-quality jobs (Fronzel et al., 2010). However, the costs of these actions are not as easily understood.

Green methods of energy extraction are not, by default, “ecologically benign” (Harte and Jassby, 1978). Renewable energy infrastructure can affect the local environment within the defined project area of a development (Lovich and Ennen, 2013); furthermore, these effects can also extend out into surrounding habitats (noise from facility operation could cause large mammals to avoid energy facilities, for example, Álvares et al., 2011). After a renewable energy development is no longer in service, landscapes may be altered by anthropogenic disturbance for hundreds of years, if they do in fact ever reach a full state of recovery (Knapp, 1992; Lutting and Rowlands, 1995). As

our country pushes forward with increased investment in large, utility-scale renewable energy infrastructure, we must take steps to implement informed development policies that balance the benefits of increased renewable energy use, against the environmental consequences of development.

Areas with high renewable energy development potential are not evenly distributed across the United States (Lopez et al., 2012). Rather, the associated environmental consequences of development will affect some regions more heavily than others. In the west, 11 states (Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming) hold more than 30 percent of the country's potential for onshore wind and rural utility-scale solar energy development (Lopez et al., 2012). In these states, federal ownership accounts for 46.9% of the total land area (Vincent et al., 2014), much of which is managed by the Bureau of Land Management (BLM). Substantial development efforts are already underway for BLM lands—a capacity of 5,608 megawatts (MW) of wind energy, 8,786 MW of solar, and 1,500 MW of geothermal was approved by BLM as of 2015 (BLM, 2015). These current infrastructure capacities must be doubled, if not tripled, to meet development goals of 28,000 to 46,000 MW of installed capacity for western states by 2020 (Haase et al., 2015).

Public lands will play an important role in future renewable energy development. And yet, these lands also provide over 550 million acres of unprotected and roadless wildlife habitats; almost half of this in the west (Karl et al., 2001). Development of public lands necessitates a thorough understanding of the costs and benefits of altering these landscapes, which comprise a large percentage of our collective natural heritage. Public

land managers will be charged with the difficult task of providing the greatest possible benefit to the widest array of beneficiaries, while simultaneously minimizing the impacts of energy development activities on the landscape.

I.2 Wind energy

Of the various renewable energy technologies, wind energy may be one of the most benign alternatives to fossil fuel (Evans et al., 2009; Harte and Jassby, 1978). Wind energy farms consist of arrays of tens to thousands of turbines that harness the kinetic energy of wind passing across the turbine blades, using attached shafts and gears to create a rotational force that powers a generator. Despite challenges with efficiency (in both use of space estimated at 3 acres per turbine (BLM, 2005), and in the low amount of realized energy output relative to installed capacity), wind technology is favored because it does not require the use of additional cooling water (Evans et al., 2009), which makes it an ideal technology for water-limited areas of the western United States. Wind farms are commercially profitable, and the technology is relatively simple to install across diverse terrain. Wind farms can also be operated simultaneously on areas used for grazing or farming, or on disturbed or reclaimed lands (Kiesecker et al., 2011).

Wind farms have a large physical footprint, although little of the total project area is directly disturbed by infrastructure. On average, wind farms require 34.5 ($SD = 22.4$) ha/MW (Denholm et al., 2009). The permanent impact of disturbance is much smaller, at an estimated 0.3 ($SD = 0.3$) ha/MW. Of this permanent disturbance, turbine areas, roads, substations, and transmission lines make up 10, 79, 6, and 2%, respectively, of the disturbed area (Denholm et al., 2009). Space is left in-between wind turbines to reduce the effects of drag from nearby structures. This increases efficiency and provides an

intact, useable habitat matrix that may be able to support wildlife populations. Total project areas for technologies such as concentration solar power or photovoltaic cells may be smaller (on average, 3.6 ha/MW), but direct disturbances comprise a greater proportion of the total project area, at an average of 3.0 ha/MW (Ong et al., 2013). Project areas at solar farms may have less value as wildlife habitat, particularly if facility grounds are cleared of vegetation (Lovich and Ennen, 2011; Menzel and Pohlmeier, 1999).

Capacity for wind energy generation is growing rapidly. From 2013 to 2014, the World Wind Energy Association (WWEA) estimated an increase in global wind energy production of 16.4%. As of 2014, wind energy provided 4% of global energy demands at 372,000 MW in 105 countries, with a projected potential of 700,000 MW globally by 2020. In the US, more than 65,800 MW of wind energy capacity have already been installed (WWEA, 2015).

It is clear that renewable wind energy development pressure will only increase in future years. However, the ability of land managers to promote responsible and informed development policies is hampered by a general lack of regulatory framework. Wind farms are subject to federal regulations (e.g., the National Environmental Policy Act, the Endangered Species Act, the National Wildlife Refuge System Administration Act, and the Migratory Bird Treaty Act) only when constructed on public lands or funded by public money (NRC, 2007). Most often, regulations pertain to compliance with national policies protecting birds and bats (NRC, 2007). There is no established policy addressing the acceptable consequences or costs and benefits of development allowable for approval of a project. Agencies have identified best practices for minimizing ecological effects of

developments (BLM, 2008; FWS, 2003); yet, the ability to reduce impacts of wind farms on the environment through effective mitigation measures is hampered by a pervasive lack of critical research documenting how wind turbines affect the environment. In order to establish effective best practices, additional research is needed to address critical data gaps (NRC, 2007).

I.3 Ecological consequences of wind energy development

What do we know about the effects of wind energy on wildlife, and on wildlife habitats?

To date, most research has addressed the potential for turbines to directly cause mortality of birds and bats as a result of collisions with wind turbines, or in bats from changes in barometric pressure near turbine blades (Cryan and Barclay, 2009; Kuvlesky et al., 2007).

Direct mortality has been documented from collision with transmission lines (electrocution) and meteorological towers, or from vehicle trampling along access roads for non-avian species (Lovich and Ennen, 2013). Estimates suggest that 20,000 to 37,000 birds are killed per year at wind farms; roughly 75% of these deaths occur in California alone (Erickson et al., 2005). Relative to other species groups, raptors, nocturnal migratory passerines, and migratory tree roosting bats have been most affected by wind turbines (Erickson et al., 2002).

Other, less-studied effects of wind farms on local wildlife include habitat loss and degradation, which may be important for the persistence of populations of volant as well as terrestrial wildlife species (Kuvlesky et al., 2007). Wind energy facilities may increase erosion and runoff (reducing water quality and altering normal hydrologic processes); increase dust, noise, electromagnetic, and chemical pollution; enhance the spread of invasive species (plants and animals); increase fire risk; or significantly modify and

fragment habitats from road and infrastructure development (BLM, 2005; Kuvlesky et al., 2007; Lovich and Ennen, 2013). These combined changes to the biotic and abiotic environment may substantially reduce habitat quality for local biota, resulting in a loss of species richness or diversity (Santos et al., 2010; Villegas-Patracca et al., 2012).

Richness declines have been documented in bird communities within wind project areas (Villegas-Patracca et al., 2012). For species that do occupy wind farms, reduced densities, lower habitat-use intensity (both within and beyond facility boundaries), and altered behavior patterns have been documented (Anderson et al., 2008; Johnson et al., 2000; Osborn et al., 1998; Winkelman, 1994). However, responses to wind-farm development will likely vary by species. Ravens and turkey vultures were found abundantly at wind farms, where they are known to scavenge carcasses (Smallwood et al., 2010). In these industrial areas, ravens and vultures had low mortality rates despite their high densities, suggesting an ability to successfully navigate this novel industrial landscape (Orlaff and Flannery, 1992). On the other hand, birds such raptors were observed using wind farm power lines, meteorological towers, and small lattice turbine towers for perching and hunting (Anderson et al., 2008); yet, this resulted in higher mortality risk (Orlaff and Flannery, 1992).

In mammals, very little is known about responses to wind turbines. Researchers have demonstrated that ground squirrels from wind farms in Southern California were more cautious and vigilant in noisy turbine environments (Rabin et al., 2006). In Germany, surveys of small game mammals showed similar levels of habitat use at wind farms and reference areas, regardless of distances from the wind farm (Menzel and Pohlmeier, 1999). For large carnivores, studies of wolves indicated avoidance (during

construction phase) and altered patterns of spatial habitat use near Portuguese wind farms; wind farms within a pack's territory reduced wolf abundance and reproductive rates (Álvares et al., 2011). Black bear showed similar avoidance during the construction phase, which did not persist post-construction (Wallin, 1998). Wolverines in Sweden showed a marginally significant reduction in use of a wind farm area during construction (Flagstad and Tovmo, 2010 in Helldin et al., 2012). Some small carnivores are known to take advantage of increased carcass abundance at wind farms (fox, coyote, badger, bobcat, skunk, and raccoon); however, the intensity of use of these habitats relative to nearby areas was not explored (Orloff and Flannery, 1992; Smallwood et al., 2010).

A number of European studies have looked at avoidance behavior of ungulates at wind farms (Helldin et al., 2012). Temporary emigration of red deer stags was documented during construction of a wind farm (Veiberg and Pedersen, 2010), and a similar construction phase avoidance was noted for reindeer—a species that also avoided access roads and power lines in other disturbed areas (Colman et al., 2008 in Helldin et al., 2012; Nellemann et al., 2001; but see Reimers et al., 2007). Wind farms did not affect home range sizes or diet quality in North American elk (Walter et al., 2006). Studies also indicated that domestic ungulates (horses, reindeer) did not avoid, or show fear of wind farms (Flydal et al., 2004; Helldin et al., 2012 and references therein).

The effects of wind turbines on reptiles must be inferred from studies of the long-lived desert tortoise (*Gopherus agassizii*); as the consequences of wind energy development for other reptiles have not yet been addressed. In the desert tortoise, under-road culverts and access roads were significant mortality threats at wind farms (Lovich et al., 2011a). Yet, tortoises also showed an increased tendency to make use of wind farm

infrastructure such as roads and concrete foundations (Lovich and Daniels, 2000). Nesting ecology was similar between wind farms and undisturbed sites (Ennen et al., 2012), and there was no effect of fires (commonly caused by turbine operation) on activity patterns or reproductive output (Lovich et al., 2011b). There were few differences in growth and demography for wind farm tortoises relative to other populations, although males exhibited lower growth rates and larger maximum body size, while females showed higher survival rates (Lovich et al., 2011c).

These previous studies comprise the entire peer-reviewed literature available in the United States regarding the effects of wind turbines on non-volant species. Most studies consider the responses of singular species; to date, there is no research addressing the potential consequences for terrestrial communities (FWS, 2003; but see Santos et al., 2010 or Villegas-Patraca et al., 2012). This lack of peer-reviewed research is even more glaring when considering the consequences of wind energy development on plant communities (however, researchers do predict that increased invasive plant abundance is likely; see Jones et al., 2015 or Lovich and Ennen, 2013). Research is needed to more clearly understand how turbines affect terrestrial species, and the mechanisms underlying species responses (FWS, 2003; Helldin et al., 2012; Lovich and Ennen, 2013; Northrup and Wittemyer, 2013; NRC, 2007; Santos et al., 2010).

I.4 Research needs and study goals

An understanding of the effects of wind farms on terrestrial wildlife species and communities is critically lacking. This knowledge gap has already promoted a general disregard for the effects of development on terrestrial organisms. For example, a 2011 review of 23 environmental impact assessments at Swedish wind farms noted that 52%

did not even discuss the potential impacts to terrestrial mammals (Lundberg, 2011, in Helldin et al., 2012). Additionally, the current lack of knowledge may substantially hinder agency and developer attempts to mitigate wind turbine effects on wildlife (Kuvlesky et al., 2007). General disturbance ecology studies (targeting singular responses to activities such as road development, noise pollution, or habitat loss) have helped land managers to better address some of these anticipated stressors for wildlife populations inhabiting wind energy developments (Lovich and Ennen, 2013). However, these disturbances may affect wildlife in synergistic ways, with potentially unique responses among taxa. Furthermore, the potential consequence of wind energy development at a community scale has been grossly ignored. A better understanding of the costs of wind farm development for terrestrial organisms, when weighed against those from conventional energy extraction, will aid in the accurate assessment of total “net biodiversity savings” realized from alternative energy production.

The goal of this study was to provide a better understanding of the dynamics of terrestrial wildlife populations and communities in altered wind-energy landscapes. In Chapter 1, I consider whether wind farms affect terrestrial communities of vertebrates including understudied taxa such as plants, invertebrates and reptiles. I compare richness, abundance, and diversity of species (at sites from wind energy farms, and from nearby reference habitats without wind turbines) to test the hypothesis that ecological stressors at wind farms result in low-diversity community assemblage patterns favoring disturbance-tolerant organisms. In Chapter 2, I estimate survival and demographic parameters for populations of side-blotched lizards (*Uta stansburiana*) to further identify whether differences exist in habitat suitability at wind farms for common terrestrial vertebrates.

Together, these results provide a baseline of research which can be used to better understand how green energy developments affect local wildlife and wildlife habitats.

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Disturbance affects biotic community composition at Southern California wind farms

ABSTRACT

Renewable wind energy infrastructure can alter biodiversity by reducing habitat suitability for flora and fauna in undeveloped habitat patches within an energy facility project area. Here, I tested whether wind farms were less diverse than nearby areas without energy development by comparing community composition and diversity of (1) predator and (2) plant communities. I used non-metric multidimensional scaling to map ecological dissimilarity across four wind farms and five reference areas in the San Geronio Wind Resource Area in Palm Springs, California. I found that wind farms were often noisier and more disturbed (measured as total road density) than sites without turbine installations. Noise and disturbances were correlated with reduced plant diversity and richness, particularly for endemic species. Non-native plant species were more likely to be present at wind farms than at natural sites. Predator communities at wind farms were also less rich, even, and diverse relative to reference areas with minor or no disturbances. Wind farms had fewer rare species, and encounter frequencies were significantly lower for loggerhead shrike (*Lanius ludovicianus*) and red-tailed hawk (*Buteo jamaicensis*). And yet, wind farm sites were at least as different from each other as they were from natural communities. Elevation and underlying habitat characteristics may be as important as disturbance intensity in driving community composition trends. Wind farms still supported many of the same species found in natural areas, suggesting that renewable wind energy facilities can provide valuable (albeit less diverse and more disturbed) habitat for wildlife.

1. INTRODUCTION

Human activities have affected more than 80% of the terrestrial surface of our planet, resulting in dramatically altered patterns of global biodiversity (McDaniels and Borton, 2002; Sanderson et al., 2002; Shochat et al., 2010). Currently, 5 to 20% of the species on earth are threatened with extinction resulting from habitat loss and degradation, exotic species invasions, pollution, and global climate change (Chapin et al., 2000; Pimm et al., 1995). Of these threats, researchers predict that land-use change will be the strongest driver of biodiversity loss over the next century, followed closely by global climate change (Sala et al., 2000). Counter-acting climate change will require substantial increases in renewable energy production potential, achieved through rapid development of renewable energy extraction infrastructure. Resulting utility-scale renewable energy developments (projects with a capacity of more than 10 megawatts (MW)) are intensive land-use strategies, requiring on average from 3.6 to 34.5 ha/MW of installed capacity for solar and wind energy, respectively (Denholm et al., 2009; Ong et al., 2013). Ironically, these developments alter local patterns of land-use, which may create local threats to biodiversity (Lovich and Ennen, 2011; 2013).

The desert southwest provides an excellent case study of this environmental tradeoff. Leasing information for desert ecosystems indicates a potential energy footprint of 26 million hectares in these shrublands (Pocewicz et al., 2011). In California, approximately 15,702 hectares have been identified as development regions for new renewable energy projects, in addition to pre-existing developments (DRECP, 2015). While the desert southwest possesses high potential for solar and wind energy development (Lopez et al., 2012), it also supports a diverse plant community with more

than 20% of the local flora comprised of species that are endemic to the region (McLaughlin, 1986). These deserts also support many species of conservation priority; for example, Southern California has one of the highest concentrations of endangered species in the United States (Dobson et al., 1997). Development in natural (un-built) landscapes may threaten large tracts of viable wildlands that have been identified as important habitats or habitat corridors for wildlife (Kiesecker et al., 2011; Spencer et al., 2010; Stoms et al., 2013). Furthermore, desert communities are slow to recover following anthropogenic disturbance; these arid regions may have altered community structure and function for centuries after human activities have ceased (Lovich and Bainbridge, 1999). Given the high risks of biodiversity loss from energy development in deserts, responsible development requires an understanding of the inherent risks to biota from changing habitat quality at energy production sites.

Wind energy is an attractive strategy for sustainable energy production in southwestern deserts (Evans et al., 2009). Relative to solar energy technology (photovoltaics), wind energy technology costs less for consumers, is more efficient to produce with fewer limitations and greater availability, produces less atmospheric pollution, requires less water consumption, and is often more socially acceptable (Evans et al., 2009). There is concern regarding the direct mortality risk for volant species such as birds and bats from collisions with wind turbines (Cryan and Barclay, 2009; Kuvlesky et al., 2007). However, these risks can be substantially reduced by designing and actively managing wind farms to reduce collision rates (Baerwald et al., 2009; May et al., 2015). A second challenge with wind energy production is that wind farms require larger tracts of land (34.5 ha per MW of installed capacity) relative to other renewable energy production

strategies (Denholm et al., 2009; Gagnon et al., 2002). However, the intensity of land-use is low, with a direct physical footprint of less than 1% of the total project area, on average (Denholm et al., 2009). For solar technologies, projects may be smaller (on average, 3.6 ha/MW), but direct disturbances comprise 83% of the total project area (Ong et al. 2013).

Few studies have addressed the consequences of wind farm development on wildlife habitats. Project areas may be able to provide suitable wildlife habitat, as facility grounds are not often cleared of vegetation as with other renewable technologies such as solar (Lovich and Ennen, 2011; Menzel and Pohlmeier, 1999). Most of the habitat loss at wind energy facilities occurs when access roads are constructed to connect turbines, rather than from turbines themselves which occupy only a few square meters each. Thus, the quality of wind farms as wildlife habitats hinges primarily on the severity of habitat degradation and fragmentation from facility construction, operation, and maintenance (Drewitt and Langston, 2006; Lovich and Ennen, 2013).

Studies have identified a number of mechanisms which may result in reduced habitat suitability for wildlife at wind farms. Turbines generate low frequency noise from generators and blade movements that may disrupt acoustic communication (Rabin et al., 2006). Habitat fragmentation and increased human activity may also disrupt habitat-use patterns, particularly in apex predators or long-lived species (Lovich and Daniels, 2000; Orloff and Flannery, 1992; Pearce-Higgins et al., 2009; Veiberg and Pedersen, 2010). Access roads for machinery and vehicles can increase mortality risk for slow-moving species (Andrews et al., 2008; Lovich et al., 2011) while promoting invasive plant establishment and increasing foraging success for scavengers and small carnivores

(Gelbard and Belnap, 2003; Lovich and Ennen, 2013; Orloff and Flannery, 2002; Smallwood et al., 2010). At the regional scale, wind farms may affect habitat suitability by altering macroclimate wind and temperature patterns (Roy and Traiteur, 2010).

If wind farms alter the quality of wildlife habitats, then we can anticipate some changes in species composition and diversity in communities inhabiting (or in proximity to) wind energy project areas. Studies in urban ecology suggest that altered patterns of land use may result in a selective filtering of species by ability to tolerate novel disturbances (Crocì et al., 2008). As the level of disturbance increases, filtering effects intensify resulting in reduced overall diversity (Crocì et al., 2008; Fischer et al., 2012; Sol et al., 2014). Species that exploit disturbed habitats will reach higher densities relative to those in similar areas without disturbance, resulting in reduced community evenness. Eventually, these species will outcompete and replace multiple native species, further reducing richness and diversity (Fischer et al., 2012). Thus, estimates of diversity, evenness, and richness can be used to make inferences about the severity of disturbances.

In this study, I addressed whether wind farms affected the composition of biotic communities by comparing nine communities at sites with ($n = 4$) and without ($n = 5$) wind energy developments in the San Gorgonio Pass Wind Resource Area (SGWRA) of Southern California. I characterized patterns of diversity and species composition in communities of (1) predators (mammals, reptiles, birds, and invertebrates) and (2) primary producers. The diversity within these two species groups can influence ecosystem function in desert systems by altering the abundances of consumers and prey species (Ayal, 2007; Fischer et al., 2012; Gordon, 1998; Sergio et al., 2008).

First, I compared overall patterns of diversity between sites with and without wind farms (“treatment effect”). I hypothesized that loss of disturbance-intolerant species and guilds (in favor of disturbance exploiters) would result in lower diversity due to declines in richness and community evenness. Second, I used ordination techniques to look at patterns of community composition. I tested for correlations between community composition and variables describing anthropogenic disturbance, or describing confounding variation from habitat or climate characteristics of selected study sites.

Third, I used life-history characteristics associated with sensitivity to disturbance to group species into functional categories of predicted tolerance to wind energy development (for example, see Lavorel et al., 1999). Using primary literature records, I developed the following alternative predictions: for predators, I expected (1) lower diversity of snakes, and greater abundance of species with (2) generalist, omnivore diets (Fischer et al., 2012; Hurst et al., 2014; Ortega-Alvarez and MacGregor-Fors, 2009). I predicted that (3) terrestrial predators would face lower mortality risk and would be more abundant than avian predators, which are often subject to high mortality from collisions with wind turbines—in the SGWRA, studies suggest an annual mortality rate of 4.71 birds and 2.17 bats per MW turbine (Chatfield et al., 2009) with a projected 6,800 birds killed annually (McCrary et al., 1986). These mortalities may result in high abundances of carcasses near wind farms; therefore, I predicted that (4) scavengers would be more abundant at wind farm sites resulting from the presence of carrion. For plant species, I predicted greater richness of (5) annual plants, with fewer (6) endemic or native species (Rejmánek, 2000; Slatyer et al., 2013; Stylinski and Allen, 1999; Vamstad and Rotenberry, 2010).

2. METHODS

2.1 Study area

The SGWRA is near Palm Springs, California (33°56'N, 116°34'W). It was developed for wind energy production starting in the mid-1980s, and is currently of the country's largest wind production regions. The wind resource area covers over approximately 18,000 ha, with more than 3,000 individual turbines providing 615 MW of installed energy capacity (NRC, 2007). The area (Figure 1) is a transition zone between the Sonoran and Mojave deserts; the region is dominated by a sparse shrub community of creosote brush (*Larrea tridentata*), brittlebush (*Encelia farinosa*), and white bursage (*Ambrosia dumosa*), with lower abundances of annual and perennial plants. There was substantial variation among the nine selected study sites, with shrub densities ranging from 0.06–0.38 per m², average shrub heights from 0.48–1.01 m, and percent shrub canopy cover of 2.95–27.5%. Study sites spanned an elevation gradient from 318–822 m, with shallow to moderate slopes from 1.51–17.4° (Table 1).

Within the SGWRA, four study sites were established at wind farms (“turbine sites”), as summarized in Table 2. The first two sites were developed more than 30 years ago, with high densities of smaller turbines with lattice tower bases. The Mesa Wind Farm (“ME”, maintained by Green Energy Maintenance Corporation), at the base of the San Gorgonio Mountains, was bordered by undeveloped federal lands to the north. A second wind farm bordered the site to the east. Sparse housing developments (1.3 km to the south and west) and undeveloped scrublands fragmented by two-track roadways separated the wind farm from major roadways (2.1 km). The second site, Painted Hills Wind Farm (“PH”, maintained by Painted Hills Wind Development), was bordered on the

south by additional wind farms and by the 1-10 Interstate (1.6 km). The west side of the site was characterized by low density two-track roads, and by housing and a quarry (1.6 km). To the north, additional wind farms continued for 0.5 km, followed by undeveloped federal lands at the base of the San Geronio Mountains. To the east, dispersed housing and residential streets filled the 1.6 km gap between the facility and Highway 62.

The third and fourth sites were characterized by newer installations of tall and tubular turbines at lower densities. The western portion of the Mountain View Wind Farm (“MV”, maintained by AES Wind Generation), was near the base of the San Jacinto Mountains, with similar facilities to the north, east and west. The site was close to the I-10 (0.3 km) and Highway 62 (1.3 km). To the south-east, constructed percolation ponds supported older model turbines, and to the south-west, the site was bordered by a railroad (0.5 km), and a highway (1.3 km). The fourth site, Dillon Wind Farm (“DI”, managed by Iberdrola Renewables), was more isolated from unaltered natural habitats; however, it supported a low development intensity surrounded by remnant patches of fragmented desert habitats disturbed by illegal off-road vehicle use and trash dumping. The eastern portion of site was located near low-density housing, multiple transmission line corridors and additional low-density wind farms. The site was 3.2 km from Highway 62 and the I-10, and flanked by a busy residential road to the east.

Five sites managed by the Bureau of Land Management were selected as reference natural areas (“non-turbine” sites). Two of these sites were low-disturbance areas that represented natural community types. These sites were at least 2.4 km from any road disturbances, with limited human activity. The first site was established to the east of the Mission Creek Preserve (“MI”) on the eastern end of the San Geronio Mountains.

The site was undisturbed except for a nearby closed two-track road (50 m), and a gravel preserve access road (0.2 km). The second site was selected within the San Gorgonio Pass Wilderness Study Area (“SG”) to the north of the ME turbine site. The study area was bisected by the Pacific Crest Trail, and characterized by minimal noise disturbance and no vehicle disturbance, at least 1.6 km from the nearest two-track road. It was noted after the site was selected that erosion and soil disturbance occurred over the course of the study, as a consequence of illegal grazing activities.

The third and fourth sites were selected from moderately disturbed areas. The third site was located south of the Whitewater Creek Preserve (“WW”), in a canyon of natural habitat between the ME and PH turbine sites. The site was 0.8 km south of residential housing, and 1.6 km north of the I-10, with wind farms to the east (0.5 km) and west (1.6 km). A busy access road flanked the site 160 m to the west. The fourth site was north of the town of Desert Hot Springs (“DH”) and south of the Big Morongo Canyon Preserve. The area is near suburban housing (0.8 km), and within 160 m of an infrequently used two-track road. The area was used frequently by residents for hiking, biking, and walking dogs. The area had a high density of pathways and off-road vehicle tracks. The final, most disturbed site (“I-10”) was located 1.3 km south of the ME wind farm and 420 m north of a rest-stop on I-10. There was substantial traffic noise, off-road vehicle use, human activity, and multiple transmission line corridors at this site.

2.2 Field surveys

To quantify predator community structure, I recorded encounter rates during timed searches for a subset of common predatory taxa consisting of reptile ($n = 11$), bird ($n = 11$), mammal ($n = 5$), and invertebrate ($n = 5$) guilds. These species (or orders, families,

and genera for invertebrates) differ in habitat requirements, dietary specialization, degree of omnivory, and trophic position, suggesting that some may be more tolerant of disturbances than others (Table 3). From June to late July of 2014, I conducted time-constrained searches targeting suitable habitat for each guild within the area surrounding study plots.

During reptile surveys, an observer (five possible on any given survey day, each surveying independently for a target guild) searched during mid-morning hours for individuals active on the surface or beneath suitable cover objects. Snake skin sheds were included as observations. During bird surveys, observers detected species in early morning hours using auditory and visual surveys. Observations included focused searches over short (3–20 min) time periods from haphazardously selected vantage points, in addition to incidental observations recorded between focused searches. A taxon observation was recorded in auditory surveys if a call was heard anytime within a 5-minute observation period; an additional encounter was recorded only if the call was heard at least 5 minutes after the previous encounter. For mammal surveys, observers mostly relied on scat sign; however, some observations included visual sightings or physical remains. A taxon was considered observed from scat if one or more scat piles were detected over a 10-minute search period. We did not record track or burrow activity, as these observations may have been biased by variation in track persistence due to differences in soil characteristics across study sites. For invertebrate surveys, observers searched under suitable cover objects and included active, silk-lined burrows as observations of tarantulas, or wolf spiders. A summary of conditions during predator surveys is presented in Table 4.

To quantify plant community structure, I estimated the richness of plant species within study plots centered at each of the nine established sites during mid-March of 2015. Each plot was 1600 m². I surveyed each site by systematically searching the study plot for undetected species until no new species were encountered for 15 minutes. This method ensured that I detected all common species; however, it may have introduced some bias against uncommon or smaller plants. Specimens were identified to species in accordance with the nomenclature of the Jepson Desert Manual and dichotomous keys therein (Baldin, 2002). For the following genera, I did not attempt to identify species: *Logfia* (Asteraceae), *Rafinesquia* (Asteraceae), *Pectocarya* (Boraginaceae), *Caulanthus* (Brassicaceae), *Cuscuta* (Convolvulaceae), *Dudleya* (Crassulaceae), and *Schismus* (Poaceae). Identifications were confirmed using pressed specimens maintained in herbaria at the University of Nevada, Reno Museum of Natural History and the Boyd Deep Canyon Desert Research Center, or by comparison with photos maintained on the CalPhotos database (<http://calphotos.berkeley.edu/>). For a list of specimens encountered, see Table 5.

2.3 Site disturbance, habitat, and climate characteristics

For each of the nine study sites, I used field surveys and GIS data to identify disturbance history, and to describe habitat and climate characteristics. I measured anthropogenic disturbance as (1) an index of road disturbance (“DISTURB”), calculated as the sum of road lengths (digitized in ArcMap from 1 m resolution California NAIP imagery) within a 1000 m buffer of plant richness plots. To better account for differences in disturbance level across road types, I weighted road lengths as 0.5 for footpaths, 1.0 for dirt roads, 3.0 for residential roads, 5.0 for secondary roads, and 10.0 for primary highways (e.g., Brady

et al. 2009). I also calculated (2) noise level (“NOISE”) as an average a-weighted decibel (dB_A) reading from a random sample of 10 readings at plant study plots taken in June and July (2015) when turbines were running and wind was ≤ 3.5 mph.

I quantified differences in habitat characteristics using data from three, 100 m point-line-intercept surveys (Herrick et al., 2005) completed in July of 2014 at plant richness study plots. I calculated non-canopy cover (“COVER”) as the proportion of random points at each meter interval per transect that did not intersect a plant canopy. I completed three, 100 m belt transects to estimate the number of shrubs within 2 m of the transect line, which I then used to calculate shrub density (“SHRUBS”) as the number of shrubs per m². I estimated the percent of the study area with a shrub canopy (“SHRUB.CANOPY”), determined from measured shrub diameters (d) obtained during belt-transect surveys, calculated as: $\pi \left(\frac{d}{2}\right)^2$. I estimated the average height of the shrub canopy (“CANOPY.HEIGHT”) from measured shrub heights (belt-transect data).

I used slope (“SLOPE”), elevation (“ELEV”), and precipitation (“PRECIP”) to approximate community differences resulting from variation in climate or topography. Slope and elevation were generated from a 2013 digital elevation model (DEM) raster at 1/3 arc-second resolution, retrieved from the National Elevation Dataset (available online from the USGS at: <http://nationalmap.gov/elevation.html>). Extracted values characterized average conditions for each 1600 m² plant study plot at the center of the predator survey area. I extracted average precipitation values for each plant study plot from precipitation models generated by the PRISM Climate Group, Oregon State University, (<http://www.prism.oregonstate.edu/>, created 4 Feb 2004), averaged across the winter

months of November through February, from 2012–2015. I also calculated the average wind speed (“WIND”) during predator encounter surveys in June and July of 2014.

2.4 Analyses

High quality habitats are thought to support a greater richness and diversity of species; these diverse habitats are often more stable, trophically complex, productive, and disturbance resistant (Johnson et al., 1996; McCann, 2000; Paine, 1966). To better understand the quality of wind farms as habitats, I compared measures of diversity for targeted plant and predator communities at turbine and non-turbine sites. For predator communities, I calculated richness (R = number of taxa); diversity (Shannon Diversity Index: $H = - \sum_{i=1}^S p_i \ln(p_i)$, where p_i equals the proportion of observations at a site belonging to taxa i); and evenness: $E = \frac{H}{\ln(R)}$. For plant communities, I calculated richness of plants by species (and by genus and family; additionally, I quantified the diversity and evenness of species **within** each plant family to address homogeneity within higher taxonomic classifications). In all three metrics, higher values indicate more diverse communities.

I visualized plant and predator community diversity using indirect ordination. I used non-metric multidimensional scaling (NMS) to characterize relationships between turbine and non-turbine communities; this technique optimizes the physical arrangement of sites along a specified number of axes, such that communities with similar species composition are close together in ordination space. NMS is a powerful tool for visualizing complex community datasets; it is well-suited for ecological data because it does not assume an underlying model of species distributions (McCune and Grace, 2002).

I performed all ordinations in Program R version 3.2.2 (R Core Team, 2015), using the default settings of the *metaMDS* function from the ‘vegan’ package version 2.3 (Oksanen et al., 2011). The *metaMDS* function follows the ordination approach outlined in Minchin (1987). In this method, up to 20 random starting configurations are allowed until a stable, global solution is reached (which requires that the same minimum stress value is returned in two consecutive iterations). I used the Bray-Curtis (Sørensen) distance estimator to quantify community dissimilarity; this measure performs well in ecological datasets (Faith et al., 1987; McCune and Grace, 2002). I compared final stress of ordinations with 2 and 3 axes to identify the simplest solution with a stress of less than 20%, and determined that a 2-dimensional solution was sufficient in all cases (McCune and Grace, 2002). I assessed the strength of ordination solutions using Shepard diagram. Output diagrams were centered, scaled to half-change units, and rotated so that the first axis maximized the variation between groups of sites with and without turbines to best represent differences resulting from the treatment effect.

I used NMS to visualize community structure independently for plant and predator data sets. For plants, ordinations were generated using presence/absence data and for predators, ordinations used the encounter rate per unit of search effort (ranging from 4.3–12.8 hours/site, Table 4). In each ordination, I used a permutation-based ANOVA to test the hypothesis that communities from turbine and non-turbine site groups were more dissimilar to the opposite group than to communities within site groupings. In ‘vegan’, this was implemented using the *adonis* function with 1000 runs to infer whether a significant amount of variation in the distance matrix was explained by the treatment effect of turbine presence or absence (Oksanen et al., 2011).

I used *envfit* ('vegan') to identify correlations between ordination structure, community diversity, and measured abiotic and biotic variables. The *envfit* function calculates correlation between site location in ordination space and selected variables using a permutation test ($n = 1000$) to determine the coefficient of determination (R^2) and a frequentist alpha value (Oksanen et al., 2011). I looked for correlations between ordination structure and richness ("RICH"), diversity ("DIVERSE"), and evenness ("EVEN"); I also calculated correlations between ordinations and disturbance, habitat, and climate variables (Table 1). All variables were z-standardized (mean = 0.0, $SD = 1.0$) to account for scale of measurement when comparing the effects of parameters (White and Burnham, 1999).

I used 'vegan' *envfit* to identify predator taxa with strong gradients in abundance along NMS Axis 1 (with one dimension, this is identical to a linear regression with abundance as a function of Axis 1 score); for plant presence and absence data, this objective was achieved using logistic regression (*glm*). I used R^2 (linear regression, predators) or pseudo- R^2 (logistic regression, plants) values to identify species where probability of presence or abundance was explained by Axis 1 score. Secondly, I used species ordination scores to explore patterns of community assembly within defined functional groupings of plants and predators, assigned using relevant taxon-specific primary literature and species accounts. I used analysis of variance to identify coefficient and standard deviation values for classes of each category that explained significant ($P < 0.05$) variation in Axis 1 score. For plants, I looked for variation along the treatment axis across the following plant habit or life form groups: shrubs, annual herbs, annual grasses, succulents, and perennial herbs/grasses. Two species were classified as ferns and were

omitted when characterizing group responses. Additionally, I classified species by origin: native or invasive to the Western United States, or endemic to California and northern Baja California (Table 5). For predators, I looked for patterns of species composition within groups of taxonomic guilds, simplified guilds separating aerial and terrestrial species, groups of species with no, low, moderate, and high levels of scavenging behavior, and across omnivore, carnivore, and insectivore diet classes (Table 3).

3. RESULTS

3.1 Predator communities

I observed 32 target taxa 884 times; an average of 13.3 taxa were encountered per site (range: 10–16), at a rate of 13.5 observations per hour (range: 7.5–20). Abundance of predators at non-turbine sites increased on average relative to abundance at turbine sites for invertebrates (46.2%), mammals (21.4%), raptors (70.1%), and songbirds (61.5%). For lizards (18.9%) and corvids (7.4%), abundances increased at turbine sites. Overall, turbine sites were 13.4% less rich ($R = 12.3$, $SD = 1.7$), 14.3% less diverse ($H = 1.8$, $SD = 0.2$), and 11% less even ($E = 0.70$, $SD = 0.06$) than non-turbine sites. On average, non-turbine sites had 14.2 ($SD = 1.3$) taxa, Shannon diversity of 2.1 ($SD = 0.2$), and evenness of 0.81 ($SD = 0.07$) (Figure 2). Of the taxa encountered, four were present at all sites (bobcat, coyote, loggerhead shrike, and Great Basin whiptail). These taxa (as well as common raven, western zebra-tailed lizard, and tarantula sign) were the most commonly encountered, with, on average, more than 0.75 encounters per hour per site.

Eleven taxa were unique to a single site; two from turbine sites (sidewinder rattlesnake, spotted leaf-nosed snake) and nine from non-turbine sites (Great Basin

collared lizard, Great Basin fence lizard, California kingsnake, southwestern speckled rattlesnake, red diamond rattlesnake, golden eagle, gray fox, American badger, and western black widow). Four of these unique taxa were observed at one high elevation non-turbine site, SG (western fence lizard, speckled rattlesnake, golden eagle, and gray fox).

The final 2-dimensional NMS solution had a stress value of 8.25%. There was a strong correlation between site dissimilarity and ordination distance for standardized data (non-metric $R^2 = 0.99$, linear $R^2 = 0.97$, Figure 3A). The presence of turbines explained 11% of the variation in Bray-Curtis distance between sites (Permutation ANOVA, $F_{1,7}=0.90$, $P = 0.534$).

Of the external variables tested, EVEN ($R^2 = 0.660$, $P = 0.044$), DISTURB ($R^2 = 0.791$, $P = 0.01$), NOISE ($R^2 = 0.786$, $P = 0.01$), SLOPE ($R^2 = 0.636$, $P = 0.05$), ELEV ($R^2 = 0.653$, $P = 0.004$), and WIND ($R^2 = 0.614$, $P = 0.07$) showed strong correlation with the 2-dimensional NMS solution at a $P < 0.10$ significance level (Figure 3B). When the solution was rotated to maximize the difference between turbine and non-turbine sites along the treatment axis, DISTURB (78.7%), NOISE (78.6%), and WIND (46.6%) were negatively correlated with Axis 1, while EVEN (64.6%), ELEV (57.2%) and SLOPE (52.6%) correlated positively with Axis 2 (Figure 3C). The resulting solution defined community structure by gradients of anthropogenic disturbance and associated high wind levels (Axis 1, “treatment axis”), and by gradients in site climate characteristics associated with greater evenness (Axis 2, “elevation axis”). Cover (canopy height, shrub and bare ground cover, and shrub density) and precipitation did not explain significant variation in predator community structure.

Moderately frequent scavenging behavior was positively correlated with treatment axis values characterized by low disturbance and noise ($P = 0.01$). Coefficients for high, low, or no scavenging behavior had large standard errors and P -values in excess of 0.35. Remaining groups explaining structure across guilds, habitat zones, and diets showed no difference in means for each group level by treatment axis value (Table 6). However, I noted that species within the mammal guild were not encountered at the lowest values of the treatment axis (Figure 4). Predator encounter rates was positively correlated ($R^2 > 0.4$) with ordination scores on the treatment axis for two of the 32 target taxa. For loggerhead shrike (*Lanius ludovicianus*) and red-tailed hawk (*Buteo jamaicensis*), encounter rates increased as total disturbance decreased; the treatment axis explained 49.9% ($P = 0.02$) and 57.2% ($P = 0.004$) of variation in encounter rates for these species, respectively (see Table 7 for correlations in remaining taxa).

3.2 Plant communities

I detected 119 plant species from 91 genera and 36 families, resulting in an average species richness of 2.5 species per 100 m² (range: 1.8–3.3). Fifteen species were endemic to California and Baja California, while 13 species were invasive. On average, wind farm plant richness was 0.8% lower for endemic species, and 124% higher for invasive plant species. Richness of plants increased at non-turbine sites relative to turbine sites for succulents (380%), shrubs (67%), annual grasses (14.3%), annual herbs (34%), ferns (none at turbine sites), and perennial herbs and grasses (50%). In total, turbine sites had 31.5% fewer species ($R = 31.8$, $SD = 3.1$) than non-turbine sites ($R = 46.4$, $SD = 4.8$). These species comprised 34.7% fewer genera at turbine sites ($R = 28.0$, $SD = 3.2$) than at non-turbine sites ($R = 42.6$, $SD = 4.2$), and 34.9% fewer plant families at turbine sites (R

= 13.8, $SD = 2.2$) than at non-turbine sites ($R = 21.2$, $SD = 3.6$). Shannon diversity was 15% lower at turbine sites ($H = 2.2$, $SD = 0.3$) than at non-turbine sites ($H = 2.6$, $SD = 0.3$). Turbine and non-turbine sites had similar evenness (turbine: $E = 0.85$, $SD = 0.06$; non-turbine: $E = 0.85$, $SD = 0.03$) of species within families (Figure 2).

I recorded 50 plant species that were present at one site only. Only six (average = 1.5 per site) of these were found at turbine sites (three native annual herbs (one endemic), two native shrubs (one endemic) and one invasive annual herb). The remaining 44 unique taxa were from non-turbine sites (average = 8.8 per site), including nine (60%) of the recorded endemic species (four annual herbs, four perennial herbs, and one shrub). Only four of the unique species at non-turbine sites were non-native (two annual grasses and two annual herbs). Of the 44 unique observations at non-turbine sites, 24 were observed at one high elevation non-turbine site, “SG”, which accounted for six of the nine recorded endemic species at non-turbine sites. Seven species were found at all nine sites (six annual herbs (two invasive, four native) and one invasive grass).

The final NMS ordination resulted in a stress of 8.42%; there was strong agreement between site dissimilarity and ordination distance (non-metric $R^2 = 0.99$, linear $R^2 = 0.97$, Figure 5A). The presence of turbines explained 16% of the variation in Bray-Curtis distance between sites (Permutation ANOVA, $F_{1,7}=1.39$, $P = 0.239$). Of the external variables tested, RICH ($R^2 = 0.747$, $P = 0.01$), DIVERSE ($R^2 = 0.772$, $P = 0.01$), DISTURB ($R^2 = 0.597$, $P = 0.06$), SLOPE ($R^2 = 0.544$, $P = 0.08$), ELEV ($R^2 = 0.899$, $P = 0.002$), and WIND ($R^2 = 0.640$, $P = 0.05$) showed strong correlation with the 2-dimensional solution at a $P < 0.10$ significance level (Figure 5B). When the solution was rotated, Axis 1 (“treatment axis”) was positively correlated with RICH (74.1%) and

DIVERSE (71.1%), and negatively correlated with DISTURB (50.0%) and WIND (56.1%). ELEV (75.8%) and SLOPE (47.7%) were negatively correlated with Axis 2 (“elevation axis”). Overall, community structure was negatively correlated with disturbance and associated high wind levels, with lower diversity and richness at high-disturbance sites. However, abiotic characteristics were important drivers of community trends as well (Figure 5C). Cover and precipitation variables did not explain significant variation in community structure.

Plants showed segregation of group means by plant status (native, endemic, or invasive) along the treatment axis (Figure 4). Presence of endemic plants was associated with low disturbance and noise scores along the treatment axis ($P = 0.001$). Similarly, scores for non-native plants were associated with high noise and disturbance on the treatment axis ($P = 0.02$). The coefficient for native plants was not significant. There were no differences in group means for plant habit levels on the treatment axis when assessed using ANOVA models (Table 6). However, when richness was partitioned by plant habit category, average richness per site at non-turbine sites always exceeded that at turbine sites.

Of the 119 encountered plants, presence or absence was strongly explained ($R^2 > 0.4$) by ordination scores on the treatment axis for ten species. Probability of presence increased with high noise and disturbance values on the treatment axis for non-native annual herbs *Oncosiphon piluliferum*, *Sisymbrium irio*, and for the weedy native perennial herb *Stephanomeria pauciflora*. Probability of presence increased with low noise and disturbance values on the treatment axis for native annual herbs *Eucrypta chrysanthemifolia*, *Eulobus californicus*, *Salvia columbariae*, and *Phacelia minor*

(endemic), as well as natives *Selaginella bigelovii* (endemic fern), *Mirabilis laevis* (perennial herb), and *Bahiopsis parishii* (shrub) (Table 8).

4. DISCUSSION

Development of renewable energy infrastructure remains a top priority in efforts to mitigate global climate change (Panwar et al., 2011). In order to maximize the net conservation gain realized from renewable wind energy production, best land-management practices should aim to reduce biodiversity loss associated with development. Collision mortality may represent only a fraction of the cumulative effects that wind farms have on biotic community function and health (Kuvlesky et al., 2007). Here, I addressed whether wind farms affected biodiversity with a focused study of plants and predators; I discuss observed community responses at disturbed, wind-farm sites in relation to nearby reference areas.

4.1 Diversity and community composition trends between sites

Study sites at wind farms were less rich, even, and diverse. Communities with low diversity are often considered to be less stable, trophically complex, productive, or disturbance resistant than sites with high levels of diversity, which suggests that wind farms as altered landscapes may provide lower quality habitats for local biota (Johnson et al., 1996; McCann, 2000; Paine, 1966).

While it is possible that differences in diversity (expressed as average richness, evenness, or diversity per site) may reflect additional confounding variation in habitat and climate traits between wind farm sites and reference areas; results from NMS ordinations suggest that these differences are likely attributable to total disturbance

intensity at wind farms sites. When community composition patterns were partitioned between the target effect (treatment axis) and confounding variation across sites (elevation axis), the final solutions for plant and predator communities showed segregation of wind farms and reference areas along the treatment axis (correlated with road density and noise level) which characterized the severity of anthropogenic disturbance at each site. Between plant and predator ordinations, wind farm sites showed similar values along the disturbance gradient (except for site PH). Ordination results strengthen the conclusion that greater disturbance intensity at wind farms results in altered community composition and reduced richness and diversity, which was most evident in plant communities.

The observed losses of diversity at wind farms were comparable in magnitude with diversity losses reported for disturbances such as natural gas extraction (Jones et al., 2014) or grazing (James, 2003), or for linear disturbances such as aquaducts (Berry et al., 2015), utility corridors (Lathrop and Archbold, 1980), or roads (Bury et al., 1977). Relative to reference sites, wind farms were more disturbed (lower score on NMS treatment axis) than all control areas except for site I-10, which was characterized by close proximity to a major interstate and off-road vehicle use. Reference sites harbored more diversity than wind farm sites, despite grazing activities, close proximity to housing developments, or paved roadways with moderate traffic volume in these areas. These types of disturbances were also sometimes present at wind farms, and may have contributed synergistically towards the overall diversity losses at these sites.

There was some evidence that wind farm communities were more homogeneous than communities at non-turbine sites. In both predator and plant datasets, fewer unique

species observations occurred at wind farm sites relative to nearby reference areas. Homogenized communities often result from high intensity of human disturbance (Proppe et al., 2013), where altered abiotic conditions can cause local extirpations of rare species with low densities, or endemic species with limited geographic ranges (McKinney and Lockwood, 1999).

Together, results indicate that wind farm developments affect more than just avian and bat species; wind farms may also affect patterns of terrestrial and non-vertebrate community composition and diversity. In this study, I noted reduced predator diversity at disturbed wind farm sites. This is an important finding because predator diversity may significantly affect ecosystem function by increasing the susceptibility of herbivore communities to trophic cascades (Finke and Denno, 2004). Similarly, changes in plant composition and richness were detected at wind farm sites. Invasion by exotic grass and herb species may alter the structure and function of entire biotic communities, for example, the trajectory of the invertebrate community (and the predators that depend upon them) can change drastically when plant invasions reduce the abundance of host plants that are important as food or as habitat (de Groot et al., 2007). Reduced plant diversity is also linked with greater potential for exotic species establishment, which can further alter the stability of plant communities, resulting in rapid declines in ecosystem diversity and function (Lyons and Schwarts, 2001; Tilman et al., 2006).

Disturbed sites were often windier than less-disturbed sites. Windy conditions might influence plant communities by creating better conditions for invasive, wind-dispersed species; however, it is more likely that the significance of this variable can be explained by strong correlations between average wind speed, and disturbance (Pearson's

$r = 0.70$, $P = 0.037$), or noise level ($r = 0.66$, $P = 0.054$). In general, developers are expected to select areas with heavy winds for wind farm development. Community structure was also explained by differences in slope and elevation, which defined the second axis gradient. For predator communities, higher elevation sites also had greater community evenness. Elevation is often an important driver of diversity trends (Richerson and Lum, 1980); it is not surprising, then, that elevation affected community composition. Abiotic factors (e.g., temperature and precipitation) can change rapidly along elevational gradients, particularly in desert communities (Beatley, 1975). For plant communities, many habitat characteristics (e.g., soil depth, solar radiation, ground temperature, or soil moisture) also vary with slope (Ehleringer and Cooper, 1988). However, there is no reason to expect that slope would directly influence predator community composition; as with the wind variable, it is possible that the importance of slope may result from its correlation with elevation ($r = 0.67$, $P = 0.048$).

In desert communities, productivity has often been linked with trophic complexity and diversity, especially for predator and plant communities (Ayal, 2007; Johnson et al., 1975). However, in this study, variables describing relative site productivity (bare ground cover, precipitation, and shrub canopy cover, density, and height) were not important predictors of ecological distances. It is possible that elevation alone was sufficient to explain overall patterns of productivity across study sites. In order to generate more precise estimates of the effects of anthropogenic disturbances on diversity trends, future studies should minimize differences among study sites to limit confounding effects of variation in habitat, climate, or topography on community composition.

4.2 Predator community composition

Encounter rates of red-tailed hawk (*Buteo jamaicensis*) and loggerhead shrike (*Lanius ludovicianus*) were strongly correlated with low disturbance (positive values along the treatment axis). On average, an individual hawk was encountered every 25.6 hours and a shrike every 3.25 hours at wind farm sites, compared with more frequent encounters of every 2.3 and 0.85 hours at non-turbine sites, respectively. Neither of these species is particularly intolerant of human disturbances (Michaels and Cully, 1998; Minor et al., 1993) and both have been documented using wind farm habitats in other California wind resource areas (Smallwood et al., 2009). However, these species also experience high mortality rates at wind farm facilities which might account for reduced encounter rates at turbine sites (Smallwood et al., 2009). Further research is needed to confirm that observed differences in abundance did not stem from confounding habitat differences between turbine and non-turbine sites.

Predator species demonstrated few associations with the disturbance axis when grouped by functional categories of scavenging propensity, foraging zone, diet, or guild. In this study, coyote (*Canis latrans*), domestic dog, (*Canis lupus familiaris*) and common raven (*Corvus corax*) were frequently encountered at turbine sites. Wind farms are known to attract a number of scavenger species (Smallwood et al., 2010); yet, there was no indication that these species were more abundant at wind farms than at nearby non-turbine sites. When grouped by frequency of scavenging, only species with moderately-frequent scavenging behavior (golden eagle (*Aquila chrysaetos*), red-tailed hawk (*Buteo jamaicensis*), American badger (*Taxidea taxus*) and gray fox (*Urocyon cinereoargenteus*)) associated with low noise and road density on the treatment axis.

These species may be less tolerant of disturbances when compared with species that scavenge more frequently. Relative to other raptor species, golden eagles are particularly intolerant of human activity (Holmes et al., 1993). Eagles and red-tailed hawks experience high mortality rates at wind farms (Orlaff and Flannery, 1992), suggesting that turbine avoidance behavior or low survival may drive some of the observed differences in community composition between wind farms and undeveloped areas. For badger and fox, however, it is unclear whether the observed association with sites by treatment effect represented a real effect or simply an artifact of inadequate abundance estimation. Whereas coyotes and domestic dogs could be identified from scat samples, observers did not assign any scat observations to either badger or fox; in fact, these species were identified from skeletal remains only; thus, their abundances are likely grossly underestimated.

There was no indication that aerial predators were more or less abundant at wind farm sites relative to terrestrial predators. Omnivores (species that are often more successful in disturbed habitats) were not encountered more frequently at wind farms than species with more specialized predator diets. When broken down by taxonomic groups, there was no evidence that any one group was more, or less successful at wind turbine sites than in nearby habitats. The highest mammal encounter rates occurred at non-turbine sites for each of the 5 species examined (Figure 4); however, the anticipated errors in estimating abundances for these species preclude any speculation regarding the success of this group at wind farms.

4.3 Plant community composition

As of 2015, this study is the first to document increased invasive plant richness at wind farm sites (see Jones et al., 2015). Of the 13 invasive species encountered, a majority (69%) were present at turbine sites. While one or more non-native plants were detected at all sites, ordination scores for non-native plants, and in particular for London rocket (*Sisymbrium irio*), were significantly correlated with the treatment axis (invasive plants were more commonly encountered at wind farm sites and non-turbine sites with high disturbance). At these sites, high road density and heavy human traffic may provide an avenue for invasive plant dispersal and establishment (Trombulak and Frissell, 2000).

Conversely, there was clear evidence that disturbed areas supported fewer plant species endemic to deserts within the California floristic province. Only four of the 15 endemic species encountered were present at disturbed sites. In particular, wild Canterbury bells (*Phacelia minor*) and Bigelow's moss fern (*Selaginella bigelovii*) showed the strongest negative association with the disturbance axis (less common at wind farms and disturbed areas). Endemic species are often characterized by narrow niche breadths; as such, they may be less able to exploit or persist in disturbed habitats (Slatyer et al., 2013). These species may mature late, produce few seeds, produce seeds infrequently, or produce seeds that disperse short-distances; together, these characteristics can reduce the pace of recolonization after severe disturbances such as road clearing or electrical fires (Brown and Boutin, 2009; McIntyre and Lavorel, 1994). Endemics with slow life-history characteristics or narrow habitat preferences may also be less able to compete with invasive species, which were more common at disturbed non-turbine sites and wind farm sites (Seabloom et al., 2003). Finally, wind farms and other disturbed

areas may be un-suitable for native species of plant pollinators or seed dispersers, resulting in the loss of species interactions required for the persistence of these endemic species (Sugden, 1985; Traveset and Richardson, 2006). Further research is needed to determine whether wind farms affect the diversity or persistence of mutualistic interactions.

There was no evidence that plant life forms, by themselves, responded differently to disturbance and energy development. Ferns were associated with non-turbine sites only; however, all remaining plant types were present in both treatments. Annual herbs were ubiquitous, and trends of abundance within this group were explained in part by plant status (a majority of non-native and endemic plants were annual herbs). Together, these results suggest that wind farm sites are able to support plant communities with diverse life-history characteristics, which may be of greater importance than total plant richness for overall ecosystem function (Hector et al., 1999).

4.4 Wind farm traits and diversity

Post-hoc examination of predator community composition data provided some evidence that community composition at wind farms with older-generation turbines (sites ME and PH) differs from that of wind farms with newer models turbines (sites DI, MV). Bird mortality rates are often higher at older, lattice tower turbine wind farms (Smallwood and Karas, 2009). These old generation wind farm facilities may thus provide greater resource subsidies (carcasses) for scavenging predators. My data indicated that ravens were found predominantly at old generation turbine sites, while coyote and dog had similar abundances at old and new generation sites. Ravens were often observed perching on non-operational lattice turbines over the course of the study; they were never observed

perching on newer tubular towers. It is unclear, however, if ravens are attracted to older generation sites because of the wide availability of perching habitat, or because of greater prey carcass subsidies.

Old-generation turbines may also suppress abundance for avian species with behaviors or habitat affinities that increase turbine-induced mortality risks (Remeš, 2000). Of the eleven targeted avian species, burrowing owl (*Athene cuniculata*), American kestrel (*Falco sparverius*), and shrike were less abundant or absent from old generation turbine sites. At the Altamont Pass Wind Resource Area in California, these species had higher than average wind turbine collision mortality rates relative to other bird species, and all three were shown to have greater mortality rates at old-generation turbine sites (Smallwood and Karas, 2009).

While there do appear to be some differences in species tolerance of older versus newer wind energy technologies, I was unable to determine the cumulative influence of turbine type on overall diversity and community composition trends. Ordination data indicated that older turbine sites were as different from each other as they were from newer turbine sites in terms of richness, diversity, evenness, and treatment axis score. In addition, there were several site characteristics that were confounded with turbine type. Old-generation turbine sites had greater road densities and noise levels, as well as greater turbine density and more time since ground disturbance—all factors that may influence diversity patterns (Adams and Geis, 1983; McAuliffe, 1984; Proppe et al., 2013; Schroeder and Perera, 2002).

4.5 Conclusions

We cannot dismiss the possibility that sites with wind farms were more disturbed than targeted reference areas **prior** to turbine installation. However, the results of this study indicate that wind farms have greater habitat disturbance and higher noise levels when compared with similar sites without turbine installations. These variables were correlated with reduced plant diversity and richness, particularly for endemic species. Probability of non-native species presence increased at wind farms, an effect that may create a detrimental stepping-stone for invasion into surrounding natural habitats (e.g., Gelbard and Belnap, 2003). Predator communities at wind farms were also less rich, even, and diverse than communities from nearby natural areas. Several avian species showed reduced encounter rates at wind farm sites, which were characterized by fewer unique species relative to nearby communities with lower disturbance intensity. For vertebrates, the potential for behavioral avoidance of wind energy facilities should be carefully considered, especially in species that demonstrate ecological interactions which are important for overall ecosystem health (Drewitt and Langstron, 2006; Pearce-Higgins et al., 2009).

There are a number of strategies that land managers can adopt to enhance the biodiversity value of wind energy facilities. Primarily, managers should promote development strategies that minimize permanent physical footprints in local habitats (particularly when rare, endemic, or conservation priority species may be affected). Ideally, wind farms should be constructed in areas of low conservation priority, and not in pristine areas or areas sensitive to disturbance (Kiesecker et al., 2011). The results from this study indicate that local diversity is affected by wind farm developments, and

so in order to avoid significant diversity loss from development of natural, un-altered habitats, wind farms may instead be situated in areas being used concurrently for ranching, farming, or resource extraction (Kiesecker et al., 2011; McDonald et al., 2009). Pre-development, managers can ensure that targeted sites are not harboring unique biological diversity. By comparing diversity across alternative wind farm siting locations, managers can identify areas where energy development will be the least detrimental. Finally, managers should continue to encourage development plans that include structural or technical specifications designed to reduce mortality risks for birds and bats.

Wind farms provide a valuable source of renewable energy that can substantially reduce national dependence on finite carbon fuels. However, this research suggests that even “green” wind energy developments affect local biodiversity. And yet, with the lowest land-use intensity of any renewable technology, wind farms provided remnant patches of habitat that are capable of supporting most of the same species found in less-developed areas. It is possible that low land-use intensity at wind farms allows for greater biological sustainability relative to other renewable technologies; however, similar research addressing biodiversity loss at solar, hydroelectric, and geothermal facilities is needed to validate this claim. In sensitive desert habitats where utility-scale renewable energy developments would create substantial risks for local biodiversity, the potential for decentralized energy production strategies should also be considered (Van Der Schoor and Scholtens, 2015).

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Table 1. Disturbance, habitat, and climate characteristics of predator and plant community study areas in the San Gorgonio Pass Wind Resource Area in southern California, north of Palm Springs.

Site (Turbine Presence) ¹	Disturbance Index ²	Noise, Ave. dBA ³	Non- canopy Cover, % ⁴	Shrub Density/ square m ⁴	Shrub Canopy Cover, % ⁴	Shrub Canopy Height, m ⁴	Slope, degrees ⁵	Elevation, m ⁵	Ave. Precip, inches ⁶	Wind, mph ⁷
SG (-)	2.61	41.16	0.84	0.06	2.95	0.61	17.40	822.73	16.95	3.80
MI (-)	22.82	45.45	0.63	0.38	27.50	0.56	5.84	640.74	13.86	3.64
WW (-)	40.23	47.62	0.75	0.15	17.43	0.66	2.72	468.75	13.99	5.65
DH (-)	24.17	48.17	0.82	0.11	18.92	0.97	4.12	500.99	10.82	2.58
I-10 (-)	67.47	54.43	0.72	0.14	19.81	0.85	8.67	390.74	11.81	13.13
ME (+)	26.48	47.92	0.76	0.23	12.97	0.62	8.84	711.98	18.43	8.40
PH (+)	59.97	66.37	0.85	0.14	13.00	0.70	13.21	464.42	13.42	8.25
MV (+)	49.89	60.49	0.87	0.09	3.88	0.48	1.92	366.12	15.26	10.15
DI (+)	34.53	49.85	0.90	0.07	12.83	1.01	1.51	318.03	6.57	6.84
-	31.46 ±	47.37 ±	0.75 ±	0.17 ±	17.32 ±	0.73 ±	7.75 ±	564.79 ±	13.49 ±	5.76 ±
Average	24.15	4.82	0.08	0.12	8.93	0.17	5.83	170.21	2.36	4.26
± SD	42.72 ±	56.16 ±	0.85 ±	0.13 ±	10.67 ±	0.70 ±	6.37 ±	465.14 ±	13.42 ±	8.41 ±
+	15.05	8.77	0.06	0.07	4.53	0.22	5.67	175.48	5.01	1.36

¹ Site coordinates: SG (33.9775, -116.6719), MI (34.0009, -116.6037), WW (33.9385, -116.6406), DH (33.9903, -116.4975), I-10 (33.9266, -116.6610), ME (33.9440, -116.6791), PH (33.9393, -116.6179), MV (33.9134, -116.6268), DI (33.9394, -116.5540)

² Total length of road (weighted by 0.5 for footpath, 1.0 for two-track, 3.0 for residential, 5.0 for secondary, and 10.0 for highway) within a 1000 m radius from plant richness study plots, digitized in ArcMap from 1 m resolution California NAIP imagery

³ Average a-weighted decibels (dBA) from 10 random readings (July 2015) when turbines were running and wind was ≤ 3.5 mph

⁴ Data from three, 100 m belt transect and point-line-intercept surveys (July 2014). Non-canopy cover: proportion of 100 random points per transect that did not intersect a plant canopy. Shrub density: number of shrubs per m². Shrub canopy cover: percent cover of shrubs, assuming a circular diameter for each shrub, averaged across all shrub species

⁵ Derived from a 2013 digital elevation model (DEM) raster at 1/3 arc-second resolution, retrieved from the National Elevation Dataset (available online from the USGS at: <http://nationalmap.gov/elevation.html>)

⁶ Retrieved from the PRISM Climate Group, Oregon State University (<http://www.prism.oregonstate.edu/>, created 4 Feb 2004), averaged across the winter months of November through February (2012–2015)

⁷ Average wind speed recorded during predator encounter surveys in June and July of 2014

Table 2. Facility characteristics for 2014–2015 predator and plant community diversity study at four target wind farm sites within the San Geronio Wind Resource Area.

Wind Farm	Wind Farm Size (km ²)	Number of Turbines	Height ¹ , Base Type ²	Turbine Density	Installation Date
ME	1.7	460	S, L	High	1983
PH	0.7	168	S, L	High	1984
MV	0.9	45	T, T	Low	2002
DI	2.5	21	T, T	Low	2007

¹ Hub height of (S) 20 to 25 m or (T) 60 to 70 m

² Tower structure of (L) steel lattice frame or (T) steel tubular tower

Table 3. Predator community surveys (2014) at turbine and non-turbine sites in Palm Springs, CA targeted 32 predator taxa within reptile, bird, mammal, and invertebrate taxonomic guilds. Taxa were classified by taxonomic guild, habitat guild, propensity for scavenging, and diet.

Guild	Latin Name	Common Name	Habitat ¹	Scavenge Behavior ²	Diet ³
Reptile (Lizard)	<i>Callisaurus draconoides</i>	Western Zebra-tailed Lizard	T	2	I
	<i>Sceloporus magister</i>	Desert Spiny Lizard	T	1	I
	<i>Sceloporus occidentalis</i>	Great Basin Fence Lizard	T	1	I
	<i>Aspidoscelis tigris</i>	Great Basin Whiptail	T	1	I
	<i>Crotaphytus bicinctores</i>	Great Basin Collared Lizard	T	1	C/I
Reptile (Snake)	<i>Coluber flagellum</i>	Red Racer	T	2	C
	<i>Lampropeltis californiae</i>	California Kingsnake	T	1	C
	<i>Phyllorhynchus decurtatus</i>	Spotted Leaf-nosed Snake	T	1	C
	<i>Crotalus ruber</i>	Red Diamond Rattlesnake	T	2	C
	<i>Crotalus cerastes</i>	Colorado Desert Sidewinder	T	2	C
	<i>Crotalus mitchellii</i>	Southwestern Speckled Rattlesnake	T	2	C
Bird	<i>Aquila chrysaetos</i>	Golden Eagle	A	3	C
	<i>Buteo jamaicensis</i>	Red-tailed hawk	A	3	C
	<i>Falco mexicanus</i>	Prairie Falcon	A	2	C
	<i>Falco sparverius</i>	American Kestrel	A	2	C/I
	<i>Athene cunicularia</i>	Burrowing Owl	A	2	C/I
	<i>Corvus corax</i>	Common Raven	A	4	O
	<i>Lanius ludovicianus</i>	Loggerhead Shrike	A	2	C/I
	<i>Geococcyx californianus</i>	Greater Roadrunner	A	2	O
	<i>Mimus polyglottos</i>	Northern Mockingbird	A	1	O
	<i>Campylorhynchus brunneicapillus</i>	Cactus Wren	A	1	I
	<i>Salpinctes obsoletus</i>	Rock Wren	A	1	I
Mammal	<i>Taxidea taxus</i>	American Badger	T	3	O
	<i>Canis latrans</i>	Coyote	T	4	O
	<i>Canis lupus</i>	Domestic Dog	T	4	O
	<i>Urocyon cinereoargenteus</i>	Gray Fox	T	3	O
	<i>Lynx rufus</i>	Bobcat	T	2	C
Invertebrate	Solifugae, combined obs.	Sun Spider	T	2	I
	Scorpiones, combined	Giant Scorpion	T	2	I
	<i>Latrodectus hesperus</i>	Western Black Widow	T	2	I
	Lycosidae, combined obs.	Wolf Spider	T	2	I
	<i>Aphonopelma</i> sp.	Tarantula	T	2	I

¹ Dominant habitat type for taxonomic guild was identified as terrestrial ("T") or aerial ("A")

² Published species accounts were used to assign predators to categories of scavenging behavior as (1) None: no records indicate scavenging behavior; (2) Low: few records documenting take of carcasses; (3) Moderate: species will take carrion when offered; and (4) High: species actively scavenges for carcasses

³ Taxa were classified as omnivores (O), carnivores (C), or insectivores (I)

Table 4. Average weather conditions and total survey effort during predator surveys conducted from mid-June to late July of 2014 at nine study sites near Palm Springs, California.

Site (Turbine Presence)	Weather ¹			Total Observation Hours (Number of Surveys)				
	Wind, mph	Relative Humidity, %	Temp, °C	Reptiles	Birds	Mammals	Inverts	Combined Guilds
SG (-)	3.80	30.94	27.99	7.88 (4)	6.58 (3)	6.92 (2)	4.57 (2)	25.95
MI (-)	3.64	35.74	29.61	8.53 (4)	6.67 (3)	6.48 (2)	7.25 (2)	28.93
WW (-)	5.65	26.31	31.63	5.75 (3)	7.88 (5)	4.28 (2)	6.28 (2)	24.20
DH (-)	2.58	41.95	30.38	7.67 (4)	7.45 (4)	7.50 (2)	7.02 (2)	29.63
I-10 (-)	13.13	57.63	26.22	6.75 (3)	12.83 (4)	8.22 (2)	8.23 (2)	36.03
ME (+)	8.40	36.25	28.57	7.10 (4)	6.45 (3)	6.32 (2)	7.42 (2)	27.28
PH (+)	8.25	25.35	31.87	8.45 (4)	7.35 (3)	7.47 (2)	7.42 (2)	30.68
MV (+)	10.15	43.05	28.95	7.97 (4)	9.68 (4)	8.50 (2)	7.30 (2)	33.45
DI (+)	6.84	35.93	32.20	6.83 (3)	8.15 (4)	7.22 (2)	7.22 (2)	29.42
Average ± SD								
-	5.8 ± 4.3	38.5 ± 12.2	29.2 ± 2.1	7.3 ± 1.1	8.3 ± 2.6	6.7 ± 1.5	6.7 ± 1.4	28.9 ± 4.5
+	8.4 ± 1.4	35.1 ± 7.3	30.4 ± 1.9	7.6 ± 0.8	7.9 ± 1.4	7.4 ± 0.9	7.3 ± 0.1	30.2 ± 2.6

¹ Averaged values taken at the start and end of each predator survey

Table 5. Plant species encountered during richness surveys (2015) at 1600 m² study plots from nine sites at wind farms and nearby reference areas north of Palm Springs, CA.

Family	Genus	Species	Native ¹	Habit	Sites Encountered ²
AGAVACEAE	<i>Hesperoyucca</i>	<i>whipplei</i>	E	shrub	SG
ASTERACEAE	<i>Acamptopappus</i>	<i>sphaerocephalus</i>	Y	shrub	I-10, WW
	<i>Ambrosia</i>	<i>dumosa</i>	Y	shrub	DI, MV, PH, I-10, DH, WW, MI
		<i>salsola</i>	Y	shrub	DI, I-10, DH, SG
	<i>Artemisia</i>	<i>californica</i>	Y	shrub	ME, MI, SG
	<i>Baccharis</i>	<i>brachyphylla</i>	Y	shrub	MI
	<i>Bahiopsis</i>	<i>parishii</i>	Y	shrub	DH, WW, MI, SG
	<i>Bebbia</i>	<i>juncea</i>	Y	shrub	DI, MV, PH, I-10, DH, WW, MI
	<i>Chaenactis</i>	<i>fremontii</i>	Y	annual herb	DI, MV, PH, ME, I-10, DH, WW, MI, SG
		<i>glabriuscula</i>	E	annual herb	MI
		<i>Encelia</i>	<i>actoni</i>	Y	shrub
	<i>farinosa</i>		Y	shrub	DI, PH, ME, I-10, DH, WW, MI, SG
	<i>Ericameria</i>	<i>paniculata</i>	Y	shrub	I-10
	<i>Erigeron</i>	<i>foliosus</i>	E	peren. herb	SG
	<i>Eriophyllum</i>	<i>wallacei</i>	Y	annual herb	WW
	<i>Lactuca</i>	sp	N	annual herb	DI, ME, I-10, WW, SG
	<i>Lasthenia</i>	<i>californica</i>	Y	annual herb	SG
		<i>coronaria</i>	E	annual herb	ME, I-10, WW, MI
	<i>Logfia</i>	sp	Y	annual herb	DI, MV, PH, ME, I-10, DH, WW, MI, SG
	<i>Malacothrix</i>	<i>glabrata</i>	Y	annual herb	DI, MV, PH, ME, I-10, DH, WW, MI
	<i>Monoptilon</i>	<i>bellioides</i>	Y	annual herb	DH
	<i>Oncosiphon</i>	<i>piluliferum</i>	N	annual herb	DI, MV, ME, I-10, WW
	<i>Rafinesquia</i>	sp	Y	annual herb	I-10, DH, MI
	<i>Senecio</i>	<i>vulgaris</i>	N	annual herb	ME
	<i>Sonchus</i>	<i>oleraceus</i>	N	annual herb	DI, PH, ME, I-10, DH, WW
	<i>Stephanomeria</i>	<i>exigua</i>	Y	annual herb	DI, MV, ME, I-10
		<i>pauciflora</i>	Y	peren. herb	DI, MV, PH, ME, I-10, WW, MI
		<i>tenuifolia</i>	Y	peren. herb	SG
	<i>Uropappus</i>	<i>lindleyi</i>	Y	annual herb	ME, SG

BORAGINACEAE	<i>Amsinckia</i>	<i>intermedia</i>	Y	annual herb	ME, MI, SG
	<i>Cryptantha</i>	<i>angustifolia</i>	Y	annual herb	DI, MV, ME, DH
		<i>barbigera</i>	Y	annual herb	DI, MV, PH, ME, I-10, DH, WW, MI, SG
		<i>maritima</i>	Y	annual herb	PH, I-10, DH, MI
		<i>micrantha</i>	Y	annual herb	DI, MV, WW
	<i>Pectocarya</i>	sp	Y	annual herb	DI, MV, PH, ME, I-10, DH, WW, MI, SG
BRASSICACEAE	<i>Brassica</i>	<i>tournefortii</i>	N	annual herb	DI, MV, PH, ME, I-10, DH, WW, MI, SG
	<i>Caulanthus</i>	sp	Y	annual herb	ME
	<i>Descurainia</i>	<i>pinnata</i>	Y	annual herb	PH
	<i>Lepidium</i>	<i>lasiocarpum</i>	Y	annual herb	PH, I-10, WW
		<i>strictum</i>	Y	annual herb	MI
	<i>Sisymbrium</i>	<i>irio</i>	N	peren. herb	ME, I-10
	<i>Tropidocarpum</i>	<i>gracile</i>	E	annual herb	PH, ME, I-10, SG
CACTACEAE	<i>Cylindropuntia</i>	<i>ramosissima</i>	Y	stem succulent	I-10, WW
	<i>Echinocereus</i>	<i>engelmannii</i>	Y	stem succulent	PH, MI
	<i>Ferrocactus</i>	<i>cylindricatus</i>	Y	stem succulent	I-10, MI
	<i>Opuntia</i>	<i>basilaris</i>	Y	stem succulent	SG
	<i>Peritoma</i>	<i>arborea</i>	E	shrub	MV
CAPPARACEAE	<i>Loeflingia</i>	<i>squarrosa</i>	Y	annual herb	DI, MV, WW
CARYOPHYLLACEAE	<i>Cuscuta</i>	sp	Y	annual herb	DH
CONVOLVULACEAE	<i>Crassula</i>	<i>connata</i>	Y	annual herb	DI, MV, ME, I-10, DH, WW, MI
CRASSULACEAE	<i>Dudleya</i>	sp	E	peren. herb	WW
CUCURBITACEAE	<i>Marah</i>	<i>macrocarpa</i>	E	peren. herb	SG
EPHEDRACEAE	<i>Ephedra</i>	<i>californica</i>	Y	shrub	MV, DH, WW, MI
EUPHORBIACEAE	<i>Ditaxis</i>	<i>neomexicana</i>	Y	peren. herb	PH, I-10
	<i>Euphorbia</i>	<i>polycarpa</i>	Y	peren. herb	DI, MV, PH, I-10, DH, WW
	<i>Stillingia</i>	<i>linearifolia</i>	Y	peren. herb	DI, PH, DH, MI
FABACEAE	<i>Acmispon</i>	<i>glaber</i>	Y	peren. herb	SG
		sp	Y	peren. herb	MI
		<i>strigosus</i>	Y	annual herb	ME, I-10, WW, MI, SG
	<i>Lupinus</i>	<i>bicolor</i>	Y	peren. herb	ME, SG
		<i>hirsutissimus</i>	Y	annual herb	WW
		<i>sparsiflorus</i>	Y	annual herb	ME, MI
		<i>Psorothamnus</i>	<i>arborescens</i>	Y	shrub

		<i>schottii</i>	Y	shrub	DI, DH
	<i>Senegalia</i>	<i>greggii</i>	Y	shrub	DH, MI
GERANIACEAE	<i>Erodium</i>	<i>botrys</i>	N	annual herb	SG
		<i>cicutarium</i>	N	annual herb	DI, MV, PH, ME, I-10, DH, WW, MI, SG
HYDROPHYLLACEAE	<i>Emmenanthe</i>	<i>penduliflora</i>	Y	annual herb	I-10, DH, WW, MI, SG
	<i>Eriodictyon</i>	<i>trichocalyx</i>	Y	shrub	SG
	<i>Eucrypta</i>	<i>chrysanthemifolia</i>	Y	annual herb	DH, SG
		<i>micrantha</i>	Y	annual herb	I-10, WW
	<i>Nama</i>	<i>demissum</i>	Y	annual herb	DH
	<i>Nemophila</i>	<i>menziesii</i>	Y	annual herb	SG
	<i>Phacelia</i>	<i>distans</i>	Y	annual herb	PH, DH, WW, SG
		<i>minor</i>	E	annual herb	DH, SG
		<i>parryi</i>	E	annual herb	PH
KRAMERIACEAE	<i>Krameria</i>	<i>bicolor</i>	Y	shrub	MV, PH, DH, WW, MI
LAMIACEAE	<i>Salvia</i>	<i>columbariae</i>	Y	annual herb	MV, PH, I-10, DH, WW, MI, SG
LILIACEAE	<i>Dichelostemma</i>	<i>capitatum</i>	Y	peren. herb	ME, MI, SG
LOASACEAE	<i>Mentzelia</i>	<i>involutrata</i>	Y	annual herb	DH
MALVACEAE	<i>Sphaeralcea</i>	<i>coulteri</i>	Y	annual herb	MI
NYCTAGINACEAE	<i>Mirabilis</i>	<i>laevis</i>	Y	peren. herb	MV, PH, I-10, DH, WW, MI, SG
ONAGRACEAE	<i>Camissonia</i>	<i>strigulosa</i>	E	annual herb	SG
	<i>Camissoniopsis</i>	<i>pallida</i>	Y	annual herb	DI, MV, PH, I-10, DH, WW, MI
	<i>Chylismia</i>	<i>claviformis</i>	Y	peren. herb	DH
	<i>Eulobus</i>	<i>californicus</i>	Y	annual herb	DI, MV, PH, DH, WW, MI, SG
PAPAVERACEAE	<i>Eschscholzia</i>	<i>californica</i>	Y	peren. herb	SG
		<i>minutiflora</i>	Y	annual herb	PH, DH
PLANTAGINACEAE	<i>Plantago</i>	<i>ovata</i>	Y	annual herb	WW
POACEAE	<i>Avena</i>	<i>barbata</i>	N	annual grass	SG
	<i>Bromus</i>	<i>diandrus</i>	N	annual grass	SG
		<i>madritensis</i>	N	annual grass	MV, PH, ME, WW, MI, SG
	<i>Hilaria</i>	<i>rigidia</i>	Y	peren. grass	PH, MI
	<i>Schismus</i>	<i>sp</i>	N	annual grass	DI, MV, PH, ME, I-10, DH, WW, MI, SG
POLEMONIACEAE	<i>Eriastrum</i>	<i>eremicum</i>	Y	annual herb	DH
	<i>Gilia</i>	<i>angelensis</i>	E	annual herb	SG
		<i>stellata</i>	Y	annual herb	DH, MI
	<i>Saltugilia</i>	<i>australis</i>	E	annual herb	WW

POLYGONACEAE	<i>Chorizanthe</i>	<i>brevicornu</i>	Y	annual herb	DI, MV, PH, I-10, DH, WW	
	<i>Eriogonum</i>	<i>elongatum</i>	E	peren. herb	SG	
		<i>fasciculatum</i>	Y	shrub	MV, PH, ME, MI	
		<i>gracile</i>	Y	annual herb	SG	
		<i>inflatum</i>	Y	peren. herb	PH, DH, WW	
		<i>palmerianum</i>	Y	annual herb	SG	
		<i>thomasii</i>	Y	annual herb	DH	
		<i>thurberi</i>	Y	annual herb	MV, PH, I-10	
		<i>Pterostegia</i>	<i>drymarioides</i>	Y	annual herb	DH, WW
	PORTULACACEAE	<i>Calandrinia</i>	<i>menziesii</i>	Y	annual herb	ME, SG
<i>Calyptridium</i>		<i>monandrum</i>	Y	annual herb	DI, DH, WW, MI	
PTERIDACEAE	<i>Pentagramma</i>	<i>triangularis</i>	Y	fern	DHS	
RHAMNACEAE	<i>Ziziphus</i>	<i>paryii</i>	Y	shrub	SG	
ROSACEAE	<i>Adenostoma</i>	<i>fasciculatum</i>	Y	shrub	SG	
RUTACEAE	<i>Thamnosma</i>	<i>montana</i>	Y	shrub	WW	
SCROPHULARIACEAE	<i>Keckiella</i>	<i>antirrhinoides</i>	Y	shrub	SG	
SELAGINELLACEAE	<i>Selaginella</i>	<i>bigelovii</i>	E	fern	WW, MI, SG	
SOLANACEAE	<i>Datura</i>	<i>wrightii</i>	Y	peren. herb	SG	
	<i>Lycium</i>	<i>andersonii</i>	Y	shrub	ME	
ZYGOPHYLLACEAE	<i>Larrea</i>	<i>tridentata</i>	Y	shrub	DI, MV, PH, I-10, DH, WW, MI	
	<i>Tribulus</i>	<i>terrestris</i>	N	annual herb	SG	

¹ Endemic to California, United States and Baja California, Mexico (E); native to the United States (Y); non-native species introduced from outside of the United States (N)

² Sites with (DI, MV, PH, and ME) and without (I-10, DH, WW, MI, SG) wind farms; for area descriptions, see Figure 1.

Table 6. Results of analysis of variance (ANOVA) models of variation in taxa scores on NMS Ordination Axis 1 (variation between turbine and non-turbine sites) explained by affinity to predator and plant community functional groups.

	Grouping ¹	Functional level	Coefficient	Standard Error	P-value ²
Plants	Status $F_{(2,116)} = 2.79$ $P = 0.07$	native	-0.145	0.093	0.121
		non-native	-0.294	0.124	0.020
		endemic	0.289	0.086	0.001
	Habit $F_{(4,112)} = 1.02$ $P = 0.40$	annual herb	-0.151	0.169	0.374
		annual grass	0.252	0.164	0.128
		perennial herb/grass	0.010	0.181	0.957
		shrub	-0.127	0.177	0.476
	succulent	-0.082	0.232	0.724	
Predators	Guild $F_{(4,27)} = 0.84$ $P = 0.51$	snake	-0.069	0.292	0.814
		lizard	-0.108	0.311	0.730
		bird	0.065	0.174	0.712
		mammal	0.268	0.311	0.396
		invertebrate	-0.386	0.311	0.225
	Diet $F_{(3,28)} = 0.36$ $P = 0.78$	insectivore	-0.174	0.257	0.504
		omnivore	0.082	0.290	0.780
		carnivore	0.080	0.186	0.670
	Zone $F_{(1,30)} = 0.12$ $P = 0.73$	carnivore/insectivore	-0.174	0.348	0.620
		aerial	0.065	0.174	0.713
		terrestrial	-0.074	0.215	0.734
	Scavenging $F_{(3,28)} = 2.73$ $P = 0.06$	none	0.102	0.220	0.648
		low	-0.120	0.132	0.372
moderate		0.833	0.295	0.009	
	high	0.035	0.332	0.916	

¹ Plants were grouped by status, and plant habit; predators were grouped by taxonomic guild, by main diet type, by habitat zone, and by propensity for scavenging

² Bold indicates significant ($P < 0.05$) effect of functional group category

Table 7. Linear correlations between NMS Ordination Axis 1 scores describing differences between turbine and non-turbine sites, and predator abundances estimated as the number of encounters per hour of search effort (summer, 2014) at each of the nine sites within the study area near Palm Springs, CA.

Taxa ¹	Direction ²	R^2	Taxa (Cont.)	Direction	R^2
Red-tailed Hawk	+	0.572	Cactus Wren	-	0.082
Loggerhead Shrike	+	0.499	Northern Mockingbird	+	0.072
Golden Eagle	+	0.394	Red Diamond Rattlesnake	+	0.060
Great Basin Fence Lizard	+	0.394	Greater Roadrunner	-	0.052
Southwestern Speckled Rattlesnake	+	0.394	Desert Spiny Lizard	+	0.049
Gray Fox	+	0.394	California Kingsnake	+	0.046
Prairie Falcon	-	0.384	Common Raven	-	0.040
Colorado Desert Sidewinder	-	0.345	Spotted Leaf-nosed Snake	-	0.035
Bobcat	+	0.280	American Badger	+	0.033
Wolf Spider	+	0.249	American Kestrel	+	0.029
Great Basin Collared Lizard	-	0.211	Western Zebra-tailed Lizard	+	0.017
Western Black Widow	-	0.211	Sun Spider	-	0.014
Rock Wren	+	0.202	Burrowing Owl	+	0.013
Giant Scorpion	-	0.189	Red Racer	-	0.008
Coyote	+	0.177	Domestic Dog	+	0.005
Great Basin Whiptail	-	0.120	Tarantula	+	0.004

¹ Bold indicated strong correlation of greater than 0.40

² Positive correlation (+) indicates higher encounter rates at high values of Axis 1 (non-turbine sites; quiet, low disturbance); negative correlation (-) indicates higher encounter rates at low values of Axis 1 (turbine sites; loud and high disturbance)

Table 8. Plant species with substantial change in probability of presence or absence along Axis 1 (treatment axis) in NMS Ordination model, from richness data collected (2015) from nine study sites near Palm Springs, CA. Only species that occurred more than once (and not at all sites) are shown.

Species ¹	Direction ³	R^2 ⁴	Species	(Cont.) Dir	R^2	Species	(Cont.) Dir	R^2
BaPar	+	0.77	CrMic	-	0.16	Lactu	-	0.05
OnPil²	-	0.77	LoSqu	-	0.16	EpCal	+	0.05
EuCal	+	0.60	Amlnt	+	0.14	ErInf	+	0.05
Silri	-	0.60	ArCal	+	0.14	BrMad	+	0.04
EuChr	+	0.56	ChBre	-	0.14	AcStr	+	0.03
PhMin	+	0.56	DiCap	+	0.14	LeLas	-	0.03
StPau	-	0.56	EuPol	-	0.14	ErFas	-	0.03
SeBig	+	0.44	EcEng	+	0.10	AmDum	-	0.02
MiLae	+	0.43	HiRig	+	0.10	BeJun	-	0.02
SaCol	+	0.43	EsMin	+	0.06	CaMen	+	0.02
StExi ²	-	0.39	ErWal	+	0.06	CaPal	-	0.02
EmPen	+	0.37	PtDry	+	0.06	LaTri	-	0.02
PhDis	+	0.31	StLin	+	0.06	LuBic	+	0.02
GiSte	+	0.30	KrBic	+	0.06	UrLin	+	0.02
SeGre	+	0.30	CaMon	+	0.06	LaCor	-	0.02
EnFar	+	0.27	AcSph	-	0.06	AmSal	+	0.02
MaGla ²	-	0.26	CyRam	-	0.06	FeCyl	+	0.00
CrCon	-	0.23	EuMic	-	0.06	TrGra	-	0.00
CrAng	-	0.18	DiNeo	-	0.05	LuSpa	+	0.00
ErThu	-	0.17	Rafin	+	0.05	PsSch	-	0.00
SoOle	-	0.16	CrMar	+	0.05			

¹ Species are abbreviated by the first two letters of the genus and first three letters of the species (if no species assigned, then first five genus letters); full names are given in Table 5. Bold indicated strong correlation of greater than 0.40

² Species responses could not be estimated because of linear separation; separation was resolved by swapping the order of presences and absences at the point of separation which results in higher residual deviances and correspondingly lower pseudo- R^2 values

³ Positive correlation (+) indicates that probability of presence increases at high values of Axis 1 (non-turbine sites; quiet, low disturbance); negative correlation (-) indicates that probability of presence increase at low values of Axis 1 (turbine sites; loud and high disturbance)

⁴ R^2 value is McFadden's Pseudo- R^2 statistic, calculated as $1 - \frac{\log(\text{model likelihood})}{\log(\text{intercept only model})}$

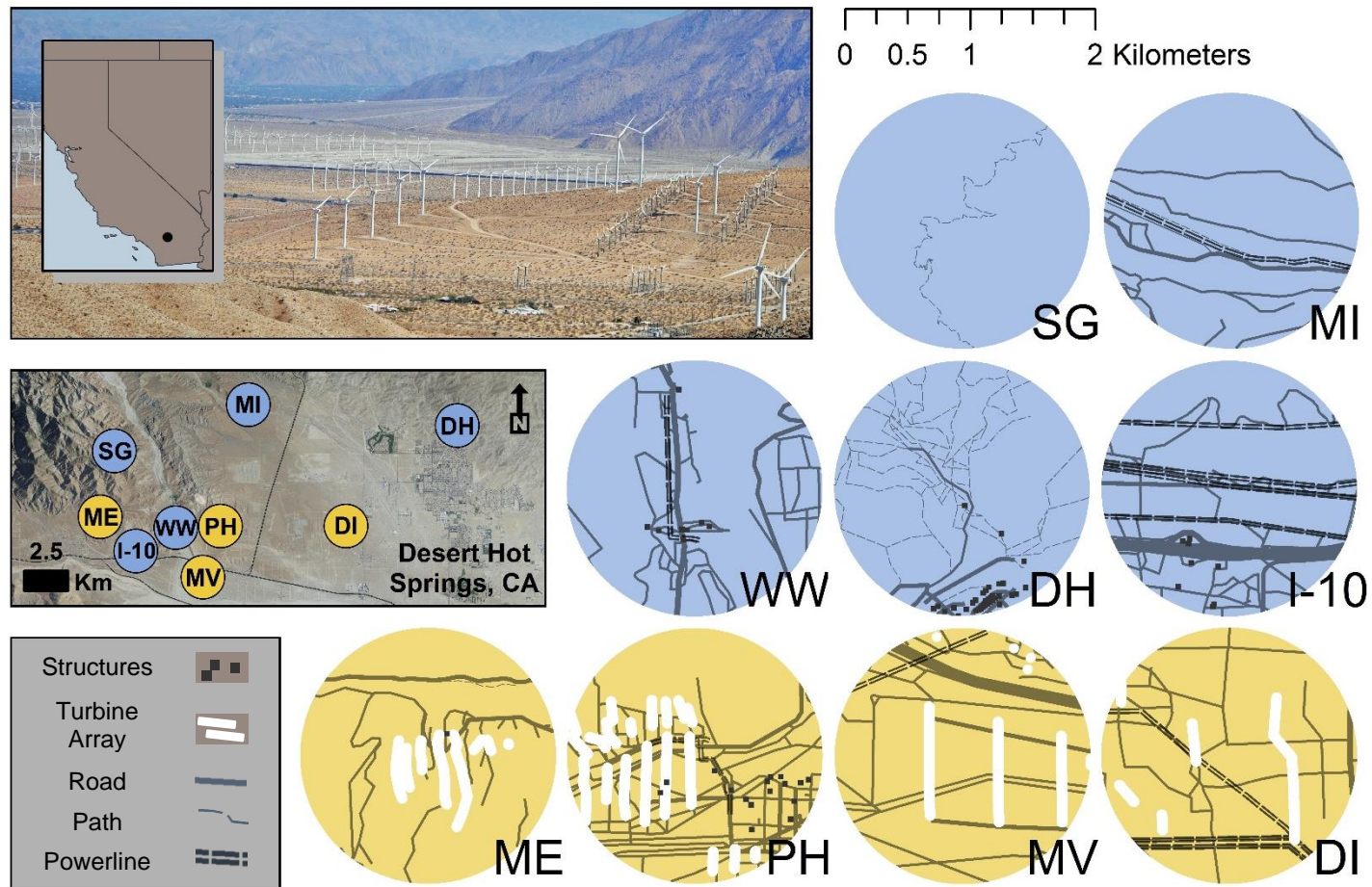


Figure 1. Study sites were located in the San Geronio Pass Wind Resource Area (SGWRA) in southern California, north of Palm Springs. Insets show the land-use history of each study area (five study areas without turbines in blue; four sites with turbines in orange) within a 1 km radius. Turbines (white), roads (grey lines; thickness reflects road traffic volume), paths (dashed line), transmission lines (double-dashed line), and buildings (squares) indicate the types of human disturbances present at each study region.

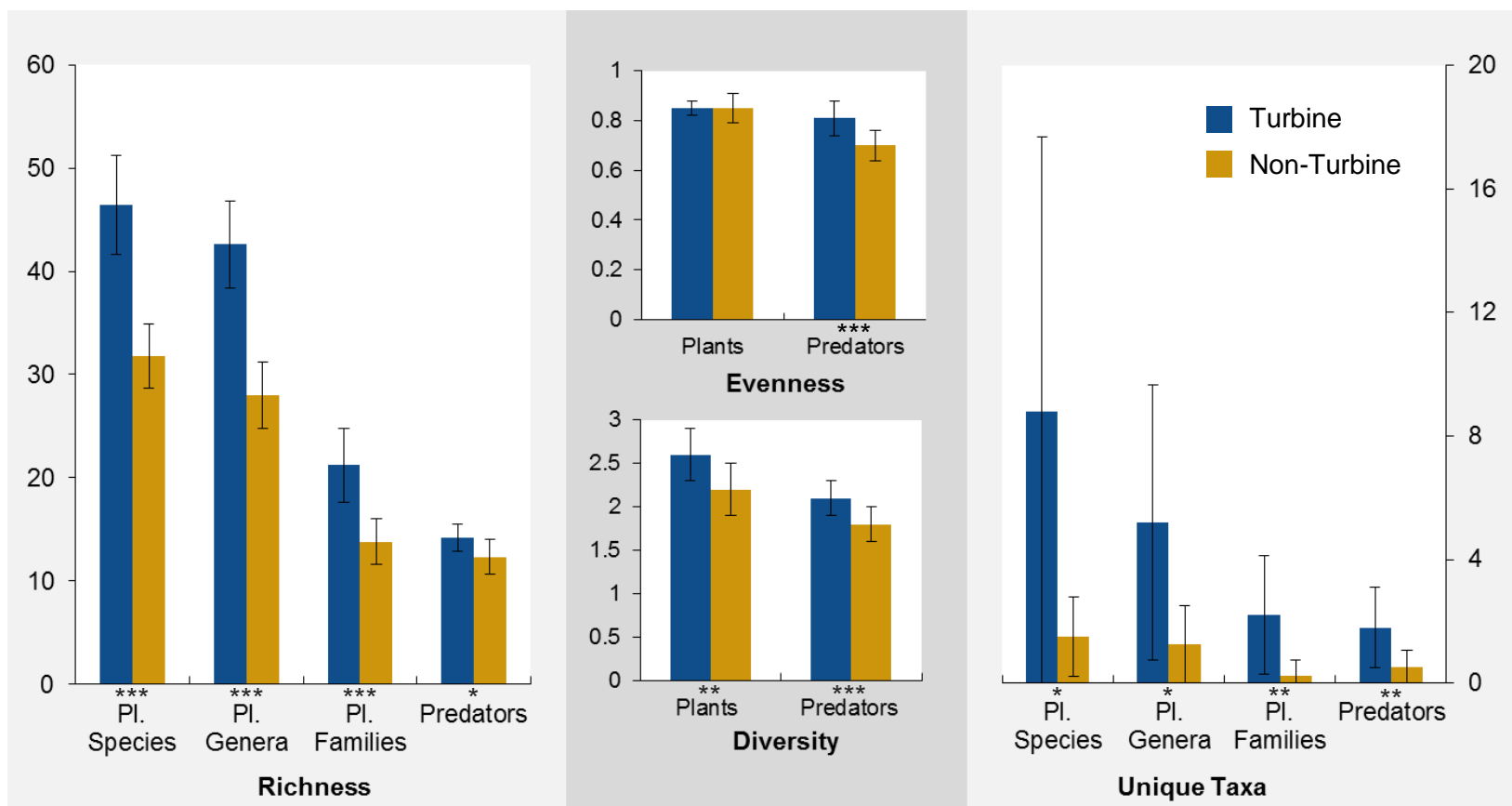


Figure 2. Total richness (left), number of unique species (right), evenness (top middle), and diversity (bottom middle) of plant and predator communities surveyed in 2014 and 2015 from sites without (blue, $n = 5$) and with (orange, $n = 4$) wind turbines in study areas near Palm Springs, CA. Bars represent the average value, with standard deviation error bars indicating variation among sites. Significance of differences (T -test) between turbine and non-turbine sites is indicated by symbols heading x-axis labels: P -values (***) < 0.05 ; (**) 0.06 – 0.10 ; (*) 0.11 – 0.15 . Diversity and evenness measures for plants represent the distribution of plant species within plant families, rather than abundances of individual species (thus, these measure are not independent data).

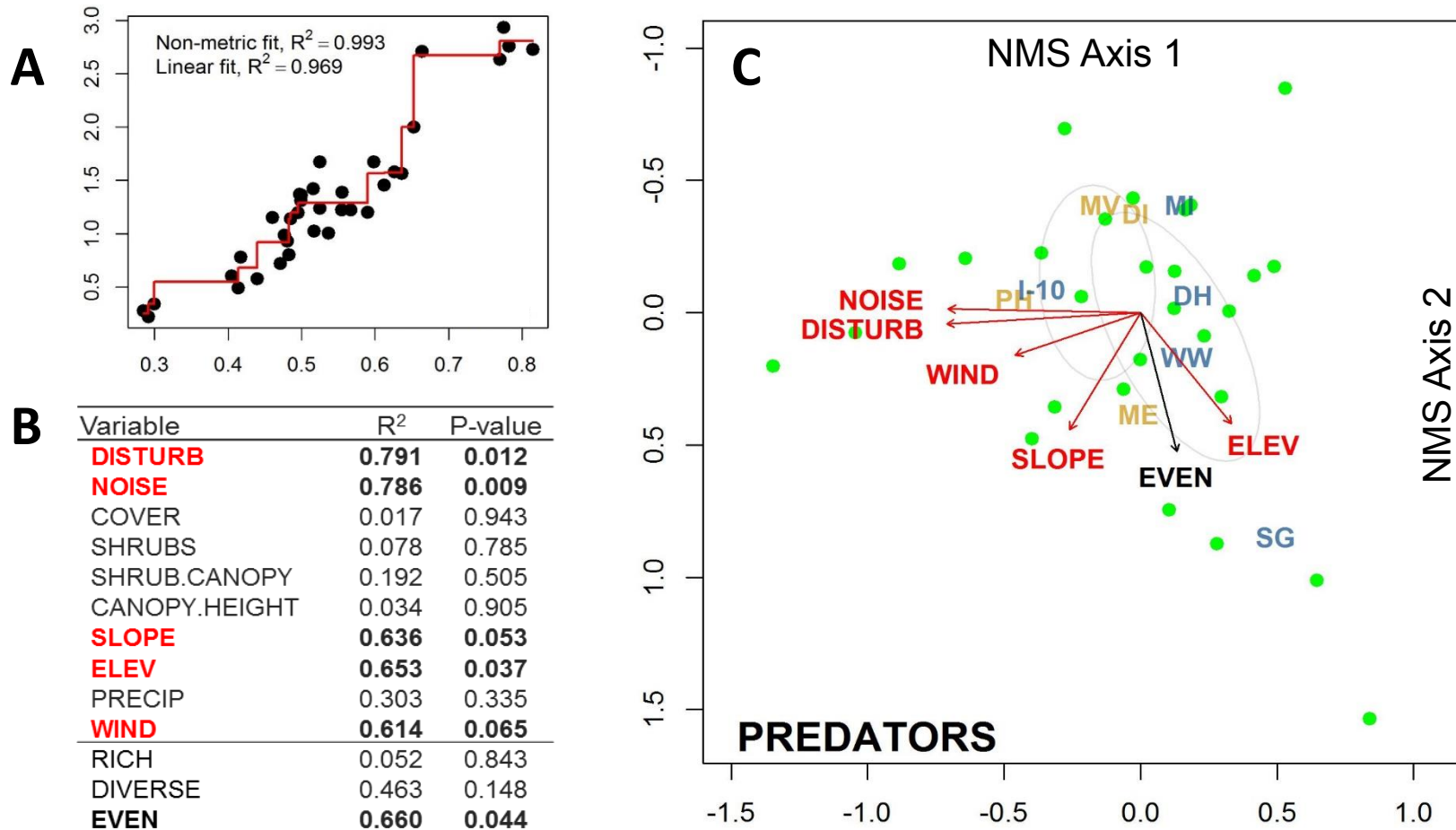


Figure 3. (A) Fit statistics for NMS ordination of 2014 predator encounter rate surveys at nine study sites near Palm Springs, CA. (B) Correlation between site variables, diversity measures, and NMS solution ($P < 0.10$ in bold; for variable descriptions, see Table 1). (C) Final NMS solution for predatory taxa (green circles) at turbine (orange text) and non-turbine sites (blue text) Plotted vectors are scaled relative to the maximum variance explained (DISTURB).

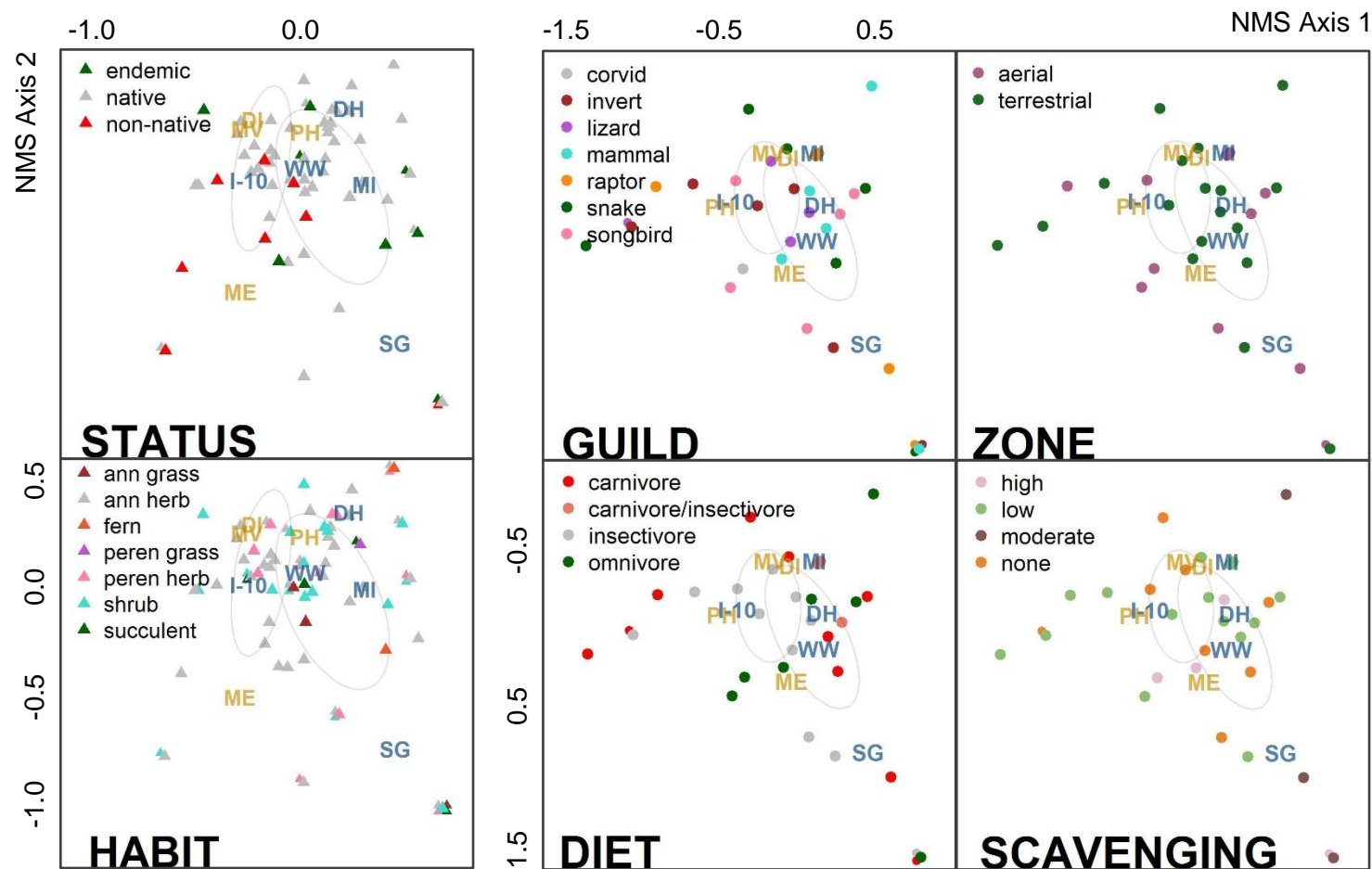


Figure 4. Correlations between plant (left panel) and predator (middle and right panels) functional groups, and NMS ordination for nine communities surveyed (2014–2015) in study areas near Palm Springs, CA. Each taxa was assigned to a functional category (see panel legends) using primary literature records. Species levels that cluster at high or low values of Axis 1 (treatment effect) indicate association between functional groups and turbine presence or absence. Clustering was tested quantitatively using analysis of variance, see Table 6.

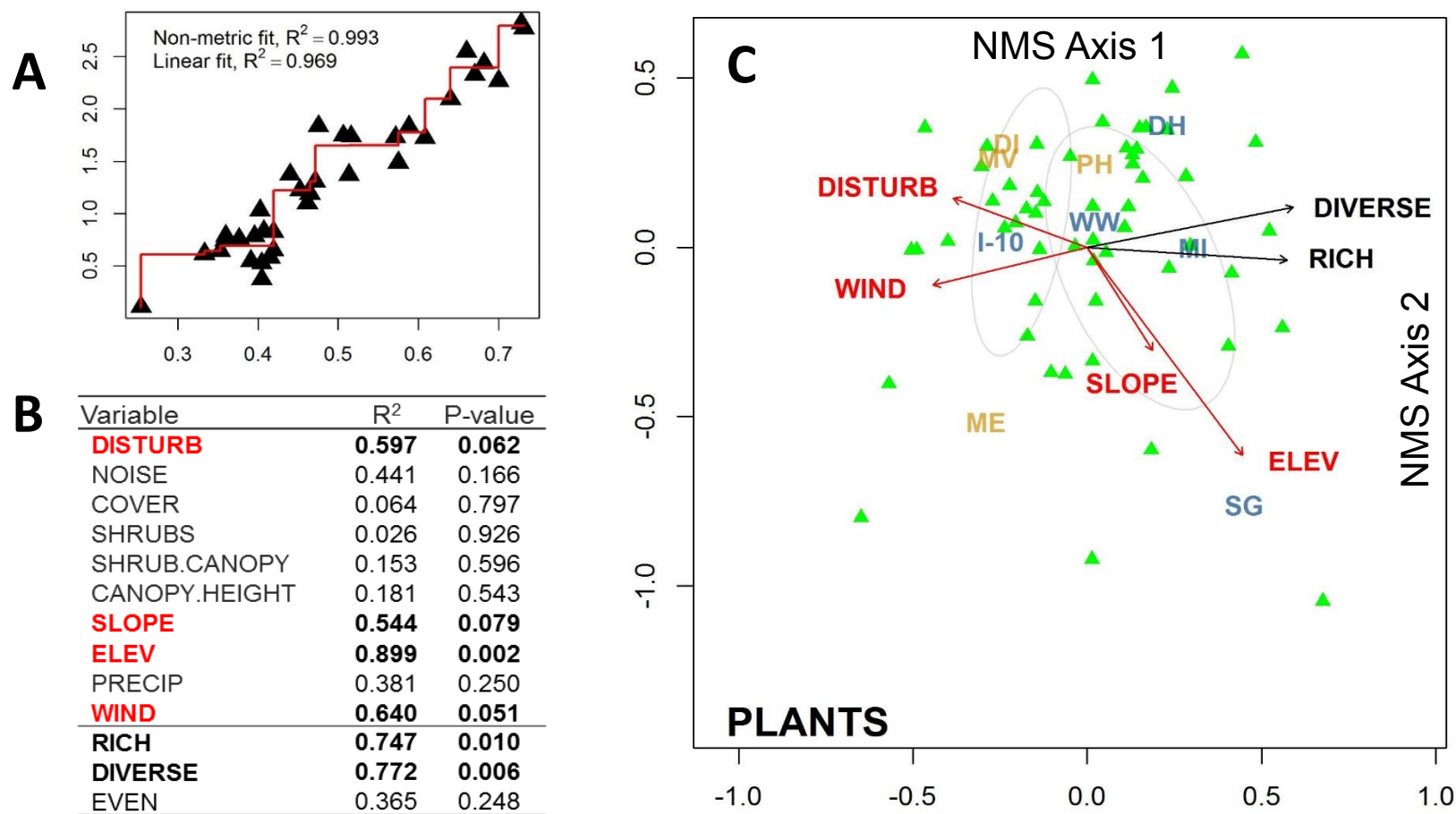


Figure 5. (A) Fit statistics for NMS ordination of 2015 plant richness surveys at nine study sites near Palm Springs, CA. (B) Correlation between site characteristics, diversity measures, and NMS solution ($P < 0.10$ in bold; for variable descriptions, see Table 1). (C) Final NMS solution for plant taxa (green triangles) at turbine (orange text) and non-turbine sites (blue text). Plotted vectors are scaled relative to the maximum variance explained (ELEV).

Effects of Wind Farms on Population Demography of Side-blotched Lizards

ABSTRACT

Fossil fuel consumption (a major driver of global climate change) can be greatly reduced by increasing renewable energy production and use. However, renewable energy infrastructure alters existing land-use strategies and land-cover characteristics. For wildlife populations, changes in local biotic and abiotic environments due to energy development may reduce the quality of remaining habitat patches in surrounding landscapes, or within the industrial-habitat matrix of wind farm project areas. Some species may not be able to persist in these areas because of declines in habitat suitability. In this study, I assessed whether land-use changes at wind farms in the San Geronio Wind Resource Area near Palm Springs (California) affected terrestrial wildlife populations by comparing key demographic parameters between populations at wind farms ($n = 4$) and at less-disturbed reference areas ($n = 5$). From 2013 to 2015, I used mark-recapture methods to test for differences in demography and survival between populations of a common reptile, the side-blotched lizard (*Uta stansburiana*). I found similar survival rate, population size, minimum and maximum age (measured as body size), and body condition of lizards at wind farms, relative to reference populations. Most of the variation in survival was explained by temporal effects; variation between populations was minor. These results suggest that wind farms do not substantially affect the suitability of habitat for side-blotched lizards. Further research is needed to address the responses of organisms with lower tolerance of habitat disturbance. These results can be used by wildlife managers to more accurately assess how wind farms affect understudied terrestrial vertebrates such as reptiles.

INTRODUCTION

Fossil fuel use is linked with climate change, an unintended consequence of developed human societies that stands to significantly alter global patterns of biodiversity (Vitousek 1994).

Slowing the rate of extinction from climate change will require substantial financial and political investment in renewable energy production (Bilgen et al. 2004). However, the sustainability of green energy technologies, in comparison with carbon-based energy sources, is affected by how much biodiversity is lost during facility construction and operation. To date, there have been few studies addressing the extent of biodiversity loss at land-based renewable energy facilities (Jones et al. 2015, but see Santos et al. 2010). Thus, it is often unclear and difficult to predict how wildlife will be affected by renewable energy development.

In the case of renewable wind energy, this lack of research is particularly problematic. Wind turbine spacing requirements demand large tracts of land to accommodate energy production needs; consequently, the environmental consequences from land-use change at wind farms may be much greater than those associated with other alternative technologies such as solar or geothermal energy (Evans et al. 2009). However, wind energy developments are unique among renewable energy technologies in that the physical footprint of a wind farm (1 ± 0.7 hectare for each MW of installed capacity) may be quite small (Hand et al. 2009). Most of the physical footprint of wind energy developments is from road infrastructure to access wind turbines (Hand et al. 2009); remaining land can be left in a natural state. Hence, if organisms can tolerate wind turbines and associated operation activities (and if species can maintain viable populations within the turbine-habitat matrix), then wind energy may present a sustainable approach for renewable energy production with minimal wildlife habitat loss (Evans et al. 2009).

Studies that address environmental consequences of wind energy for wildlife focus primarily on direct mortality risk for birds and bats (Kuvlesky et al. 2007, Cryan and Barclay 2009). However, the potential for behavioral avoidance and loss of habitat at wind farm sites may result in greater biodiversity loss when compared to the effects of turbine-induced mortality alone (Drewitt and Langstron 2006, Pearce-Higgins et al. 2009). Habitat loss may exclude important native species while benefiting a number of urban-tolerant organisms—an effect with potentially dire consequences including altered patterns of ecosystem structure and function (Gordon 1998, Finke and Denno 2004, Hansen et al. 2005). Researchers have begun to address whether fitness is reduced for some birds and mammals inhabiting wind energy developments (Rabin et al. 2006, Helldin et al. 2012, Lovich and Ennen 2013). However, information is sorely lacking for many other organisms. Only a handful of studies have been published regarding impacts to plants, invertebrates, or reptiles.

Here, I begin to address this research gap with a focused study of population level responses of reptiles to wind farm developments within California's San Geronio Wind Resource Area [SGWRA]. By addressing whether these facilities still support viable reptile populations, this research can aid efforts to identify and mitigate the environmental consequences of wind farms. Reptiles are key components of desert food webs, providing energy for many endothermic mesopredators (Ayal 2007). Reptiles are also important predators of invertebrates and rodents, and some species may act as ecological keystones (Spiller and Schoener 1990, Polis 1991, Vitt 1991, Brown et al. 1994). Thus, an absence or exclusion of reptiles from wind energy sites may affect species interactions, community composition and diversity, and ecosystem function. To date, responses of reptiles to wind farms have only been documented in a single species of specialist herbivore (the desert tortoise, *Gopherus agassizii*).

Researchers found differences in habitat use as well as novel causes of mortality (Lovich and Daniels 2000, Lovich et al. 2011). However, as a long-lived, k-selected species, the desert tortoise may not respond to wind energy development in a manner that is representative of other native reptile species.

To assess the suitability of wind farms as habitat for common reptile species, I estimated daily survival rate, population sizes, capture probabilities, age structure (using minimum and maximum body size information), and body condition for populations of the common side-blotched lizard, *Uta stansburiana*. Side-blotched lizards are ideal focal organisms for population studies because they maintain large populations across small areas, are present in high densities, and often yield high recapture rates making it easy to follow individuals over time (e.g., Tanner 1972). In addition, the side-blotched lizard is arguably the most demographically well-known lizard of North American deserts, providing useful context for understanding selective pressures that define population persistence (Tinkle 1967a, Fox 1978, Turner et al. 1982, Wilson and Cooke 2004). With a biomass of roughly 84 grams per acre, this species provides a quick meal to a diverse community of predators from loggerhead shrikes and roadrunners, to leopard lizards, rattlesnakes, and even scorpions (Tinkle et al. 1962, Tinkle 1967a, Parker and Pianka 1975, Wilson 1991); researchers acknowledge the importance of this species in desert food webs across the western United States (Tinkle 1967a, Parker 1974, Wilson 1991).

Survival rate and population size can reflect the ability of a species to persist; in disturbed habitats, altered patterns of predation pressure or resource abundance can influence these parameters (Johnson et al. 1975, Polis et al. 1998, Gosselink et al. 2007). For example, in poor quality habitats with substantial invasive plant cover, lizards may encounter fewer prey items, with greater costs of movement (Pearson 2009, Rieder et al. 2010). These factors increase

mortality risk, either by reducing energy reserves, or by increasing exposure to predators (Nussbaum and Diller 1976, Fox et al. 1981, Zani 2005). A loss of suitable habitat can result in smaller population sizes if carrying capacity of a given environment is reduced (Thomas et al. 2001).

Here, I hypothesize that wind farm developments reduce the quality of wildlife habitat through changes in predator and plant community characteristics. In Chapter 1, I identified a substantial change in plant community composition with increased abundance of non-native plants, and fewer endemic and rare species at wind farm sites. If increased non-native plant abundance and diversity reduces quality of lizard habitat at wind farms, then I predict that mortality will be higher, with lizards of younger average age (smaller size) and poorer body condition, ultimately resulting in smaller populations when compared to natural sites. Chapter 1 also identified changes to predator community diversity, with decreased abundance of red-tail hawks (*Buteo jamaicensis*) and loggerhead shrikes (*Lanius ludovicianus*). If wind farms cause a decrease in the diversity and abundance of lizard predators such as shrikes, then lizards at wind farms will be (on average) older and in better body condition with larger population sizes and higher survival rates. To test these alternative predictions, I track the fates of individuals and populations using a mark-recapture study design to monitor plots located at sites within wind farms, relative to reference sites without wind energy development.

STUDY AREA

I established study plots (60 x 60 m) across a gradient of habitats within the SGWRA (Figure 1). I selected 4 study populations at wind farms (sites ME, PH, DI, and MV). Sites ME and PH were developed in the mid-1980s; these sites have high densities of smaller turbines with lattice tower

bases. Sites DI and MV were developed in the mid-2000s; these sites have tall, tubular turbines at lower densities (turbine site characteristics are summarized in Table 1).

I chose 5 sites without wind turbines as reference study populations (sites MI, SG, WW, DH, and I-10). Sites MI and SG were low-disturbance areas at least 2.4 km away from any paved road disturbances, with limited human activity (however, site SG experienced substantial erosion and soil disturbance over the course of the study from illegal grazing). Sites WW and DH were moderately disturbed areas, with nearby paved roads used frequently for access to recreation opportunities (WW), and substantial human activity such as hiking and dog walking (DH). Finally, site I-10 was highly disturbed, 420 m north of a rest stop on a major interstate, the I-10. There was substantial traffic noise, off-road vehicle use, human activity, and multiple transmission line corridors at this site.

METHODS

Mark-recapture Field Surveys

For each of the 9 populations, I tracked the fates of adult and sub-adult lizards (>35 mm snout-to-vent length [SVL]) within study plots from July 2013 to June 2015. I visited each site on 10 primary trapping sessions within 5 seasonal periods (overwinter, late spring, early summer, mid-summer, and late summer, Figure 2). The length of each primary session ranged from 7 to 12 days. Within each primary session, I visited each of the 9 sites twice on consecutive days (“secondary occasions”); except during the late spring survey period which consisted of 1 rather than 2 secondary occasions. Site PH was added in late spring of 2014 (surveyed for 7 primary trapping sessions).

I surveyed for lizards during morning or evening activity periods, except on cool or cloudy weather days when lizards were active all day. I recorded the following at both the start and the end of each survey: time of day, wind speed, cloud cover, substrate temperature (in the open, under cover, and on basking rocks), air temperature, relative humidity, and whether turbines on site were active. I avoided surveying on days that were unsuitable for lizard activity (i.e., rainy, high winds, or cold temperatures). During each survey, 1 to 4 observers walked at random through the plot area; observers caught any unmarked adult or sub-adult lizards within the survey plot by hand or with a noose. Observers continued to catch lizards from the time of lizard emergence, until lizards retreated either at dusk or during mid-day high temperatures. On average, surveys took 2.92 ($SD = 1.04$) hours to complete. I recorded locations for all marked lizards encountered and prior to analyses I used GPS locations and scale patterns of lizards from field photos (when available) to verify recaptures across years. I used maps of lizard capture localities to verify that all individuals encountered were within the survey area.

For each newly-captured lizard, I recorded mass, total length [TL], and SVL. I determined sex, palped females for yolked follicles, identified whether the lizard had previously lost a tail, noted any additional injuries, and counted the number of external parasites. In 2013, I individually marked lizards by removing combinations of up to 3 toes per lizard to create unique permanent identifiers. I applied temporary numbers on the dorsum with paint markers to allow for re-sighting of lizards without recapture. I found that recapture rates were low using this method; natural toe-loss rates were high and previously marked lizards were difficult to recapture when paint marks were lost after shedding. Beginning in the early summer of 2014, I instead marked lizards using a unique combination of 3 plastic beads secured with surgical steel to the base of the tail posterior to the hemipenes (Fisher and Muth 1989). This increased

encounter rates, and eliminated the need to handle lizards more than once. All capture and handling procedures were approved by the Institutional Animal Care and Use Committee at the University of Nevada, Reno (Protocol 00586).

Analyses

I compared lizard age and body condition between wind farms and natural reference areas using a linear mixed-effect model with a random intercept for study sites (*lme4* package version 1.1–7, implemented in program R Version 3.1.2, www.r-project.org, accessed 31 Oct 2015). I modeled age as a function of body size (SVL), allowing for fixed effects of years (2013, 2014, and 2015), seasons (overwinter, late spring, early summer, mid-summer, and late summer), and sex. I then evaluated whether wind farms affected age and condition by estimating the significance (using *p*-values generated in *lmerTest* version 2.0–29, which estimated degrees of freedom using Satterthwaite's approximation) of a unique intercept for sites with wind turbines. I used the same approach to model lizard body condition for males and females independently. Condition was modeled as a function of lizard weight in grams, after correction for body size (TL), allowing for variation across years and seasons.

I estimated survival rates using a robust-design framework in Program MARK version 8.0 (White and Burnham 1999). Prior to analyses, I removed capture records of lizards with significant injuries or deaths resulting from capture or processing ($n = 2$). Estimates of population size were derived from a conditional likelihood using the Huggins' estimator, which also allowed for individual covariates in models of survival (S), capture (p), and recapture (c) (Huggins 1991). Robust design is a valuable analysis framework because it allows for reduced parameter bias by combining modeling technique for both open (survival and emigration estimated between primary sessions) and closed (encounter rates estimated between secondary

occasions) populations (Kendall et al. 1997). Probability of being unavailable for capture (γ' , γ'') was not estimated in survival models (parameters fixed to zero). Under the fixed emigration model formulation, the survival parameter is confounded between probability of surviving, and probability of being present for capture. *Uta stansburiana* has limited dispersal, with migration rates estimated at 4–16% (Tinkle 1967b); 90% of the time, lizards are found less than 24 m from the center of their territory (Spoecker 1967). Thus, it is unlikely that a fixed emigration model introduced substantial bias in estimates of survival.

Model Structures

I selected the most competitive model substructures using sample-size corrected Akaike Information Criterion values ([AICc], Burnham and Anderson 2002). Preliminary analyses indicated that models with unique values of survival for each period were too complex given the data; therefore, I constrained time trends to be an effect of year (2013, 2014, 2015) and of season (overwinter, late spring, early summer, mid-summer, and late summer). Before testing models with standardized covariate effects, I determined the most competitive substructure of survival between all possible additive models with unique values per site (site), year (t_A), and season (t_S), against the null model of no variation (.). I then used the top survival structure to model p and c with variation by site (s), with unique values per primary period (t), or as a linear trend across periods (t_L). I compared each of these structures using either the null model (.) of $p = c$, or the intercept model (I) of $p = c$ for time and group effects, but with unique intercepts for each parameter.

I then used this top-ranking model to compare models with covariate or group effects on p , c , and S . I created 3 model subsets comprised of the null (no covariate) model, and all possible combinations of 1 or 2 variables within each covariate category: (1) survey conditions for p and

c , (2) individual covariates for S , and (3) site or group variation for S . For a summary of model structures, see Table 2. Covariates within each subset that attained a cumulative model weight $[\sum w_i]$ of ≥ 0.5 were used to build a predictive model for population-level inferences (Barbieri and Berger 2004). Model weights indicate likelihood of a given model; when a balanced set of models is used, the cumulative weight for a covariate is calculated as the sum of individual AICc weights for all models containing that variable (Burnham and Anderson 2002, Doherty et al. 2012).

Encounter probability model covariates.— For p and c , I allowed for variation in encounter rates as a function of survey conditions during mark-recapture surveys (for a summary of survey conditions, see Table 3). I included wind speed, which may reduce surface activity of reptiles by unfavorably increasing the rate of heat loss (Stevenson 1985). I also included substrate temperature, which can limit ectotherm surface activity (Huey 1982). I distinguished between evening surveys, relative to morning surveys, as evening surveys may be characterized by warmer and less-variable substrate temperatures, potentially suppressing lizard activity (Grant 1990). I included an effect of humidity, as low humidity can increase evaporative water loss, and may also affect insect prey activity levels (Norris 1953, Tanner 1972, Holm and Edney 1973, Jaworski and Hilszczański 2013). Lastly, I allowed for linear change in p and c across levels of cloud cover (none, low, moderate, or high) to account for differences in activity resulting from weather.

Survival models for individual variation covariates.— I modeled survival as a function of individual variation in age, as smaller adult lizards (less than reproductive size of 46 (male) or 42 (female) mm SVL) often experience higher mortality rates (Tanner and Jorgensen 1963, Zani 2008). I allowed for variation in survival between sexes, as male *U. stansburiana* may have

reduced survival rates under some conditions (Tinkle 1967a, Wilson and Cooke 2004). I considered the effect of stress on survival as a function of recent tail loss. Lizards with regenerated tails incur a substantial social and physical cost that may increase mortality risk (Bateman and Fleming 2009). I allowed for an effect of poor body condition on survival probability by quantifying condition as the residual body mass after correcting for size, assuming a linear correlation, with male and female lizards as unique groups. Low mass relative to body size may indicate low fat reserves for reproduction, high stress, or high parasite loads (Hahn and Tinkle 1965, Wilson 1991, Moore et al. 2000, Klukowski and Nelson 2001). I did not correct for the influence of female reproductive condition on body condition because I lacked robust data regarding female gravidity. Nor did I account for variation in survival across lizard throat color phenotypes (Sinervo and Lively 1996), as this characteristic could not be reliably assessed in the field.

Survival models for site-specific covariates and group structure.— Next, I considered whether survival differed by location. I compared a null model with no site variation, a model allowing unique values of survival at each site, models with variation across site habitat and climate characteristics, and models with variation as a function of anthropogenic disturbance (for a summary of these variables by site, see Table 4). For habitat and climate, I considered whether survival variation reflected differences in site elevation, as a variable that may indicate differences in plant community composition, presence of key predators or competitors, or patterns of climate (Turner et al. 1973, Nussbaum and Diller 1976). Second, I estimated total winter precipitation from data retrieved from PRISM climate models (<http://www.prism.oregonstate.edu/>). Winter precipitation may increase annual plant growth, resulting in more abundant invertebrate prey, which in turn may influence survival probability by

affecting growth rate and reproductive output of lizards in the following spring (Turner et al. 1973, 1974; Worthington 1982). Third, I estimated average bare-ground cover using field data collected in July of 2014 at 100 m point-line-intercept plant transect surveys (3 per site) (Herrick et al. 2005). I calculated bare-ground cover as the proportion of random points at each meter interval per transect that did not intersect a plant canopy. In deserts, total plant cover is often correlated with habitat quality (i.e., resource abundance, or vulnerability to predators) (Ayal 2007).

I generated models with an effect of anthropogenic disturbance on survival using 3 alternative variable structures: (1) groups of sites with and without wind turbines; (2) a gradient of anthropogenic noise disturbance, measured as an average A-weighted decibel reading [dB_A] from a random sample of 10 readings taken at plots in June and July (2015) when turbines were running and wind was ≤ 3.5 mph; and (3) a gradient of human land-use disturbance, estimated as the total linear road footprint (digitized in ArcMap from 1 m resolution California NAIP imagery) within 1000 meters of the established study plots. To account for differences in disturbance with road size, I allowed for a weighted effect of 0.5 for footpaths, 1.0 for dirt roads, 3.0 for residential roads, 5.0 for secondary roads, and 10.0 for primary highways (disturbance effect may increase with traffic volume and road width, see Vos and Chardon 1998).

RESULTS

Turbine sites were windier ($t_{139,4} = 2.55$, $P = 0.012$) relative to non-turbine sites; otherwise, survey conditions were similar between groups (Table 3). Over 3 years (10 capture periods), I encountered 394 lizards ($n = 25$ –88 per site), 1172 times. On average, I observed 8.4 lizards at each site per secondary occasion (site \bar{x} range: 4–16). Mixed-effects models of body condition

and age (accounting for variation over time and between males and females) did not support an effect of turbine presence or absence on body condition (female: $t_{7.0} = 0.01$, $P = 0.99$; male: $t_{6.1} = -1.12$, $P = 0.31$) or age and size (both sexes: $t_{6.7} = -0.84$, $P = 0.43$); thus, I found no evidence that wind farms affected lizard condition (as length-corrected mass) or age (as SVL) (for a summary of demographic characteristics by population, see Table 5).

I extracted derived population size estimates and model beta parameter estimates from the predictive model ($\Delta\text{AICc} = 2.34$) with important covariate effects only. Model estimates of capture and recapture probabilities are shown in Figure 3. The model supported a linear increase in encounter probability over time ($\beta = 0.13$, $\text{SE} = 0.02$), with an intercept allowing higher recapture rates relative to capture rates ($\beta = 0.59$, $\text{SE} = 0.12$). The model included a negative effect of windy survey conditions on encounter probabilities ($\beta = -0.25$, $\text{SE} = 0.06$, $\sum w_i = 0.99$). The model did not support differences in encounter rates for lizards at turbine relative to non-turbine sites. Nor did population size estimates for 2013–2015 differ between populations at turbine and non-turbine sites (Figure 4).

Confidence intervals for survival parameter beta estimates in the predictive model did not substantially overlap zero (except in survival difference between years 2014 and 2015, Figure 5). Models of survival indicated increased survival for adult lizards, relative to sub-adults ($\beta = -0.46$, $\text{SE} = 0.10$, $\sum w_i = 0.86$) (Figure 6A). Survival varied across season relative to the spring capture period (early summer: $\beta = 1.11$, $\text{SE} = 0.42$; mid-summer $\beta = 0.66$, $\text{SE} = 0.35$; late summer: $\beta = 2.02$, $\text{SE} = 1.05$; overwinter: $\beta = 1.86$, $\text{SE} = 0.30$) and year relative to conditions in 2015 (2013: $\beta = -1.08$, $\text{SE} = 0.32$; 2014: $\beta = -0.19$, $\text{SE} = 0.22$) (Figure 6B). Models allowing unique survival probabilities for each site did not estimate; nor was there support for differences

n survival across noise or disturbance gradients. However, I found support for higher survival at sites with substantial bare ground cover ($\beta = 0.14$, $SE = 0.07$, $\sum w_i = 0.62$) (Figure 6B).

The top model (by AICc rank) outperformed the remaining 28 candidate models, including the predictive model ($\Delta AICc = 2.34$, Table 6). The top model was identical to the predictive model, except for an additional effect grouping sites by turbine presence, which reduced survival ($\beta = -0.34$, $SE = 0.16$). However, this covariate was not identified as an important predictor of survival, with a cumulative model weight of 0.36. Weights for remaining covariates are summarized in Table 7.

DISCUSSION

In this study, I addressed the environmental effects of renewable energy developments on wildlife habitat quality by comparing mortality and population size of wind farm side-blotched lizard populations, with those from nearby reference habitats. Side-blotched lizard survival probabilities and population sizes are affected by territory quality which is linked with overall cover, composition, and diversity of local plant communities (Bury and Busack 1974, Fox et al. 1981). Predator diversity and abundance are also important predictors of population size and survival probability; increased predation resulted in a 15% reduction in survival over a 6-month activity period in one study of adult side-blotched lizards (Turner et al. 1974). And yet, demographic metrics were not different between populations at wind farms and at reference sites. Similarly, survival and population size were not affected by the magnitude of anthropogenic disturbance at each of the study sites. Therefore, evidence does not support a reduction in habitat suitability for lizard populations at wind farms, despite the observed changes in plant and predator communities at these sites (Chapter 1).

In disturbed habitats (with no resource supplementation), reduced habitat quality and associated stress can result in poor body condition, which may have demographic consequences for local populations (Wilson 1991, Hoare et al. 2006, Kight and Swaddle 2007). Alternatively, changes in predation pressure may affect stress as well as time spent foraging (rather than scanning for predators) which may ultimately affect body condition (Amo et al. 2006). Here, I found no evidence for differences in body condition (as length-corrected mass) between lizard populations at wind farms relative to natural areas; nor was body condition an important predictor in survival models.

I also predicted that changes in habitat quality or predation pressure would affect average lizard age at wind farms, as lizards would be less likely to survive long enough to attain large body sizes in poor quality habitats with few prey or many predators. Model results indicated no difference in average age, modeled as SVL, between wind farm and reference populations. Thus, I find no evidence for differences in individual quality resulting from wind farm presence or absence; this suggests either similar levels of predation and quality of habitat, or compensation for the presence of these effects by some additional means; for example, by increasing wariness behavior to avoid predators or by altering patterns of habitat use.

Annual survival projections for side-blotched lizards in this study (14.7–29.3%, Table 5) were well within the range of values reported for this species, and were higher than average for southern populations (Tinkle 1967*a*, Parker 1974, Turner et al. 1974). Of the covariates used to define differences in biotic or abiotic characteristics at study sites (canopy cover, elevation, precipitation, physical disturbance, and noise), cumulative model weights indicated greater survival probability at sites with low plant canopy cover. Typically, plant cover is thought to **increase** probability of survival for reptiles by providing refuge from predators (Parker 1974) or

from extreme temperatures (Fitch 1955). Researchers have shown that areas with heavy habitat loss from off-road vehicle use support smaller side-blotched lizard populations, as a possible consequence of loss of habitat cover (Bury and Busack 1974). Alternatively, plant cover may be habitat or food for prey (Bury and Busack 1974, Vitt and Ohmart 1974, Jones 1981). However, some evidence suggests that areas with heavy plant cover also support more endothermic mesopredators, which may result in lower survival for ectotherms such as lizards (Ayal 2007). Alternatively, the cover variable may have supported a model with variation in survival by site that better reflected additional important, but unmeasured drivers of survival differences among populations.

In this study, sites with low shrub cover were mostly wind farms; thus, the cover parameter allowed for higher daily survival rates at wind farms. Yet, models with an intercept allowing variation in survival probability between groups of wind farm and non-wind farm sites supported lower survival probabilities at wind farms. A model including both cover and group terms outperformed (by AICc score) all other models including the predictive model selected using cumulative model weights; the model with both terms predicts similar survival rates between wind farms and natural areas (the cover and turbine intercept parameters cancel each other out). Thus, it is difficult to make inferences regarding the effects of wind farms on survival rates for lizards, as conclusions differ with the top model used.

Rather, these results suggest that selective factors defining demography in side-blotched lizards are more variable over time than they are spatially, despite substantial anthropogenic disturbance at wind farm sites. Indeed, population-level responses to severe wind energy landscapes were minor when compared with the drastic changes observed in survival rate and population size over the active season (Parker 1974). Researchers have attributed high annual

population turnover in this species (Tinkle 1967*a*) to a number of factors; primarily high predation pressure (Parker 1974, Turner et al. 1982, Ferguson and Fox 1984, Wilson and Cooke 2004), competition (Fox 1978), and low winter rainfall (Turner et al. 1982), with associated meager spring productivity and low prey abundances (Ferguson and Fox 1984). It is unlikely that these important selective forces remain constant across the target study area; In Chapter 1, richness and diversity of predators and plants varied across the study region as a function of disturbance and elevation. Further studies over a narrow seasonal activity period, repeated annually, and using study sites with similar amounts of total bare ground cover would help to identify any differences in survival between populations. Direct quantification of predation rate and prey or habitat availability would also strengthen any conclusions regarding selective pressures at wind farms relative to reference sites.

Severe drought conditions in the SGWRA during the summers of 2013–2015 may have reduced prey availability and increased competition, affecting intensity of natural selection and possibly obscuring demographic differences between wind farms and natural areas. Further studies should verify that the patterns observed during this study remain consistent during periods of high resource availability. A longer study period may reveal differences in demography that were not observed over a 3-year study period. Additionally, larger study plots may provide a better picture of population dynamics at the scale of wind-farm projects. Finally, windy conditions and non-permanent identification methods reduced capture and recapture rates; these sources of variation should be eliminated from future studies to increase precision of parameter estimates.

MANAGEMENT IMPLICATIONS

Wind farms appear to be suitable habitats for side-blotched lizards; development and maintenance of wind energy facilities has minimal (if any) influence on population persistence as indicated by survival and population size estimates. As a species with moderate tolerance of disturbance, r-selected life-history characteristics (e.g., high density and reproductive rate) may provide an advantage in marginal wind farm habitats (Castellano and Valone 2006). The side-blotched lizard is an abundant and widely exploited prey species (Wilson 1991), suggesting that Southern California wind farms should be able to sustain the prey biomass needed for many other native predator species (assuming that these species are as tolerant of human disturbances as their prey). To better understand whether wind farm habitats are truly suitable for terrestrial vertebrate taxa, research is needed to verify the value of wind farm habitats for disturbance-intolerant species. In the meantime, land managers can reduce the potential for unforeseen consequences to local biodiversity by promoting conservative development scenarios; for example, by minimizing physical disturbance, or by favoring development alternatives in low-conservation priority habitats (Pearce-Higgins et al. 2009, Kiesecker et al. 2011). Avian and bat mortality rates (and associated carnivore attraction) can be reduced by using larger turbines and slower cut-in speeds (Barclay et al. 2007, Baerwald et al. 2009). Multiple land-use strategies that combine wind energy extraction with other land uses (e.g., ranching, farming) can maximize land value while minimizing additional biodiversity loss (McDonald et al. 2009, Janke 2010, Kiesecker et al. 2011). Alternatively, decentralized methods of energy production create additional opportunities to efficiently generate renewable energy, with minimal effects on local biodiversity (Van Der Schoor and Scholtens 2015).

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Table 1. Facility characteristics for each of the four target wind farm study populations within the San Geronio Wind Resource Area near Palm Springs, CA.

Wind Farm	Wind Farm Size (km ²)	Number of Turbines	Height ¹ Base Type ²	Turbine Density	Installation Date
ME	1.7	460	S, L	High	1983
PH	0.7	168	S, L	High	1984
MV	0.9	45	T, T	Low	2002
DI	2.5	21	T, T	Low	2007

¹ Hub height of (S) 20 to 25 m or (T) 60 to 70 m

² Tower structure of (L) steel lattice frame or (T) steel tubular tower

Table 2. Description of covariates used to model probabilities of survival (S), capture (p) and recapture (c) for side-blotched lizards surveyed in 2013–2015 from nine study populations near Palm Springs, CA. Models were build using a Huggins robust-design model framework, implemented in program MARK. Each variable was scaled and centered to z-score units.

Parameter	Category	Variable	(Model Nomenclature) Description
Capture/Recapture (pc)	Null	No Variation	(.) Probability equal for c and p ; p equal within periods and occasions
		Unique Intercept	(I) Unique intercepts for c and p , but model structure is equivalent for c and p across time, group, and survey covariates; p equal between secondary occasions
	Group	Site	(s) Unique probability for each location
	Time	Full Variation	(t) Unique probability for each period
		Linear Trend	(t _L) Capture probability increases linearly
	Survey Condition Covariates ¹	Wind Speed	(Wind) Average wind speed (measured with Kestrel meter)
		Activity Period	(Type) Categorical variable allowing unique probability for morning or evening activity periods
		Temperature	(Temp) Temperature of substrate in direct sunlight
		Relative Humidity	(RH) Relative humidity at breast height (measured with Kestrel meter)
		Cloud Cover	(Cloud) Probability of c and p decreases linearly as cloud cover increased from low (0–24%) to moderate (25–49%), high (50–75%), or very high cover (75–100%)
Survey Effort		(Time) Time from first adult lizard emergence to end of survey, not adjusted for number of surveyors	
Survival (S)	Time	Annual	(t _A) Categorical, probability of S is unique across study years (2013, 2014, 2015)
		Seasonal	(t _S) Categorical, unique probabilities of overwinter (mid-August to mid-March), late spring (late-March to late-May), early summer (June), mid-summer (July) and late summer (late-July to early-August) survival

Survival (S)	Individual Covariates	Age	(Age) Categorical, survival differs between subadults, and adult female and male (42, 46 mm) lizards
		Sex	(Sex) Categorical, allows unique probability for male and female lizards
		Stress	(Tail) Recent tail loss was used as an index of social or physiological stress, expressed as the length of the tail relative to total body length
		Body Condition	(BCI) Body condition was measured as residual, length-corrected mass from a regression of mass and body size (SVL)
	Group Covariates	Site	(Site) Unique survival values for each location
		Turbine Presence	(Treat) Categorical variable allowing unique survival for turbine and non-turbine site groups
		Elevation	(Elev) Elevation in meters was obtained from a digital elevation model
		Noise	(Noise) Average A-weighted decibels [dBA] measured on non-windy days
		Non-canopy Cover	(Cover) Bare ground cover was measured at each site using a random-point-intercept survey protocol
		Precipitation	(Precip) Winter precipitation data (mm) was retrieved from the PRISM Climate Group, Oregon State University (http://www.prism.oregonstate.edu/ , created 4 Feb 2004)
Human Disturbance	(Roads) Survival at each site as a function of total physical human disturbance, measured as total road length within a 500 m radius		

¹ For quantitative variables, values represent the maximum of recorded values from the beginning and the end of each mark-recapture or resight survey

Table 3. Conditions during 10 side-blotched lizard mark-recapture or resight surveys completed from 2013–2015 for nine 0.36 ha study areas near Palm Springs, CA (see Figure 1 for site descriptions). Values represent the calculated average of recorded conditions across all surveys (standard deviation in parentheses).

Site (Turbine Presence)	Cloud Cover ¹	Relative Humidity, %	Substrate Temp, °C	Wind Speed, mph	Survey Effort, Decimal Hours	Afternoon Surveys, %
SG (–)	1.44 (0.86)	28.65 (15.33)	47.24 (7.54)	6.14 (4.28)	3.74 (1.02)	0.00
MI (–)	1.78 (1.22)	26.24 (14.28)	41.68 (7.11)	6.96 (3.91)	3.19 (1.78)	66.67
WW (–)	1.12 (0.33)	30.52 (13.68)	45.96 (5.14)	10.34 (3.63)	2.57 (0.79)	72.22
DH (–)	1.82 (1.29)	32.10 (14.59)	46.59 (5.40)	4.10 (2.67)	2.84 (1.04)	29.41
I-10 (–)	1.61 (1.14)	34.06 (15.08)	42.69 (3.28)	12.36 (5.33)	3.03 (1.20)	55.56
ME (+)	1.39 (0.85)	29.32 (11.88)	43.19 (4.81)	8.34 (3.49)	2.89 (0.88)	66.67
PH (+)	1.42 (0.79)	24.38 (6.62)	46.19 (4.04)	13.01 (4.72)	2.38 (0.54)	84.62
MV (+)	1.47 (0.87)	29.65 (13.48)	44.71 (6.00)	13.37 (4.70)	2.92 (0.72)	0.00
DI (+)	1.47 (0.72)	28.34 (11.66)	43.49 (5.82)	6.54 (4.19)	2.85 (0.44)	0.00

¹ Categorical, values from 1 (none) to 4 (full cover)

Table 4. Characteristics describing the biotic and abiotic environment of nine target study areas for a side-blotched lizard mark-recapture study in the San Gorgonio Pass Wind Resource Area of Southern California, north of Palm Springs.

Site (Turbine Presence) ¹	Disturbance Index ²	Noise, Average dBA ³	Non-canopy Cover, % ⁴	Elevation, m ⁵	Ave Precip, inches ⁶
SG (-)	2.61	41.16	0.84	822.73	16.95
MI (-)	22.82	45.45	0.63	640.74	13.86
WW (-)	40.23	47.62	0.75	468.75	13.99
DH (-)	24.17	48.17	0.82	500.99	10.82
I-10 (-)	67.47	54.43	0.72	390.74	11.81
ME (+)	26.48	47.92	0.76	711.98	18.43
PH (+)	59.97	66.37	0.85	464.42	13.42
MV (+)	49.89	60.49	0.87	366.12	15.26
DI (+)	34.53	49.85	0.90	318.03	6.57
-	31.46	47.37	0.75	564.79	13.49
Average	(24.15)	(4.82)	(0.08)	(170.21)	(2.36)
(SD)	42.72	56.16	0.85	465.14	13.42
+	(15.05)	(8.77)	(0.06)	(175.48)	(5.01)

¹ Site coordinates: SG (33.977467, -116.671905), MI (34.000944, -116.603721), WW (33.938483, -116.640581), DH (33.990331, -116.497455), I-10 (33.926645, -116.660955), ME (33.943971, -116.679064), PH (33.939314, -116.617866), MV (33.913365, -116.626786), DI (33.939366, -116.553998)

² Total length of road within a 1000 m radius from the center of the study area, digitized in ArcMap from 1 m resolution California NAIP imagery. Road length was weighted by 0.5 for footpath, 1.0 for two-track, 3.0 for residential, 5.0 for secondary, and 10.0 for highways

³ Average A-weighted decibels reading from random samples of 10 readings taken June–July (2015) when turbines were running and wind was ≤ 3.5 mph

⁴ Data from three, 100 m point-line-intercept surveys (July 2014) used to calculate proportion of 100 random points per transect that did not intersect a plant canopy

⁵ Derived from a 2013 digital elevation model (DEM) raster at 1/3 arc-second resolution, retrieved from the National Elevation Dataset (available online from the USGS at: <http://nationalmap.gov/elevation.html>)

⁶ Retrieved from the PRISM Climate Group, Oregon State University (<http://www.prism.oregonstate.edu/>, created 4 Feb 2004), averaged across the winter months of November through February (2012–2015)

Table 5. Demographic summary for nine side-blotched lizard target populations in 0.36 ha study plots surveyed from 2013–2015 in the San Geronio Pass Wind Resource Area of Southern California, north of Palm Springs.

Site (Turbine Presence)	Individuals Marked (♂,♀)	Weight		Size		Maximum Adult Size, mm	Minimum Hatchling Size, mm	Sex Ratio (percent ♀)
		Average ± SD		Average ± SD				
		♂	♀	♂	♀			
SG (-)	48, 40	3.98 ± 1.49	3.49 ± 1.09	46.9 ± 6.6	45.8 ± 5.5	58	23	0.42
MI (-)	34, 20	4.21 ± 1.35	3.28 ± 0.79	47.9 ± 5.4	45.5 ± 4.0	54	25	0.29
WW (-)	23, 18	4.40 ± 0.47	3.11 ± 0.71	49.6 ± 2.0	46.9 ± 2.6	54	23	0.39
DH (-)	15, 17	3.79 ± 0.86	2.78 ± 0.40	46.5 ± 3.9	43.4 ± 2.4	52	24	0.57
I-10 (-)	18, 17	4.70 ± 0.41	3.12 ± 0.64	49.9 ± 2.4	46.4 ± 3.5	54	22	0.47
ME (+)	24, 20	4.36 ± 1.24	3.49 ± 0.95	48.5 ± 5.8	45.9 ± 4.8	54	21	0.42
PH (+)	13, 21	4.63 ± 0.62	3.11 ± 0.55	50.4 ± 2.3	46.5 ± 3.3	53	25	0.81
MV (+)	17, 23	4.08 ± 0.82	2.99 ± 0.56	48.4 ± 3.6	44.7 ± 2.6	52	22	0.68
DI (+)	10, 15	2.58 ± 1.22	2.99 ± 0.81	41.3 ± 6.3	44.1 ± 3.2	52	22	0.75

Site (Turbine Presence)	Capture Probability ¹ Average ± SD (Max)	Population Size ¹ Average ± SD (Max)	Average Daily Survival, % ¹ Average ± SD (Max)	Annual Survival Projection, % ¹
SG (-)	51.26 ± 2.63 (67.74)	31.7 ± 4.9 (50.4)	99.57 ± 0.14 (99.87)	25.79
MI (-)	50.48 ± 2.56 (67.74)	16.2 ± 3.1 (33.3)	99.39 ± 0.22 (99.81)	14.69
WW (-)	47.14 ± 2.54 (62.10)	15.4 ± 3.4 (22.2)	99.50 ± 0.16 (99.85)	20.82
DH (-)	53.08 ± 2.68 (70.29)	11.9 ± 1.8 (31.5)	99.55 ± 0.15 (99.86)	24.47
I-10 (-)	45.12 ± 2.77 (59.26)	12.2 ± 2.8 (17.2)	99.47 ± 0.17 (99.84)	19.25
ME (+)	49.14 ± 2.52 (65.53)	12.6 ± 2.6 (18.9)	99.50 ± 0.16 (99.85)	21.18
PH (+)	45.50 ± 2.66 (64.10)	19.7 ± 2.5 (23.3)	99.57 ± 0.14 (99.87)	26.17
MV (+)	44.10 ± 2.76 (56.40)	17.3 ± 3.9 (25.5)	99.58 ± 0.14 (99.87)	27.12
DI (+)	50.89 ± 2.64 (70.39)	10.1 ± 2.0 (20.2)	99.61 ± 0.14 (99.88)	29.25

¹ Values are extracted from a predictive Huggins robust-design model generated in program MARK: $S(I + t_{AS} + \text{Age} + \text{Cover}); pc(I + t_L + \text{Wind});$ see Table 2 for a full description of model structures

Table 6. Competitive models used to test for differences in survival rate across nine study populations of side-blotched lizard surveyed from 2013–2015 in the San Geronio Pass Wind Resource Area of Southern California, north of Palm Springs. All models were tested with fixed emigration, and encounter probabilities modeled as $pc(I + t_L + \text{Wind})$. For full model parameter descriptions, see Table 2.

Survival (<i>S</i>)	AICc	Δ AICc	AICc Weight	Num. Par
<i>S</i> (I + t_{AS} + Age + Cover + Treat)	3133.843	0.000	0.286	14
<i>S</i> (I + t_{AS} + Age + Cover)	3136.184	2.341	0.089	13
<i>S</i> (I + t_{AS} + Age + Cover + Noise)	3136.469	2.626	0.077	14
<i>S</i> (I + t_{AS} + Age + Roads + Elev)	3136.470	2.627	0.077	14
<i>S</i> (I + t_{AS} + Age + Cover + Roads)	3136.786	2.943	0.066	14
<i>S</i> (I + t_{AS} + Age + Cover + Precip)	3136.905	3.063	0.062	14
<i>S</i> (I + t_{AS} + Age)	3137.862	4.019	0.038	12
<i>S</i> (I + t_{AS} + Age + Roads)	3137.988	4.146	0.036	13
<i>S</i> (I + t_{AS} + Age + Cover + Elev)	3138.102	4.259	0.034	14
<i>S</i> (I + t_{AS} + Age + Precip + Site)	3138.395	4.552	0.029	21
<i>S</i> (I + t_{AS} + Age + Treat)	3138.765	4.922	0.024	13
<i>S</i> (I + t_{AS} + Age + Precip)	3138.832	4.989	0.024	13
<i>S</i> (I + t_{AS} + Age + Noise)	3139.004	5.161	0.022	13
<i>S</i> (I + t_{AS} + Age + Roads + Precip)	3139.628	5.785	0.016	14
<i>S</i> (I + t_{AS} + Age + Treat + Roads)	3139.759	5.916	0.015	14
<i>S</i> (I + t_{AS} + Age + Roads + Noise)	3139.789	5.946	0.015	14
<i>S</i> (I + t_{AS} + Age + Elev)	3139.870	6.027	0.014	13
<i>S</i> (I + t_{AS} + Age + Treat + Precip)	3139.927	6.084	0.014	14
<i>S</i> (I + t_{AS} + Age + Precip + Noise)	3140.309	6.466	0.011	14
<i>S</i> (I + t_{AS} + Age + Elev + Noise)	3140.494	6.651	0.010	14
<i>S</i> (I + t_{AS} + Age + Precip + Elev)	3140.673	6.830	0.009	14
<i>S</i> (I + t_{AS} + Age + Treat + Elev)	3140.700	6.857	0.009	14
<i>S</i> (I + t_{AS} + Age + Treat + Noise)	3140.706	6.863	0.009	14
<i>S</i> (I + t_{AS} + Age + Site)	3143.502	9.659	0.002	20
<i>S</i> (I + t_{AS} + Age + Cover + Site)	3143.502	9.659	0.002	20
<i>S</i> (I + t_{AS} + Age + Elev + Site)	3143.502	9.659	0.002	20
<i>S</i> (I + t_{AS} + Age + Noise + Site)	3143.502	9.659	0.002	20
<i>S</i> (I + t_{AS} + Age + Roads + Site)	3143.502	9.659	0.002	20
<i>S</i> (I + t_{AS} + Age + Treat + Site)	3143.502	9.659	0.002	20

Table 7. Significance of covariates used to model probabilities of survival and capture or recapture rates for populations of side-blotched lizards studied from 2013–2015 near Palm Springs, CA (Huggins robust design, program MARK). Each variable was scaled and centered to z-score units. A cumulative model weight of 0.5 or greater was used to identify important variables (in bold).

Parameter	Category (# models) ¹	Variable (# models)	$\sum w_i$ ¹
Survival Probability	Group Covariates (37)	Cover (7)	0.620
		Treat (7)	0.360
		Roads (7)	0.226
		Precip (7)	0.165
		Elev (7)	0.156
		Noise (7)	0.146
		Site (7)	0.043
	Individual Covariates (11)	Age (4)	0.855
		Tail (4)	0.250
		BCI (4)	0.173
		Sex (4)	0.172
Capture/Recapture Probability	Survey Conditions (22)	Wind (6)	0.990
		Type (6)	0.433
		Temp (6)	0.135
		RH (6)	0.081
		Cloud (6)	0.077
		Time (6)	0.076

¹ Cumulative model weights, $\sum w_i$, were calculated from a model subset of all possible combination of 0, 1, and 2 covariates within a model category; covariate descriptions are given in Table 2

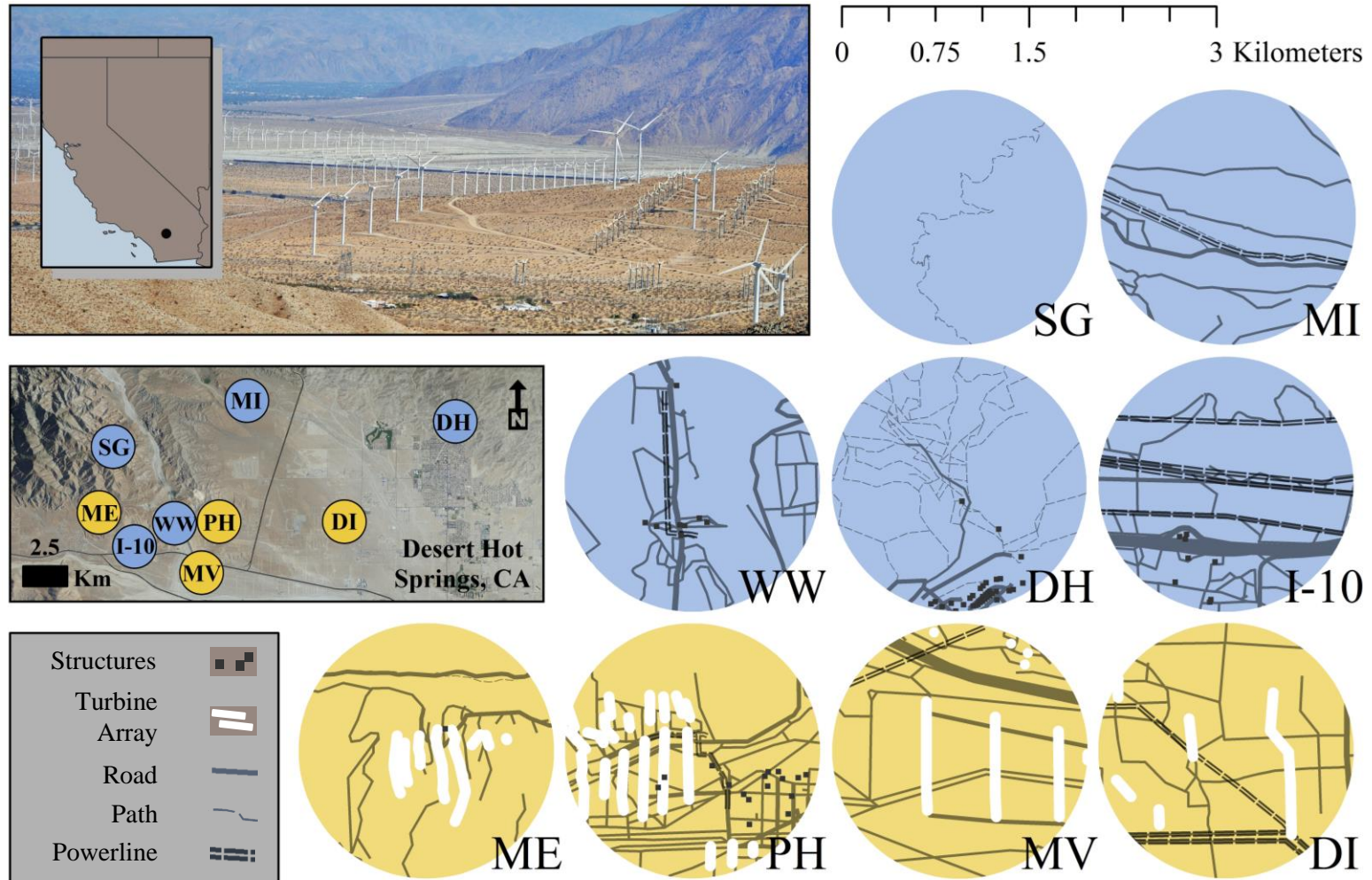


Figure 1. Study sites were located in the San Geronio Pass Wind Resource Area in southern California, north of Palm Springs. Insets show the land-use history of each study area (five study areas without turbines in blue; four sites with turbines in orange) within a 1 km radius, as indicated in the figure legend. Thickness of roads reflects road traffic volume.

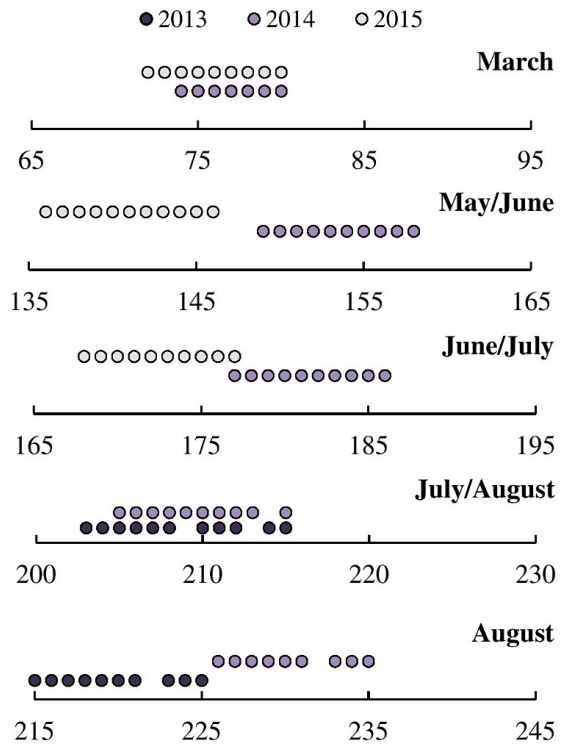


Figure 2. Survival probability of side-blotched lizards was estimated between 10 mark-recapture or resight survey occasions at nine study populations near Palm Springs, California. Surveys were conducted from late July of 2013 to early July of 2015, over five seasonal survey periods from March through August. Julian date is given for each season on the x-axis; circles represent secondary occasions within a seasonal survey period.

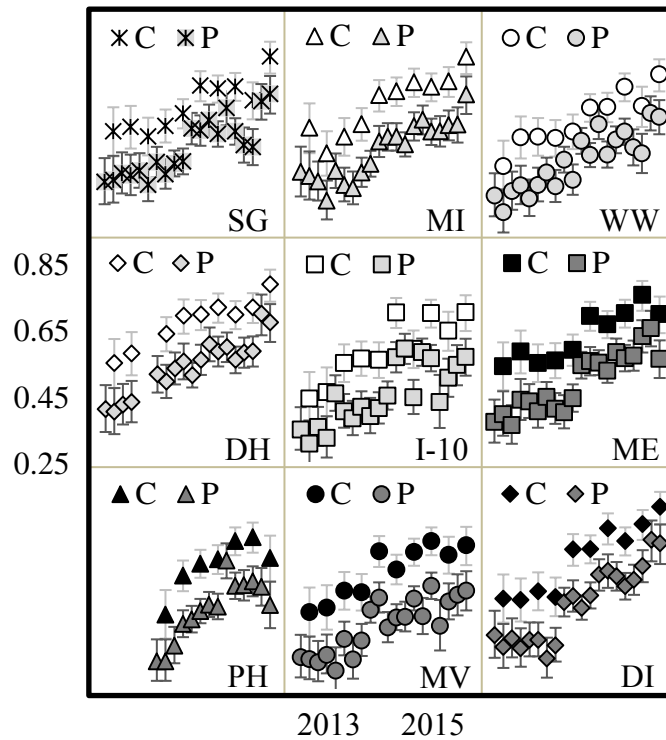


Figure 3. Encounter probability (y-axis) of side-blotched lizards for capture (p) and recapture (c) events during mark-recapture surveys (10 sessions, 2 occasions each) from 2013–2015 (x-axis) in the San Gorgonio Wind Resource Area near Palm Springs, extracted from the predictive model: $S(I + t_{AS} + \text{Age} + \text{Cover})$; $pc(I + t_L + \text{Wind})$. Insets provide estimates for each of the nine study areas; closed symbols represent sites ($n = 4$) with wind farms, open symbols/ asterisk ($n = 5$) indicate reference sites. Site PH was not visited on sessions 1–3, and site DH was surveyed only 1 occasion on session 3. Error bars indicate 95% confidence intervals for model estimates.

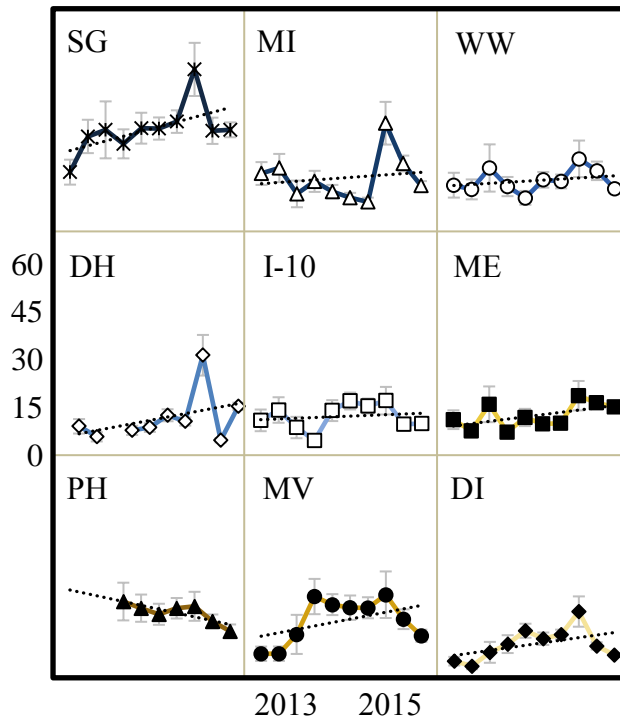


Figure 4. Model derived estimates of population size, from Huggins robust design model: $S(I + t_{AS} + \text{Age} + \text{Cover})$; $pc(I + t_L + \text{Wind})$. Population size (y-axis) is for side-blotched lizards in 0.36 ha plots surveyed from 2013–2015 in Palm Springs, California (10 sessions, 2 occasions each, x-axis). Insets provide estimates for each of the nine study areas; closed symbols represent sites ($n = 4$) with wind farms, open symbols/asterisk ($n = 5$) indicate reference sites. Note that site PH was not visited on sessions 1–3, and site DH was surveyed only 1 occasion on session 3. Error bars indicate standard deviation of $N\text{-hat}$ estimates.

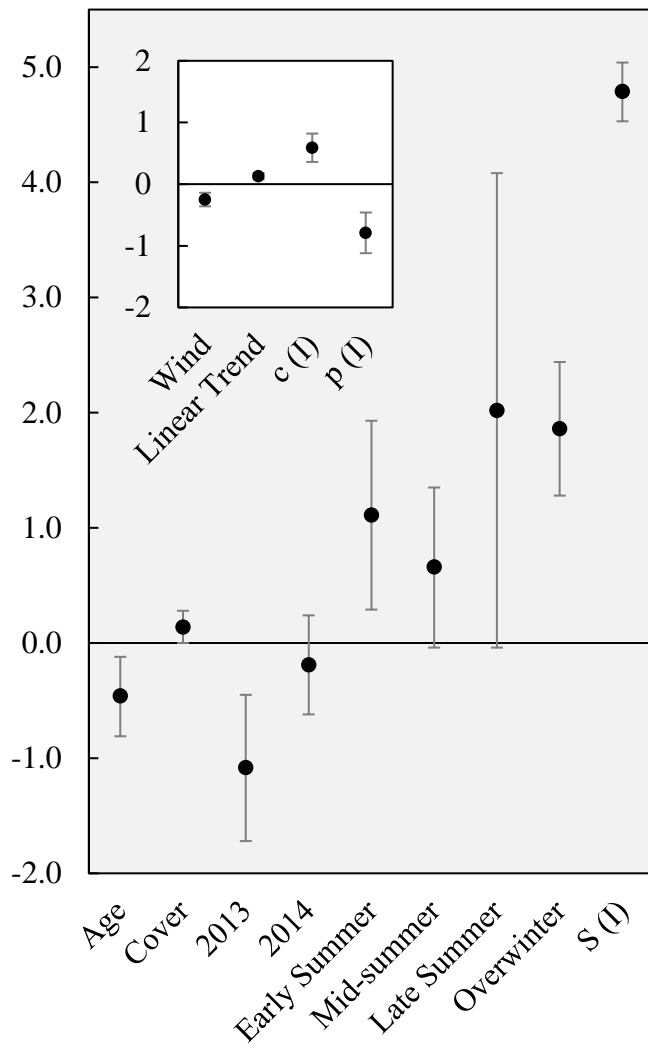


Figure 5. Parameter estimates for survival and encounter probability (inset) of side-blotched lizard populations surveyed in study areas near Palm Springs, CA from 2013–2015. Beta values were extracted from the predictive model, selected using a Huggins robust design model in program MARK: $S(I + t_{AS} + \text{Age} + \text{Cover})$; $pc(I + t_L + \text{Wind})$. For parameter descriptions, see Table 2. Error bars indicate 95% confidence intervals for parameter estimates. Reference conditions were for adult lizards in late spring of 2015.

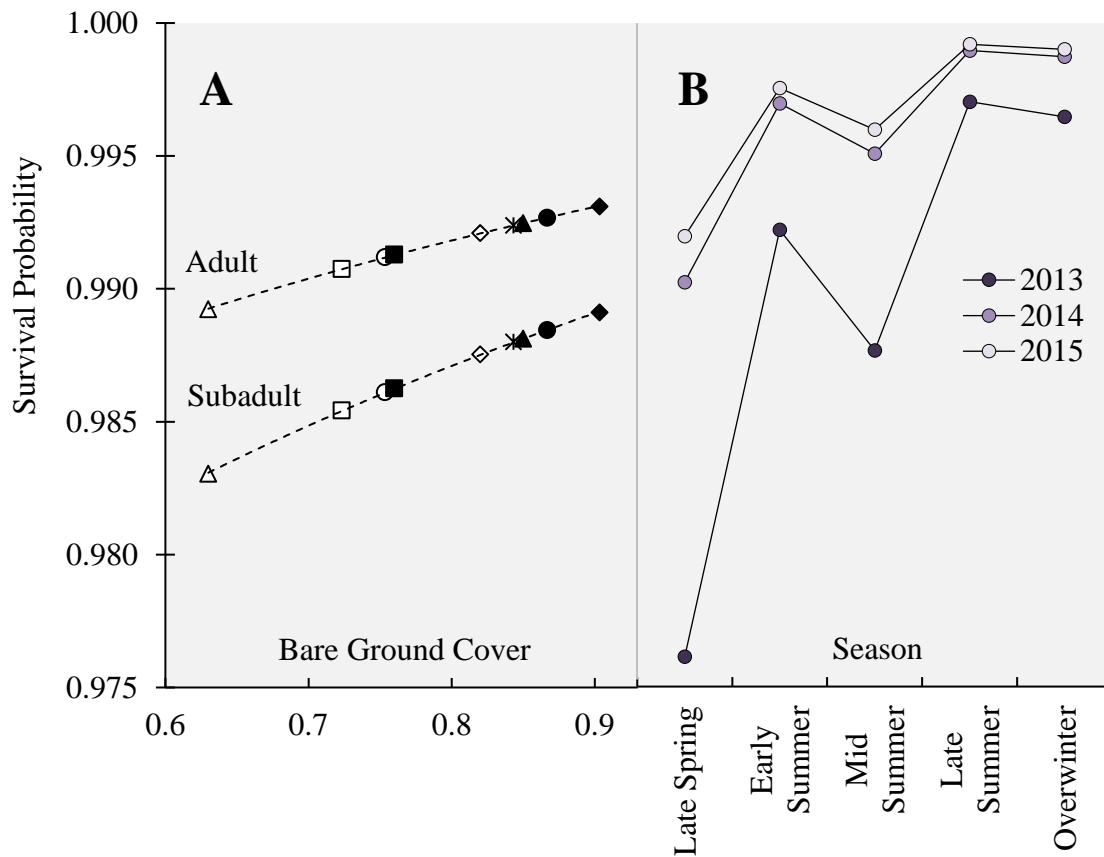


Figure 6. Estimated effects of individual covariates and temporal variation on daily survival probability estimates (y-axis) for side-blotched lizard populations surveyed from 2013–2015 in the San Geronio Wind Resource Area of southern California. Closed symbols give values for sites ($n = 4$) with wind farms; open symbols/asterisk ($n = 5$) indicate reference sites. Survival was modeled as a function of lizard age and site bare ground cover (A) and as a function of temporal variation (B). Estimates are generated by back-transformed beta parameters, from the model $S(I + t_{AS} + \text{Age} + \text{Cover})$; $pc(I + t_L + \text{Wind})$.

CONCLUDING REMARKS

The austere deserts of western North America are some of the wildest, and most irreplaceable landscapes remaining in the United States. Despite their vast and humble appearances, American deserts are teeming with a diversity of life. The Mojave, Great Basin, and Sonoran deserts support many species that occur nowhere else in the world, from pupfish inhabiting tiny isolated pools, to endemic plants clinging to the sides of sand dunes (Dobson et al., 1997; McLaughlin, 1986). And yet, desert habitats are rapidly disappearing in the face of human development pressures such as renewable energy development (Wilshire et al., 2008). In developing our deserts, we must acknowledge that we gamble with unique desert wildlife and habitat that define our American heritage, in exchange for energy resources to support human populations. Our decision to develop these areas for human profit is not something that can be taken lightly.

The issue of renewable energy development is exceedingly complex. With the growing threat of global climate change, it is virtually uncontested among scientists that we must take steps to reduce our carbon footprint in order to protect global biodiversity, and to allow for our own persistence (Bilgen et al., 2004). The vast renewable energy potential of the desert southwest makes development of energy infrastructure a nearly inevitable future (Pocewicz et al., 2011). But what can be done to safeguard our cultural and biological resources?

One approach is to minimize ecological disturbances from energy developments by choosing technologies with low biodiversity costs. The low land-use intensity, small physical footprint, and intact habitat matrix of wind farms suggests that this energy production strategy may be a suitable option. In this study, I established that community

diversity and composition patterns were affected by wind energy development (also, see Villegas-Patraca et al., 2012); however, wind energy sites were able to support many of the same species found in natural areas, including viable populations of important prey. A comparison of environmental consequences from other promising alternative energy production strategies is still needed before deciding whether wind energy is the most ecologically sustainable solution for meeting energy needs. Decentralized energy production should be considered as well; it presents an opportunity to generate electricity with minimal disturbance of natural, cultural, or aesthetic resources (Alanne and Saari, 2006).

Ecological costs of wind energy development can be minimized by placing wind farms in areas of low biodiversity and with pre-existing human disturbances (Kiesecker et al., 2011). Technological solutions can help to reduce bird and bat mortality risk; however, research is needed to determine the causes of avian behavioral avoidance of wind energy sites (Baerwald et al., 2009; Pearce-Higgins et al., 2009). Finally, it is clear from this research that wind energy sites may exclude disturbance-intolerant organisms; however, by identifying the changes in abiotic and biotic selective pressures at wind farms that are most likely to exclude these species, research can aid with facility design and maintenance strategies that further reduce the risk of local biodiversity loss. While development may be in the near future for these important desert habitats, a conservative approach towards wind energy technology may help to increase renewable energy potential, without needlessly harming valuable desert habitats.

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Supplementary data: predation rate estimates and lizard wariness behavior

S1. LIZARD PREDATION RATES AT DISTURBED WIND FARM SITES

S1.1 Introduction and Methods

In Chapter 1, I established that predator communities were less diverse at sites without wind turbines. I also found lower encounter rates of red-tailed hawk (*Buteo jamaicensis*) and loggerhead shrike (*Lanius ludovicianus*) at wind farm sites. In an effort to better understand the consequences of altered predator abundances at wind farm sites, here I estimated site-level predation intensity within avian, reptilian (lizard), and mammalian guilds for a common terrestrial vertebrate with many predators, the side-blotched lizard (*Uta stansburiana*). I compared attack rates of model side-blotched lizards at wind farm sites (“turbine sites”), against attack rates for models placed in nearby sites without energy development (“non-turbine sites”) in the San Geronio Wind Resource Area of Southern California (Figure S1.1). Clay models have been successfully used in a range of studies to infer predation pressure on different reptile color morphs, sexes, subpopulations, and within different habitats (Gifford et al., 2008; Husak et al., 2006; Vervust et al., 2007).

If red-tailed hawks and loggerhead shrikes are less abundant at wind farm sites, then I predict that lizard populations will experience lower avian predation pressure and thus fewer model lizard attacks at wind farm study sites. To test this prediction, I created lizard replicas using Sculpey™ polymer clay, a sculpting medium that remains soft unless exposed to high temperatures. Using a museum specimen of an adult male *U.*

stansburiana (specimen UNR5028, University of Nevada, Reno), I created a plaster mold of the body and tail. I pressed softened clay into the mold to form the body of the model, and then added clay to form arms and legs by hand, shaping them into a natural alert stance (Figure S1.2). I used orange, blue, and yellow spray paint to color the models in a pattern similar to an adult male's breeding colors (it should be noted that colors matched to the human eye may not accurately approximate the colors perceived by non-human predators).

In June of 2014, I deployed models in open, visible areas within 50–100 m of the nine established study plots. I placed models ($n = 37$ per site) at least 10 m apart along four parallel 100 m rows. I retrieved models after 10–13 days; however, due to high temperatures experienced in June in the Palm Desert region (during lizard activity hours, substrate temperatures of up to 55°C were recorded), it is likely that hardening of the clay prevented additional imprints after 4–5 days.

S1.2 Results and Discussion

Data from site SG were discarded because many clay models were trampled by cows prior to retrieval. Of the 296 models deployed, only 10.8% of models ($n = 32$) were attacked. Of the 32 attacks, 9.4% could not be assigned to any taxa, and 50% were from non-predatory small mammals (i.e., rodents, as indicated by characteristic small imprints from front incisors). In total, 13 models were attacked by target predators: birds (9), reptiles (3), or non-rodent mammals (1). Most predator attacks occurred at non-turbine sites (85%). Results are summarized by site in Figure S1.3.

The observed average avian model attack rate of 3.0% was within, albeit on the low end, the range of documented clay model attack rates (Brodie, 1993; Hinman et al.,

1997). A greater proportion of models were attacked at non-turbine sites, supporting our prediction of lower avian predation pressure at wind farms. However, the strength and significance of differences between site groups cannot be fully assessed given the sparsity of attacks per location. Larger samples or replicated experiments are needed in order to make inferences regarding predator activity and predation pressure across sites and within groups of wind farms and natural sites.

I noted that most beak prints were small and narrow, characteristic of smaller species such as the rock wren or the shrike. Many predators such as snakes and large invertebrates are less visually oriented than birds; predation intensity by these guilds is poorly approximated using a model lizard attack rate proxy. Finally, I noted that beetles and ants fed on clay models; it is unclear whether this drew the attention of additional predators.

S2. ANTIPREDATOR FLIGHT BEHAVIOR OF COMMON SIDE-BLOTCHED LIZARDS, *UTA STANSBURIANA*, AT WIND ENERGY SITES

S2.1 Introduction and Methods

Disturbances can affect abundances of predators, intensity of predation experienced by prey species, or perceived risk of predation for prey in altered habitats. For example, in disturbed wind farm habitats, changes in anti-predator behaviors related to foraging, alertness, and time spent away from predator refugia were demonstrated in California ground squirrels (Rabin et al., 2006). In reptiles, it is possible that increased human activity at wind farms affects perceived predation risk, resulting in more frequent (or greater intensity of) anti-predator behavior (Amo et al., 2006). Alternatively, reduced

prey or predator abundance, or lower intensity of predation may suppress predator avoidance behavior in favor of increased foraging activity (Lima and Dill, 1990).

In 2014, I quantified predator avoidance behavior in a common terrestrial vertebrate, the side-blotched lizard (*Uta stansburiana*). This species is abundant at study sites within the San Geronio Wind Resource Area (SGWRA). Previous research has documented variation in anti-predator behavior for side-blotched lizards in response to differing levels of predation intensity (Zani et al., 2013). In this study, I expected that lizards would be less predator-adverse at wind farm sites, resulting from lower predator diversity and lower abundance of some avian predators in these study areas (see Chapter 1). I quantified anti-predator behavior as the average flight initiation distance (FID) of individual lizards, following the procedure of Zani et al. (2013). From August 5–14 (2014), I conducted FID trials on unmarked lizards in the vicinity of eight established study plots in the SGWRA (Figure S1.1); surveys were completed at a ninth site (SG), but results were excluded from final analyses because of inconsistencies in FID measurements among observers.

For each FID trial, an observer simulated a predator approach by walking (a standardized approach pace of approx. 0.4 m/sec) towards a lizard from a starting point of within 45 degrees of the snout (i.e., facing the lizard) at a distance of 4 m from the lizard. The approach ended when the target lizard fled from the observer. At the end of the approach, I recorded FID as the linear distance, to the nearest cm, between the lizards' starting perch and the observer at the time of flight. I recorded time of day, perch temperature, and perch height to account for potential variance in FID resulting from variable trial or perch conditions. I conducted trials during the morning activity period

between 7:00–11:00 am, with substrate temperatures between 30–45°C. I targeted only adult lizards, and only lizards perched on rocks that were not obscured from the approaching observer by an overhead canopy. A number of surveys could not be completed because lizards fled at distances greater than 4 m; for each observer survey, I calculated the number of trials not completed, relative to the total number of trials.

To minimize potential differences in FID resulting from observer bias, each of the four possible observers dressed in similar colors during trials (white shirt, blue jeans, hardhat, and sun glasses), except on August 5th at site WW when observers lacked hardhats. I compared FID values for trials with ($n = 20$) and without ($n = 35$) hardhats at site WW and found no significant difference ($t_{41.2} = -0.34$, $P = 0.74$) between surveys. Therefore, both trials were included in final analyses.

I compared FID at turbine and non-turbine sites using a mixed-effects model (*lmer* in *lme4* package version 1.1–7) in Program R version 3.2.2 (R Core Team, 2015). I used a random intercept of site and compared models incorporating fixed effects of trial time, perch temperature, and perch height (and the interactions of the former three variables) using the *dredge* function (MuMIN package version 1.15.1). I ranked models by AICc score, and used the top ranking model for inference (Burnham and Anderson 2002). Secondly, I used a linear mixed-effects model with a random site intercept to compare the percentage of trials not completed per observer-survey between turbine and non-turbine sites, where incomplete trials were defined as lizard FIDs greater than 4 m.

Lizards were not captured after trials; therefore, I was not able identify if sex, physical condition, or size/age influenced results in FID trials. Instead, I explored the effects of these variables using FID observations on known-condition, marked lizards at

each of the nine study plots from June 26–July 6 (2014). In total, 14 males and 14 females were observed, 1–3 times each. Body condition was rated as good for 13 of these lizards and poor for 15, where condition was defined by whether or not the tail length was greater than or less than 1.7 times the body length, respectively (indicating a recent tail loss which may represent a substantial physiological stress event). Body sizes ranged from 40–54 mm. Trial conditions mirrored those of unmarked lizard trials except that both morning and evening activity periods were included, and hardhats were only worn at turbine sites for logistical reasons. I inferred significant effects of categorical sex and condition variables, and a continuous size variable on FID from linear mixed-effects models with single-term fixed effects and a random intercept for repeated observations on individual lizards. I assessed significance of fixed effects from P -values generated using the Satterthwaite's approximation in the *lmerTest* package, version 2.0–29. Given the sparsity of this dataset, I did not test for additive, interactive, or polynomial effects for any of the three condition variables, nor did I allow for site variation in FID.

S2.2 Results and Discussion

Final FID trials compared behavioral responses between sites with ($n = 4$) and without ($n = 4$) wind turbines (328 observations, 28–55 per site). Flight responses were highly variable (Figure S2.1). The random effect of study site explained 3.8% of the total variance in FID (null model: $\text{FID} \sim 1 + (1|\text{Site})$, Table S2.1). The null model was the most competitive model with an adjusted R^2 of 0.01, followed by a model with a turbine vs. non-turbine group effect ($\Delta\text{AICc} = 2.73$). The standard deviation of the turbine coefficient did not overlap zero (-0.16 , $SD = 0.13$); however, the P -value for the turbine effect was not significant ($P = 0.25$).

I recorded 213 incomplete trials, or 1 incomplete trial for every 2 complete trials. Percentage of incomplete trials ranged from 8–56% across 29 surveys (Figure S2.2). In the null model, a random site intercept explained 45% of the total variance with an adjusted R^2 of practically 0.00. Adding a group effect for surveys at turbine and non-turbine sites reduced model performance ($\Delta\text{AICc} = 5.80$). The group effect parameter was small, overlapped zero, and was not significant (-0.04 , $SD = 0.07$, $P = 0.58$).

I used a dataset with 47 observations on 28 marked individuals to test whether sex, condition, and size influence FID. Linear mixed-effect models detected small effect sizes for each of these three variables, with parameter values overlapping zero. None of the effects were significant when P -values were calculated (Table S2.2). The best performing model consisted of a random intercept only with no additional effects (adjusted $R^2 = 0.24$); it outperformed the nearest model by ΔAICc of 1.92. Variance in FID between lizards accounted for 67% (0.54 , $SD = 0.73$) of the total variance, suggesting a consistent individual response to simulated predator approach that did not vary as a function of individual condition.

Most of the observed variation in antipredator behavior was between individual lizards, rather than between populations or sites. These results suggest that (1) predation pressure does not differ between wind farm sites and reference sites, or (2) individual flight responses were influenced by a number of additional underlying factors that may have obscured any potential differences between sites. Studies have identified sex (Lailvaux et al., 2003), size and age (Lopez et al., 2005), and recent tail loss (Cooper, 2003) as important drivers of wariness response; yet, I found no difference in lizard wariness resulting from these characteristics. Alternatively, a lizard's environment may

affect its perceived predation risk. Lizard sprint speeds decline when they experience non-optimal temperatures (such as during the early morning) which may increase wariness to predators (Bulova, 1994). Or, lizards perched at greater heights may flee sooner because of greater distances to refugia, or because of an increased visibility between predators and prey (Cooper, 2006). In this study, neither temperature nor perch height explained antipredator response.

Additional unmeasured factors may have played an important role in observed wariness behavior of individual lizards. For example, lizards closer to potential refugia may be less wary when approached by simulated predators (Zani et al., 2009). Variation in health (Lopez et al., 2005) or reproductive condition (Bauwens and Thoen, 1981) may affect flight response. Finally, prior experience with predators (Olla and Davis, 1989), as well as innate genetic (Storfer and Sih, 1998) or hormonal (Johnson et al., 1996) differences may affect individual response to predators. Given the substantial variation in flight response observed between individuals, it seems likely that any number of these unmeasured characteristics plays a substantial role in driving variation in wariness responses within and between sites. Such individual level variation, if heritable, would create an opportunity for lizards to adapt to differences in predation pressure between sites.

S3. LITERATURE CITED

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Table S2.1. Mixed-effects models were used to test whether fixed effects of size, sex, or body condition influenced antipredator behavior, measured as flight initiation distance (FID, $n = 47$ observations), for 28 marked side-blotched lizards (*Uta stansburiana*) from study populations (Figure S1.1, excluding site WW) within the San Geronio Wind Resource Area, CA.

Model ¹	AICc	Δ AICc	Parameter		
			Estimate	SE	<i>P</i> -value
FID = (1 ID)	118.3	0.00			
FID = Body Condition + (1 ID)	120.2	1.92	-0.306	0.323	0.352
FID = Sex + (1 ID)	120.7	2.44	-0.199	0.325	0.546
FID = Size + (1 ID)	124.5	6.19	-0.036	0.041	0.395

¹ Three covariates were tested regarding individual variation: (1) body condition as a categorical variable separating lizards with long, intact tails from those with recent tail regenerations; (2) sex as a categorical variable grouping male and female lizards; and (3) unstandardized snout-to-vent length (mm) to quantify body size. Models included a random intercept of individual identity, ID.

Table S2.2. Mixed-effects models were used to test whether lizard (*Uta stansburiana*) perch characteristics influenced antipredator behavior, measured as flight initiation distance (FID), for unmarked side-blotched lizards from study populations within the San Geronio Wind Resource Area (Figure S1.1). Each site ($n = 8$, site SG excluded) was modeled as a random intercept, with additional fixed effects and a group covariate (Treatment) to identify populations at study areas with ($n = 4$) and without ($n = 5$) wind turbines.

Model ¹	Adjusted R-squared ²	AICc	Δ AICc	Model Weight
FID = (1 Site)	0.012	762.2	0.00	0.595
FID = (1 Site) + Treatment	0.018	765.0	2.73	0.152
FID = (1 Site) + Height	0.012	765.0	2.75	0.151
FID = (1 Site) + Height + Treatment	0.018	767.7	5.49	0.038
FID = (1 Site) + Time	0.013	768.2	6.00	0.030
FID = (1 Site) + Temp	0.014	770.6	8.37	0.009
FID = (1 Site) + Time + Height	0.013	771.0	8.73	0.008
FID = (1 Site) + Time + Treatment	0.018	771.1	8.88	0.007

¹ Models tested fixed effects of turbine presence or absence (Treatment), perch height of lizard (Height), time of day (Time), and perch temperature (Temp)

² Calculated as Nagelkerke's modified statistic (not equivalent to classical R²)

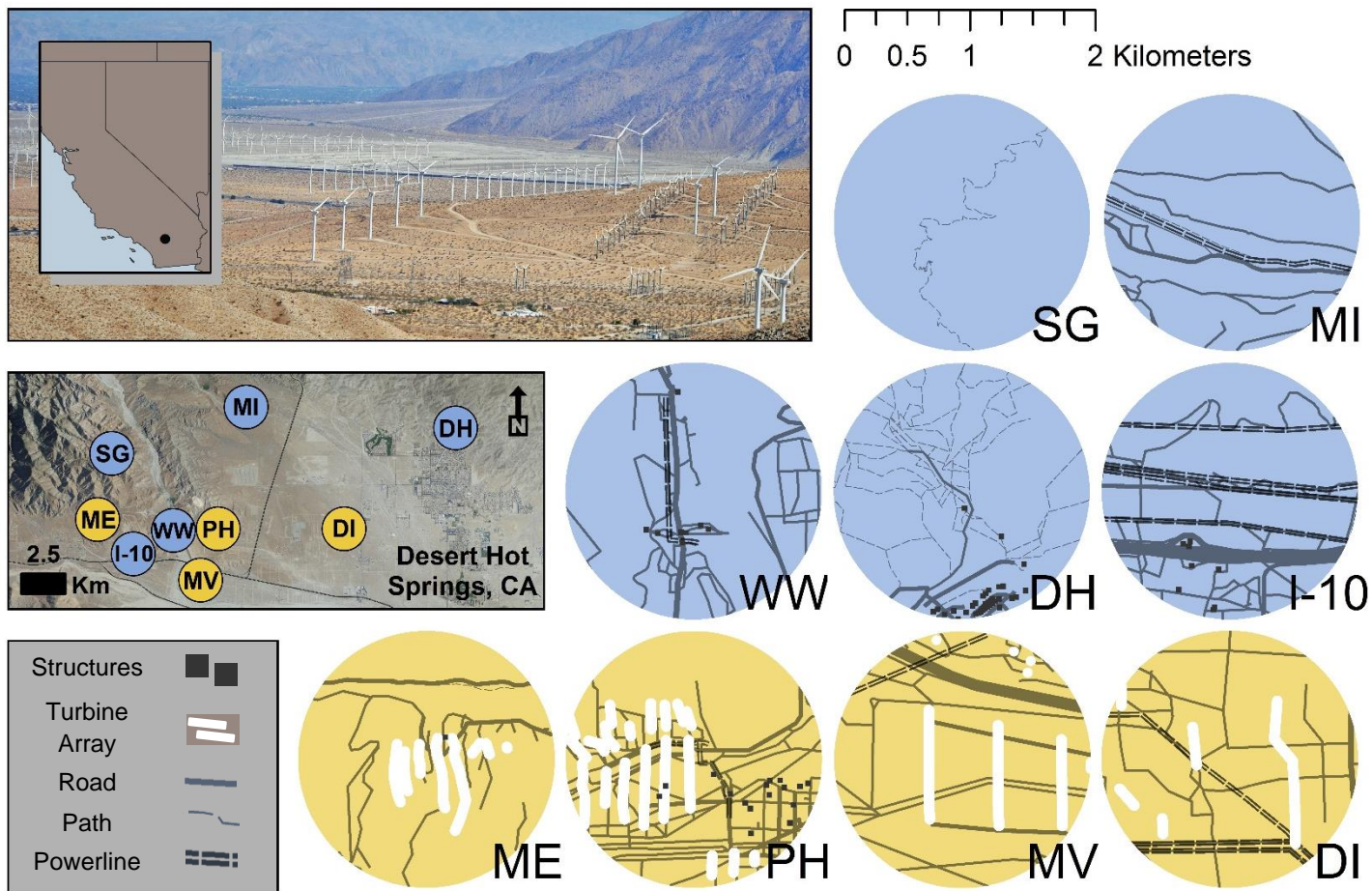


Figure S1.1. Study sites were located in the San Geronio Pass Wind Resource Area (SGWRA) in southern California, north of Palm Springs. Insets show the land-use history of each study area (five study areas without turbines in blue, top middle panels; four sites with turbines in orange, bottom panel) within a 1 km radius. Turbines (white), roads (grey lines; thickness reflects road traffic volume), paths (dashed line), transmission lines (double-dashed line), and buildings (squares) indicate the types of human disturbances present at each study region.

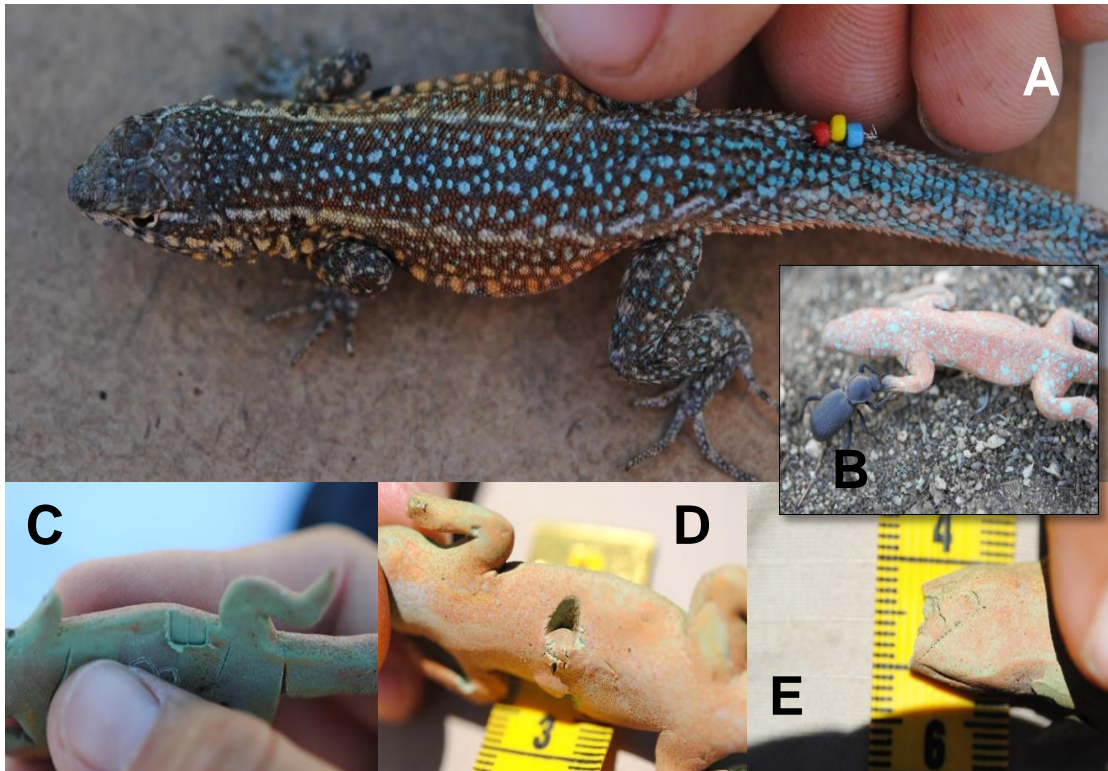


Figure S1.2. Adult male side-blotched lizards, *Uta stansburiana* (A) were used to create model lizard replicas (B) to assess attack frequencies of mammal (C), bird (D), and reptile (E) predators on clay models. Attack rates were quantified in 2014 at eight sites (four wind farms and four natural areas) in the San Geronio Pass Wind Resource Area of Southern California (see Figure S1.1) to test whether habitat disturbance at wind energy developments affects reptile predation pressure.

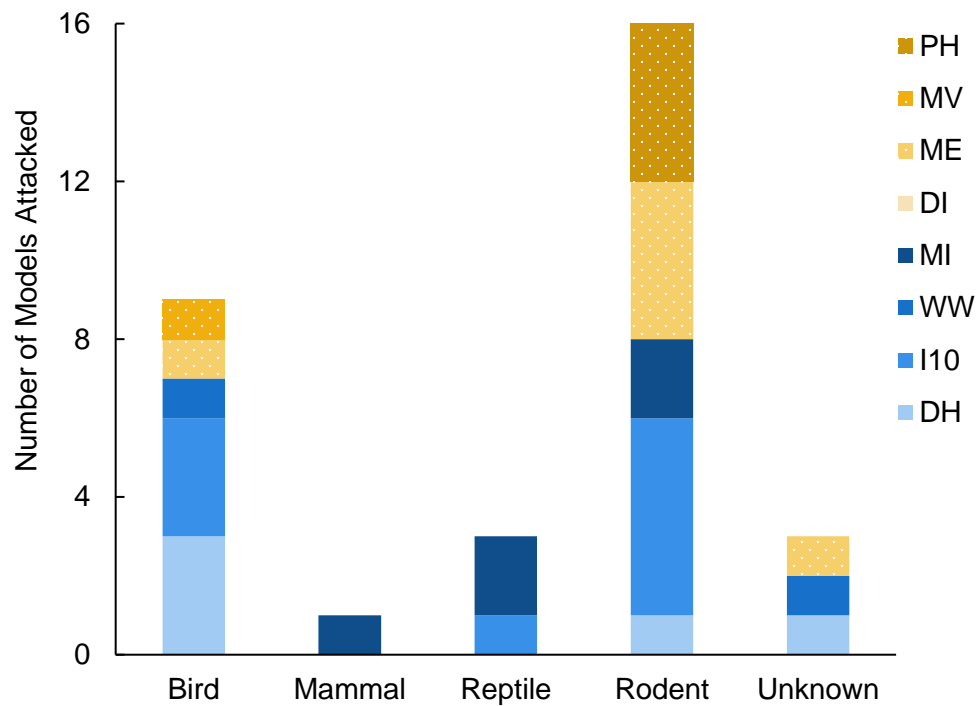


Figure S1.3. Clay model attack rates were used to estimate predation pressure for lizards (*Uta stansburiana*) at wind farms. The number of model clay lizards attacked by predatory birds, mammals, and rodents was quantified in summer of 2014 at four wind farm sites (orange, patterned) relative to four natural reference areas (blue) in the San Geronio Wind Resource Area near Palm Springs, CA, (see Figure S1.1 for site descriptions). Models were also attacked by non-target rodent predators; some attacks could not be assigned by taxa and were classified as unknown.

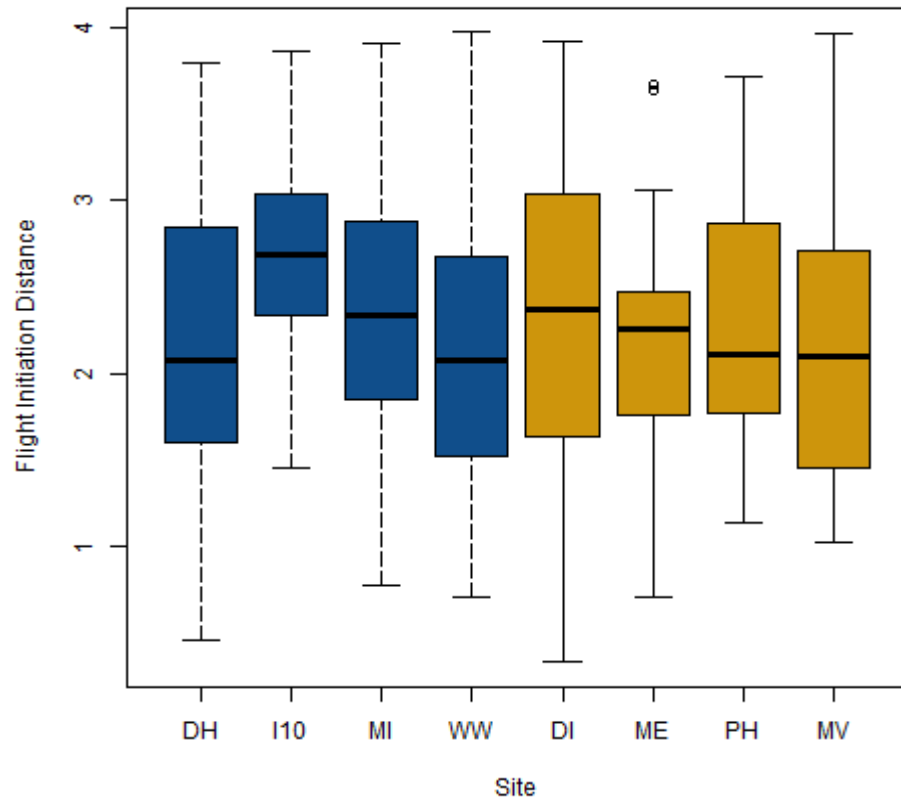


Figure S2.1. Boxplots of flight initiation distance (FID, in meters) for side-blotched lizards (*Uta stansburiana*) at nine study sites within the San Geronio Wind Resource Area near Palm Springs, CA in summer of 2014 (see Figure S1.1 for site descriptions). FID was highly variable within sites for both natural areas (dark blue, dashed whiskers) and wind farms (orange, solid whiskers).

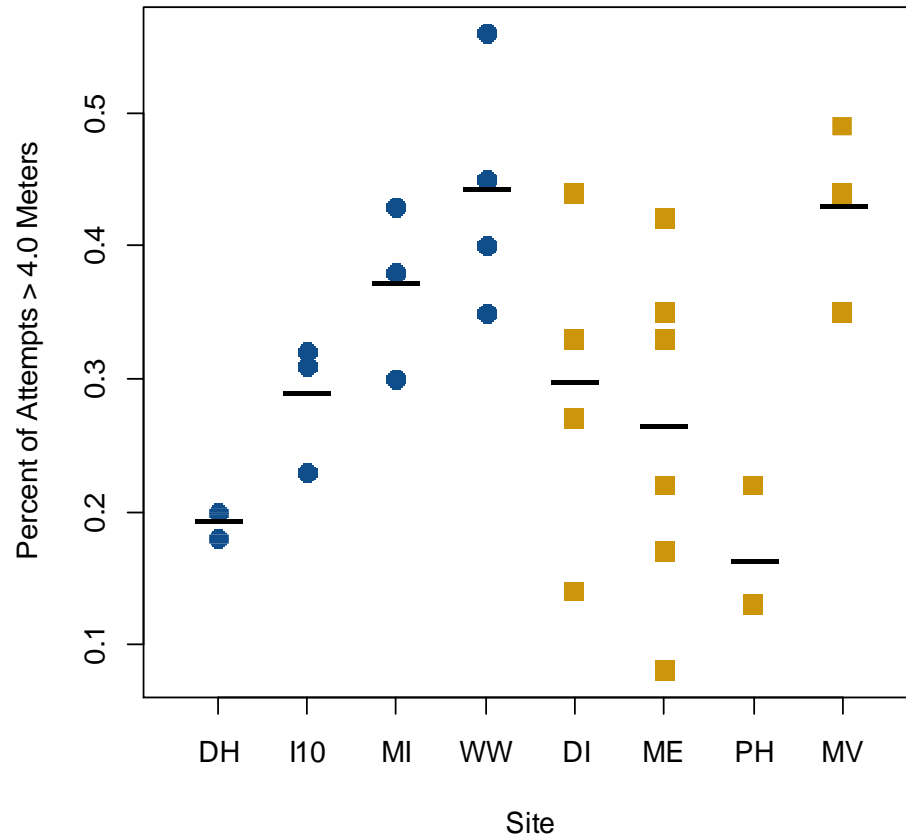


Figure S2.2. Proportion of side-blotched lizard (*Uta stansburiana*) flight initiation distance (FID) trials exceeding a 4 meter distance between lizards and observers at nine study sites within the San Geronio Wind Resource Area near Palm Springs, CA in summer of 2014 (see Figure S1.1 for site descriptions). Natural reference (blue circle) and wind farm (orange square) study areas are shown; points represent values for individual surveys; bars indicate mean values.