PREDICTORS OF BAT MORTALITY RATES AT NORTH AMERICAN WIND FACILITIES AND AN EVALUATION OF BIASES INFLUENCING MORTALITY ESTIMATES

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MAUREEN THOMPSON

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Thesis Approved:

Dr. Scott Loss

Thesis Adviser

Dr. Karen McBee

Dr. Monica Papes

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Name: MAUREEN THOMPSON

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Abstract: Wind energy helps close the gap between energy production and energy consumption; however, it is not entirely environmentally neutral as there are several indirect and direct impacts to birds and bats. The primary direct impact for bats is mortality by collisions with wind turbines. Several cross-site syntheses of bat collision data have resulted in national fatality estimates that vary by more than an order of magnitude (33,000 to 888,000 bats per year). However, no research has taken a national-scale approach to assess spatial, seasonal, and taxonomic variation in bat mortality and to evaluate the methodological factors that influence mortality estimation. I completed an exhaustive review of the published and unpublished literature on bat collisions with wind turbines and extracted data from 218 post-construction mortality reports representing 100 U.S. facilities and 12,000 bat fatality records. This database was used to conduct a national-scale meta-analysis to (1) Identify wind facility-scale correlates of bat collision mortality rates, (2) Assess how differences among studies in study design, data collection, and statistical analysis contribute variation to wind facility-scale estimates of mortality, and (3) Based on those analyses, identify specific study design, data collection, and analysis steps that should lead to relatively unbiased estimates of bat fatality rates. Information from these systematic data-driven analyses — the first of their kind to be based on a national data set of this size — will be useful in considering locations of future wind farms, designing mortality monitoring protocols, and ultimately, improving our understanding of impacts of wind facilities to bat populations.

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CHAPTER I

INTRODUCTION

Energy use past, present, and future

Since the 1960's, energy consumption has exceeded energy production in the United States, though the distance between the two has begun to decrease in the last 7 years. The production and use of renewable energy sources doubled between 2000 and 2014, largely because of state and federal government mandates and incentives for renewable energy development (EIA 2015 a). Globally, renewable electricity generation is now commensurate with natural gas (IEA 2014). In 2014, renewable sources of energy accounted for about 10% of total U.S. energy consumption and 13% of electricity generation (EIA 2015b). The electricity produced through wind power accounted for 4.3% of all electricity generated in the U.S. from July 2013 - July 2014 (EIA 2014), and the U.S. Department of Energy estimates that 20% of the electricity in the United States could be provided from wind energy by 2030 (U.S. DOE 2008).

Wind-generated energy offers the opportunity to reduce carbon and other greenhouse gas emissions that contribute to climate change. However, wind energy is not entirely environmentally neutral because wildlife and their habitats can be directly and indirectly impacted by development. Ecologists have expressed concern regarding these impacts, including bird and bat collisions with wind turbines (Kunz et al. 2007a,b; Kuvlesky et al.

2007), habitat loss, and creation of movement barriers (Kuvlesky et al. 2007; Pruett et al. 2009; Kiesecker et al. 2011. Studies of the Collision Mortality of Volant Species at Wind Facilities

Collision mortality of birds at wind facilities has been an issue of concern since the first commercial wind facilities were constructed in the 1970's (Rogers et al. 1977). Concern over bat mortality is more nascent but rapidly emerging as a major consideration for ecologists and developers. Bat fatalities have been reported anecdotally at wind facilities since the 1970's as well (Hall and Richards 1972, Durr and Bach 2004, Kuntz et al 2007, Kuvlesky et al 2007, Arnett et al 2008). However, bat collisions generally received little attention until 2003, when an estimated 1,400-4,000 bat collision fatalities (~48 bats/turbine) were reported at the Mountaineer Wind Energy Center in West Virginia over an eight month period (Kerns and Kerlinger 2004). Relatively high mortality rates have also been documented at other facilities along forested mountain ridges in the eastern United States, where rough estimates of mortality rates range from ~15-53 bats per megawatt per year (Arnett et al. 2008). Data from the Midwestern U.S. and Canada also suggest that moderate to high bat mortality rates occur across a variety of landscapes, including agricultural lands, prairies, deciduous forests and coniferous forests (Jain et al 2011, Barclay et al. 2007, Kuntz et al. 2007, Arnett et al. 2008).

Uncertainty about the Biological Significance of Collision Mortality

Concerns are mounting regarding the potential cumulative effects of wind energy development and other anthropogenic activities on bat populations. Bats are long-lived mammals with low reproductive rates (~1 pup/pair/year) that require high adult survivorship to avoid population declines (Barclay and Harder 2003). Furthermore, many bat species are migratory, and this life history trait exposes bats to a variety of anthropogenic and natural mortality sources that may make them more vulnerable to population decline and extinction than non-migratory species (Pimm et al. 1988). As such, bat populations are generally not able to recover quickly from broad-scale anthropogenic impacts, and many species are already known or suspected to be in decline, including the Eastern red bat (*Lasiurus borealis*), Northern long-eared bat *(Myotis septentrionalis*), and Little brown bat (*Myotis lucifugus*) (Pierson 1998, Racey and Entwistle 2003, Winhold and Kurta 2006, Jones et al. 2009, Frick et al. 2010). Wind turbine-related mortalities are problematic given limited knowledge about bat population sizes and trends, as well as the magnitude of other anthropogenic threats. Without information as basic as the distribution and movements of affected bat species, determining the cumulative effects of wind facilities on bat populations is difficult.

Of particular concern for bat conservation efforts is the emergence of white-nose syndrome, an infectious fungal pathogen that is decimating populations of multiple bat species. As of 2012, white nose syndrome had killed over 5.5 million bats (Froschauer 2012) in twenty states and has since continued spreading westward across the continent. Understanding and ameliorating the risks that wind farms pose to bat population sustainability is especially important in light of the perilous state of bat populations as a result of white-nose syndrome.

Impetus for Research

The vast majority of studies of wind energy impacts on bats have focused on quantifying mortality rates at the scale of individual wind facilities. Several authors have used facility-scale data to estimate the number of bats killed at wind energy facilities at the scale of the entire United States. The models and data used by Kunz et al. (2007b) represent an early effort to estimate annual U.S. bat fatalities; they estimated that 33,000–111,000 bats are killed annually at wind energy facilities. Cryan (2011) estimated that roughly 450,000 bats are killed each year in North America. Smallwood (2013) used previously unpublished data to derive an estimate of 888,000 bats killed in the United States in 2012. Despite this advance in the analysis of collision mortality data, no research has used raw data from multiple individual wind facilities to conduct a meta-analysis of the factors that affect bat collision mortality rates (Hayes 2013).

Estimates of bat mortality rates are strongly affected by sampling biases, by the method used to quantify sampling biases, by the estimators used to account for biases and calculate bias-adjusted mortality estimates. In fact, as much variation in mortality estimates can be attributed to variation in field methods as by actual variation in mortality (Smallwood 2013). Despite previous recommendations to standardize data collection approaches among individual studies, no research has used a meta-analysis to assess the methodological factors that most influence mortality rate estimates in individual studies.

Statement of Problem

The purpose of this thesis is to address the above research gaps by conducting meta-analyses using an extensive collision mortality database first compiled as part of a study of bird collisions with wind turbines at the scale of the contiguous United States (Loss et al. 2013). My specific objectives under this broad goal include:

1) Identification of wind facility-scale correlates of bat collision mortality rates.

2) Assessment of how methodological differences (in study design, data collection, and statistical analysis) contribute variation to wind facility-scale estimates of mortality, and, based on this analysis, identification of specific study design, data collection, and analysis steps that should lead to relatively unbiased estimates of bat mortality rates.

Information from these systematic data-driven analyses — the first of their kind to be based on a national data set of this size (41 studies representing 41 wind facilities and 1,379 bat fatality records) — will be useful in considering locations of future wind farms, designing mortality monitoring protocols, and ultimately, improving our understanding of impacts of wind facilities to bat populations.

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CHAPTER II

FACTORS AFFECTING BAT COLLISION MORTALITY AT WIND FACILITIES IN THE U.S.

INTRODUCTION

Since the 1960’s, energy consumption has exceeded energy production in the United States, with the gap between the two only beginning to decrease in the last 7 years (EIA 2014). In 2014, cumulative output from wind, solar, and other renewable power sources was greater than in any previous year (EIA 2015). With 65 gigawatts (GW) of installed generating capacity, wind energy currently provides nearly 5% of all U.S. electricity generation. Utility-scale wind facilities are operating in 39 states, and increases in efficiency are expected to allow the industry to expand into all 50 states (Zayas et al. 2015). Although wind generated electricity is renewable and generally considered clean, wind facilities can have numerous indirect and direct adverse effects on wildlife. Major indirect effects include loss of habitat, disturbance and displacement of individuals or populations, and creation of barriers to short or long-distance movements. The primary direct effect of wind energy is the collision of wildlife with wind turbines, a phenomenon that has been recorded worldwide (Amorim et al. 2012, Escobar et al. 2015, Doty & Martin 2013, Vilegas-Patraca et al. 2012). For bats, collision with wind turbines is a relatively understudied and potentially significant additive source of mortality, with hundreds of thousands of bats estimated to be killed in the U.S. each year (Kunz. et al. 2007, Cryan 2011, Smallwood 2013).

This mortality is of particular concern because bats have low reproductive rates (~1 pup/pair/year) and require high adult survivorship to avoid population declines (Barclay and Harder 2003), because little is known about bat population sizes and trends, and because bats face an increasing variety of anthropogenic threats. Many bat species are already known or suspected to be in decline, including the Eastern red bat (*Lasiurus borealis*), Northern long-eared bat (*Myotis septentrionalis*), and Little brown bat (*Myotis lucifugus*) (Pierson 1998, Racey and Entwistle 2003, Winhold and Kurta 2008, Jones et al. 2009, Frick et al. 2010). Without information as basic as the distribution and movements of affected bat species, determining the cumulative effects of wind facilities on bat populations is difficult (Cryan 2011). Gathering additional large-scale information about the drivers of bat collision mortality at wind turbines is crucial to advancing bat conservation efforts.

Significant research has been conducted to identify predictors of bird-turbine collision rates and to implement management to reduce this mortality (Barclay et al.2007, Johnson et al 2002, Smallwood and Karas 2009, Loss et al. 2013). However, bat collisions at wind turbines are relatively understudied despite evidence that more bats than birds are killed (Ellison 2012, Smallwood 2013). Most U.S. bat fatalities appear to occur between late July and mid-September and to primarily affect a few tree-roosting species, including the Hoary bat (*Lasiurus cinereus*), Eastern red bat, and Silver-haired bat (*Lasionycteris noctivagans*) (Johnson 2005, Kuntz 2007a, Arnett et al. 2008). Previous studies have assessed correlates of bat mortality at individual wind facilities, such as spatial proximity to hibernacula, ravines, and/or wetlands (Piorkowski and O'Connell 2010, Ferreira 2015), and one study used data from 25 wind facilities (in 8 U.S. states and 3 Canadian provinces) to document that larger and taller turbines likely increase bat mortality rates (Barclay et al. 2007). However, to date, relatively little systematically derived information exists about the characteristics of wind facilities and their surrounding landscapes that can be used to reliably predict bat collision mortality rates at a national scale (Hayes 2013). Additionally, in both local studies and large-scale quantitative reviews, inference about impacts to bats has been limited because different studies use different sampling methods and statistical estimators that preclude direct data comparisons and limit cross-facility identification of mortality rate correlates. I conducted a national-scale quantitative review of the factors driving rates of bat collisions with U.S. wind turbines. To do this, I compiled a national bat collision database that included mortality data from 218 published and unpublished reports representing 100 wind facilities. I combined this data with meta-data on wind facility characteristics and with landscape variables derived from satellite imagery. To reduce biases associated with different studies using different methods, I ran all mortality data through a common statistical estimator, and I addressed the following objectives: (1) provide a descriptive summary of how bat collision mortality varies by species and season and (2) identify facility-related and landscape-level variables that can be used to predict bat-turbine collision mortality. Information from this analysis, which is based on the largest data set compiled to date, will be useful for developing wind facility siting guidelines, for informing placement decisions by wind energy companies, and to improve our understanding of the impacts of wind facilities on bat populations.

METHODS

Literature Search

I searched for published and unpublished literature related to bat mortality monitoring at wind facilities. Many of these reports were available from a previous analysis of bird collision mortality at wind farms in the contiguous United States (Loss et al 2013). To find additional reports, I used the search engine Google to search the terms “bat(s) and wind turbine(s),” ‘‘wind farm bat mortality,’’ ‘‘post-construction bat and bird mortality,” “bird and bat fatality turbine,’’ and “wind facility bird and bat mortality.” Google was used because most reports were unpublished and therefore not indexed in academic literature databases such as Google Scholar and Web of Knowledge. I included “birds” in some search terms because I found that many of the studies surveyed for, and reported, both bird and bat fatalities but did not include “bat” in the title. I initially observed that all reports located through Google were PDF documents, so I added “PDF” to the above search terms and repeated the same search process. In addition to the formal literature search, I directly accessed some reports listed in an online bibliography (Bats and Wind Energy Cooperative 2014) and contacted authors directly when reports were not hyperlinked on this site. I also received additional reports through personal contacts including Brianna Gary (New York State Department of Environmental Conservation), Jennifer Szymanski (U.S. Fish and Wildlife Service), and Julie Beston (USGS). Finally, I searched reference lists of acquired studies to find other relevant reports, and I used Google to search specifically for these titles.

Data Extraction

The above literature search resulted in a compilation of 218 published and unpublished reports representing 100 wind facilities across the U.S. and Canada—by far the largest data set compiled for this topic to date. Extracted and compiled meta-data included variables related to the wind facility and to survey methodology. Facility-level meta-data included the facility name, location, mean elevation, elevation range (i.e., maximum elevation minus minimum elevation), land cover type in the area surrounding wind turbines, number of turbines in the facility, turbine lighting system, turbine model, turbine type (i.e., monopole vs. lattice tower), height to turbine hub (i.e., nacelle), and height to upper turbine blade tip (i.e., maximum height). The extracted data include raw mortality records for more than 12,000 bats catalogued both by wind farm and species. For most reports (n=126), I was also able to extract the exact date each individual bat was found, and for a small subset of reports (n=12), I could extract age and sex information. Studies provided neither bias-adjusted estimates of mortality by species, nor species-level corrections for searcher efficiency or carcass persistence. Therefore, I did not generate bias-adjusted mortality estimates for each bat species. I did, however, compile descriptive summaries of the raw mortality data by species and month to illustrate major emergent patterns in bat mortality across the U.S. (Figures 2.1 - 2.3).

Inclusion Criteria

I standardized and reviewed all extracted data to the extent possible, and I applied inclusion criteria to minimize biases and therefore increase the rigor of my analyses. I only included data from mono-pole wind turbines (i.e., those with a solid tower) because: (1) lattice-style turbines (i.e., those with a hollow cage-like tower) have largely been phased out of use in the U.S. in favor of taller, higher energy producing monopole turbines, and (2) the vast majority of reports we reviewed were conducted at facilities that only had mono-pole turbines. Lattice turbines, because they provide perches for birds, may attract and cause high mortality rates of birds (Kerlinger, 2002; Orloff and Flannery, 1992; Osborn and others, 1998). However, the taller mono-pole turbines may be responsible for high bat mortality rates (Barclay 2007). This may be because the newer mono-pole turbines have longer blades that reach to a height through which many migratory bats travel (Barclay 2007).

The estimator I used (described in detail below) requires either the specific dates on which each turbine was surveyed or the average number of days between surveys. Therefore, I excluded reports that provided neither of these parameters nor associated information from which these parameters could be derived (i.e., specific beginning and end dates of the study in addition to the number of completed surveys during that period). Studies were also excluded if searches were reported to occur at varying intervals (e.g., 3.5-7 days), if the number of turbines searched was not reported, or if no methods were used to account for searcher efficiency and/or scavenger removal. Many studies used surrogate carcasses for bats (e.g., birds or mice) during experimental scavenger removal and searcher efficiency trials. The use of surrogate carcasses undoubtedly adds bias because both surveyor and scavenger detection of carcasses likely varies by the species used. However, I did not exclude data from studies that used surrogates because this would have reduced my sample size nearly to zero.

I excluded studies that searched turbines of more than one size or design (e.g. if both lattice and mono-pole turbines were searched) without reporting bat mortality separately for each group of turbines. I also excluded studies that only presented a range of searcher efficiency and carcass persistence rates and not a point estimate because the distribution of values in that range might be skewed and not accurately characterized by an average. If hub height or blade length was not listed in the study, I procured this information from a database of wind facility information associated with the recently released USGS windfarm map (Diffendorfer et al. 2014), and I excluded studies for which nacelle height and blade length information was still not available. After all inclusion criteria were applied, there remained 41 studies (representing 41 wind facilities) available for analysis.

Among the extracted data, there were many instances of multiple studies being conducted at the same wind farm. To ensure that only independent samples were included in my analysis, I chose one study to use for each facility based on which study had (in progressive order, only continuing to the next successive criterion if studies were the same with regard to the previous criterion): (1) the larger proportion of the year covered by surveys, (2) the larger number of turbines sampled, and (3) the larger number of total turbine surveys completed. If studies were equivalent with regard to all 3 of these criteria, I used the RandBetween function in Microsoft Excel to randomly select a study to use.

Accounting for Variable Seasonal Coverage of Sampling

Bat mortality is predominantly observed in the late summer and early fall (Johnson 2005, Kuntz et al. 2007a, Arnett et al. 2008, Baerwald 2014), and seasonal coverage of this peak period and the entire year was inconsistent among the studies meeting inclusion criteria. I therefore accounted for varying annual coverage of mortality surveys using a method developed for a similar analysis of bird-turbine collisions (J. Beston personal communication) to estimate year-round mortality for all wind facilities. First, I only considered studies that met the above inclusion criteria, except I allowed for multiple studies from the same facility if together they covered a year. Second, I used those above qualifying studies to estimate the average total observed bat mortality by month for that facility (Table 2.6). Third, using only studies that sampled throughout the entire year, I produced summed raw counts of monthly fatalities across all year-round studies. Fourth, I multiplied the summed total of fatalities for each month by the probability of finding a carcass at that wind facility (as described under "Variables incorporated into pfOngoingMortality"). Fifth, I used these corrected estimates of monthly mortality to determine the average percentage of mortality that occurred in each month of the year across all facilities. Sixth, I used these monthly average percentages to standardize mortality estimates for each partial-year study to a full-year estimate. For example, if a study sampled across 4 months that were estimated to capture 75% of annual mortality on average, I added 25% to generate the full year estimate. The method does not account for regional- and site-specific differences in bat abundance, timing of bat migration, and seasonal patterns of bat mortality, but making this adjustment is an improvement over the assumption that mortality does not occur during the unsampled remainder of the year.

Choice of Common Estimator

Methodological variation, including the choice of statistical estimator used to calculate mortality, can contribute as much variation to mortality rate estimates as does actual spatiotemporal variation in mortality rates (Smallwood 2013). Although, no meta-analysis can be completely free of bias, applying a single estimator to all mortality data removes the significant bias that arises from different studies using different estimators. To choose the specific estimator I would use, I ran all 41 datasets through several variations of a Markov Chain statistical estimator recently developed by Etterson (2013), as well as the Huso (2010), Erickson (2003a), and two Shoenfield (2004) estimators (Eqs. 1 and 1p). These are the estimators most commonly used to estimate bird and bat mortality rates at individual wind facilities.

To estimate the bias associated with each estimator, I conducted a replicated mortality simulation (repeated 100 times) in which the "true" amount of mortality was known and an average bias value was estimated for each estimator as the relative difference between true mortality and estimated mortality (Appendix Table 4). For the data that met my inclusion criteria, the Etterson estimators produced average bias values that were closer to zero (-0.05) than any other estimator, that were significantly lower than bias values for the Huso estimator (results of Tukey pairwise comparisons: $F=71.28$, $df = 7$; $p<0.001$), and that were statistically similar to bias values for the other estimators. All variants of the Etterson estimator performed equally well (mean bias for all = -0.05). Therefore, I chose the one that allowed me to declare whether a study covered a short or long period of time (+/-180 days), and that assumed search intervals were variable (hereafter referred to "pfOngoingMortality" estimator), because these assumptions most closely matched the methods under which my included data were collected.

I incorporated the raw extracted mortality data from each included study into pfOngoingMortality, with a slightly varied code (Appendix Table 1) that produces an output representing the probability that a dead bat that has fallen in the search area during regular or irregular survey intervals will be found by a searcher (hereafter, "pf"). The required variables for "pfOngoingMortality" are carcass removal rate per day, experimental searcher efficiency rate, the number of surveys conducted per turbine, the interval between surveys, and whether or not a "clean sweep" survey was conducted (i.e., a pre-season carcass search to remove carcasses deposited prior to the sampling period). If information about whether a clean sweep survey was not reported, I assumed none was conducted.

As described under “Data Extraction,” values for most of these estimator inputs had already been extracted from studies. However, to derive the carcass removal rate, I used information presented in the report (usually mean persistence time in days or persistence probability after a certain number of days) and converted it to a carcass removal rate (the inverse of carcass persistence, as it is usually reported) using either: (1) (1- persistence probability)^(1/carcass duration (days)) OR (2) 1/mean persistence time. (e.g., for (1) a study reporting a carcass persistence probability of 0.87 on the 4th day would have a daily scavenger removal rate of 1-0.87^(1/4) = 0.034; for (2) a study with a 31.9 day persistence time would have a daily scavenger removal rate of 1/31.9= 0.03).

Calculation of Bat Mortality Dependent Variables

To calculate annual per turbine and per megawatt (MW) mortality for each included wind facility, I first divided the total number of bats estimated to have been killed per year at the facility (based on the above adjustment for partial-year sampling) by pf. To estimate annual per turbine mortality, I then divided that value by the number of turbines searched at the facility. To calculate the annual per megawatt mortality, I divided the mortality per turbine/year by the stated MW capacity of each turbine. For example, a facility with estimated mortality of 32 bats per turbine/year and per turbine capacity of 1.5 MW, mortality per MW would be 21.3 bats per MW/year (i.e., 32/1.5).

Identification of Factors Influencing Bat Mortality Rates

To identify factors that explain variation in bat collision mortality rates, I conducted two model selection analyses, one with mortality per turbine per year as the dependent variable and one with mortality per MW per year as the dependent variable. Because distributions of per turbine and per MW bat collision mortality were not normal and included zero values, I (ln+1) transformed both dependent variables for the following model selection exercise. For both analyses, the statistical replicates were the full-year mortality rates (either per turbine or per MW) for each included wind facility data record. All analyses were conducted in Program R Version 3.1.2 “Pumpkin Helmet” (R Core Team 2014).

To rank alternative models, I used Akaike’s Information Criteria, corrected for small sample sizes (AICc, Burnham and Anderson 2002). I compared ΔAICc values, which capture the tradeoff between strong model fit and parsimonious model structure (ΔAICc=0 indicates the “best” model) and AIC weights (ωi), which indicate relative model support (ωi=1 indicates maximum support). I conducted correlation analysis to ensure that strongly correlated variables were not included in analysis. Turbine hub height and total height (height to blade tip) were the only strongly correlated variable pair (R= 0.9725), so I chose to use hub height because more wind facilities reported this variable and because this value is more commonly investigated and discussed in the literature.

For both per turbine and per MW mortality, I ran a series of single-variable linear regression analyses, one each for each for the following variables and one for the intercept-only (i.e., null) model. Variables assessed included the number of total turbines in the facility, turbine nacelle height, mean facility elevation, and elevation range in the facility. I also included several landscape-level percent land cover variables that were calculated at 500 meter and 1 kilometer scales surrounding wind facilities using ArcGIS (Environmental Systems Research Institute [ESRI], Redlands, California). Landscape variables calculated included 4 cover classes that were based on redefinitions of classes in the National Land Cover Database (NLCD, Homer et al. 2015), including: percent forest (includes deciduous, evergreen and mixed forest), percent scrubland (includes dwarf and shrub scrub), percent graminaceous (includes grassland/herbaceous, sedge/herbaceous, lichens, and moss), and percent agriculture (Appendix Table 3). I included only the four most common land cover types and did not include other land cover variables (e.g., open water, wetlands, and urban development) that comprise only a small proportion of the land cover across the data set. While there are many factors that affect mortality rates, I was limited by the variables documented in the reports available to me. My analysis illustrated strong support for a relatively small number of independent variables (see Results); therefore, I did not consider it necessary to construct and rank more complex multi-variable (either additive or interactive) models.

RESULTS

Descriptive Summary of Data

My literature review of bat mortality data from wind farms across North America resulted in a total of 218 studies from 100 facilities that span 39 years (1976 to 2014) and represent 26 states and 2 Canadian provinces (Fig 2.7). The raw mortality information summed across studies, provides the largest bat mortality data set compiled to date (12,532 total bat fatality records). The vast majority (82%) of bat fatalities have been found during the late summer and early fall period between the months of July and October (Figures 2 and 3). Bat mortality of nineteen species has been documented but three bat species comprise the vast majority of all documented bat fatalities, including the Hoary Bat (35%), the Eastern Red Bat (27%), and the Silver-haired bat (17%) (Figure 2.1).

Factors Influencing Bat Mortality Rates

Out of the thirteen candidate models for mortality per MW per year, the model containing percent graminaceous cover at the 500 meter scale was overwhelmingly the most strongly supported (ΔAICc >11 better than the next best model; ωi 0.992)(see Table 2.1 and Figure 2.4). Specifically, mortality per MW is estimated to decrease with increasing amount of graminaceous cover at the 500 m scale (coefficient estimate = -1.759, 95% CI: -2.882 to -0.636). Across the range of percent graminaceous cover values (at the 500 m scale) in my data set (0.000 – 0.986), mortality was estimated to decrease by 7.8%.

For the mortality per turbine per year analysis, two models received relatively equivalent strong support, the model with percent graminaceous cover at the 1 km scale (ωi 0.493; coefficient estimate = -1.3319; 95% CI = -2.298562 to -0.3651492) (Figure 2.5) and the model with percent graminaceous cover at the 500 meter scale (ωi 0.362; coefficient estimate = -1.3910; 95% CI = -2.357125 to -0.4248542)(Table 2.2; Figure 2.6). As indicated by coefficient confidence intervals that do not overlap zero, the inverse relationship between graminaceous cover and per turbine mortality appears to be biologically important at both spatial scales. Across the range of percent graminaceous cover values (at the 1 km scale) in my data set (0.000 – 0.987%), mortality was estimated to decrease by 7.64%. At the 500 meter scale mortality was estimated to decrease by 6.23% between low and high values of graminaceous cover (0.000 – 0.986%).

DISCUSSION

While isolating for a major source of bias, I found that bat mortality per turbine and per MW are both inversely correlated to graminaceous cover in the area immediately surrounding (500m – 1km) wind turbines. Using the most extensive data set compiled to date I confirmed that most mortalities (82%) occur between July and October, and are primarily composed of migratory tree roosting bats, likely returning from their breeding grounds in Canada. None of these species is classified as threatened or endangered in the U.S. Population size and demographic trends are only available for the Eastern Red Bat (Vonhof and Russell 2015), so interpreting significance of turbine caused mortalities is difficult. As wind development continues to grow, fatalities and thus the potential for biologically-significant impacts to local populations increases (NAS 2007; Erickson et al. 2002; Manville 2009).

Factors Influencing Bat Mortality Rates

The results of my analysis indicate that percentage cover of grassland in the area surrounding wind turbines is inversely related to bat collision mortality. There are at least three potential non-mutually exclusive explanations for this relationship. First, the relationship could have an underlying geographic explanation, with facilities that have a large proportion of graminaceous vegetation in the landscape also tending to occur in a particular region characterized by one or more factors that limit mortality rates. Qualitative assessment of the data roughly supports this explanation, as the 15 sites with the most graminaceous cover were all in the western half of the U.S., although covering a variety of states and regions including the southern Plains (Texas), Pacific Northwest (Oregon and Washington), interior west (Wyoming), and California. The sites with the least graminaceous cover were more scattered, including the northeast (Maine), Appalachian Mountains (Pennsylvania, New York, Maryland, and West Virginia), upper Midwest (Illinois and Wisconsin), interior west (Wyoming), and California. However, these high graminaceous cover sites were somewhat biased towards the Appalachian Mountains, with 10 of the 15 sites with least graminaceous cover in this region. Appalachian ridgelines are already known to be associated with some of the highest documented bat mortality rates (Hayes 2013, Arnett 2005).

Second, in addition to variation in the amount of graminaceous cover between regions, the general turbine placement locations relative to terrain and the type of turbine arrays vary somewhat predictably by region, and this could also influence bat collision risk. In much of the west, wind facilities are often spread across open flat areas, while in the east, strings of turbines are typically placed along ridgelines. Regional differences in the interactions between terrain type, turbine array, and bat abundance and movement patterns may all underlie the documented relationship between bat mortality and graminaceous cover. I did consider including a “region” model in our analysis, but did not because: (1) the bat species killed in greatest numbers have ranges that cover greater than half of North America (Harvey et al. 2011), and therefore transcend regional boundaries, and (2) in the U.S., there are not clearly defined and ecologically distinct regions containing similar bat assemblages as there are for birds (i.e., the so-called "Bird Conservation Region's;" US FWS 2008).

Third, bat abundance and/or bat mortality may simply be lower in areas with lower graminaceous cover, independent of region, turbine array, and terrain. Many bat species, including the tree-roosting species most affected by turbine collisions, are speculated to be either less abundant or less concentrated in broad grassy areas, especially during migration seasons (Baerwald and Barclay 2009, Johnson et al. 2004). Previous studies have generally reported lower bat collision rates at wind turbines in the interior of North America, but some exceptions do exist (Jain 2005; Piorkowski and O'Connell 2010). The pattern I documented may occur as a complex combination of the above explanations, and additional a priori-designed research would help parse out these potentially interacting predictors of bat mortality rates.

In a previous study summarizing data from nine proposed or active wind facilities, Baerwald and Barclay (2009) assessed geographic variation in bat activity at wind facilities and speculated that bat migration routes and travel distances are limited by the distance to appropriate stopover sites (trees) (Berthold 2001; Catry et al. 2004; Cryan and Veilleux 2007; Fleming and Eby 2003; Richardson 1978 & 1998). All facilities used in their analysis were relatively similar to each other with regard to landscape features (mostly in agriculture and native grass landscapes without trees), but bat activity was higher at sites closer to other landscape features (woodlands, rivers, and mountain foothills). If greater migratory activity occurs near these landscape features, it follows that bat mortalities would be similarly associated. Baerwald and Barclay (2009) also suggested that wind facility size (i.e., number of turbines) was a potential correlate of observed bat mortality rates. They suggested that bats may be attracted to turbines (Cryan 2008, Cryan and Barclay 2009, Kuntz et al 2007) but that larger facilities may experience a "dilution" effect of fewer mortalities per turbine. I found no support for the importance of wind facility size for explaining bat mortality rates.

Barclay et al. (2007) found that some variation in bat mortality was explained by turbine height, with taller turbines killing more bats. A relationship between structure height and collision rate has repeatedly been found for birds—including at wind turbines (Loss et al. 2013), buildings (Loss et al. 2014), and communication towers (Longcore et al. 2012). Additionally, radar and other studies support that taller wind turbines reach into the airspace used by a large number of migrating bats (Fiedler 2004, Mabee and Cooper 2004, Plissner et al. 2006). I found no support for the importance of turbine height in explaining bat mortality rates. Barclay et al. (2007) included data from older and significantly shorter lattice-style turbines that were excluded from my analysis for being incomparable to most current facilities in my dataset and for largely being phased out of use in the U.S. Thus, my finding of no effect of turbine height may have arisen if all of the mono-pole turbines in my data set (which included turbines ranging from 38 to 100 meters in height) were above a height threshold beyond which bats generally use the airspace. This equal use of the airspace may not actually occur, as bats are known to fly at significantly higher heights than the tallest turbines in my dataset (Mabee and Cooper 2004). Whether turbine height will become more important as new turbines extend even higher remains unclear. New turbines are proposed to be as tall as 140 meters (Zayas et al. 2015). Additional research to address this question will be necessary once mortality monitoring has been conducted at the new larger turbines. Regardless of the relationship between mortality and turbine height, the qualitative patterns emerging from the literature—and supported by the descriptive component of my analysis—demonstrate that bat mortality rates and the species most affected, differ by geographical region, and that bat mortality rates are generally higher than bird mortality rates (Smallwood 2013, Barclay et al. 2007, Arnett and Baerwald 2013).

Limitations and Research Needs

I found that percent graminaceous cover was the most important variable regardless of whether the dependent variable was mortality per megawatt or mortality per turbine. This further supports the robustness of my finding. Ideally, I would have used a more refined denominator that represents the actual time turbines are in operation and/or the volume of airspace at risk, variables that would better reflect variation in the actual risk of collision across locations and time. Potential variables to use would include: (1) mortality per amount of actual energy generated (e.g., in gigawatt-hours), a variable that better reflects the time of turbine operation than the potential MW of generating capacity, or (2) energy generated per area or volume of airspace affected by turbine blades, a variable that allows comparability of turbines of different sizes while also accounting for actual operation time. Use of these denominators is limited by lack of public availability of energy generation outputs, information that is often considered to be proprietary at the level of individual wind facilities (Johnson et al. in press).

As with any quantitative review, my results are limited by the availability and quality of the data used for analysis. I attempted to minimize biases by using rigorous data inclusion criteria that excluded studies that didn’t document methods clearly, account for various survey-related biases or conduct appropriate bias trials. However, even with these criteria specified, the data I used may not be representative of all U.S. wind turbines, wind facilities, field methods, statistical estimators, or bat collision mortality rates. Furthermore, the data included in my analysis were pooled from different years, and there may be significant interannual variation in mortality rates, both within individual wind facilities (e.g., Pearce-Higgins et al. 2012) or across regional and national scales as bat populations fluctuate. Despite controlling for many factors in the input data, some bias surely contributed to the analysis because the included studies still varied substantially with regard to their study design and data collection protocols. Nonetheless, my analysis greatly advances on previous understanding of the predictors of bat collision mortality at U.S. wind turbines.

CONCLUSIONS

My results provide some insight useful in future wind turbine placement decisions and suggest that wind facilities in areas with a high proportion of graminaceous cover experience reduced bat collision mortality rates. However, I do not suggest that placing turbines in intact grassland or shrubland ecosystems is necessarily ideal. Previous research has suggested that achieving energy development goals (US DOE 2008, Arnett and Baerwald 2013) may be possible even by placing new wind turbines in already-disturbed agricultural areas, and this may also contribute to reducing bat collision mortality by avoiding such high mortality areas as the eastern U.S. mountains. Ideally, ecologically informed decisions regarding placement of wind facilities should not only incorporate information from my analysis for bats but also include a weighting of the factors influencing bird collision mortality rates and the indirect habitat-related effects to birds, bats, and other wildlife species, as well as the overall impacts to the ecosystem.

Wind energy's ability to generate electricity without many of the environmental impacts associated with other energy sources (e.g., air pollution, water pollution, mercury emissions, and greenhouse gas emissions) could benefit bats and many other wildlife species. However, the adverse impacts of wind facilities on wildlife, including bat collisions with wind turbines, remain a significant conservation issue. Populations of many bat species are experiencing long-term declines, due in part to a fungal pathogen (white-nose syndrome), habitat loss, and numerous other anthropogenic impacts (deforestation, habitat loss, noise pollution) (Racey and Entwistle 2003, Winhold et al. 2008, Frick et al. 2010, Bunkley and Barber 2015). Bat fatalities at turbines raise additional concerns, and the issue may become even more pressing as the expansion of wind energy development is expected to continue throughout North America. In particular, turbines are projected to get larger and thus allow for efficient wind capture and energy generation in new regions (e.g., the southeastern U.S.) and at lower wind speeds (Zayas 2015). Further research will be needed to document correlates of mortality rates with this shifting turbine design and placement forecast.

Figure 2.1 Summed raw counts of bat mortality by species across North American studies at 100 wind facilities

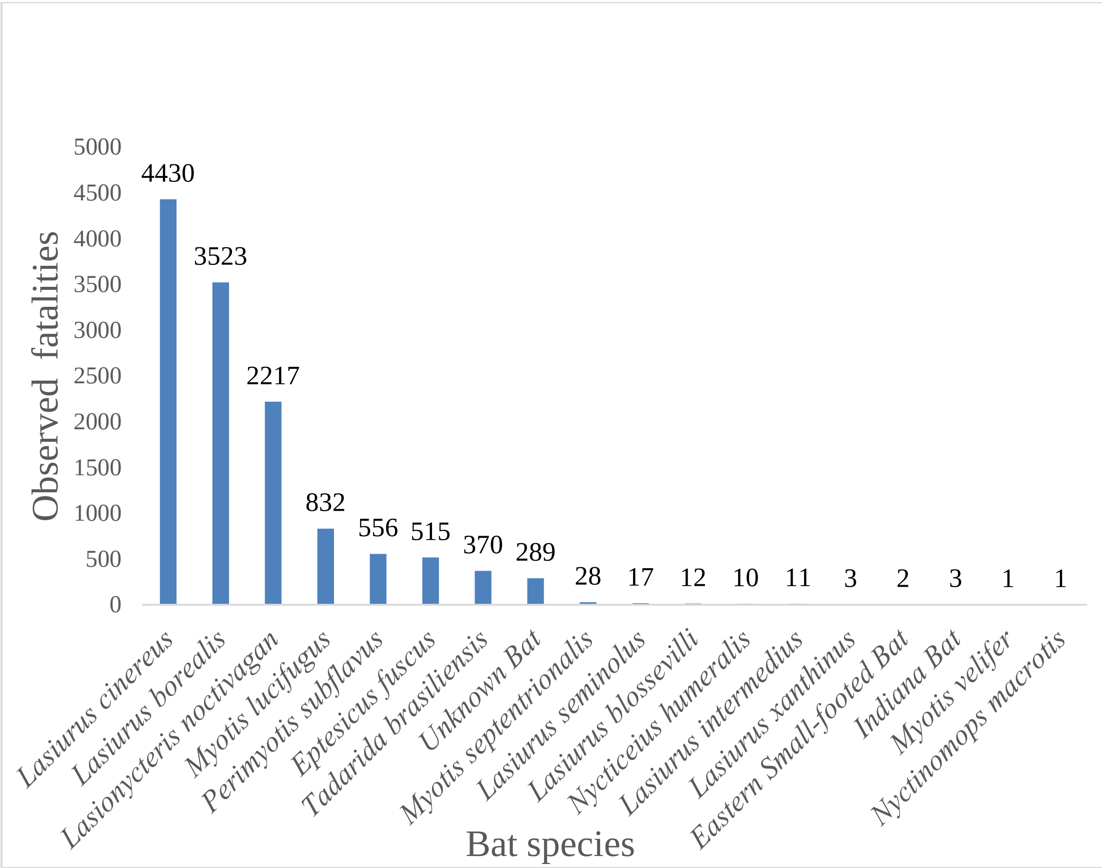


Figure 2.2 Counts of All Observed Bat Mortality by Month

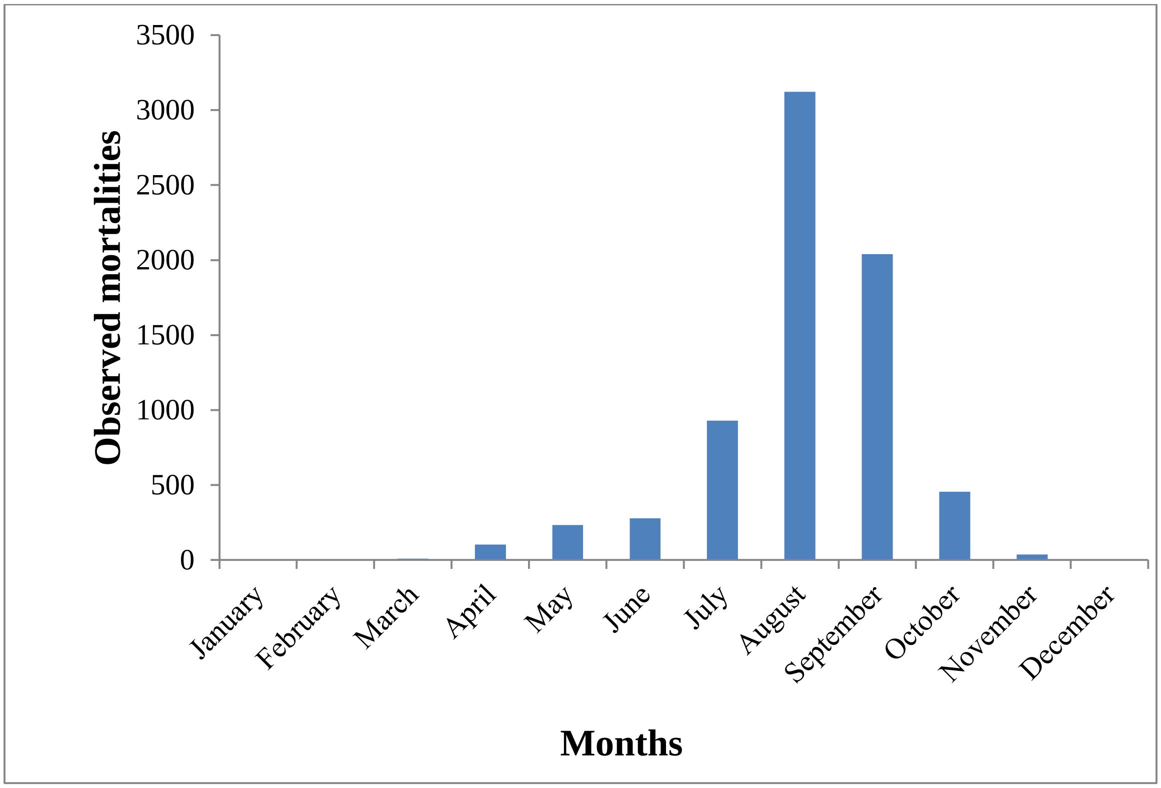


Figure 2.3 Raw Bat Mortality by Month among the Four Most Common Species

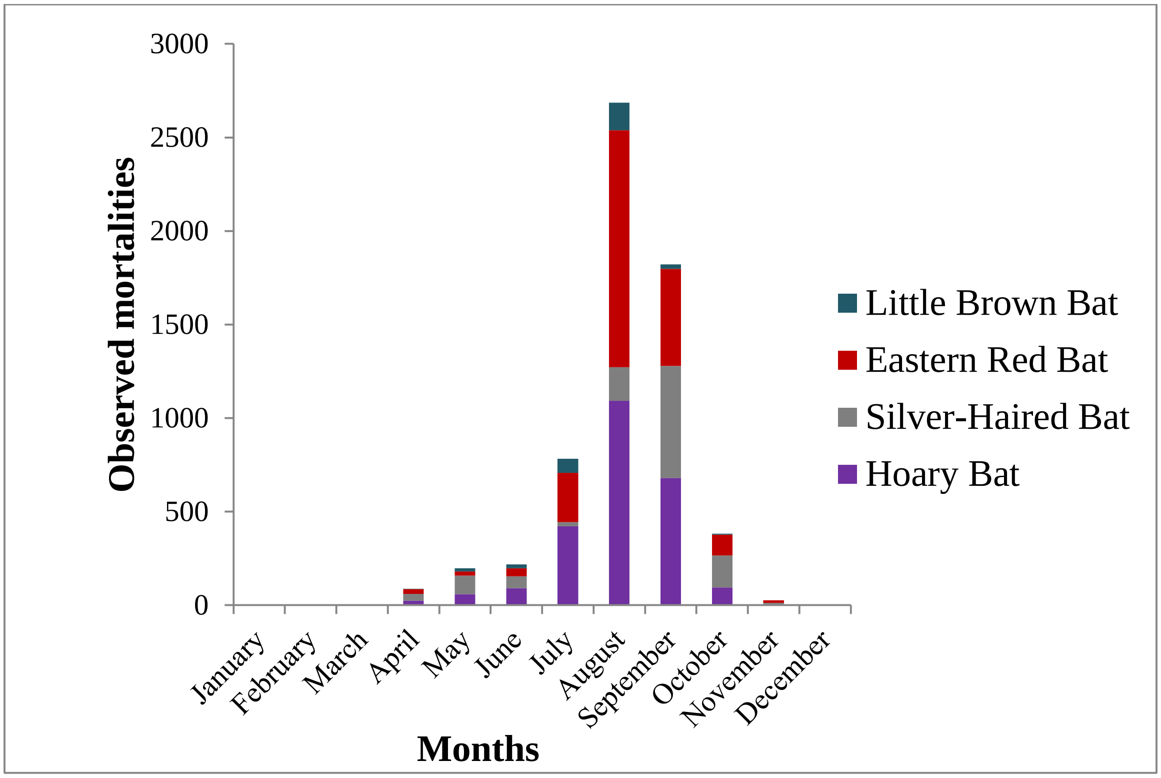


Figure 2.4 Relationship between bat mortality per megawatt per year and graminaceous cover at the 500 m scale surrounding wind turbines.

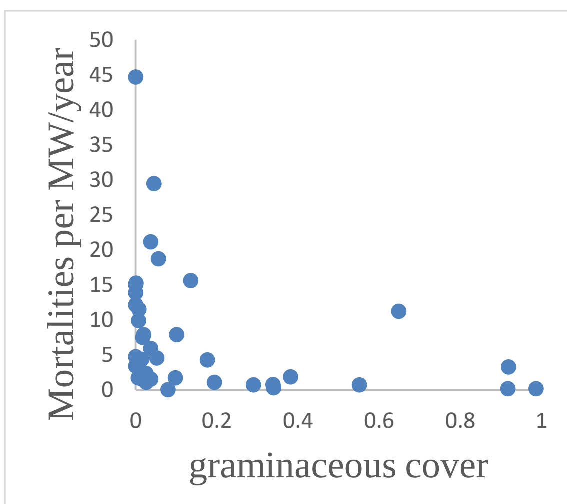


Table 2.1 Model selection results for analysis of factors related to bat collision mortality (per megawatt per year) as derived from Akaike's Information Criteria, corrected for small sample size (AIC_c). All candidate models include estimated mortality per megawatt per year as the dependent variable.

Candidate Model	K[a]	ΔAIC_c[b]	ω_i[c]
Graminaceous cover at 500 m	2	0.00	0.992
Graminaceous cover at 1 km	2	11.004	0.004
Mean elevation	2	11.765	0.003
Elevation change	2	15.265	0.000
Forest cover at 1 km	2	16.065	0.000
Forest cover at 500 m	2	16.391	0.000
Scrub at 500 m	2	20.303	0.000
Scrub at 1 km	2	20.325	0.000
Agriculture at 1 km	2	20.366	0.000
Agriculture at 500 m	2	20.373	0.000
Hub height	2	35.577	0.000
Null model	2	37.187	0.000
Facility size	2	37.550	0.000

[a] Number of parameters in the model

[b] Difference in AICc value between model and the most strongly supported model

[c] IAC weight – relative support for model

Figure 2.5

a. The relationship between bat mortality per turbine per year and graminaceous cover at a 1 kilometer radius around turbines.

b. The relationship between bat mortality per turbine per year and graminaceous cover at a 500 meter radius around the turbines.

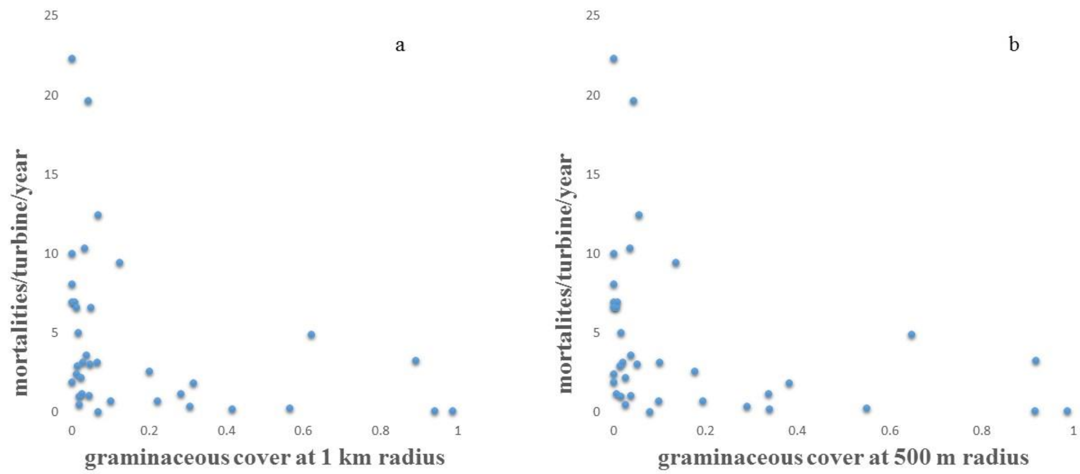


Table 2.2 Model selection results for analysis of factors related to bat collision mortality (per turbine per year) as derived from Akaike's Information criteria, corrected for small sample size (AIC_c). All candidate models include estimated mortality per turbine per year as the dependent variable.

Candidate Model	K[a]	ΔAICc[b]	ωi[c]
graminaceous cover at 1km	2	0.00	0.493
graminaceous cover at 500 m	2	0.617	0.362
forest cover at 1 km	2	4.208	0.060
forest cover at 500 m	2	4.642	0.048
scrub cover at 500 m	2	7.906	0.009
scrub cover at 1 km	2	7.946	0.009
agriculture at 500 m	2	7.998	0.009
agriculture at 1 km	2	8.005	0.009
mean elevation	2	18.888	0.00
elevation change	2	21.617	0.00
hub height	2	21.862	0.00
facility size	2	25.206	0.00
Null model	2	25.479	0.00

[a] Number of parameters in the model

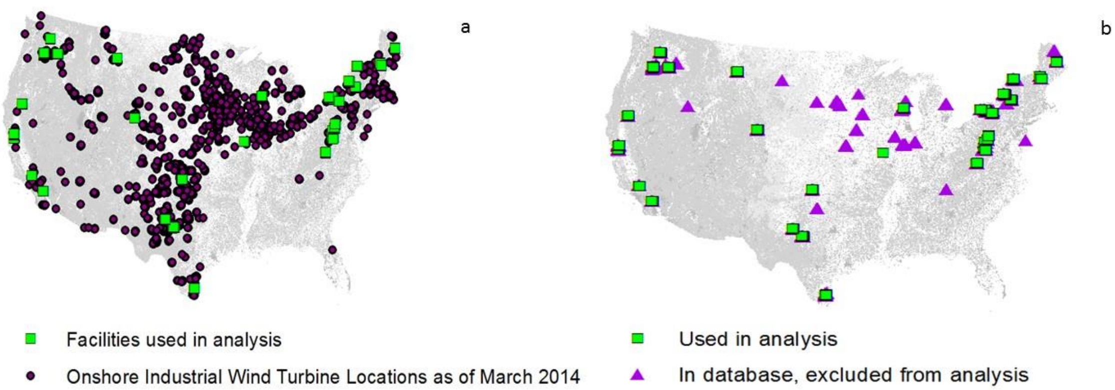
[b] Difference in AICc value between model and the most strongly supported model

[c] IAC weight – relative support for model

Figure 2.6

a Map of all wind facilities in the U.S. as of March 2014 (Diffendorfer 2014) and sites represented in this analysis

b Map of sites with studies catalogued in my database, and sites used in analyses



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CHAPTER III

INFLUENCE OF STUDY DESIGN, DATA COLLECTION, AND STATISTICAL ANALYSIS ON WIND FACILITY-SCALE ESTIMATES OF BAT MORTALITY

INTRODUCTION

Global installed capacity and production from all renewable technologies have increased substantially in the last decade, and supporting policies have continued their spread to countries in all regions of the world. In the United States, wind power added significantly more new electricity than any other resource sector in 2014 (Zayas et al. 2015). While wind energy is generally considered clean, there are direct and indirect effects to wildlife associated with wind development (Kunz et al. 2007a and b, Kuvlesky et al. 2007, Pruett et al. 2009, Kiesecker et al. 2011). Bat collisions with wind turbines are a significant conservation and management concern because they represent an additional source of mortality to already diminishing bat populations (Racey and Entwistle 2003, Winhold et al. 2008, Frick et al. 2010, Bunkley and Barber 2015). Wind turbines are a relatively new source of bat mortality and a global issue, as mortality has been documented almost everywhere wind turbines have been installed (Kunz et al. 2007, Rydell et al. 2012, Doty and Martin 2012, Hull and Cawthen 2013) and because the most common species found dead at wind turbines (Chapter 2) have ranges that cover much of North America. Bats are long-lived mammals with low reproductive rates; therefore their populations do not quickly rebound from large-scale impacts such as those posed by wind energy.

Wind facility-scale studies to document the amount of bat collision mortality (Arnett 2009, Grodsky 2012, Miller 2008, Piorkowski and O'Connell 2010) are important to provide a rough magnitude of the threat wind development poses to bats and for informing local siting decisions and mitigation approaches. However, gaining a large-scale understanding of cumulative population impacts to bats requires analysis of mortality data from multiple facilities that collectively span regional and continental scales. Such analyses are hampered by discrepancies among studies regarding study design approaches, data collection protocols, and the statistical methods used for estimating mortality (Kuntz 2007b, Warren Hicks et al. 2013, Smallwood 2007 & 2013). Post-construction mortality surveys vary greatly with regard to study duration, the number of turbines monitored and surveys conducted, and the way incidentally found carcasses (i.e., those outside of official surveys) are treated, among other factors. Facility-scale studies are also highly variable with regard to the methods used to account for biases that lead to underestimation of mortality rates (e.g., removal of carcasses by scavengers and observer inefficiencies) (Ellison 2012). For example, scavenger removal trials are variable with regard to duration, season, and species deployed. Similarly, experimental searcher efficiency trials are variable with regard to the number of carcasses deployed, number of test repetitions, species deployed and searcher expectation (whether or not the searcher knows they are being tested).

In addition to the above methodological variation, different studies employ different statistical estimators to combine mortality counts with scavenger removal and searcher efficiency rates into bias-adjusted mortality estimates. The most commonly used estimators include those developed by Huso (2010), Erickson (2003), Shoenfeld (2004), as well as so-called naïve estimators (i.e., several similar estimators that use fairly simple assumptions and calculations). Each estimator is characterized by a set of assumptions, which if violated, can lead to biased mortality rate estimates (reviewed by and examples below from Warren-Hicks 2013). For example, the Huso estimator assumes that a carcass missed on an initial search will not be found on subsequent searches. In practice, all carcasses found in mortality surveys are generally included in estimation regardless of time since death. This type of assumption violation contributes overestimation bias when intervals between surveys are short. Likewise, both the Shoenfeld and Erickson estimators assume that any carcasses missed on an initial search will have the same probability of discovery on all subsequent searches. Violation of this assumption contributes underestimation bias as carcass detection decreases with time (Smallwood 2007). Warren-Hicks et al. (2013) documented that both searcher efficiency and scavenger removal rates depend on time since death, a factor that has not yet been accounted for in industry-standard estimators. These and other contrasting assumptions (and the associated biases that arise when assumptions are violated) highlight the difficulty of comparing mortality rates among, and drawing large-scale inferences from, multiple studies that use different estimators. The above methodological variation can contribute as much variation to mortality rate estimates as does actual spatiotemporal variation in mortality rates (Smallwood 2013). However, no study to date has used a common statistical estimator to systematically assess how different methodological steps influence the percentage of carcasses found.

Several studies have assessed how mortality estimates for individual wind farms are affected by field biases such as imperfect detection and scavenging of carcasses (Smallwood et al. 2010; Huso 2010; Korner-Nievergelt et al. 2011). However, no research has taken a national multi-site approach to investigating how varying approaches to account for these biases, and to designing and conducting post-construction mortality surveys in general, influence facility-scale estimate bias and the proportion of total carcasses found. To conduct such an analysis, I completed a systematic review of the published and unpublished literature (including 218 studies representing 100 wind farms), extracted bat mortality data and methods meta-data. I implemented a common statistical estimator for all extracted data (Etterson 2013). My objectives were: (1) to identify methodological steps that lead to high proportion of carcasses being found, and (2) based on this analysis, and a simultaneous qualitative review of the literature, to make specific recommendations about how future studies can be conducted to result in minimally biased mortality estimates. My analysis is specific to bat collision mortality at wind turbines, but many of the recommendations also apply to studies of bird-turbine collisions and likely apply to studies of wildlife collisions at other types of manmade structures.

METHODS

Detailed Background on Methodological Variation Among Studies

As introduced above, many sources of methodological variation exist among different post-construction monitoring studies. Comparisons among bat mortality rates are therefore likely to be biased unless these numerous differences in approach are accounted for. Here, I provide further details and examples of this methodological variation to provide context for the variables extracted and analyzed in my quantitative review.

First, study durations and intensities (i.e., frequency of fatality surveys, size of search plots, and distance between transects) are highly variable. Post-construction mortality surveys were originally designed to monitor avian mortality rates with a focus on raptors. Many procedures from these original studies have been adopted for bats; however, these methods may be skewed toward detection of large carcasses (e.g., raptors) and under-detection of small carcasses (e.g., bats). Second, variation among studies in how incidental carcasses (i.e., those found outside of standardized search periods or outside of defined search areas) are handled produces an additional source of underlying variability in mortality estimates. Some protocols assume that all carcasses found incidentally on survey plots would have been found during the next scheduled standardized search. Other studies exclude incidental fatalities from estimates of mortality. Both of these approaches are likely to be biased because searcher efficiency trials generally indicate that fewer than half of such incidental carcasses would be found in any given standardized trial (Warren-Hicks et al. 2013).

Third, searcher efficiency trials vary in duration, intensity (i.e., how many replicate trials are conducted), and in the types of carcasses used. For bat mortality estimation, quail chicks, mice, and "small birds" (usually not described to species) are commonly used as surrogates. Casting doubt on the efficacy of using surrogate species, Warren-Hicks et al. (2013) found that observer detection of bats is roughly half that of small birds; therefore, using these, and potentially other carcass types, as surrogates for bats likely contributes bias to estimates of bat detectability and mortality. Because of concerns about the spread of disease and fungi, mice and small birds are often used as surrogates for bats.

Other factors not uniformly treated across studies include the season during which carcass persistence trials are conducted, the number of carcasses used in each trial, and whether the experimentally placed carcasses are frozen, freshly killed, or a mix of both. All of these factors can affect the rate of carcass persistence by affecting the rate of carcass decay and the likelihood that scavengers will be attracted to and able to locate the carcass.

Literature Search

I searched for published and unpublished literature on bat mortality monitoring at wind facilities. A large sample of reports were available from a previous analysis of bird collision mortality at wind farms in the contiguous United States (Loss et al 2013). To find additional reports, I used the search engine Google to search the terms "bat(s) and wind turbine(s)," ‘‘wind farm bat mortality,’’ ‘‘post-construction bat and bird mortality," "bird and bat fatality turbine,’’ and "wind facility bird and bat mortality." Google was used because most reports were unpublished and therefore not indexed in academic literature databases such as Google Scholar and Web of Knowledge. I included "birds" in some search terms because I found that many of the studies sampled for and reported bat fatalities but did not include “bat” in the report title. I also found that all reports located through Google were PDF documents, so I added “PDF” to the above search terms and repeated the same search process. In addition to the Google search, I directly accessed some reports listed on an online bibliography (Bats and Wind Energy Cooperative 2014) and contacted authors directly when reports were not hyperlinked on this site. During the research, I also became aware of colleagues with access to additional reports, including Brianna Gary (New York State Department of Environmental Conservation), Jennifer Szymanski (U.S. Fish and Wildlife Service), and Julie Beston (USGS). Through these three women I acquired many new reports. Finally, I searched reference lists of acquired studies to find other relevant reports, and I used Google to search specifically for these titles.

Data Extraction

The above search resulted in a compilation of 218 published and unpublished reports representing 100 wind facilities across the U.S. and Canada—by far the largest data set compiled on this topic to date. Extracted and compiled meta-data included variables related to survey methodology, experimental scavenger removal, searcher efficiency trial methodology, and post-survey statistical methods used to estimate mortality rates.

Extracted mortality data included both raw bat fatality counts and estimated mortality rates (per megawatt and per turbine, if reported) for each study. Estimated mortality rates generally are based on combining raw mortality data with values of bias adjustments and in some cases adjustments for un-sampled time periods and/or turbines, though methods vary. Raw mortality records consist of over twelve thousand recorded bats by farm and species. Records associated with most reports (n=126) provide exact dates each individual was found, and a small subset (n=12) provide age and sex information.

In addition to extracting overall bat mortality rate estimates for each study, I extracted data for individual bat species. However, because studies do not provide estimates of mortality by species, nor species-level corrections for searcher efficiency or carcass persistence, I did not generate species-specific mortality estimates. I did, however, compile descriptive summaries by species and month to illustrate major emergent patterns in bat mortality across the U.S. (Chapter 2 Figures 2.1 – 2.3).

Inclusion Criteria

I standardized and reviewed extracted data to the extent possible, and I applied inclusion criteria to minimize biases and therefore increase the rigor of my analyses. I included only mono-pole wind turbines because: (1) lattice-style turbines have largely been phased out of use in the U.S. in favor of taller, higher energy producing turbines and (2) the vast majority of reports reviewed were conducted at facilities that only had monopole turbines. Lattice turbines, because they provide perches for birds, may attract and cause higher mortality rates of birds (Kerlinger, 2002; Orloff and Flannery, 1992; Osborn and others, 1998). However the taller mono-pole turbines may be responsible for the increased mortality of bats (Barclay 2007). It has been suggested that this is because the newer turbines, with longer blades, reach up to a height at which migrants travel (Barclay 2007).

The estimator I used in the below analysis required either the specific dates on which each turbine was visited or the average number of days between searches. Therefore, I excluded reports that provided neither of these parameters nor associated information from which these parameters could be derived (i.e. specific begin and end dates in addition to the number of completed searches). Studies were also excluded if searches were reported to occur at varying intervals (e.g., 3.5-7 days), if the number of turbines searched was not reported, or if no methods were used to account for searcher efficiency and/or scavenger removal. Many studies used surrogate carcasses. This adds bias. However, I included these studies because removing them would have reduced my sample size nearly to zero.

I excluded studies that searched turbines of more than one size or design (i.e., both lattice and monopole turbines were searched) without reporting bat mortality separately for each. I also excluded reports that only presented a range of searcher efficiency rates and carcass persistence rates because the distribution of values in that range might be skewed and an average might not be an accurate characterization. After all inclusion criteria were applied there were 41 studies available for analysis.

Among the extracted data, there were many instances of multiple studies being conducted at the same wind farm. To ensure that only independent samples were included in my analysis, I chose which study to included based on (in progressive order, only continuing to the next successive criterion if studies were the same with regard to the previous criterion): (1) proportion of the year covered by sampling, (2) number of turbines sampled, and (3) number of total turbine surveys completed. If studies were equivalent with regard to all 3 of these criteria, I used the RandBetween function in Microsoft Excel to randomly select which study to use.

Choice of Common Estimator

Methodological variation, including the choice of statistical estimator used to calculate mortality, can contribute as much variation to mortality rate estimates as does actual spatiotemporal variation in mortality rates (Smallwood 2013). Although, no meta-analysis can be completely free of bias, applying a single estimator to all mortality data removes the significant bias that arises from different studies using different estimators. To choose the specific estimator I would use, I ran all 41 datasets through several variations of a Markov Chain statistical estimator recently developed by Etterson (2013), as well as the Huso (2010), Erickson (2003a), and two Shoenfield (2004) estimators (Eqs. 1 and 1p). These are the estimators most commonly used to estimate bird and bat mortality rates at individual wind facilities.

To estimate the bias associated with each estimator, I conducted a replicated mortality simulation (repeated 100 times) in which the "true" amount of mortality was known and an average bias value was estimated for each estimator as the relative difference between true mortality and estimated mortality (Appendix Table 4). For the data that met my inclusion criteria, the Etterson estimators produced average bias values that were closer to zero (-0.05) than any other estimator, that were significantly lower than bias values for the Huso estimator (results of Tukey pairwise comparisons: $F=71.28$, $df = 7$; $p<0.001$), and that were statistically similar to bias values for the other estimators. All variants of the Etterson estimator performed equally well (mean bias for all = -0.05), therefore, I chose the one that allowed me to declare whether searches were conducted over a short (less than 180 days) or long (180 days or greater) period of time, and assumed that search intervals were variable (hereafter referred to "pfOngoingMortality" estimator) because these assumptions most closely matched the methods under which my included data were collected.

I incorporated the raw extracted mortality data from each included study into "pfOngoingMortality", with a slightly varied code (Appendix Table 1) that produces an output representing the probability that a dead bat that has fallen in the search area during regular or irregular survey intervals will be found by a searcher (hereafter, "*pf*"). The required variables for "pfOngoingMortality" are carcass removal rate per day (the inverse of carcass persistence, as it is usually presented), experimental searcher efficiency rate, the number of surveys conducted per turbine, the interval between surveys, and whether or not a "clean sweep" survey was conducted (a pre-season carcass search to remove carcasses deposited prior to the sampling period). If information about whether a clean sweep survey was not reported, I assumed none was conducted.

As described under “Data Extraction,” values for most of these estimator inputs had already been extracted from studies. However, to derive the carcass removal rate, I used information presented in the report (usually mean persistence time in days or persistence probability after a certain number of days) and converted it to a carcass removal rate using either: (1) persistence probability)^(1/duration of carcass (days)) or (2) 1/mean persistence time. (e.g., for (1) a study reporting a carcass persistence probability of 0.87 on the 4th day would have a daily scavenger removal rate of 1-0.87^(1/4) = 0.034; for (2) a study with a 31.9 day persistence time would have a daily scavenger removal rate of 1/31.9= 0.03).

Identification of Methodological Factors Influencing Probability of Finding a Bat Carcass

To identify factors that explain variation in bat collision mortality rates, I conducted a model selection analysis with pf as the dependent variable. All analyses were conducted in Program R Version 3.1.2 “Pumpkin Helmet” (R Core Team 2014). To rank alternative models, I used Akaike’s Information Criteria, corrected for small sample sizes (AICc, Burnham and Anderson 2002). I compared ΔAIC_c values, which capture the tradeoff between strong model fit and parsimonious model structure (ΔAIC_c=0 indicates the “best” model) and AIC weights ($\omega_{i,}$), which indicate relative model support (ω_i=1 indicates maximum support). I ran a series of single-variable linear regression analyses, one each for each for the following variables and one for the intercept-only (i.e., null) model. Variables assessed included interval between visits, number of searches per turbine, searcher efficiency, search radius, transect width, and a null model. While there are many factors that affect carcass detection, I was limited by the variables documented in the reports available to me. I then investigated these strongly-supported models for the nature of relationships to assess how different study design approaches contribute variation to facility scale mortality estimates. This analysis illustrated strong support for one independent variable (see Results); therefore, I did not consider it necessary to construct and rank more complex multi-variable (either additive or interactive) models.

RESULTS

Methodological Factors Influencing Probability of Finding a Bat Carcass

My literature review of bat mortality data from wind farms across North America resulted in a total of 218 studies from 100 farms that span 38 years (1976 to 2014) and represent 26 states and 2 Canadian provinces (Fig 2.7). Studies included for the analysis (n=41) were highly variable with regard to the candidate variables we assessed. Search interval varied from 1 to 30 days (average = 9.78 days; mode = 7 days). Number of searches per turbine varied from 7 to 258 (average = 48.71 modes = 7 and 9). Transect width varied from 3.5 to 11 meters apart (average = 6.09 mode =5). Radii ranged from extending 20 meters from the turbine to 125 meters (average = 74.38, mode = 60). Searcher efficiency rates varied from 23 to 90 percent (mean = 52.3%, mode = 50%).

Despite variation among studies in all of the candidate variables assessed, the model containing interval between mortality searches received the strongest support by far for explaining the probability of finding a carcass (ΔAIC_c >3 better than the next best model; ω_i 0.832; Table 3.1 and Figure 3.1). As evidenced by the coefficient estimate (coefficient estimate = 0.017; 95% CI = -0.025 to -0.008), the probability of finding a carcass decreases with increased time between searches (Figure 3.1). Across the entire range of search intervals in my data set (1 to 30 days), *pf* was estimated to increase 7.18% . Although receiving much less support than the search interval model, number of surveys per turbine also received some support in our model selection analysis (ΔAIC_c = 3.331 and >6 better than the next best model; ω_i 0.158, Table 3.1 and Figure 3.2).

Many of the variables considered as variables affecting probability of finding a carcass are also inputs into the equation estimating probability of finding a carcass. So there is no surprise that they are related to, and influence, the outcome. This investigation can be viewed as a sensitivity analysis - and the results are useful in assessing what changes in methodology would have the greatest impact.

DISCUSSION

The Importance of Short Search Intervals

My assessment of methodological influences on carcass discovery rates suggests that the most influential factor driving the probability of finding a bat carcass under a wind turbine is the frequency of mortality surveys. This finding is particularly useful because the recommendation that arises from it—to decrease search interval if the objective is to increase *pf*—can be easily implemented in future studies of bat collisions at wind turbines. Based on the relationship (Figure 3.1), this benefit appears to occur across all ranges of search intervals, including whether shortening from a 4 week to a 1 week interval or when shortening from a 2 day to a 1 day interval.

Shortening the search interval of mortality monitoring surveys is more likely to be immediately successful than other methodological steps to increase the proportion of carcasses found. For example, searcher efficiency rates could be indirectly improved by training searchers; however, this training may be time intensive and have varying effectiveness depending on the searcher, the carcass species, and the type of substrate searched. The search radius of mortality surveys could also be expanded, but generally, a larger search radius is more likely to reach into areas that may otherwise have been excluded because of relatively low carcass detectability rates (e.g., due to high tree density) or safety concerns (e.g., steep or uneven terrain). Unlike these other changes, increasing the frequency of carcass searches simply requires changing the schedule of fieldwork. This will almost always necessitate increased cost to accommodate the increased person-hours spent searching, but unlike alternative steps, the benefit on cost should be seen in nearly all cases and almost immediately upon implementation. Furthermore, simultaneously reducing the search interval and reducing costs for other methodological steps that have less influence on *pf* may still result in a net overall improvement in the proportion of carcasses found. Further replicated and controlled experimental research to quantify *pf* under varying field methods and in varying regions and land cover contexts would contribute further insight into the development of optimal search protocols.

In addition to increasing the proportion of total carcasses found, increasing the frequency of searches also provides at least three other benefits. First, as evidenced by a multi-study quantitative review of bird-turbine collisions, short search intervals lead to quicker identification of the full range of species killed at wind turbines, including rare species that are infrequently killed but more likely to experience population level impacts from wind turbine collisions (Beston et al. 2015). Thus, increasing search frequency could have significant implications for quantifying the presence and rate of mortality for imperiled species and for informing their management and conservation. Second, even though most studies quantify and account for scavenger removal of carcasses and searcher detection rates, these corrections may become less accurate over long search intervals because both scavenger removal and searcher detection may be highly variable depending on the time since an animal died (Warren-Hicks 2013) and this variation is generally not accounted for in post-construction mortality studies. Thus, even bias-corrected estimates may become more biased with longer search intervals, and short search intervals should effectively control for the above age-related variation. Third, studies increasingly indicate that the statistical estimators used to combine the above bias estimates with raw mortality counts may themselves be less biased at short search intervals (Korner-Nievergelt et al. 2011; Smallwood 2013). This likely occurs because most statistical estimators make simplified assumptions about the temporal distributions of mortality (for studies of turbine collisions, usually that mortality rates are constant through time), when in reality, mortality is typically highly variable and often occurs in a few major episodic events that are not captured by most sampling schemes (Etterson 2013). Short-interval monitoring would be more likely to capture this temporal variation, would preclude the reliance on assumed temporal models by allowing actual data to replace assumptions, and would better inform assumptions about temporal models of mortality for studies that can't search on short survey intervals. With frequent searches the relationship between mortality rates and weather data (Cryan and Brown 2007, Rydell et al. 2010) could be analyzed to inform mitigation measures, such as seasonal shut-downs or modifications to cut-in speed (Arnett et al. 2008).

Potential Influence of Other Methodological Factors on Mortality Estimation Bias

Numerous methodological practices that are common to many mortality monitoring studies result in biases that have largely remained unassessed and unaccounted for to date. These unaccounted for factors contribute bias to my (and all other) analyses that combine data from multiple studies. Unfortunately, I was unable to assess the impact of most of these methodological steps on pf because not enough studies meeting my inclusion criteria provided information to have sufficient replication of these potential independent variables. For example, the use of surrogate species in scavenger removal and searcher detection trials (e.g., using: mice, small birds, chicks, or an undocumented mix of many carcass types instead of bats) is variable and can be unrepresentative of the decay, scavenger removal, and searcher efficiency rates of the species they are meant to represent (Warren-Hicks 2013, Arnett 2008 & 2009, Kerns 2005, Smallwood 2007). The use of frozen or frozen then defrosted carcasses in carcass persistence trials may reduce the ability of scavengers to locate a carcass (Smallwood 2007, Kerns 2005). Vegetation height and density can influence searcher efficiency rates, so detection bias trials conducted in one space and time may not apply to others (Warren-Hicks 2013, Arnett 2005 & 2006, Smallwood 2007). Some studies use dogs in addition to human surveyors, and this step has been shown to increase the probability of finding carcasses and to reduce detection variability between individual surveyors (Arnett 2006, West (2010) Gulf Wind, Stantec (2009) Munnsville).

In addition to the above ways in which studies differ in the methods employed, the statistical estimators used are also highly variable and vary in efficacy depending on many factors related to many other protocol procedures (Warren-Hicks 2013, Huso 2010, Smallwood 2007). As a result of all of these variations among studies, caution is encouraged when making comparisons of bat mortality estimates between studies (Warren-Hicks 2013, BHE Environmental 2011, Smallwood 2007).

Conclusions and Management Recommendations

All of the above types of methodological variation complicate comparisons in mortality estimates among wind facilities and extrapolation of results to a regional and national scale. Additionally, interpretation of the biological importance of these estimates is limited by a relative lack of information about bat populations and life histories (Librandi-Muma 2012; Cryan 2011; Kuntz 2007b). In particular, more information is needed on species' migratory routes and migratory behavior to better make facility siting decisions (Librandi-Muma 2012, Larkin 2006, Reynolds 2006, Cryan and Brown 2007, Johnson et al 2011a & 2011b, Cryan 2011, Kuntz 2007b). Finally, data sharing and increased transparency in the industry would promote more informed decision making and facilitate sharing of ideas and productive methodological shifts (Smallwood 2007, Kuvlesky 2007, Loss 2013, Vonhof 2015).

Standardization of study design and data collection protocols across studies would reduce biases in cross-study analyses and provide improved insight into the impacts of wind turbine collisions on bat populations. My *quantitative* review—based on the largest data set compiled to date—suggests that reduction of search intervals, and perhaps standardization to short intervals (e.g. twice per week) may provide the greatest improvements in the proportion of carcasses found and therefore in our understanding of both local and large-scale impacts to bats. However, based on my *qualitative* observations of the hundreds of reviewed post-construction mortality reports, I am able to recommend additional methodological and data organization suggestions that have either been: (1) quantitatively shown to reduce mortality estimation bias in previous studies, (2) or hypothesized to be important contributors to estimation bias and should be empirically tested in the future (see Table 3.2 for specific recommendations).

Fatalities of bats at wind turbines remain a significant conservation issue, especially given the important ecosystem services they provide. The combined losses of bats from wind energy development and other anthropogenic threats, such as white-nose syndrome (Frick et al. 2010), could cost the North American agricultural sector billions of dollars in insect pest control services each year (Boyles et al. 2011). To fully understand the ecological, economic, and social significance of bat collision mortality and to minimize any adverse impacts, we must know both the total number of dead bats and the population level impacts of these deaths for individual species. Data collected in post-construction mortality monitoring is integral to this process and provides a unique and valuable resource for conservation efforts. The value of this resource could be even further amplified through standardization of methods and adoption of my, and previous researchers', study design and data collection recommendations related to shortened search intervals and other methodological steps.

Table 3.1 Model selection results for analysis of factors related to probability of finding bat carcasses as derived from Akaike's Information criteria, corrected for small sample size (AICc). All candidate models include pf as the dependent variable

Candidate Model	**K[a]**	**ΔAICc[b]**	**ωi[c]**
Interval (days)	2	0.000	0.835
Searches per Turbine	2	3.331	0.158
Searcher efficiency	2	9.731	0.006
Null Model	2	14.477	0.001
Search Radius	2	14.474	0.001
Transect Width	2	36.345	0.000

[a] Number of parameters in the model

[b] Difference in AICc value between model and the most strongly supported model

[c] IAC weight – relative support for model

Figure 3.1

a Relationship between probability of finding a carcass and interval (in days) between searches, the top model selected in AICc analysis.

b Relationship between probability of finding a carcass and searches per turbine, the second strongest candidate model in selected in AICc analysis.

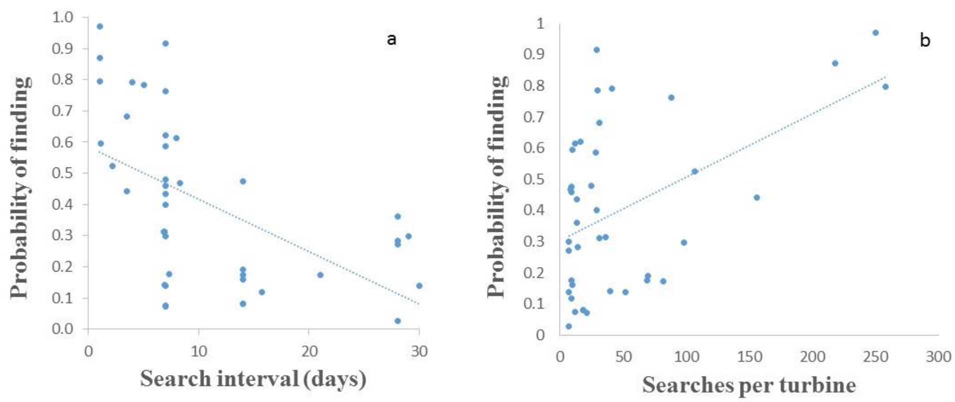

Table 3.2 Specific recommendations for and observations of post-construction monitoring studies to improve our understanding of impacts to bats. Recommendations have either been quantitatively supported or are hypothesized to reduce bias (see citations).

Recommendation	Relevant Citations
Consider that the use of statistical estimators can vary in efficacy depending on local context and in some scenarios may produce unacceptable levels of bias	Warren-Hicks (2013) Huso (2010) Smallwood (2007)
Given current methodological variation, use caution when comparing mortality estimates between wind facilities	Warren-Hicks (2013) BHE Environmental (2011) Cedar Ridge Smallwood (2007)

Increase survey design and data collection protocol consistency across studies	Warren Hicks (2013) Arnett (2007) Kuntz (2007b) Kuvlesky (2007) BHE (2011) Cedar Ridge Smallwood (2007)
Standardize bias trials (with regard to number of carcasses deployed)	Smallwood (2007)
Improve data sharing/transparency and data pooling/repositories	Smallwood (2007) Kuvlesky (2007) Loss (2013) Vonhof (2015)
Observations	**Relevant Citations**
Surrogate carcass species contribute positive and negative bias to search efficiency and scavenger trials	Warren-Hicks (2013) Arnett (2008) & (2009) Kerns (2005) Smallwood (2007)
Use of dogs to complement human surveyors reduces variability of carcass detection and increase likelihood of finding carcasses	Arnett (2006) West (2010) Gulf Wind Stantec (2009) Munnsville
Carcass age affects search efficiency (increased time since death reduces likelihood of finding)	Warren-Hicks (2013) Smallwood (2007)
Carcass persistence changes over time, as local scavengers learn about novel food source	Arnett (2005)
Freezing of experimentally placed carcasses can reduce scavenger attraction and increase persistence times	Smallwood (2007) Kerns (2005)
Increased vegetation height can significantly reduce searcher efficiency	Warren-Hicks (2013) Arnett (2005) & (2006) Smallwood (2007)
Researchers have no control over carcass persistence times, and little control over searcher efficiency, but they do have control over search interval	Huso (2010)

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CHAPTER IV

CONCLUSION: IMPLICATIONS AND SIGNIFICANCE

Because collisions with wind turbines are an additional source of mortality to already diminishing bat populations, they are a significant conservation and management concern. Facility-scale mortality studies are important, though not the final step if we want to know the cumulative impacts of collision morality on bat populations. Comparisons among post-construction mortality monitoring studies in North America are difficult given the discrepancies among study methodologies and statistical methods used for estimating mortality. The use of a common estimator (Etterson 2013) applied to raw data from across the continent allowed for an assessment of current survey methodology while isolating a significant source of bias. My systematic quantitative analysis of the factors that drive bat collision mortality should contribute to a broader understanding of how bat species are being impacted directly by the growing wind energy industry. My systematic assessment of monitoring approaches that lead to the least biased estimates of mortality should help streamline efforts to ameliorate those impacts.

The vast majority of species killed at wind facilities are migratory tree-roosting bats. Because this group of bats is comprised primarily of solitary tree dwellers that do not hibernate (and therefore do not congregate in easily observable groups) it has been difficult to develop suitable field methods to estimate their population sizes (Carter et al. 2003, Schorr et al. 2014). As a result of this lack of information about population sizes, the population-level impacts of wind turbine collisions are highly uncertain for tree-roosting bat species. Not only are there virtually no reliable current or historical population estimates available for common bat species

(Schorr et al. 2014), there is also little known about bat migratory pathways and habits (e.g., exact timing of migration, habitat use during migration, etc.). Long distance migratory species are being killed by wind turbines in the greatest numbers, so understanding what routes they take, what landscape features are of significance to them, and under what conditions they fly will make efforts to reduce the negative impact of wind facilities more effective and more efficient.

To add gravity to estimates of bat mortality at wind facilities, additional research will be required to determine the extent of the impact on migratory tree bat populations. Specifically, population size estimates similar to those available for the Eastern red bat (*Lasiurus borealis*) (Vonhof and Russell 2015) need to be generated for the other species that appear to collide most frequently with wind turbines, including: the Hoary bat (*Lasiurus cinereus*) and Silver-haired bat (*Lasionycteris noctivagan*), as well as for the species that are most imperiled due to white-nose syndrome (WNS), including the Big brown bat (*Eptesicus fuscus*), Eastern small-footed bat (*Myotis leibii*), Indiana bat (*Myotis sodalis*), Little brown bat (*Myotis lucifugus*), and Northern long-eared bat (*Myotis septentrionalis*). The relatively new source of bat mortality from wind turbine collisions is a continent-wide issue because the three most common species found as mortalities have ranges that cover much of the U.S., and many of the affected populations likely breed in the boreal forests of Canada.

With unprecedented levels of population decline due to WNS, more bat species are likely to be listed as threatened or endangered at the state or federal level (Librandi Mumma et al. 2011). Seven bat species, including two federally endangered species and one recently listed as threatened, have been confirmed to be afflicted with WNS. Five of the seven species, Big brown bat, Eastern small-footed bat, Indiana bat, Little brown bat, and Northern long-eared bat have been found dead at wind facilities. The Indiana bat has been on the Endangered Species List since 1967. The Northern long-eared bat was listed as threatened in April, 2015. The causative fungus of WNS, *Pseudogymnoascus destructans*, has been found on an additional five species, including one endangered species, without confirmation of the disease. Two of these species, the Eastern red bat and Silver-haired bat, are also two of the three most common bat mortalities found at wind facilities across most of North America (Chapter 2). Considering the magnitude and diversity of anthropogenic threats that affect bat populations, minimizing adverse impacts to bats is a significant concern for ecologists and the wind energy industry.

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Appendix Table 1

PfOngoing Mortality R script

```
pfOngoingMortality <- function(pr,pd,J,variable,long,sweep){
  #This function calculates the value of pf (probability that a carcass
  #is found) from Etterson, M. Hidden Markov models for estimating
  # animal mortality from anthropogenic hazards. Note this function calls
  #other functions that require the R package expm to be installed.
  #
  #Inputs:
  #   pr = estimated daily scavenging rate
  #   pd = estimated per-search discovery rate
  #   J = row-vector of intervals between searches
  #   variable = TRUE/FALSE (TRUE indicates variable search schedule, FALSE = constant
  #       search schedule)
  #   long = TRUE/FALSE (TRUE indicates experiment assumed to be long, FALSE = short)
  #   sweep = TRUE/FALSE (TRUE indicates a sweep was performed, FALSE = no sweep)
  #
  #Outputs:
  #   pf = probability that a carcass is found.
   gf<-gfOngoingMortality(pr,pd,J,TRUE,FALSE,sweep)
  U <- sum(J)
```

```
  pf <- gf/U
  return(pf)
}
```

Appendix Table 2

Average proportion of bat collision mortality occurring across year-round studies in the U.S. Proportions were used to adjust partial-year estimates to full-year estimates

Expected mortality	month	% of 100
2.10352277	January	0.13
10.44576247	February	0.66
47.0059311	March	2.98
32.95746339	April	2.09
80.83587826	May	5.12
63.50256699	June	4.02
113.4031423	July	7.19
363.485778	August	23.03
581.8122893	September	36.87
230.3690191	October	14.60
50.58460415	November	3.21
1.632328994	December	0.10
1578.138287		100.00

Appendix Table 3

NLCD Land Cover Codes and my land cover as grouped by NLCD (left) and as regrouped and analyzed (right).

Old Class	**New Class**
11 Open Water	10 – Water
12 Perennial Ice/Snow	
21 Developed, Open Space	20 – Developed
22 Developed, Low Intensity	
23 Developed, Medium Intensity	
24 Developed, High Intensity	
31 Barren Land	30 – Barren
41 Deciduous Forest	40 – Forest
42 Evergreen Forest	
43 Mixed Forest	
51 Dwarf Scrub	50 Scrubland
52 Shrub Scrub	

71 Grassland/Herbaceous	70 Herbaceous
72 Sedge/Herbaceous*	
73 Lichens*	
74 Moss*	
81 Pasture/Hay	80 Agriculture
82 Cultivated Crops	
90 Woody Wetlands	90 Wetlands
95 Emergent Herbaceous Wetlands	

Appendix Table 4

Description of Simulations to estimate Bias

To examine potential biases associated with various estimators I simulated mortality and carcass discovery at wind turbines under conditions reported for each of the reviewed studies. Specifically:

1. I assumed:
    a. constant daily mortality rate (λ)
    b. constant daily probability of carcass scavenging (p_r)
    c. constant per-search probability of carcass discovery (p_d)
2. In (1) above, lambda was arbitrarily set to 1
3. Estimates of p_r and p_d and search schedule (J) were taken from the individual study reports.
4. Each dataset was simulated 100 times assuming that the reported experimental conditions and estimated probabilities from each study were correct and the

number of simulated birds killed in each simulation (indexed by i, N_{ki}) and number of bats found (N_{fi}) from each iteration was recorded.

5. I applied versions of each estimator to estimate p_{fi} specific to that estimator and applied this value to the values of N_{fi} from each iteration to produce estimates of N_k: ($\hat{N}_{ki} = N_{fi} / p_{fi}$)
6. Bias was estimated as the average difference between the estimated and simulated number of bats killed: $bias = \frac{1}{1000} \sum_{i=1}^{1000} \left(\hat{N}_{ki} - N_{ki} \right)$
7. I used published R-code for each estimator from Etterson (2013) and Korner-Nievergelt (2014).

VITA

Maureen McCormack Thompson

Candidate for the Degree of

Master of Science

Thesis: PREDICTORS OF BAT MORTALITY RATES AT NORTH AMERICAN WIND FACILITIES AND AN EVALUATION OF BIASES INFLUENCING MORTALITY ESTIMATE

Major Field: Natural Resource Ecology and Management

Biographical:

Education:

Completed the requirements for the Bachelor of Science at The Evergreen State College, Olympia, WA in 2009.

Experience:

Over the last decade my resume has grown to include a variety of wildlife and sensitive species surveys, as well as avian and small mammal research. The past two summers I led a Beaver ecosystem monitoring crew in Eastern Washington. In this capacity I trained interns in plant identification, methods of plant community monitoring, and quality data maintenance. I also designed and implemented an amphibian inventory program. In this position I was in charge of staffing, scheduling, procuring and maintaining field gear, and producing summary reports. For the past two years, as a consultant with Tetra Tech, I conducted raptor point counts, aerial surveys, and ground-based habitat evaluations for Lesser Prairie-Chicken. I am currently working as a Researcher with my graduate alma mater to investigate facility scale correlates of bat mortality rates and assess how methodological differences contribute variation to wind facility-scale estimates of mortality. In this position I manage a large database, conduct analyses, prepare manuscripts for publication, and communicate results to a variety of interest groups. Additionally, I am the Citizen Science Biologist for Utah's Hogle Zoo. In this position I engage members of the public in state-wide conservation efforts and coordinate with agency personnel to conduct field surveys for at-risk species. I am also the co-founder of Dongs Inc.

Professional Memberships:

Association for Women in Science & Oregon Native Plant Society