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Review

Estimates of bird collision mortality at wind facilities in the contiguous United States

Scott R. Loss [a,*], Tom Will [b], Peter P. Marra [a]

[a] Migratory Bird Center, Smithsonian Conservation Biology Institute, National Zoological Park, P.O. Box 37012 MRC 5503, Washington, DC 20013, USA
[b] U.S. Fish and Wildlife Service, Division of Migratory Birds, Midwest Regional Office, 5600 American Blvd. West, Suite 990, Bloomington, MN 55437-1458, USA

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ABSTRACT

Wind energy has emerged as a promising alternative to fossil fuels, yet the impacts of wind facilities on wildlife remain unclear. Prior studies estimate between 10,000 and 573,000 fatal bird collisions with U.S. wind turbines annually; however, these studies do not differentiate between turbines with a monopole tower and those with a lattice tower, the former of which now comprise the vast majority of all U.S. wind turbines and the latter of which are largely being de-commissioned. We systematically derived an estimate of bird mortality for U.S. monopole turbines by applying inclusion criteria to compiled studies, identifying correlates of mortality, and utilizing a predictive model to estimate mortality along with uncertainty. Despite measures taken to increase analytical rigor, the studies we used may provide a non-random representation of all data; requiring industry reports to be made publicly available would improve understanding of wind energy impacts. Nonetheless, we estimate that between 140,000 and 328,000 (mean = 234,000) birds are killed annually by collisions with monopole turbines in the contiguous U.S. We found support for an increase in mortality with increasing turbine hub height and support for differing mortality rates among regions, with per turbine mortality lowest in the Great Plains. Evaluation of risks to birds is warranted prior to continuing a widespread shift to taller wind turbines. Regional patterns of collision risk, while not obviating the need for species-specific and local-scale assessments, may inform broad-scale decisions about wind facility siting.

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* Corresponding author. Current address: Department of Natural Resource Ecology and Management, Oklahoma State University, 008C Agricultural Hall, Stillwater, OK 74078, USA. Tel.: +1 405 744 4607.
E-mail addresses: scott.loss@okstate.edu (S.R. Loss), Tom_Will@fws.org (T. Will), MarraP@si.edu (P.P. Marra).

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1. Introduction

Wind energy has emerged globally as a promising alternative to fossil fuels. As of June 2013, more than 270 gigawatts (GW) of power generation capacity were installed across the world's >13,000 wind facilities (The Wind Power, 2013). Roughly 20% of this capacity is installed in the United States (American Wind Energy Association, 2013), providing enough energy to power 18 million households. A continued increase of U.S. wind energy development is expected in response to the Department of Energy's (DOE) goal to have 20% of total energy generated from wind power by 2030 (U.S. DOE, 2008). Conservationists have expressed concern about direct and indirect impacts of wind energy development on wildlife, including bird and bat collisions with wind turbines (Kunz et al., 2007a, 2007b; Kuvlesky et al., 2007), habitat loss, and creation of barriers to wildlife movement (Drewitt and Langston, 2006; Kuvlesky et al., 2007; Pruett et al., 2009; Kiesecker et al., 2011). Despite the decommissioning of many lattice-tower turbines that have caused large numbers of bird collisions, such as those at Altamont Pass in California (California Energy Commission, 1989; Smallwood and Karas, 2009), bird collisions still occur at turbines with solid monopole towers (e.g. Johnson et al., 2002; Kerns and Kerlinger, 2004), which now comprise the vast majority of U.S. turbines.

Wildlife mortality from collisions with wind turbines is the most direct, visible, and well-documented impact of wind energy development. However, conclusions about collision rates and impacts of collisions on bird populations are tentative because most of the mortality data is in industry reports that are not subjected to scientific peer review or available to the public (Piorkowski et al., 2012). The accessible data—which could provide a non-representative sample of all studies—suggests that bird collision rates at turbines are lower than at other structures, such as communication towers, buildings, and power lines (Drewitt and Langston, 2006), and that mass collision events are rare at wind facilities (but see Johnson et al., 2002; Kerns and Kerlinger, 2004; American Bird Conservancy, 2011). Pre-construction assessment of collision risk at proposed wind facilities has been unreliable, with no clear link documented between predicted risk levels and post-construction mortality rates, likely due to substantial variation in collision rates among turbines and a failure to consider risks at individual proposed turbine sites (de Lucas et al., 2012a, 2012b; Ferrer et al., 2012). In addition, most risk assessments focus on the total numbers of birds predicted to be present at a site. A failure to consider species-specific risks may result in relatively high post-construction rates of mortality for some species even if total bird mortality is relatively low (Ferrer et al., 2012).

Mean estimates of annual U.S. mortality from wind turbine collisions range between 20,000 and 573,000 birds (Erickson et al., 2001, 2005; Manville, 2009; Sovacool, 2012; Smallwood, 2013). Earlier estimates were generated by summarizing a small sub-set of industry reports and extrapolating mortality rates across all turbines (Erickson et al., 2001, 2005), by using small samples of preliminary data (Sovacool, 2012), or by using undocumented methods (Manville, 2009). A recent study estimates annual U.S. collision mortality at 573,000 birds and greatly improves upon earlier efforts by using data from a large sample of wind facilities and by accounting for several methodological differences among the studies used (Smallwood, 2013). However, this study did not distinguish between lattice and monopole turbines. Because monopole turbines comprise the vast majority of all installed U.S. wind turbines, it is important to separately estimate mortality and assess correlates of mortality for this turbine type.

We reviewed the wind energy literature, including both peer-reviewed articles and unpublished industry reports, and extracted data to systematically estimate bird collision mortality and mortality correlates at monopole turbines in the contiguous U.S. Specifically, we (1) defined inclusion criteria to ensure a baseline level of rigor for studies used in the estimate, (2) fitted a predictive model that includes correlates of mortality, accounts for differences among studies in the proportion of the year during which collision events were sampled, and includes estimate uncertainty, and (3) implemented the fitted model to estimate bird collision mortality for wind facilities in the contiguous U.S.

2. Material and methods

2.1. Literature search

We searched Google Scholar, the Web of Science database (using the Web of Knowledge search engine), and the National Renewable Energy Laboratory's Wind-Wildlife Impacts Literature Database (http://wild.nrel.gov/) to identify studies documenting bird collisions with wind turbines. We also searched Google because most industry reports are not indexed in databases. We used the search terms "bird AND wind turbine" with "collision," "mortality," "fatality," "carcass," and "post-construction"; all terms with "bird" replaced by "avian" and "wildlife"; and "turbine" replaced by "farm, "facility " and "energy." We checked reference lists and three online bibliographies (National Wind Coordinating Committee, 2005; Johnson and Arnett, 2011; Ellison, 2012) to identify additional studies. In addition to articles we were able to access, we found citations for 47 reports/articles in reference lists that appeared to contain collision mortality data but could not be located using the above search strategy. We requested many of these reports from either the authors that conducted studies or the companies that commissioned studies. However, for some reports we could not find contact information that could be used to request reports, and for some reports that we requested, we received no response to our inquiry. We were therefore only able to acquire 18 (38%) of these additional studies.

Cursory review indicated that studies of collision mortality at wind facilities vary substantially with regard to study design and sampling protocol. These differences must be considered when combining results from multiple studies to estimate mortality (Loss et al., 2012). As described in the following sections, we controlled for much of this variation by implementing inclusion criteria that created a baseline of rigor that studies had to meet to be included in analysis and by accounting for the proportion of the year during which carcass surveys were conducted, the number of turbines in wind facilities, and the size of carcass search plots.

2.2. Inclusion criteria

We only included studies for in-depth review if they were conducted in the U.S. or Canada, provided data on bird collisions with wind turbines, did not repeat findings of earlier studies, and were available as a formal report (i.e. we excluded one legal testimony and one conference presentation). We included Canadian studies to increase the sample of data for the analysis of mortality correlates (Section 2.5), and we thus assume that correlates do not differ between the U.S. and Canada. We only applied our mortality estimation model across wind turbines in the contiguous U.S. (Section 2.6) After in-depth review of remaining studies, we excluded those that focused only on a particular bird group (e.g. raptors), that sampled at fewer than three turbines, and that grouped turbine collisions with collisions from other objects, such as power lines and vehicles. Because raw counts underestimate true mortality (Korner-Nievergelt et al., 2011), we only included studies that corrected counts for searcher detection and scavenger removal rates as estimated in trials at the same facility. All studies that accounted for these factors also adjusted mortality estimates for the proportion of the wind facility's turbines that were not surveyed (or they surveyed all of the facility's turbines), thus resulting in an adjusted mortality estimate for the entire wind facility. Three studies that documented zero fatalities met our inclusion criteria. Because no methods currently exist to adjust zero counts for scavenger removal and imperfect detection, we assumed that zero was the actual number of birds killed at these sites. Finally, studies were included only for turbines with solid monopole towers (i.e., we excluded studies of lattice turbines or of multiple turbine types that did not differentiate among types). We excluded lattice turbines because they have largely been decommissioned in the U.S. and because our objective was to provide a mortality estimate relevant to current and future turbine technology. Our estimate therefore does not incorporate high mortality rates historically recorded at some lattice turbines (e.g., those at Altamont Pass), but we do include data from monopole turbines at some of these same sites. After implementing these inclusion criteria, 68 studies remained from which we extracted data (Appendix A), while 27 studies were excluded from further analysis (Appendix B).

2.3. Data extraction

For each wind facility, we extracted the total estimated amount of mortality across all turbines (after correction for scavenger removal and searcher efficiency). When studies only reported estimated per turbine mortality rates, we multiplied these by the number of turbines in the facility. If the study did not provide the number of turbines, we extracted this information from an online database (Open Energy Info, 2013). Several facilities had more than one study conducted over non-consecutive periods or had a single study that provided data for different periods without providing a collision estimate over the study's entire duration. In these cases, we extracted separate mortality estimates, which resulted in a total of 76 mortality estimates (Appendix A) being extracted from the 68 studies meeting inclusion criteria. However, because of non-independence of separate mortality estimates from the same facility, we only used a single mortality estimate for each wind facility. This resulted in 58 mortality estimates being carried forward for further analysis (see Appendix C for methods describing selection of a single mortality estimate for facilities with more than one estimate).

In addition to extracting total mortality estimates, we extracted data for individual bird species. However, because studies did not provide estimates of species mortality that were corrected for species-specific values of scavenger removal and searcher detection rates (instead, either a single set of adjustments was applied across all birds, or separate adjustments were made, but only for coarse size groupings), we did not generate species-specific mortality estimates. Such estimates would be strongly biased toward larger more detectible species. Nonetheless, we did summarize raw counts to illustrate the species composition of fatalities that have been found at wind facilities in the contiguous U.S. We extracted species data from some studies that were excluded from the total mortality estimate for not adjusting for searcher detection and scavenger removal rates, but we still excluded species records from studies that included lattice turbines, that were from fewer than three turbines, and that did not quantify mortality for all bird groups or distinguish among multiple mortality sources. We also excluded fatalities found incidentally (i.e., not during standardized surveys). The additional included studies are shown in Fig. 1 and Appendix B.

Finally, we also extracted information about wind facilities, including the height of turbine hubs (i.e., the top of the tower where the rotor is mounted; hereafter "hub height"), height to the upper blade tip (hereafter "top height"), and blade-swept area (Appendix A). If hub height was not provided, we calculated this value using one of three approaches: (1) if the turbine model was provided, we looked up hub height online or in a turbine specifications database (The Wind Power, 2013), (2) if the turbine model was not provided but top height and rotor radius were, we calculated hub height as top height minus rotor radius, and (3) if neither the turbine model nor rotor radius were provided but top height was, we used the hub height from turbines with the same top height.

Fig. 1. Regions defined for calculation of annual bird mortality from collisions with monopole turbines in the contiguous U.S. and locations of wind facilities with mortality data used in analysis (solid circles—wind facilities with data used to generate estimates of total mortality *and* with data extracted for species count summary; open circles—wind facilities with data used only for species count summary; open circles with black dot—wind facilities with data used only for total mortality estimates). The two wind facilities on the Texas Gulf Coast were classified as being in the East because of the biological dissimilarity of this region from the rest of the Great Plains.

Table 1
Model selection results for analysis of characteristics related to bird collision mortality at monopole turbines as derived from Akaike's information criteria, corrected for small samples (AIC_c). All candidate models included total mortality estimates for the entire wind facility as the dependent variable (estimates adjusted for scavenger removal, searcher efficiency, and search radius), assumed a Poisson error structure, and included offsets for the number of days out of the year covered by sampling and the number of wind turbines in the facility.

Model	K[a]	Δ AIC_c[b]	w_i[c]
Nacelle height + region	5	0.00	0.521
Nacelle height * region	8	0.73	0.361
Region	4	3.86	0.076
Nacelle height	2	5.06	0.042
Null model	1	16.17	0.000

[a] Number of parameters in the model (including intercept parameter).
[b] Difference in AIC_c value between model and the most strongly supported model.
[c] AIC weight – relative strength of support for model.

2.4. Adjusting mortality estimates for varying search radius

Both turbine height and the radius of carcass search plots influence the proportion of carcasses found out of the total number killed (Smallwood, 2013). Although turbine hub height and search radius co-vary, there is little standardization among studies for the manner in which search radius is selected, and this contributes bias when comparing studies. To account for this among-study variation, Smallwood (2013) calculated the proportion of predicted fatalities found in the maximum search radius for several combinations of hub height and search radius. The proportions were based on a logistic function that predicts a distance asymptote for the cumulative number of fatalities found. We used the hub height and search radius for each mortality estimate in our data set and the proportions in table 3 of Smallwood (2013) to adjust mortality estimates and to account for varying search radius.

For nine of the mortality estimates, hub height information was unavailable; in these cases, we applied the average proportion for the matching search radius across different hub heights in table 3. For estimates with either hub height or radius not exactly matching values in table 3, we used the proportion for the height-radius combination nearest to the non-matching value, or if the non-matching value was equidistant between two values in table 3, we averaged the two proportions. For estimates in our data set with neither the height nor radius matching the values given in table 3, we used the proportion that was the best match that could be derived from the table by first selecting the set of proportions with the nearest search radius and then, among proportions for that radius, selecting the one with the nearest height (e.g. if the mortality estimate came from a study with search radius = 60 m and hub height = 68.5 m, we first referenced all proportions corresponding to a radius of 63 m, the closest value to 60 m, and then, among proportions for that radius, we selected the one for a hub height of 50 m, the closest value to 68.5 m). We took this approach of first referencing radius and then referencing height because Smallwood (2013) found that search radius explains greater variation than height in the predicted distance asymptote.

2.5. Analysis of mortality correlates

We conducted an analysis to identify correlates of collision mortality using previously suggested important variables (Barclay et al., 2007; Kuvlesky et al., 2007; Pearce-Higgins et al., 2012). Because not all variables were available for all mortality estimates, and because we sought to only include variables with sufficient replication to allow rigorous modeling, we only included variables for analysis if they were available for a minimum of 50 mortality estimates. Variables meeting this criterion included: hub height, top height, rotor diameter, and geographical region. We defined four geographical regions, including California, the West (excluding California), the Great Plains, and the East (Fig. 1). Hub height, top height, and rotor diameter were all strongly correlated with each other ($r \geqslant 0.81$). Because hub height was the most commonly reported variable in the studies we reviewed, we removed top height and rotor diameter from further analysis, thus avoiding multicollinearity in the following model selection exercise. Thus, the remaining variables included hub height and region. From the 58 mortality estimate replicates, we removed five records that lacked information about one of these predictor variables because the following model selection approach required that the same number of replicates and the same set of response variable data be used for each candidate model. Thus, we used 53 mortality estimates for the following model selection exercise (Appendix A). For a full description and rationale for every decision made prior to reaching this final data set used for analysis, see Appendix C.

All analyses were conducted in Program R. We used an information theoretic approach for model selection. In a preliminary analysis that assumed a normal distribution of errors (R function lm), visual assessment of the response variable residuals indicated unequal variance of residuals. We therefore used a generalized linear modeling approach (R function glm) for all candidate models, and we assumed a Poisson error distribution because the response variable was based on count data.

A common assumption for studies that do not sample throughout the year is that mortality is negligible during the un-sampled time period—typically the months outside of migration seasons. However, collision mortality can be substantial during summer (Osborn et al., 2000; Gritski et al., 2010; Stantec, 2011) and winter (Kerlinger et al., 2007; Young et al., 2007). Furthermore, in a preliminary analysis of all mortality estimates that met our inclusion criteria, estimated mortality increased significantly with an increasing proportion of the year sampled ($\beta = 0.081$, $\pm 95\%$ $CI = 0.013–0.149$, $df = 76$, $p = 0.020$). Annual mortality estimates derived from a partial year of sampling may therefore substantially underestimate mortality. To account for sampling of varying proportions of the year and to generate a mortality estimate that includes the potential for mortality in un-sampled periods, we specified an offset term in all candidate models that was the log

Table 2
Estimates of bird mortality from collisions with monopole wind turbines in the contiguous United States.

Region	Total # of turbines	Total MW capacity	Total mortality			Mortality per turbine			Mortality per MW		
			Mean	LCI[a]	UCI[b]	Mean	LCI[a]	UCI[b]	Mean	LCI[a]	UCI[b]
California	13,851	5796	108,715	56,095	161,335	7.85	4.05	11.65	18.76	9.68	27.84
East	6418	11,390	44,006	34,749	53,262	6.86	5.41	8.30	3.86	3.05	4.68
West	5757	9590	27,177	19,671	34,682	4.72	3.42	6.02	2.83	2.05	3.62
Great Plains	18,551	29,896	54,115	29,923	78,307	2.92	1.61	4.22	1.81	1.00	2.62
Total U.S.	44,577	56,852	234,012	140,438	327,586	5.25	3.15	7.35	4.12	2.47	5.76

[a] Lower bounds of estimate 95% confidence interval.
[b] Upper bounds of estimate 95% confidence interval.

Table 3
Comparison of characteristics of wind facilities in the data set used to generate estimates of contiguous U.S bird mortality from collisions with monopole wind turbines and in larger database of U.S. onshore wind facilities (The Wind Power, 2013).

Region	Data analyzed (53 facilities)				All U.S. facilities in database		
	# Of facilities	Ave. # of turbines in facility	# Of turbines surveyed	Ave. hub height	# Of facilities	Ave. # of turbines in facility	Ave. hub height (m)
California	4	74.00	246	64.6	154	89.94	81.0
East	22	57.77	569	77.1	228	28.15	76.5
West	17	85.41	721	67.1	149	38.64	78.2
Great Plains	10	81.80	271	63.7	469	39.55	75.6
Total U.S.	53	72.40	1807	70.4	1000	44.577	76.7

of the count duration (number of days of the year sampled with maximum = 365 for year-round studies). In addition, because the number of turbines in the wind facility influences the total number of carcasses found, and therefore the total estimated amount of mortality at the wind facility, we also specified an offset term that was the log of the number of turbines in the wind facility.

Within this generalized linear model structure (assumption of a Poisson error distribution, specification of offsets for sampling duration and number of turbines in the facility, and response variable = fatality count adjusted for scavenger removal, searcher efficiency, and search radius), we defined a candidate set of models, including a null model, single-variable region and hub height models, a 2-variable additive model including both region and height and a 2-variable region-height multiplicative model (i.e. the global model). We used Akaike's Information Criteria corrected for small sample sizes and the residual deviance from each candidate model to compare relative support for each model. We considered models to be strongly supported if they had ΔAIC values <2.0 (Burnham and Anderson, 2002).

2.6. Model used for estimation of collision mortality

We used the additive 2-variable model that included hub height and region—the most strongly supported model from the above model selection procedure (see details of AIC analysis results in Section 3.1)—to predict mortality across all U.S. wind turbines with location data available as of May 2013 (R function predict). We used a database that included entries for 1000 onshore wind facilities in the contiguous U.S. (i.e. excluding Alaska and Hawaii) either in production or currently under construction (i.e., facilities that will be in production within two years) that also referenced the number of turbines in the wind facility (44,577 total turbines; 56,852 total MW capacity; The Wind Power, 2013). We could not determine whether each individual wind turbine was a lattice or monopole model. Our mortality estimates, which are based only on collision data from monopole turbines, therefore represent the amount of expected mortality if all current U.S. wind turbines were updated to monopole models. We excluded wind facilities in Alaska and Hawaii because we found no mortality data for these states, and it is unclear whether mortality data from the contiguous U.S. can be reliably extrapolated to these regions. We also found no data for the southwestern U.S.; we assumed that the amount of mortality per turbine in Arizona, Nevada, and Utah was the same as the rest of the West (excluding California) and that mortality in New Mexico was similar to mortality in the Great Plains due to proximity of this state to two western Texas facilities with data. We also classified two facilities on the Texas Gulf Coast as being in the East because of the biological dissimilarity of this region from the Great Plains. For 29% of the 1000 wind facilities in the database, the hub height or turbine model was listed. Turbine model information was used to cross-reference another portion of the same database that included specifications, including average hub height, for different types of wind turbines. For the remaining 71% of turbines, we used the average height of other turbines in the same region. We provide a discussion of the assumptions made for the mortality estimate in Section 4.4.

3. Results

3.1. Correlates of mortality

The additive 2-variable model that included turbine hub height and region was the most strongly supported model in our analysis, followed by the multiplicative height-region model. Because the relative strength of support was greater for the additive model (Table 1), we used this model for mortality prediction. The univariate region and hub height models each received less support than the 2-variable models; however, because both variables were included in the best-supported model, we compared estimated mortality across turbine heights and among regions. Bird collision mortality was modeled to increase significantly with increasing hub height (univariate hub height model: $\beta = 0.039$ [95% $CI = 0.037–0.40$], $z = 51.5$, $df = 52$, $p < 0.001$) (raw data back-calculated to per turbine mortality rates in Fig. 2A). Across the range of hub heights in our data set (36–80 m), and when accounting for varying proportions of the year being surveyed, annual model-predicted mortality increased nearly ten-fold (from 0.64 to 6.20 birds per turbine). After accounting for varying proportions of the year being sampled, annual per turbine mortality was modeled to be highest in the East (median = 8.16 birds), followed by California (median = 4.82 birds), the West excluding California (median = 3.64 birds), and the Great Plains (median = 2.43 birds) (raw data back-calculated to per turbine mortality rates in Fig. 2B).

3.2. Estimates of total bird collision mortality

Using the top model from the previous analysis and incorporating region and height data for the 1000 onshore wind facilities generated a mean estimate of 234,012 birds (95% $CI = 140,438–327,586$) killed annually by collisions with monopole wind turbines in the contiguous U.S. Mortality estimates varied geographically due to variation in both numbers of turbines and mortality rates. There was no statistically significant difference among regions in turbine hub height ($F = 2.278$, $df = 285$, $p = 0.080$); therefore, this factor was unlikely to explain substantial regional variation in mortality estimates. We estimate that 46.4% of total mortality at monopole wind turbines occurs in California, 23.1% occurs in the Great Plains, 18.8% occurs in the East, and 11.6% occurs in the West (Table 2).

Regional rankings changed when estimated mortality was evaluated on a per turbine or per MW basis. California still ranked above all regions with a mean annual collision rate of 7.85

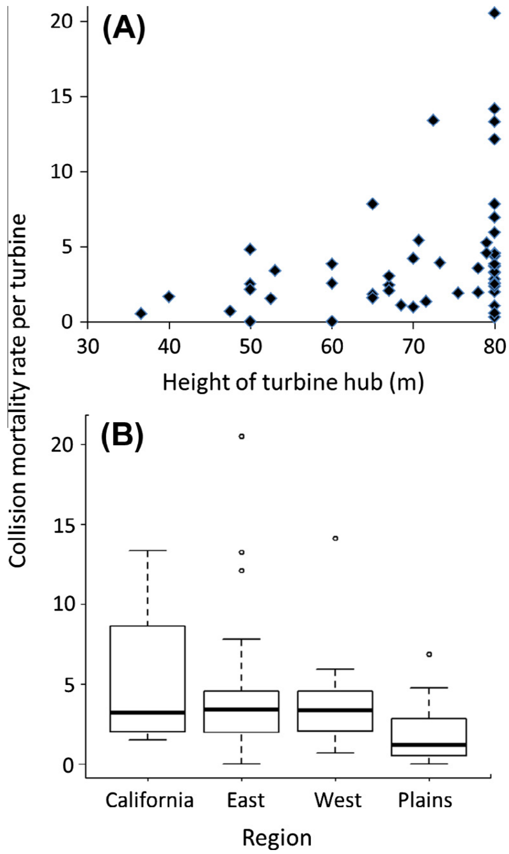

Fig. 2. Univariate relationships between annual per turbine bird mortality at monopole turbines (adjusted for scavenger removal, searcher efficiency, and search radius) and variables appearing in the top model selected in the AIC_c analysis: (A) turbine hub height (i.e. height to the hub where the rotor is mounted) and (B) geographic region.

birds/per turbine (95% *CI* = 4.05–11.65), followed by the East (6.86 birds/turbine; 95% *CI* = 5.41–8.30), the West (4.72 birds/turbine; 95% *CI* = 3.42–6.02), and the Great Plains (2.92 birds/turbine; 95% *CI* = 1.61–4.22). On a per MW basis, California had a mean collision rate of 18.76 birds per MW (95% *CI* = 9.68–27.84), followed by the East (3.86 birds/MW; 95% *CI* = 3.05–4.68), the West (2.83 birds/MW; 95% *CI* = 2.05–3.62), and the Great Plains (1.81 birds/MW; 1.00–2.62). Regional differences based on the additive region-height model are different from those based on the univariate region model (Section 3.1) because the latter were calculated independently of turbine height data.

The wind facilities for which we compiled mortality data may not be representative of all wind facilities (see Table 3 for comparisons between wind facilities with and without bird mortality data). For example, California data came from facilities that had fewer turbines than average for wind facilities in this state, and data from the East, West, and Great Plains came from facilities that had more turbines than average for these regions. Mortality data also came from wind facilities with turbine hub heights that were shorter than average for California, the West, and the Great Plains. For the East, facilities with mortality data had turbine heights that were characteristic of average heights for this region.

For the species count summary, we found 3605 fatality records representing at least 218 bird species from 73 studies (raw counts in Appendix D). As mentioned in Section 2.3 and further discussed in Section 4.4, these counts may be non-representative of species-specific mortality because counts are influenced by rates of scavenger removal and searcher detection.

4. Discussion

4.1. Comparison to other mortality estimates

Our mean projected estimate of 234,012 annual bird collisions in the contiguous U.S. – and even our low-end estimate (140,438) – is greater than most previous estimates, including ~20,000 birds/yr (Sovacool, 2012), 10,000–40,000 birds/yr (Erickson et al., 2001; Manville, 2005), and 20,000–40,000 birds/yr (Erickson et al., 2005). Two recently published annual estimates exceed our upper estimate of 327,586 birds: 440,000 (Manville, 2009) and 573,000 (Smallwood, 2013). We provide the first mortality estimate specific to monopole turbines. Our focus on this turbine type could explain why our estimate is lower than the estimate of Smallwood (2013) that also includes data from lattice turbines. Lattice turbines can kill relatively large numbers of birds (Smallwood and Karas, 2009), but they are largely being decommissioned in the U.S. Our modeling approach, including data extraction from 68 studies and prediction of mortality based on turbine height and geographic region, provides a more systematic approach than most previous estimates (but see Smallwood, 2013). Furthermore, we accounted for variable sampling coverage among studies (by defining an offset term in our model for the number of days out of the year sampled). This approach improves upon the assumption that mortality is negligible during periods of the year that are not surveyed.

4.2. Correlates of mortality

Bird collision rates at communication towers are known to increase with increasing tower height (Longcore et al., 2008, 2012). Meta-analyses of collision mortality at wind turbines have found either an increase in mortality with height, but only for bats (Barclay et al., 2007), or a decrease in mortality with turbine size for birds (Smallwood, 2013). Our finding of support for a positive relationship between bird collision mortality and turbine hub height may be a result of: (1) using a mortality data set that is larger and more comprehensive than those used in previous studies (but see Smallwood, 2013), (2) accounting for variation among studies in the proportion of the year sampled, and/or (3) only including data from solid monopole turbines.

Our finding of a positive relationship between turbine height and bird mortality appears to be the opposite of the Smallwood (2013) finding of reduced mortality rates with increasing turbine size for raptors (nationwide) and for all birds (within the Altamont Pass Wind Resource Area). However, we used a different metric for turbine size (hub height) than the previous study (MW of energy generation capacity), and this difference in approaches could explain our apparently contradictory result. In addition, our analysis only included wind turbines with solid monopole towers, whereas previous authors have included both monopole and lattice-towered turbines, including lattice turbines from the Altamont Pass Wind Resource Area (APWRA). Lattice turbines in the APWRA are relatively small (with total height between roughly 18 and 45 m; Orloff and Flannery, 1992) and characterized by relatively high per turbine mortality rates. This mortality likely occurs due to a combination of turbine design (lattice turbines provide perches that attract raptors near spinning blades) and turbine placement near areas of high bird movement (mountain ridgelines) (Smallwood and Thelander, 2008; Smallwood and Karas, 2009). Thus, our exclusion of lattice turbines could contribute to the unique finding of a positive relationship between turbine height and mortality.

The average hub height of U.S. wind turbines has increased 50% between 1998 and 2012, and further up-scaling in both hub height and rotor size is expected in the future (U.S. DOE, 2013). Because we found a strong correlation between turbine hub height and rotor diameter, it is important to note that increased bird mortality may be a result of both increased turbine height and increased rotor diameter. As turbines get taller, greater mortality may occur due to turbines extending further into altitudes that contain large numbers of flying birds. As rotor diameter increases, a greater area of airspace is swept by the turbine blade and therefore exposed to collision risk. Recent well-publicized research indicates that larger wind turbines may provide more efficient energy generation (Caduff et al., 2012; National Geographic, 2012). Given that we found evidence for increased bird mortality with increasing height of monopole turbines along with a move toward increasing turbine size, we argue that wildlife collision risk should be incorporated with energy efficiency considerations when evaluating the "greenness" of alternative wind energy development options.

Our finding of some evidence for mortality rates differing among geographic regions (both when based on our mortality predictions and model selection results) suggests that coarse-scale decisions about siting of wind facilities may benefit from information about bird collision risk. Because average per turbine collision rates in the Great Plains may be relatively low, wind energy development in this region could potentially result in comparatively lower collision risk for wildlife. Previous research illustrates that the development potential for wind energy in the Great Plains is sufficient to meet the output capacity of the DOE's 20% goal, even if development occurs only on lands that are already disturbed (Kiesecker et al., 2011). However, precaution must be taken when making broad-scale siting decisions, especially in regions where no publicly available mortality data exists, including many Great Plains states (North Dakota, Kansas, Nebraska, and Colorado) and the U.S. Southwest. Furthermore, little is known about the potential for indirect impacts of wind energy development in both disturbed and undisturbed areas (Kuvlesky et al., 2007; Kiesecker et al., 2011; Piorkowski et al., 2012). Finally, the cumulative effects of wind energy must still be considered because a large number of turbines that each cause a small number of collisions can still result in a large overall amount of mortality.

4.3. Data biases and model limitations

Study design and sampling methods varied among the studies we used in our analysis, and it is unclear how this variation influenced our mortality estimates. Although we were unable to account for all sources of variation, we accounted for variation in seasonal coverage of surveys, and our inclusion criteria and adjustments accounted for searcher efficiency, scavenger removal, and varying search plot radius. Other methodological differences could not be accounted for. For example, whereas some studies clearly distinguished between fatalities found during scheduled searches and those found incidentally, others combined all fatality observations without differentiating among types of records. Some studies corrected for incomplete searching of survey plots (e.g., due to obstructions, dense vegetation, or safety concerns), while others did not make these corrections or did not present information to determine whether corrections were made or even necessary. Furthermore, different statistical estimators were used to generate mortality estimates, and some of these consistently under-estimate mortality (Huso, 2010). Standardization of methods for carcass searches, searcher efficiency trials, scavenger removal trials, and statistical estimators will reduce biases in comparisons of multiple studies (for further discussion of biases affecting mortality estimates, see Kunz et al., 2007b; Smallwood, 2007; Smallwood et al., 2010; Loss et al., 2012).

Extrapolation of data from the western U.S. and Great Plains to the southwestern U.S. may influence our mortality estimates. Further research at wind facilities in these areas is needed to estimate regional mortality rates and identify species and locations that are at elevated risk of bird mortality from wind energy development. We collected data from wind turbines with hub heights ranging from 36 to 80 m, but we applied our predictive model to turbines with hub heights up to 100 m. The accuracy of extrapolating our model to taller turbines remains untested because there are no publicly available studies of mortality for U.S. turbines taller than 80 m. Notably, 8.2% of U.S. wind facilities with turbine hub height data available (24 of 290 facilities) included turbines in excess of 80 m in height.

Because we were unable to determine the specific turbine type for every wind turbine in the database to which we extrapolated the fitted model, we assumed that all turbines had monopole towers. This assumption assumption is likely valid given that all U.S. facilities for which we found mortality data—except for three facilities in California—solely use monopole turbines. Nonetheless, given that a small number of the turbines to which we extrapolated our model were lattice turbines, our estimates represent the amount of mortality expected if all U.S. turbines were updated to monopole models. Additional estimate bias may have occurred due to uncertainty about hub heights for some turbines to which we extrapolated the fitted model. However, we are unaware of a wind turbine data set that includes hub height information for every U.S. wind turbine; development and use of such a database would improve future mortality estimates and assessments of mortality correlates.

Finally, as illustrated in Table 3, the wind facilities from which we extracted mortality data may be of non-representative size and have non-representative turbine heights for their respective region. In addition, it is unclear whether the mortality estimates that we used provide a representative sample of all collision mortality data that has been collected at U.S. wind facilities. Despite numerous calls for an increase in the transparent reporting of study results and availability of reports to the public and scientists (Kunz et al., 2007b; Stewart et al., 2007; Piorkowski et al., 2012), collision data largely remains confidential and/or offline. Furthermore, reports that have been released to the public (e.g. on the internet) are often difficult to locate. We join previous authors in calling for increased transparency in data reporting. Requiring industry reports to be made publicly available would greatly improve understanding of wind energy impacts to wildlife.

4.4. Study design improvements and research needs

Our findings do not obviate the need for pre-construction risk assessments for proposed wind facilities and post-construction studies at existing facilities. Even within regions predicted to have relatively low risk, local mortality rates may be substantially higher along or near migratory routes, such as rivers and ridgelines, or in areas with high bird abundance or sensitive species. Ideally, pre-construction studies should be conducted for at least one entire year prior to wind facility siting decisions. As suggested by Ferrer et al. (2012), these risk assessments are likely be effective only when based on investigation of species-specific risks and locations of individual proposed turbines, as opposed to assessment of risks to all birds combined and for locations of entire wind facilities. Post-construction studies ideally should extend for at least three years with sampling conducted throughout the year. If year-round sampling is not possible, inferences about site-specific mortality rates should not be extended beyond the period of sampling coverage. Post-construction studies should identify the age and sex of carcasses when possible because this information can be used to inform understanding of population dynamics relative to mortality at wind turbines. (For a thorough discussion of needed protocol improvements, see also Smallwood (2013)).

Determining whether individual bird species are vulnerable to population declines as a result of wind turbine collisions is a major conservation objective. However, little evidence exists to infer whether turbine collisions cause population declines (Stewart et al., 2007). In the wind energy literature that we reviewed, we found a relatively small sample of mortality data with species information available. Furthermore, these mortality counts are likely influenced by detectability and scavenger removal rates that vary by species, with larger species more likely to be detected and less likely to be removed by scavengers than smaller species (Smallwood, 2007). Most studies do not provide the species-level detectability and scavenger removal information that is needed to calculate bias-adjusted estimates for individual species. Thus, any mortality estimates for large species would be inflated relative to those of small species. Further research is needed to increase the sample size of species-specific mortality data at U.S. wind facilities and to clarify how species-specific biases influence estimation of mortality and assessment of population impacts. In addition, intensive research of local and regional-scale population impacts to raptors and other slow-reproducing, long-lived species (e.g., waterbirds) is needed. As illustrated by studies in Europe, even low rates of turbine collision mortality have the potential to be associated with significant population declines for raptors in some localities (Carrete et al., 2009; Dahl et al., 2012).

Wind facility siting decisions should also incorporate risks to bats, which experience high collision rates, primarily along forested mountain ridgetops in the eastern U.S. but also in isolated portions of western North America (Kunz et al., 2007a). Post-construction mortality studies have often focused on bird collisions with only incidental reporting of bat fatalities. Study designs and sampling protocols for investigating bird fatalities may be inadequate for quantifying bat collision rates because factors that affect collision rates (e.g., time of day, season, weather, and turbine and wind facility characteristics) may differ between the two taxa. With increased quantity and rigor of bat studies (Arnett et al., 2008), our approach can be applied to generate total and region-specific estimates of bat mortality.

4.5. Conclusions

Development and production of U.S. wind energy represents a promising opportunity to decrease global carbon emissions and increase energy independence. However, our results suggest that the amount of U.S. bird mortality caused by collisions at monopole wind turbines is non-trivial. Furthermore, the projected trend for a continued increase in turbine size coupled with our finding of greater bird collision mortality at taller turbines suggests that precaution must be taken to reduce adverse impacts to wildlife populations when making decisions about the type of wind turbines to install. Despite an apparent lower magnitude of bird mortality at wind turbines compared to other anthropogenic mortality sources (e.g., windows/buildings, Klem, 2009; Loss et al., 2014; communication towers, Longcore et al., 2012, 2013; feral and pet cats, Loss et al., 2013), mortality at wind facilities should not be dismissed offhand. Instead, we stress the importance of considering species-specific and location-specific risks and the potential for cumulative impacts of multiple wind facilities and multiple mortality threats.

The total amount of bird collision mortality at U.S. wind facilities will likely increase with increased wind energy development in the coming decades. Scaling our estimates to the scenario projected to meet the DOE's 20% goal (a six-fold increase from current generation capacity, U.S. DOE, 2008) produces a mean annual mortality estimate of roughly 1.4 million birds. This estimate assumes that average wind turbine height will not increase. Installation of increasingly larger turbines could result in a greater amount of mortality. Multi-scale decisions about where to site wind facilities and individual wind turbines in the context of risks to individual bird species will be crucial to minimizing this mortality. Mortality estimates can be updated using our approach as more wind facilities are constructed, more regions are studied, and additional mortality data is compiled and made publicly available.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2013.10.007.

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