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Avian fatalities at wind energy facilities in North America: a comparison of recent approaches

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Abstract: Three recent publications have estimated the number of birds killed each year by wind energy facilities at 2012 build-out levels in the United States. The 3 publications differ in scope, methodology, and resulting estimates. We compare and contrast characteristics of the approaches used in the publications. In addition, we describe decisions made in obtaining the estimates that were produced. Despite variation in the 3 approaches, resulting estimates were reasonably similar; about a quarter- to a half-million birds are killed per year by colliding with wind turbines.

Key words: birds, fatalities, human–wildlife conflicts, turbines, wind energy facilities, wind farm

The current pace of wind energy development and its projected growth have prompted questions about the environmental effects of this renewable energy source. Primary concerns are the consequences to birds and bats, although other taxa may also be influenced. In this paper, we focus only on issues related to birds at onshore wind energy facilities, because offshore wind energy development has not yet occurred in North America. Two primary issues of wind energy development are fatalities of birds that collide with wind turbines and avoidance of wind turbines by birds, potentially reducing quality of surrounding habitat for the birds. We do not discuss the avoidance issue here, because of the paucity of information available from well-designed studies.

Most previous research examining effects of wind energy development on birds focused on individual wind facilities; this limited scope precluded cumulative fatality estimates and large-scale inferences. Fortunately, 3 recent publications based on systematic compilation and analysis of a large number of data sets (Smallwood 2013, Loss et al. 2013, and Erickson et al. 2014) presented estimates of numbers of birds killed annually by wind turbines in North America. The 3 publications (hereafter, reviews) differ in scope, methodology, and resulting estimates. The objective of our paper is to clarify distinctions among the 3 approaches. In addition, we describe decisions made in obtaining the estimates that were produced. Our hope is to provide a clearer understanding of differences among the reviews and to stimulate thinking about improvements that might be feasible for future estimates of wildlife fatalities from wind turbines. In this paper, we sometimes include information about the 3 reviews that was not in the original publications but was added following discussions with co-authors.

Distinctions among the 3 approaches can be viewed as falling into one of 5 categories: (1) scope of study—types of species and turbine models included and geographic range for which projections were made; (2) criteria for inclusion—decisions made by the authors about which studies were to be included in their analyses; (3) adjustments for biases—what was done to reduce biases in estimates of fatalities caused by imperfect availability and perceptibility (discussed in detail below); (4) statistical procedures—how results from the reviewed studies were combined to derive estimates; and (5) estimates of total fatalities—how results of the authors' reviews were used to draw inferences about the collision issue.

Figure 1. Examples of lattice turbines, right, and monopole turbines, left. (*Photo by Shawn Smallwood*)

Scope of study

Knowledge of the species included in a fatality estimate is important in that species vary in population size and trajectory, geographic range at various times of the year, and vulnerability to anthropogenic fatalities. Knowledge of the geographic region of coverage is also important for policy planning activities and assessing range-wide impacts to bird populations. Loss et al. (2013) and Smallwood (2013) estimated the number of fatalities for all avian species. Erickson et al. (2014) restricted attention to small passerine species, although an analogous estimate for raptors is planned.

Smallwood (2013) and Loss et al. (2013) estimated the number of fatalities for the conterminous United States. Erickson et al. (2014) also included Canada. Erickson et al. (2014) recognized the absence of data from the southwestern United States, which has 7% of the continent-wide operating capacity. Accordingly, they multiplied estimated numbers of fatalities by 1.07. (Note that it would have been more accurate to use $1.00 + 0.07/(1.00 – 0.07) = 1.075$, because the non-southwestern capacity is 93% of the total.)

Loss et al. (2013) and Erickson et al. (2014) considered only wind turbines of the monopole design, arguing that earlier wind turbines with lattice structures are being phased out. They believed that lattice designs provide perch sites for raptors and other birds, whereas monopole designs do not. Also, lattice designs (Figure 1) were used with early wind turbines, many of which had lower generating capacity than newer models, thus skewing metrics such as wildlife fatalities per megawatt capacity. Smallwood (2013) included all design types, including lattice, monopole, and vertical, although the number of turbines with vertical designs is very small (Figure 2). It should be recognized that lattice-structure wind turbines remaining in place, even if nonfunctional, can provide perch sites for raptors and other birds and, thus, increase risk of collision with nearby wind turbines, regardless of their design. Ideally, all types of designs should be included in an analysis, but in proportion to the number of each design in the entire universe of wind generators. Loss et al. (2013) and Erickson et al. (2014) underrepresented lattice designs in their analyses, and Smallwood (2013) likely overrepresented them in one of his 2 comparative analyses because of the extensive investigations conducted at turbines with lattice designs at Altamont Pass.

Criteria for inclusion

All reviews included information from the conterminous United States and Canada (although relatively few sites in Canada were used), because all authors assumed that sites in Canada have similar relationships between fatality rates and variables used to estimate those rates. Erickson et al. (2014) noted that data were lacking from the southwestern United States. This could create a serious deficiency in the national fatality estimates if relationships between fatality rates and wind turbine metrics in the Southwest differ from those elsewhere.

Each of the 3 reviews incorporated as many reports as feasible, with modest differences in selection criteria (Table 1). Loss et al. (2013), for example, included reports only if results had been adjusted for bird-carcass availability and perceptibility. The other authors used reports with original data and applied adjustments to those data as part of their analyses.

The number of studies and number of wind energy facilities used to develop national fatality estimates varied. Loss et al. (2013) included 53 studies involving 53 wind energy facilities. Smallwood included 72 studies covering 71 wind energy facilities, 19 of which were at Altamont Pass, California. Erickson et al. (2014) summarized 116 studies at >70 wind energy facilities.

Loss et al. (2013) excluded studies with <3 turbines investigated. Because Loss et al. (2013) based their analysis on estimated number of fatalities for entire wind farms, this step was important to ensure that aberrant results from a small sample of turbines were not extrapolated to a larger scale. Smallwood (2013) and Erickson et al. (2014) based their analyses on an estimated number of fatalities per megawatt capacity and, thus, did not need to be restrictive regarding number of turbines investigated.

Loss et al. (2013) included studies that reported no fatalities, and Erickson et al. (2014) included 1 facility with no small-bird fatalities but other avian fatalities. Smallwood (2013) did not encounter any such studies.

Adjustments for biases

All wildlife fatality estimates reviewed were based on systematic searches for wildlife carcasses near wind turbines. Carcasses found usually are presumed to represent fatalities caused by collisions with the nearest wind turbine. The number of carcasses found is a biased estimator of the actual number of fatalities for 4 reasons, as follow.

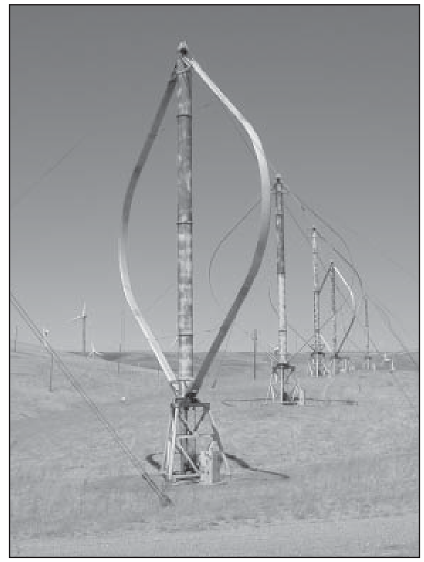

Figure 2. Example of vertical-axis design turbine, the only type of its kind put into industrial-scale operation. A few others appeared singly or as demonstration projects. The ones pictured were 150 kW. This type also occurred as a 250 kW version. Both types were in the Altamont Pass until 2000, and both were removed in 2002. (*Photo by Shawn Smallwood*)

1. Spatial incompleteness. Most (about 86% by capacity) wind farms are not investigated, according to Erickson et al. (2014) at the time of their study. At many of the investigated wind farms, carcass searches were conducted at only a fraction (about 24%) of the turbines, according to Erickson et al. 2014. However, implementation of the 2012 Wind Energy Guidelines (U.S. Fish and Wildlife Service 2012) does appear to have increased availability of studies recently. Further, at some turbines, carcass searches do not cover the entire area over which a carcass could come to rest.

2. Temporal incompleteness. Many fatality searches are conducted for only part of the year.

3. Incomplete availability. Due to removal by scavengers or humans, some carcasses do not persist long enough to be detected during fatality searches.

4. *Imperfect perceptibility.* Some wildlife carcasses that are available to be detected by searchers are missed. Clearly, an interaction between availability and perceptibility exists in that remains of a partially scavenged carcass can be so minimal as to virtually eliminate the possibility of detection.

Regarding spatial incompleteness, the fact that only some wind turbines at some wind farms were included in studies is a sampling issue. In Smallwood's (2013) sample, 27% of the studies were at Altamont Pass, California, a well-known hot spot for wildlife collisions with turbines. Accordingly, Smallwood (2013) presented 2 sets of fatality estimates, one including Altamont Pass and one omitting Altamont Pass.

The other aspect of spatial incompleteness is the possibility that carcasses fell outside the searched area. Search areas varied widely among studies, including circles of radius ranging from 20 to 90 m around a turbine and rectangles ranging from 110 m × 110 m to 252 m × 252 m (Erickson et al. 2014). Smallwood adjusted estimates based on proportion of all fatalities found for classes of turbine tower height paired with plot size derived in Smallwood (2013) from raw data contained in previous studies. Loss et al. (2013) also used Smallwood's (2013) method. Erickson et al. (2014) acknowledged the issue but made no adjustment for this spatial incompleteness. They noted that larger search areas were more likely to include birds killed from other causes (i.e., background fatality) in addition to birds killed by colliding with a wind turbine. Notably, crippling bias is a potentially major type of spatial incompleteness bias that has not yet been addressed in any study of anthropogenic bird fatality. This bias arises from birds that are severely injured but live long enough to move outside of the surveyed area and that consequently may be missed by searchers.

Temporal incompleteness arises from searches not being conducted throughout an entire year. Most studies covered 12 months (or longer), but some were 6 to 9 months long (Erickson et al. 2014). Spring and fall migration periods are the most critical for most wind farms, although fatalities can be common at other times of the year at some facilities (e.g., Osborn et al. 2000). Smallwood (2013) and Erickson et al. (2014) proceeded on the assumptions that search periods covered the times when birds would be present in substantial numbers and that virtually all fatalities would occur during these times. These requirements are similar to many state guidelines, which do not require winter surveys. Accordingly, resulting fatality estimates would be biased low to the extent that birds colliding with turbines (bird strikes) occur outside of the surveyed period. In contrast, Loss et al. (2013) included the logarithm of search duration as an offset variable in their regression model. The net effect of this modeling step is to assume that fatality rates during the unsearched portion of the year are the same values as during the searched portion. Accordingly, their estimates likely are biased high, but by only about 13% overall, based on the total duration of studies (1,371 months) versus total duration of studies had they all been 12 months or longer (1,550 months; Table 2).

Incomplete availability of data arises when a bird carcass becomes undetectable between the time of its death and the search for fatalities. For example, a carcass could be consumed or carried off by a predator or scavenger, or it might decay. The rate at which a carcass becomes undetectable clearly depends on the predator and scavenger community, temperature, humidity, and other local variables. The best way to heighten carcass availability is to conduct searches frequently and early in the day. Doing so also increases the chance to document the full array of bird species killed, including any rare species that might be killed only occasionally but could experience disproportionate population-level impacts of collisions (Beston et al. 2015). Summarizing studies in Erickson et al. (2014), we find the most frequent intervals between searches are 14 days and 7 days (Table 3). More studies had 30 days between searches than daily searches. Only rarely would a small carcass be expected to persist for as long as 30 days.

To adjust for incomplete availability, many studies include carcass removal trials in which carcasses are placed in locations similar to the search area, and observers visit them periodically to determine how long they remain

Table 1. Characteristics of three recently published analyses of wind-related fatalities of birds in North America.

Decision	Article		
	Loss et al. (2013)	Smallwood (2013)	Erickson et al. (2014)
Scope of study			
Species included	All birds	All birds	Small passerines (excluding large corvids)
Geographic area of estimates generated	Conterminous U.S.	Conterminous U.S.	Conterminous U.S. and Canada.
Tower designs included	Monopole	Lattice and monopole	Monopole
Criteria for inclusion			
Geographic area of studies reviewed	Conterminous U.S. (data from Canada included only in covariate analysis)	Conterminous U.S. and Canada	Conterminous U.S. and Canada (2 facilities).
Status of reviewed studies	Formal reports only, not conference presentations, or legal testimony; bias adjustments required.	All available that author could find that relied on periodic searches.	Excluded some studies because: (1) duration of studies was too brief, (2) no bias trials had been conducted, 3) field methods or statistical analyses were conducted inappropriately (e.g., searchers in efficiency trials were aware of carcasses).
Number of studies	53	72	116
Number of wind energy facilities studied	53	71 (19 in Altamont Pass)	>70
Minimum number of turbines	3	None	None
Studies with no fatalities	Included	None found	Included
Studies with multiple fatality estimates	Estimate usually based on estimate with longest sampling time.	NA	If multiple estimators given for a year, the largest value was used. If annual estimates were available, their average was used.
Adjustments for biases			
Adjustment for size of search plot	Adjusted estimate based on proportion of all fatalities found for classes of turbine tower height paired with plot size, as derived by Smallwood (2013) from raw data contained in previous studies.	Adjusted estimate based on proportion of all fatalities found for classes of turbine tower height paired with plot size derived in Smallwood (2013) from raw data contained in previous studies.	Did not adjust. Expressed concern that background mortality (carcasses found that were not caused by collision with wind turbines) is likely an important positive bias, and limited studies suggest this bias may partially or even completely offset any bias associated with plot size.

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Adjustment for background fatality	None	None	None. Recognized as an issue but did not adjust; suggested that larger search plots are likely to include more wind turbine fatalities but also more fatalities from other causes.
Adjustment for portion of year sampled (duration)	Implicitly assumed fatality rate during non-sampled portion of year was the same as during sampled portion.	None	None. Recognized issue but considered omission acceptable due to relative absence of birds outside the part of year studied.
Bias adjustment(s)	Required for inclusion; no specific method.	Applied a single estimator, dependent on body size and type of bird (raptor, non-raptor) developed by author, to original data in all studies.	Created a customized adjustment factor for each study based on: (1) estimator method used, (2) search interval (e.g., weekly, bi-weekly, etc.), and (3) classification of both bias trial results. They classified the overall average value for carcass removal as fast (0 to 10 days), moderate (11 to 23 days), or slow (24 or more days). Searcher efficiency rates (proportion found) within each study were averaged and categorized as low (0 to 0.375), medium (0.375 to 0.65), or high (0.65 to 1.00). For each combination of these four factors, they determined the lowest and highest bias adjustment values, based on trial simulations presented in Erickson et al. (2014).
Statistical procedures			
Stratification	East, Plains, West, California	None	Five avifaunal biomes (Rich et al. 2004)
Metric	Fatalities $turbine^{-1}$ $year^{-1}$, adjusted for turbine height	Fatalities MW^{-1} $year^{-1}$	Fatalities MW^{-1} $year^{-1}$
Adjustment for number of turbines sampled	Included log(number of turbines in wind farm) as offset variable in regression model.	Not applicable	Not applicable
Carcass removal (or persistence) rate	From original reports	Persistence: $R_C = \sum R_i / I$, where R_i is the smoothed predicted proportion of carcasses remaining each day, and I is average search interval. Done separately for small birds and large birds (and rock pigeons)	Estimated average number of days for the removal of small-bird carcasses in 70 studies; this ranged from 1.64 to 27.8 days. For projects that did not report mean removal rates, they attempted to determine average duration of carcasses remaining from the data in the report. Regional carcass removal rates were calculated using all values for each avifaunal biome.

Searcher efficiency (detection) rate	From original reports	Average values by bird size (small, medium, large), type (raptor or not), and visibility provided by substrate.	When possible, searcher efficiency values estimated with multiple opportunities to detect a carcass were adjusted to reflect single-search values to compare to other studies that provided values from a single search. When multiple years of study were conducted, all data were combined into a single searcher efficiency value estimate for that project. For projects that did not report searcher efficiency rates, they attempted to determine searcher efficiency from data in the report. All searcher efficiency values in each biome were combined to obtain regional searcher efficiency rates.
		Results	
Fatality estimate, per year	140,000-328,000	573,093	134,000-230,000 (small passerines) 214,000-368,000 (all birds)
Estimates of population size for comparison.	Comparison not made.	Comparison not made.	Partners in Flight (2013) land bird population estimates database.

detectable (e.g., Smallwood 2007).

This leads to the final bias of observed counts—imperfect detectability. Some carcasses that are available to be detected by searchers are not detected. Clearly an interaction between availability and perceptibility exists in that remains of a scavenged or decayed carcass can be present, but so minimal as to virtually eliminate the possibility of detection. Perceptibility is likely affected by numerous other variables, such as skill and attentiveness of searchers, width of search transects, weather conditions, size and color of carcass, substrate, and vegetation (Smallwood 2007).

Attempts to account for imperfect perceptibility involve searcher detection trials, in which carcasses are placed in the search area and the proportion of those that are detected by searchers is computed. Complications to such trials include limited ability for researchers to use carcasses of similar size, coloration, and state of decay as actual fatalities. Species typically used in carcass removal trials and searcher detection trials, such as mallard (*Anas platyrhynchos*), quail (*Coturnix* spp.), ring-necked pheasant (*Phasianus colchicus*), rock pigeon (*Columba livia*), and house sparrow (*Passer domesticus*), may not be representative of fatalities because of differences in size or coloration (Smallwood 2007, Erickson et al. 2014). All 3 reviews used adjustments to biases in availability and perceptibility when generating national bird mortality estimates.

Statistical procedures

Procedures used to address these sampling issues are described above. Loss et al. (2013) and Erickson et al. (2014) used different stratifications to develop their estimators of the total number of bird fatalities (Table 1), whereas Smallwood (2013) did not use a stratification approach other than including or excluding Altamont Pass estimates.

The reviews varied in their approach to estimating the total number of bird fatalities. We present only summaries here; the original publications should be consulted for details. All used ratio estimation, involving the ratio of fatalities per energy unit (i.e., operating capacity or number of turbines)

Table 2. Number of studies with specified duration (from Erickson et al. 2014).

Duration (months)	Number of studies
6	10
6.5	2
7	6
7.5	2
8	1
9	21
10	1
12	63
14	1
24	6
36	2

Table 3. Number of studies with specified search intervals (from Erickson et al. 2014).

Interval between searches (days)	Number of studies
1	23
2	1
3	6
3.5	3
4	2
7	50
14	55
21	2
30	36
90	1

where the total of energy units for the entire geographic area was known. Smallwood (2013) and Erickson et al. (2014) used the nameplate generating megawatt (MW) capacity, the stated capacity of a wind turbine, as the energy unit. These values were publicly available for all wind farms. Loss et al. (2013) used the number of turbines, adjusted for turbine height, as the energy unit. Because turbine height generally correlated closely with generating capacity, the 2 approaches were not as dissimilar as might appear. An energy unit that would seem preferable to megawatt, turbines, or turbines adjusted by height is the actual output of a wind turbine (e.g., in megawatt hours; this unit should help account for differences in the potential for wind turbines to cause fatalities, reflecting the amount of time that a generator's turbine blades were spinning. However, data on operating output were generally considered proprietary and not publicly available at the scale of individual wind facilities. Further, Smallwood et al. (2010) used that metric and found no improvement in predicting mortality rates over the generating capacity. A more precise energy unit might involve a combination of turbine rotor-swept area, hours of operation, and, especially, hours of operation during high bird activity in the area. Obtaining this information, however, would require further development of approaches to quantify bird abundance near turbines with a high degree of spatial and temporal resolution. Also, some fatalities occur with non-moving turbines, and Longcore et al.'s (2012) estimates for bird collision mortality at communication towers with strobe lights and without guy wires were similar to estimates for wind turbines of similar height. Because Loss et al. (2013) used the estimated total number of fatalities for an entire wind farm, they included as an offset term in their model the logarithm of number of turbines in the wind energy facility.

Several mathematical methods have been proposed for using results from carcass removal and searcher efficiency trials to adjust counts of fatalities for incomplete availability and imperfect perceptibility. The most commonly used methods are: Shoenfeld's (2004), used in 74 studies reviewed in Erickson et al. (2014); Jain's (2005), 22 studies; Huso's (2010), 9 studies; and an older, so-called naïve method, 10 studies. Each method relies on a set of assumptions about relations between detectability, search intervals, and results from the bias trials. See Smallwood et al. (2013) or Korner-Nievergeldt et al. (2011) for more-detailed discussions and comparisons of the methods.

In their review, Loss et al. (2013) used adjusted estimates from each included study, regardless of the method adopted in the study. Smallwood (2013) adjusted observed counts of fatalities by a number he developed that varied by body size and type of bird (i.e., raptor, nonraptor). Smallwood's (2013) approach assumed that a carcass missed on the first search after the fatality occurred would also be missed on all subsequent searches, so the estimator is biased somewhat high when search intervals are short. Erickson et al. (2014) created a customized adjustment factor for each reviewed study based on: (1) the estimator method originally used; (2) search interval (e.g., weekly, bi-weekly, etc.); and (3) classification of carcass removal trials and searcher detection trials. They classified the overall average value for carcass removal as fast (0 to 10 days), moderate (11 to 23 days), or slow (≥24 days). Searcher efficiency rates (proportion found) within each study were averaged and categorized as low (0 to 0.375), medium (0.375 to 0.65), or high (0.65 to 1.00). For each combination of these 4 factors, they determined the lowest and highest bias adjustment values, based on trial simulations.

Estimates of total fatalities

Loss et al. (2013) estimated the annual number of birds (all species) killed at wind energy facilities in the conterminous United States to be between 140,000 and 328,000. Smallwood's (2013) estimate for the conterminous United States was 573,093 total fatalities for all species. Erickson et al. (2014) estimated annual number of fatalities for all species for the conterminous United States and Canada to be between 214,000 and 368,000 birds. For small passerines, the range was between 134,000 and 230,000. Erickson et al. (2014) took a further step by comparing their estimates to estimated sizes of bird populations, based on Partners in Flight (2013) estimates.

Discussion

Several differences among the approaches taken in the 3 studies can be identified (Table 1). Loss et al. (2013) and Smallwood (2013) included all species of birds, whereas Erickson et al. (2014) focused on small passerines, which would result in a reduced estimate of total fatalities. Smallwood's (2013) review included an over-representation of data from Altamont Pass in 1 set of estimates, which likely would increase his estimate compared with the others. The number of studies included in each analysis varied from 53 to 116. Smallwood (2013) and Loss et al. (2013) used an adjustment for size of the search plot that Smallwood (2013) developed; Erickson et al. (2014) recognized the potential problem of search plot size but commented that larger plots likely would contain more fatalities from other causes, such as predation. Some studies in the Altamont have suggested that burrowing owl (*Athene cunicularia*) mortality may in part be due to predation.

Smallwood (2013) and Erickson et al. (2014) made no adjustment for studies not conducted for a full year. Their estimates accordingly would not include any fatalities that occurred during non-search periods, most of which presumably were when birds were largely absent from the area of the wind energy facility. Loss et al. (2013) made such an adjustment, which, if birds actually were not at risk during non-search periods, would inflate their estimate, but by only about 13%. Loss et al. (2013) used fatality estimates based on adjustments for imperfect availability and perceptibility used by original authors. The other analyses used consistent sets of adjustment values (Table 1). Accuracy of the various adjustment methods differ in relation to a number of variables, and cannot be ascertained readily.

All reviews used ratio estimation to obtain estimates of total numbers of fatalities. Ratios used by Smallwood (2013) and Erickson et al. (2014) were estimated number of fatalities per megawatt capacity. Loss et al. (2013) instead used the number of fatalities per turbine, adjusted for height, which correlates strongly with capacity. Capacity may not be an ideal denominator because, for example, a 3.0 megawatt turbine with 90-m-long blades has only 37% more rotor-swept area than a 1.5 megawatt turbine with 77-m-long blades. Even the rotor-swept area has issues as a denominator, because a smaller fraction of that area is occupied by a blade at any moment in time for a larger rotor than for a smaller one (Tucker 1996*a, b*). Other denominators to consider might be (1) actual output from a turbine or (2) the turbine's rotor-swept area combined with hours of operation, weighted by bird activity at the time. Unfortunately, these data are generally not available. Further investigation into this issue is warranted.

Variation in which studies were included in each review raises a major question about data availability. Each author took great pains to find as many studies as possible, subject to selection criteria mentioned. That the reviews differed in studies found indicates a major lack of transparency in accessibility of reports on fatality studies conducted at wind energy facilities. The overriding question is how representative the data sets were. Random, systematic, or stratified sampling of wind farms, turbines, and years, along with large sample sizes, would ensure with high probability a representative sample. However, none of these methods was used. Each sample is a hodge-podge of sampling units. A statistician would approach such a sample either with considerable caution and a host of caveats, or with a blindfold of optimism.

Evidence that the sample is not representative is ample. The issue of whether fatality rates at monopole and lattice design towers differ and should be included or not is an example. Some lattice towers remain, and others may be built, so they should be included in the sample in appropriate proportions. Clearly the geographical representation of sampled wind farms differs dramatically from the universe of wind farms that exist. For example, in the Loss et al. (2013) stratification, 53% of the capacity was in the Great Plains, versus 15% of the surveyed turbines. Erickson et al. (2014) noted the absence of surveyed wind farms in the Southwest, despite the region hosting 7% of wind capacity. Texas is the leading state in terms of wind energy production, but wind farms there are poorly represented in the available data. However, due to the recent U.S. Fish and Wildlife Service wind energy guidelines (USFWS 2012), it appears that a much larger percentage of projects are collecting avian fatality data. Since these review papers were written, >50 additional studies are now available for inclusion in future meta-analyses, including several studies in areas where data were lacking at the time.

Because fatality studies generally are conducted by or financially supported by the wind industry, a skeptic might question if results of studies demonstrating high rates of fatalities are made as easily available as results from innocuous wind farms. Legal requirements for wind energy developers to ensure accessibility of study results would resolve many problems associated with analyses, such as those reviewed here.

A study analogous to the 3 studies we reviewed was conducted for Canada by Zimmerling et al. (2013), who estimated the total number of fatalities in 2011 to be 23,300 birds. That number was based on an estimate of 8.2 fatalities per turbine annually and included adjustments for the biases we discussed above and direct extrapolation of that value across all wind turbines in Canada.

Despite the differences among the 3 reviews, all of them estimated annual number of bird fatalities from wind developments within the same magnitude, roughly a quarter- to a half-million birds per year at 2012 build-out levels. Of course there are more turbines now and many more are planned. Along with the associated power lines and towers, number of fatalities will increase. Prompted by the U.S. Fish and Wildlife Service (2012) guidelines, more studies are being conducted, resulting in additional information to understand impacts, risk, and siting concerns.

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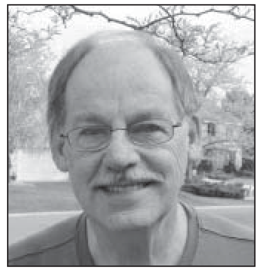

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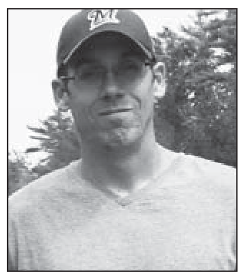

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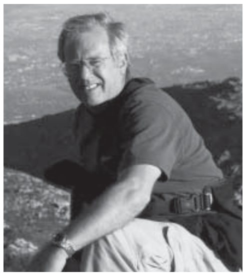

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