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Research Article

Bat Mortality Due to Wind Turbines in Canada

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ABSTRACT Wind turbines have been hypothesized to affect bat populations; however, no comprehensive analysis of bat mortality from the operation of wind turbines in Canada has been conducted. We used data from carcass searches for 64 wind farms, incorporating correction factors for scavenger removal, searcher efficiency, and carcasses that fell beyond the area searched to estimate bat collision mortality associated with wind turbines in Canada. On average, 15.5 ± 3.8 (95% CI) bats were killed per turbine per year at these sites (range = 0–103 bats/turbine/yr at individual wind farms). Based on 4,019 installed turbines (the no. installed in Canada by Dec 2013), an estimated 47,400 bats (95% CI = 32,100–62,700) are killed by wind turbines each year in Canada. Installed wind capacity is growing rapidly in Canada, and is predicted to increase approximately 3.5-fold over the next 15 years, which could lead to direct mortality of approximately 166,000 bats/year. Long-distance migratory bat species (e.g., hoary bat [*Lasiurus cinereus*], silver-haired bat [*Lasionycteris noctivagans*], eastern red bat [*Lasiurus borealis*]) accounted for 73% of all mortalities. These species are subject to additional mortality risks when they migrate into the United States. The little brown myotis (*Myotis lucifugus*), which was listed as Endangered in 2014 under the Species At Risk Act (SARA), accounted for 13% of all mortalities from wind turbines, with most of the mortality (87%) occurring in Ontario. Population-level impacts may become an issue for some bat species as numbers of turbines increase.

KEY WORDS bat populations, correction factors, mortality, wind turbines.

Although wind power is widely viewed as a clean alternative to fossil fuel-based energy generation, there has been some concern regarding the impact of wind farms on bats (Cryan 2011, Arnett and Baerwald 2013, Hayes 2013, Smallwood 2013). Bats can be killed either through direct collisions with turbine blades (Horn et al. 2008, Rollins et al. 2012) or barotrauma (Baerwald et al. 2008, Grodsky et al. 2011, but see Rollins et al. 2012). Unlike mortality associated with migratory birds, for which collisions with wind turbines represent a small percentage of total mortality caused by collisions with human structures (Longcore et al. 2012, Calvert et al. 2013, Zimmerling et al. 2013), for bats, wind turbines represent one of the largest sources of anthropogenic mortality (Cryan and Brown 2007, Cryan 2011, O'Shea et al. 2016). Results from mortality studies at various sites in the United States and Europe generally suggest that annual bat mortality ranges from 0 to >50 fatalities/turbine, although data collection protocols, experimental design, and analysis methods varied substantially among wind farms, making data difficult to compare (Arnett et al. 2008, Cryan 2011, Arnett and Baerwald 2013, Hayes 2013, Smallwood 2013). Kunz et al. (2007) predicted that as many as 110,000 bats may be killed per year by 2020, assuming there would be 48,000 turbines in the United States, based on an average of 2.3 bats/turbine/year. Cryan (2011) suggested that this may be a considerable underestimate, and provided a number of 450,000 bats/year based on an average published mortality rate of 11.6 bats/megawatt (MW)/year and an estimated 40,000 MW of installed capacity in Canada and the United States at the time. Using a different approach, Smallwood (2013) estimated 17.2 bats/MW/year (or 34.4 bats/turbine for a typical modern 2-MW turbine), which would represent about 888,000 bats killed each year across an installed capacity in 2012 of 51,630 MWs.

The reason for the high mortality rate of bats at wind turbines is uncertain. A relatively high percentage of migrating bats may fly at altitudes <120 m, within the rotor swept zone of a turbine. Bats may be unable to recognize moving blades as a threat, and even if they avoid a collision with the turbine blade, they may suffer barotrauma (Baerwald et al. 2008, Grodsky et al. 2011), although Rollins et al. (2012) questioned the evidence that bats are killed by barotrauma based on empirical data from forensic

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analyses. Alternatively, bats may actually be attracted to turbines (Cryan and Barclay 2009), perhaps because insect concentrations around turbines provide feeding opportunities, or because they mistake the monopole of wind turbines for a tall roost tree (Hensen 2004, Cryan and Barclay 2009). Behavioral observations of bats around turbines using thermal and infrared imagery suggest that bats may investigate the nacelles (the housing that encloses the gears and generating components at the top of the tower) of wind turbines (Horn et al. 2008, Cryan et al. 2014, Jameson and Willis 2014), thus increasing their risk.

North American long-distance migrants (e.g., hoary bat [*Lasiurus cinereus*], silver-haired bat [*Lasionycteris noctivagans*], eastern red bat [*Lasiurus borealis*]) typically comprise $\geq$66% of all fatalities at wind turbine installations (Kunz et al. 2007, Arnett et al. 2008), although at some individual sites, mortality of mouse-eared bats (*Myotis* spp.) may be similar to some of the long-distance migrants (Arnett et al. 2008, Arnett and Baerwald 2013). The long-distance migrants use linear landscape features (Hensen 2004, Cryan and Barclay 2009), and fly longer over potentially unfamiliar terrain, all of which may increase the risk that they interact with a turbine. Other species, such as little brown myotis (*Myotis lucifugus*), may also fly significant distances between their summer roosts and winter hibernacula sites, and it is possible that some of the sites with high mortality of these species are along these movement corridors.

Whether the reported levels of mortality could be having population-level impacts on some species of bats has not been assessed (Arnett and Baerwald 2013), although such impacts are possible (Cryan 2011, Hayes 2013, Smallwood 2013, Baerwald et al. 2014). Bats have low reproductive rates so even low levels of mortality could lead to population declines (Barclay and Harder 2003). Mortality due to wind turbines is likely to be cumulative to other sources of mortality, and could be particularly important for populations that are already stressed by other factors such as the fungal disease that causes white-nose syndrome (WNS; *Pseudogymnoascus destructans*). This disease has caused population declines of $\geq$95% for little brown myotis, northern myotis (*Myotis septentrionalis*), and tri-colored bat (*Perimyotis subflavus*) in many regions of the eastern United States and Canada (Blehert et al. 2009). Mortality due to wind turbines could accelerate declines and slow recovery of bats (if they develop resistance to WNS), thus threatening the long-term viability of populations.

Wind energy development in Canada has grown in recent years, but a comprehensive analysis of bat mortality data from Canadian sites has not yet been undertaken. Installed capacity of commercial wind power in Canada increased 7.5 times between 2005 and 2013; as of December 2013, the country had >167 wind farms with 4,019 wind turbines, and this number is expected to increase 3.5-fold over the next 15 years (Canadian Wind Energy Association [CANWEA] 2016). Wind energy development has been subject to considerable effort for environmental assessment, relative to many other forms of industrial development. The high effort required for environmental assessments has been due partly to uncertainty about the impacts of a relatively new and changing technology and concerns about wildlife impacts observed at some of the early facilities (e.g., California, USA). As a result, many wind farm developers have undertaken lengthy baseline studies or collaborative research for multiple seasons and years to determine wildlife use of the site prior to development (A. C. Pomeroy and M. V. d'Entremont, Stantec, unpublished report). The results of these studies, which have focused on birds and bats, have, at some sites, influenced the placement of turbines within wind facilities and occasionally influenced the decision whether or not to proceed with development of a wind facility (National Wind Watch 2011). Once constructed, many sites have carried out extensive and costly post-construction studies to monitor direct mortality caused by wind turbines to birds and bats. Our objectives were to analyze available data on bat mortality from post-construction monitoring reports in Canada to derive national and provincial estimates of total bat mortality associated with wind turbines, to compare them with published estimates from other countries, and to estimate whether this level of mortality may be having potential population-level impacts on some species.

STUDY AREA

We quantified the extent of bat collisions with wind turbines at 64 wind farms across 9 provinces in Canada. The number of wind farms analyzed per province varied from 2 wind farms in the provinces of British Columbia, Manitoba, and New Brunswick to 31 in Ontario (Table 1). We were unable to obtain data for wind farms in the Yukon Territory, Northwest Territories, or the province of Newfoundland and Labrador. There are currently no wind farms operating in Nunavut Territory.

Most of the wind farms were sited in agricultural landscapes, where grains (corn, soya bean, and wheat) or pasture were the predominant vegetation type. Some wind farms were sited in forested areas, with variable size clearings around the base of the turbine. The immediate area around the base of each wind turbine was surrounded by a small gravel pad.

Data from wind farms were generally collected in spring (Apr–May) and fall (Aug–Nov), but at some sites, data were also collected in the summer (Jun–Jul; Appendix 1). Bats are not typically active in the winter (Dec–Mar) in Canada and so data from these months were not included. Most data were collected between 2007 and 2013, although data for 7 wind farms were from 2002–2006 (Appendix 1).

METHODS

We obtained available post-construction monitoring reports for sites across Canada that had been submitted to Natural Resources Canada as part of the post-construction monitoring requirements of an environmental assessment. We also obtained data from post-construction monitoring reports from the Bird Studies Canada Wind Energy Bird and Bat Database (www.bsc-eoc.org/birdmon/wind/main.jsp) and data from developers or their consultants. These reports provided the results of carcass searches conducted around the

Table 1. Estimated bat mortality per turbine from collisions at 64 Canadian wind facilities (2002–2013) with available carcass search data, and predicted total mortality/province based on the number of installed turbines.

Province or territory	No. wind farms	No. turbines	No. wind farms analyzed	Estimated mortality/turbine	Predicted annual mortality
Yukon[a]	2	2	0	1.5	3
Northwest Territories[a]	1	4	0	1.5	6
British Columbia	4	162	2	4.2	680
Alberta	30	760	11	10.9	8,284
Saskatchewan	6	133	3	11.7	1,556
Manitoba	3	123	2	23.3	2,866
Ontario	50	1,270	31	24.5	31,115
Quebec	23	1,150	3	2.1	2,415
New Brunswick	4	113	2	0.6	68
Nova Scotia	30	181	6	0.5	91
Prince Edward Island	10	95	4	2.1	200
Newfoundland[a]	4	26	0	1.4	36
Total	167	4,019	64		47,397

[a] Where no data were available, we calculated estimated mortality per turbine based on the averaged estimates from 3 adjacent provinces for Newfoundland, and the northernmost wind farm in British Columbia for the Territories of Northwest and Yukon.

base (35–85-m radius) of turbines at operating wind farms in 9 provinces. Bat carcass data collected by environmental consultants were collected humanely and ethically and consultants followed guidelines accepted at the time data were collected.

Estimating Mortality Rates

Data collection methodologies used to estimate mortality from carcass searches were not standardized and therefore varied between wind energy developments, especially prior to 2007 when federal environmental assessment guidelines for birds were published (Environment Canada 2007*a*,*b*). Similar federal guidelines in Canada have not been published for bats because they fall under provincial jurisdiction. The lack of standardization in data collection protocols means that the raw carcass counts may not be comparable among studies. Some carcasses, for example, may be removed by scavengers, others may land outside the search area, and others may be overlooked by the searcher. These factors may vary among studies depending upon the terrain and vegetation type, the area searched, the interval between searches, the timing of the searches, and turbine height (Hull and Muir 2010). Nevertheless, by using a standardized analytical approach that takes into account all relevant correction factors (i.e., searcher efficiency, scavenger removal, proportion of area searched within a 50-m radius, proportion of carcasses expected to fall within a 50-m search radius, proportion of carcasses expected during the times of year that surveys took place), and using data specific to each wind farm, it is possible to estimate collision mortality rates, and to make direct comparisons among studies.

Searcher efficiency (Se) and scavenger removal (Sc).—The estimated values for searcher efficiency (range = 0.30–1.0) and scavenger removal (range = 0.10–0.91) varied substantially, highlighting the importance of using site-specific values whenever possible (Appendix 1). Differences in searcher efficiency are expected because of variation in the vegetation type being searched, which varied from gravel pads to agricultural fields to regenerating vegetation and differences in observers and their experience. Estimates of scavenger removal are also expected to vary among sites, depending both on the vegetation type, which affects the ease with which scavengers can find carcasses, and the scavenger community in any given area, which potentially includes birds, mammals, and invertebrates such as ants and burying beetles (Labrosse 2008).

We generally accepted the values for searcher efficiency and scavenger removal provided in the post-construction monitoring reports. However, we recalculated searcher efficiency and scavenger removal values where obviously inappropriate data were used (e.g., data from the winter season, when bats are not actively foraging in Canada, or when large conspicuous birds such as gull carcasses [family Laridae] were the only test carcasses used for trials).

We acknowledge the possibility that some estimates of detection probabilities and scavenger removal may be biased in various ways. In some studies, multiple observers with different levels of experience may have carried out the carcass searchers, but only a single value of searcher efficiency was provided, and it was unclear whether this was an appropriately weighted average across observers. Some reports indicated that carcasses used for searcher efficiency and/or scavenger removal trials were not always placed in the same vegetation types where carcass searches were conducted; if they were concentrated in vegetation types that were easier or harder to search, this could have created a bias. The type of carcasses used in the searcher efficiency and scavenger removal trials also varied among projects; most studies used bats and birds that had previously been found around turbines, but a few used young rats and mice of varying ages. Usually, data from different types of carcasses were pooled in the studies so bat-specific searcher efficiency or scavenger removal values were unavailable. Labrosse (2008) demonstrated that detection probability associated with carcasses increases with the contrast and color against the background. If domesticated birds or rodents are used for searcher efficiency trials, this could lead to biased estimates if they are more conspicuous than typical wild birds or bats. Furthermore, searcher efficiency trials should be conducted when the searcher is unaware when they are being tested to avoid changes in search patterns on testing days; if the searcher finds a domestic bird or rodent, he/she would immediately be aware of the trial. Scavenger removal trials may also have been biased if they were only carried out during part of the season, domestic rodents instead of bats were used, they were not in all of the vegetation types being searched, or some of the carcasses had already been dead for a while (freshly dead bats and birds are likely to be scavenged much faster than older, partly dehydrated or frozen carcasses; Van Pelt and Piatt 1995). The implications of these potential biases are considered further in the discussion.

Proportion of area searched within a 50-m radius of turbines (Ps) and proportion of carcasses expected to fall within a 50-m search radius (Pr).—Although all post-construction monitoring studies provided estimates of the proportion of area searched within a 50-m radius of turbines (Pr), most did not estimate the proportion of bats expected to fall beyond 50 m, so we used standardized estimates based on 9 studies that monitored a larger search area. The distribution of carcasses in relation to distance from the turbines is affected mainly by the height of the turbines (Hull and Muir 2010, but see Klug and Baerwald 2010, Huso and Dalthrop 2014*a*), which is very similar for most Canadian turbines, and is not affected by vegetation type, searcher efficiency, or scavengers. Hull and Muir (2010) used a Monte-Carlo approach based on ballistics to model the proportion of carcasses that would be thrown various distances from a turbine, assuming that bats acquire a forward momentum based on the speed of the blade and are equally likely to be hit anywhere along the length of the blade. For turbines with an 80-m nacelle and 45-m-long blades, similar to most Canadian turbines, they suggested that 99% of bats with a weight of 14 g (similar to a big brown bat [*Eptesicus fuscus*]) would land within 57 m of the turbine base. Because all bats killed at Canadian turbines were small to midsized (range = 4–30 g), we assumed that a radius of 80 m would include nearly all individuals, and used empirical data from post-construction monitoring reports that had searched areas up to 80 m around turbines. All of these studies used equal-width transects to provide uniform coverage through the search area. In 3 studies, an 80-m or larger radius was searched around each turbine. In 5 studies, a 120×120-m square grid was searched around each turbine, whereas 1 used a 160×160-m square grid. All of these studies provided complete coverage out to 60 m and partial coverage at greater distances. To estimate the number of carcasses that fell between 60–80 m from the turbine, we extrapolated the number of carcasses found per square meter in the corners of the square search grid to a circle in 10-m increments from 60 m out to a radius of 80 m. Averaged across these 9 studies, we estimated that 82% of carcasses fell within a 50-m radius (Pr). This estimate is similar to an estimate based on our own unpublished study in spring 2013 that searched the entire 85-m radius around turbines in an agricultural area and found 65 carcasses of which 83% were within a 50-m radius. We thus applied a correction factor of 82% to the estimated mortality for the 55 wind farms that only searched up to a 50-m radius.

Proportion of carcasses expected during the times of year that surveys took place (Py).—Most sites were surveyed for only part of the year, generally in seasons when the highest risk of mortality was anticipated, usually the spring and autumn migration periods. At 1 site, only 3 months (i.e., 1 season) of post-construction monitoring data were available. Most of the reports provided the number of carcasses and estimated total mortality per site only for seasons that were surveyed. To extrapolate estimates to an annual total (Py), we used data from 4 wind farms in Alberta and 2 in Ontario that were surveyed for 2 years throughout the annual cycle, and for which mortalities were reported for each month. Using these data, we estimated the monthly distribution of mortality throughout the year. We then estimated annual mortality for other sites by dividing the corrected number of bat mortalities per turbine by the proportion of mortalities expected during the actual dates that surveys took place (e.g., in Alberta, 100% of mortalities occurred between 01 Apr and 31 Oct, whereas 100% of mortalities occurred between 01 May and 31 Oct in Ontario). We acknowledge that there are limitations to extrapolating results from one season to other seasons based on only 6 studies because seasonal patterns of mortality rates may vary among locations. However, because most studies concentrated their efforts during seasons with the highest expected mortality, and because bats are not normally active in Canada during the winter months, any error associated with extrapolation to these seasons is unlikely to have much impact on our estimates.

We used data specific to each of the 64 wind farms, and applied the following standardized formula so that data were analyzed the same way for each site and could be compared:

$$\mathrm{C} = \mathrm{c}/(\mathrm{Se} \times \mathrm{Sc} \times \mathrm{Ps} \times \mathrm{Pr} \times \mathrm{Py}) \tag{1}$$

where C = corrected number of bat mortalities, c = number of carcasses found, Se = proportion of carcasses expected to be found by searchers (searcher efficiency), Sc = proportion of carcasses not removed by scavengers over the search interval (scavenger removal), Ps = proportion of area searched within a 50-m radius of turbines, Pr = proportion of carcasses expected to fall within a 50-m search radius, and Py = proportion of carcasses expected during the times of year that surveys took place. Our analyses assumed that all carcasses found were killed as a result of interactions with turbines. Where multiple years of data were collected at a site, we used the average of all years for the analysis. In many studies, there were insufficient data to apply separate correction factors and estimate mortality for each season (i.e., spring, summer, and fall), so we used average factors. At 1 wind farm in Alberta, searcher efficiency was not reported and, at 2 wind farms, scavenger removal was not reported. In these cases, we applied the average values for wind farms in Alberta (i.e., $Se = 0.65$ and $Sc = 0.61$). We did not include data from any reports in our analysis where both searcher efficiency and scavenger impact data were absent, or where surveys occurred for <3 months throughout the year.

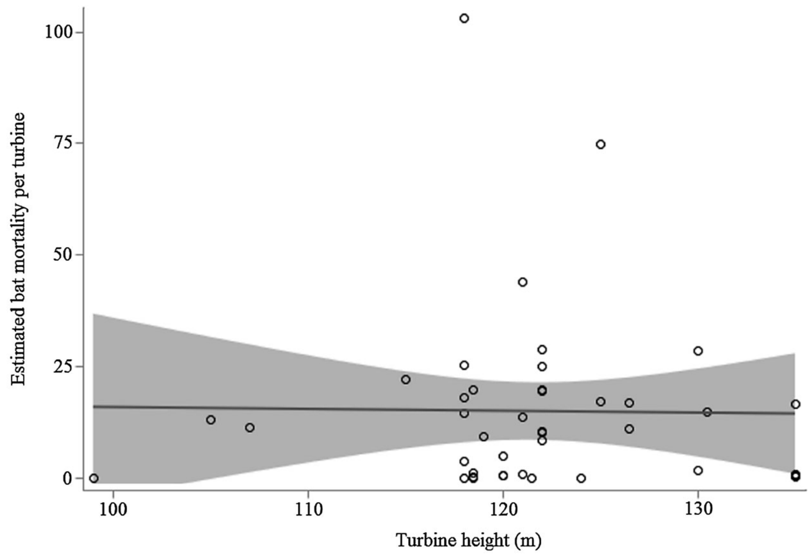

Figure 1. Regression of estimated annual bat mortality per wind turbine in relation to turbine height (i.e., hub height plus blade length) based on data from 47 wind facilities in Canada that were commissioned after 2007. Confidence intervals (95%) are shaded.

We estimated total bat mortality for each province based on estimated average mortality per turbine in that province for studies for which we had data, multiplied by the total number of turbines in the province. We were unable to obtain collision data for the province of Newfoundland and Labrador, and so used the average estimated mortality per turbine for the 3 other adjacent provinces (i.e., New Brunswick, Nova Scotia, Prince Edward Island). For the territories of Northwest and Yukon, we used the estimated mortality for the northernmost wind farm in British Columbia. We estimated total collision mortality for wind farms across Canada as the sum of the provincial estimates.

To determine if there was any significant variation in estimated mortality among the provinces for which we had collision mortality data, we conducted a 1-way analysis of variance (ANOVA) using estimated average mortality for each of the 64 wind farms as the sampling unit and province as the predictor variable. We did not analyze whether specific turbine model types or characteristics affected the risk to bats, with the exception of turbine height, where we conducted a regression to determine if there was a correlation between turbine height and estimated bat mortality per turbine.

RESULTS

Estimates of collision mortality among the 64 wind farms varied between 0 and 103 bats/turbine/year (Appendix 1). The average estimated mortality was 15.5 ± 3.8 bats/turbine/year (95% CI: 11.7–19.3). Estimated mortality per turbine differed among provinces in Canada ($F_{8,\ 55} = 2.56$, $P = 0.019$), with the highest mortality rates in Ontario followed by the prairie provinces (Table 1). Sixty-six percent of all mortalities occurred in the province of Ontario, which had the highest mortality rates and the most turbines installed (Table 1). Based on 4,019 installed turbines (the no. installed by Dec 2013), the estimated annual mortality across Canada would be 47,400 bats (95% CI: 32,100–62,700).

There was no relationship between bat mortality/turbine and height of wind turbines ($r^2 = 0.0002$; Fig. 1). There was relatively little variation in the height of wind turbines (i.e., hub height plus rotor radius) in the sample for which we had data. Of the wind farms analyzed in this study, 89% started their post-construction monitoring studies after January 2007, and based on a review of turbine specifications, we found that the height (i.e., tower and blades) of nearly all of the turbines erected since 2007 was between 117 m and 136 m with a capacity of 1.5 MW to 3.0 MW.

Overall, the 58 reports that recorded species composition during post-construction mortality surveys provided information on 4,615 bat carcasses of 9 species. The most frequently recovered species were hoary bat, silver-haired bat, eastern red bat, little brown myotis, and big brown bat, comprising 99% of all identified fatalities, of which the first 3 are long-distance migratory bats comprising 73% of the mortalities (Table 2). Of the 3 bat species listed as Endangered under the Species At Risk Act, an estimated

Table 2. Species-specific reported mortality and predicted annual bat mortality based on 4,615 bat carcasses from 64 wind facilities across Canada collected between 2002 and 2013. We used only the number of turbines within a species' range to estimate total predicted annual mortality.

Species	No. carcasses	Proportion of total	Predicted mortality
Hoary bat	1,553	0.34	16,345
Silver-haired bat	1,149	0.25	11,093
Eastern red bat	677	0.15	6,998
Little brown myotis	586	0.13	5,832
Big brown bat	411	0.09	4,075
Northern myotis	35	0.01	464
Eastern small-footed myotis	3	0.00	32
Tri-colored bat	3	0.00	15
Long-legged myotis	1	0.00	2
unknown bat species	197	0.04	2,498

87% of all little brown myotis mortality across Canada occurred in the province of Ontario. In contrast, the northern myotis and tri-colored bat comprised only 1% of all mortality.

DISCUSSION

Turbine-Related Mortality Estimates

Our estimates for average annual mortality per turbine at wind farms in Canada of 15.5 bats is higher than many estimates presented in reports from individual studies in Canada. This is due mainly to incorporation of an additional correction factor: the proportion of bats likely to fall outside a 50-m search radius. Correction for the proportion of bats killed at other times of year also contributed to the difference.

Nevertheless, our estimates are lower than some recent estimates for bat mortality in the United States. Smallwood (2013) undertook a detailed assessment of correction factors based on data from 60 different reports, and estimated an average mortality of 17.2 bats/MW/year, implying 34 bats/turbine for a 2-MW turbine. Extrapolated to an installed capacity of 51,630 MW in the United States, this implies 888,000 bat fatalities/year. Most of the difference in Smallwood's (2013) estimates, compared to ours, appears to be due to differences in the correction factors rather than a difference in the number of carcasses found in the data he analyzed. For example, he used larger corrections for bats falling outside a 50-m radius. He extrapolated from data collected primarily within 50 m of the turbines assuming a logistic distribution of carcasses in relation to distance from turbines. This model assumes that carcasses could fall up to 126 m away from an 80 m turbine. This is 2 times farther than Hull and Muir (2010) suggested is likely based on aerodynamic theory. Furthermore, his analysis assumed that mortality is proportional to the rated capacity of the turbines, but particularly for newer turbines this seems unlikely; for example, there is only a 19% increase in the blade swept area between a 1.5 MW and 3.0 MW turbine (https://www.gerenewableenergy.com/wind-energy/turbines/product-specs.html, accessed 05 Nov 2015). Hayes (2013) used distribution-fitting analysis to determine which distribution curve best fit bat fatality data from 21 sites in the United States. Based on his analysis, Hayes (2013) derived an estimate of 11.7 bats/MW/year, implying 23.4 bats/turbine for a 2-MW turbine. Extrapolated to an installed capacity of 51,630 MW in the United States, this implies 604,000 bats fatalities/year. However, Hayes used a relatively small number of studies ($n = 21$) that used different methodologies, which were not a representative sample of wind farms across the United States (Huso and Dalthrop 2014*b*).

We found substantial variation among sites in the estimated mortality/turbine, ranging from 0 to 103 bats/year with significant variation among provinces. Variation in mortality estimates is expected because of site-specific characteristics that may concentrate migratory bats in some areas and not in others. For example, landscape features such as promontories and large bodies of water may be more likely to concentrate bats along the shoreline (Diehl et al. 2003). This may be why the province of Ontario, where the majority of wind farms are located within 20 km of shoreline, had the highest estimated mortality rate of any province or territory. Variation is also likely to be influenced by differences in population densities of bats among regions, but no data are available on geographic variation in bat population densities in Canada.

The accuracy of collision mortality estimates depends on the amount of carcass search data and the accuracy of correction factors used to account for incomplete carcass detections. If search effort is only sufficient to detect a few (or no) carcasses, estimates will be unreliable regardless of correction factors used. Several factors could lead to overestimates of searcher efficiency, including use of inappropriate carcasses that may be more conspicuous or larger than species that would be expected to be found during carcass searches (Labrosse 2008), concentrating carcasses in more exposed vegetation types within the search area, and failing to ensure that searcher efficiency trials occur without the knowledge of the observer. Scavenger removal may be biased if carcasses used in the trials are not fresh (Smallwood 2013), are not representative of the species being detected, or if too many carcasses are used at one time for trials (i.e., scavenger swamping; Smallwood et al. 2010). All of these could result in underestimates of mortality. On the other hand, many studies estimate scavenger removal over the total search interval (e.g., 3 days). This may lead to an overestimate because, on average, one would expect carcasses to be exposed to potential scavengers for only half the search period (e.g., for a typical 3-day search interval bats would be equally likely to be killed 1, 2, or 3 days before the search, leading to an average exposure of 1.5 days). Our correction factors may also be biased low if some bats fall beyond 80 m (Jain et al. 2007, 2009; Smallwood 2013), although ballistic modeling suggests very few bats are expected beyond that distance (Hull and Muir 2010) given the height of turbines for wind energy projects we analyzed. On the other hand, we assumed that all bats found during carcass searches died as a result of interacting with the turbines. If some of these bats died from other sources of mortality unrelated to wind turbines, this would lead to an overestimate of the impacts of turbines. The net effect of these various potential biases, both positive and negative, is difficult to predict, because some may cancel each other; given the available data and assumptions, our estimates are reasonable.

Species-Specific Population Impacts

Mortality from wind power could potentially affect the dynamics of some regional bat populations in Canada, either now or in the near future. In Canada, the little brown myotis, northern myotis, and tri-colored bat were all listed as Endangered in 2014 under the federal Species at Risk Act (http://canadagazette.gc.ca/rp-pr/p2/2014/2014-12-17/html/sor-dors274-eng.php, accessed 05 Nov 2015), because of population declines associated with WNS. One of these, the little brown myotis, had the fourth highest mortality due to wind turbines of bat species in Canada. Based on United States

Fish and Wildlife Service (2014) estimates, the little brown myotis population size in eastern North America was around 6 million individuals prior to the arrival of the WNS fungus, with the present day population about 600,000 bats, half of which would occur in Canada assuming an equal distribution of little brown myotis in Canada and the United States. Our estimates of mortality for little brown myotis suggest that current wind turbine collision mortality could be up to 1.4% of the total eastern population, which could have a significant impact on likelihood of recovery. This is likely to be an overestimate if the number of fatalities has declined over time as a result of WNS. However, the biological significance of any remaining mortality could still be important because the mortality of even a few individuals has the potential to affect the ability of little brown myotis populations in eastern Canada to recover in the future if they do develop resistance to the fungus, especially with the projected increase in numbers of wind turbines. Numbers of mortalities were much smaller for the eastern small-footed myotis (*Myotis leibii*), tri-colored bat and long-legged myotis (*Myotis volans*), but it is not known whether these species are less vulnerable to impacts from wind turbines because of differences in flight, foraging behavior, or habitat, or whether they simply have small populations and are therefore generally uncommon around wind farms. Nevertheless, even low rates of mortality have the potential to be biologically significant for relatively rare species, and it is possible that future wind farms, if accidentally sited near important concentration areas such as maternity roosts or hibernacula, could cause higher mortality.

For long-distance migrants, the most commonly recorded bat species in carcass searches, our estimates suggest potential population impacts could be expected in the future. In Canada, the population size of hoary bat is estimated to be about 2.5 million individuals (E. F. Baerwald, American Wind Wildlife Institute, personal communication). At present mortality rates, wind turbines in Canada may be affecting 0.66% of the current population. With a projected 3.5-fold increase in turbine installation over the next 15 years (CANWEA 2016), this could lead to direct mortality in Canada of approximately 2.3% of the hoary bat population annually. Furthermore, long-distance migrants, such as the hoary bat, that move south through Canada, potentially encounter some of the more than 23,000 wind turbines in the United States, where an estimated 604,000 bats are killed each year by turbines (Hayes 2013) of which about 33% of the mortalities are hoary bats (Arnett and Baerwald 2013). In the absence of data on the origin of bats that are killed at wind facilities in the United States, if we assume a proportion (e.g., 0.33) of the hoary bats killed there originated or migrated from Canada, this represents an additional 66,000 mortalities per year for Canadian hoary bats, which could increase to 396,000 with a projected 6-fold future growth in turbine installations in the United States. This suggests that, at present, the combined turbine related mortality in Canada and the United States could affect 3.3% of Canadian hoary bat populations, and could increase to 11.5% in 15 years. These levels of mortality are unlikely to be sustainable over the long-term given that bats are long-lived, with low reproductive rates (Barclay and Harder 2003).

MANAGEMENT IMPLICATIONS

Cumulative turbine-related mortality could have population-level impacts on some bat species. As a precautionary principle, at sites and in provinces where bat mortality is high, mitigation measures to reduce mortality (Baerwald et al. 2009, Arnett and Baerwald 2013) should be implemented and follow-up studies conducted to confirm their effectiveness. Implementation of bat population monitoring programs should also be implemented to allow more rigorous evaluation of population impacts to better guide management.

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APPENDIX 1

Number of carcasses (c) and correction factors (Se = searcher efficiency, Sc = scavenger removal, Ps = percent area searched, Py = proportion of carcasses expected during the times of year that surveys took place, Pr = proportion of carcasses expected to fall within a 50-m search radius) used to estimate bat mortality per turbine (EM/turbine) at wind farms in Canada (2002–2013). We gave wind farms a unique number (WF Id), grouped provinces by region (eastern = Prince Edward Island, Nova Scotia, New Brunswick, and Quebec; central = Ontario and Manitoba; western = Saskatchewan, Alberta, and British Columbia), and categorized number of turbines as small (1−10 turbines), medium (11−40 turbines), or large (>40 turbines).

WF Id	Region	Year	Size	Monitoring dates	c	Se	Sc	Ps	Py	Pr	EM/turbine
1	Eastern	2012	Medium	4 Apr–28 Oct	4	0.45	0.77	0.61	1	0.82	1.1
2	Eastern	2012	Medium	19 Apr–2 Nov	1	0.51	0.89	0.68	1	0.82	0.2
3	Eastern	2012	Medium	13 Apr–29 Oct	11	0.85	0.59	1.00	1	0.82	1.0
4	Eastern	2011	Small	01 Apr–01 Oct	0	0.79	0.86	0.82	1	0.82	0.0
5	Eastern	2008	Small	01 Jun–01 Nov	0	0.69[a]	0.71	1.00	0.87	0.82	0.0
6	Eastern	2010	Medium	08 Apr–01 Nov	0	0.65[a]	0.35	0.43	1	0.82	0.0
7	Eastern	2011	Medium	01 Apr–01 Nov	4	0.84	0.59	1.00	1	0.82[a]	0.9
8	Eastern	2009	Medium	15 Apr–15 Oct	5	0.84	0.91	1.00	1	0.82[a]	0.4
9	Eastern	2008	Small	01 Apr–07 Jun, 23 Aug–30 Oct	0	1.00	0.28	1.00	0.6	0.82	0.0
10	Eastern	2002–2006	Medium	16 Apr–01 Nov	0	0.69	0.10	0.22	1	0.82	0.0
11	Eastern	2011	Small	02 May–02 Oct	9	0.84	0.19	1.00	1	0.82	17.1
12	Eastern	2011	Medium	01 May–30 Jun, 01 Aug–31 Oct	1	0.45	0.82	0.81	0.93	0.82	0.4
13	Eastern	2008	Large	30 Apr–11 Oct	11	0.90	0.65	1.00	1	0.82	0.7
14	Eastern	2008	Large	08 May–04 Oct	8	0.77	0.65	1.00	0.98	0.82	0.6
15	Eastern	2008	Large	01 May–15 Oct	33	0.65	0.60	1.00	1	0.82	4.1
16	Central	2010	Large	01 Apr–31 Oct	34	0.81	0.76[a]	0.45	1	0.82	5.6
16	Central	2009	Large	01 Jul–31 Oct	111	0.80	0.56[a]	0.23	0.88	0.82	23.0
17	Central	2007	Large	01 Apr–31 Oct	192	0.60	0.50	0.81	1	0.82	25.5
17	Central	2008	Large	01 Apr–31 Oct	236	0.60	0.45	0.81	1	0.82	10.4
18	Central	2007	Large	01 May–15 Jun, 15 Aug–01 Nov	163	0.76	0.45	0.64	0.8	0.82	31.5
19	Central	2007	Large	10 May–03 Jun, 02 Aug–22 Sep	41	0.70	0.53	0.50	0.87	0.82	6.9
20	Central	2002–2006	Small	15 Apr–31 Oct	8	0.69	0.83	0.53	1	0.82	32.2
21	Central	2008	Medium	14 Apr–30 May, 02 Jul–17 Oct	120	0.43	0.75	0.82	0.97	0.82	15.0
22	Central	2009	Large	21 Apr–31 May, 01 Jul–23 Oct	270	0.72	0.58	0.32	0.97	0.82	77.2
23	Central	2011	Small	01 Apr–01 Nov	139	0.36[a]	0.94[a]	0.81	1	0.82	103.1
24	Central	2010	Large	15 Apr–31 May, 01 Jul–31 Oct	25	0.62[a]	0.53[a]	0.53	0.97	0.82	4.0
25	Central	2012	Medium	01 Jul–31 Oct	167	0.58	0.66	1.00	0.84	0.82	26.4
26	Central	2010	Small	15 Apr–31 May, 01 Jul–Oct	36	0.71	0.77	0.91	0.97	0.82	18.2
27	Central	2009	Large	01 May–31 May, 01 Jul–01 Oct	47	0.30	0.79	1.00	0.97	0.82	11.4
27	Central	2010	Large	01 Jun–30 Oct	68	0.49	0.67	1.00	0.87	0.82	19.4
28	Central	2011	Medium	18 Apr–21 Oct	51	0.64	0.81	0.73	1	0.82	14.9
29	Central	2010	Small	01 Apr–01 Oct	51	0.69	0.63	0.99	1	0.82	28.9
30	Central	2010	Small	01 Apr–01 Oct	33	0.70	0.60	0.98	1	0.82	19.6
31	Central	2011	Small	03 May–31 Oct	48	0.62	0.72	1.00	1	0.82	16.5
32	Central	2010	Small	01 Apr–01 Oct	16	0.68	0.68	1.00	1	0.82	8.4
33	Central	2011	Small	01 Jul–15 Oct	16	0.73	0.65	1.00	0.84	0.82	12.3
34	Central	2008	Small	01 Apr–29 Sep	9	0.60	0.90	0.25	1	0.82	13.6
35	Central	2010	Small	01 Jul–01 Oct	23	0.68	0.82	1.00	0.84	0.82	12.0
36	Central	2011	Large	01 Apr–15 Oct	148	0.68	0.76	1.00	1	0.82	24.9
37	Central	2011	Medium	01 May–31 Oct	47	0.57[a]	0.62	0.98	1	0.82	9.7
37	Central	2012	Medium	18 May–30 Sep	99	0.63	0.55	0.98	1	0.82	20.9
38	Central	2012	Small	15 Jul–30 Sep	70	0.70	0.76	0.92	0.83	0.82	42.0
39	Central	2012	Small	15 Jul–30 Sep	43	0.62	0.60	0.99	0.83	0.82	33.9
40	Central	2012	Small	15 Jul–30 Sep	59	0.77	0.32	0.98	0.83	0.82	71.7
41	Central	2012	Small	15 Jul–30 Sep	36	0.76	0.85	0.99	0.83	0.82	13.8
42	Central	2012	Medium	01 May 1–05 Jun, 15 July–30 Sep	24	0.52	0.65	0.37	0.97	0.82	24.0
43	Central	2013	Small	01 May–31 Oct	27	0.90	0.48	0.63	1	0.82	24.3
44	Central	2012	Small	01 Jul–31 Oct	9	0.52	0.51	0.14	0.84	0.82	72.5
44	Central	2013	Small	01 May–31 Oct	8	0.77	0.69	0.35	1	0.82	5.3
45	Central	2013	Large	01 May–31 Oct	146	0.72	0.72	0.85	1	0.82	18.5
46	Central	2013	Small	01 May–31 Oct	63	0.67	0.91	1.00	1	0.82	12.6
47	Central	2012	Small	08 Aug–26 Sep 26	86	0.75	0.78[a]	1.00	0.86	0.90	31.7
57	Central	2007	Small	08 Aug–26 Sep 26	98	0.53	0.78[a]	1.00	0.86	0.82	11.6
48	Central	2011	Large	09 May–03 Jun, 02 Aug–28 Oct	114	0.47	0.79	1.00	0.89	0.82	21.0
48	Central	2012	Large	02 Apr–29 May, 03 Jul–28 Sep	181	0.41	0.78	1.00	0.96	0.82	36.1
49	Western	2012	Medium	01 Apr–31 Oct	46	0.78	0.61	1.00	1	0.82	19.7
50	Western	2011	Large	01 Apr–31 Oct	85	0.61	0.61	1.00	1	1.00	11.4
51	Western	2002–2006	Small	17 Apr–01 Nov	2	0.42	0.56	1.00	1	1.00	0.9
52	Western	2011	Large	01 Apr–15 Oct	16	0.59	0.61	1.00	1	1.00	1.8
53	Western	2002–2006	Large	01 Apr–01 Nov	19	0.65	0.61	0.46	1	0.82	2.1
54	Western	2002–2006	Medium	01 Apr–15 Oct	45	0.50	0.61	1.00	1	1.00	7.4

(*Continued*)

(*Continued*)

WF Id	Region	Year	Size	Monitoring dates	c	Se	Sc	Ps	Py	Pr	EM/turbine
55	Western	2002–2006	Large	01 Apr–31 Oct	54	0.70	0.61	0.46	1	0.82	2.9
56	Western	2002–2006	Medium	01 Apr–01 Nov	532	0.72	0.75	0.46	1	0.82	67.3
57	Western	2008	Medium	01 Apr–01 Nov	72	0.68	0.61	0.46	1	0.82	23.0
58	Western	2007	Large	01 Apr–15 Oct	47	0.70	0.55	1.00	1	1.00	6.1
58	Western	2008	Large	01 Apr–15 Oct	87	0.75	0.61	1.00	1	1.00	9.5
58	Western	2009	Large	01 Apr–15 Oct	155	0.75	0.61	1.00	1	1.00	16.9
59	Western	2012	Large	01 May–30 Jun, 01 Aug–30 Sep	34	0.46	0.85	1.00	0.93	1.00	3.5
60	Western	2011	Medium	01 Apr–31 Oct	48	0.72	0.65	1.00	1	1.00	4.7
61	Western	2013	Small	04 Aug–29 Sep	8	0.61	0.91[a]	1.00	0.81	1.00	5.9
62	Western	2010	Medium	01 Apr–15 Oct	31	0.88	0.27	1.00	1	1.00	5.9
62	Western	2011	Medium	2 Apr–15 Oct	26	1.00	0.27	1.00	1	1.00	4.4
63	Western	2009	Medium	13 Aug–19 Oct	4	0.52	0.50	0.49	0.78	0.82	1.5
63	Western	2010	Medium	20 Apr–27 Sep	53	0.91	0.30	0.81	1	0.82	17.2
64	Western	2011	Large	01 Apr–20 Jun	2	0.65	0.68	0.47	0.14	0.82	3.5

[a] We adjusted the reported correction factor because it was miscalculated or inappropriate data were used.