

# Effects of turbine height and cut-in speed on bat and swallow fatalities at wind energy facilities

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## Abstract

Understanding the relationship between the height of wind turbines and wildlife fatalities is important for informing and mitigating wildlife collisions as ever taller and denser arrays of wind turbines are erected across the landscape. We examined relationships between turbine height and fatalities of bats and swallows at 811 turbines in Ontario, Canada, ranging from 119 to 186 m tall. We accounted for cut-in speeds, operational mitigation, and taller turbines projecting carcasses farther from the turbine base than shorter turbines. Fatalities of hoary bats (*Lasiurus cinereus* Palisot de Beauvois, 1796), silver-haired bats (*Lasionycteris noctivagans* Le Conte, 1831), and big brown bats (*Eptesicus fuscus* Palisot de Beauvois, 1796) increased with increased maximum blade height of turbines. In contrast, fatalities of little brown bat (*Myotis lucifugus* Le Conte, 1831) and eastern red bat (*Lasiurus borealis* Müller, 1776) decreased with increased turbine height. Fatalities of purple martins (*Progne subis* Linnaeus, 1758) and tree swallows (*Tachycineta bicolor* Vieillot, 1808) were higher at taller turbines than shorter turbines. However, fatalities of cliff swallow (*Petrochelidon pyrrhonota* Vieillot, 1817) and barn swallow (*Hirundo rustica* Linnaeus, 1758) were not associated with turbine height. Our results suggest that varying flight height among species may be one factor affecting collision risk.

**Key words:** aerial insectivore, fatality estimator, Ontario, turbine height, wildlife mortality, wind turbine

## Introduction

Renewable energy sources, such as wind energy, are a critical tool in divesting from fossil fuels and slowing the rate of climate change (Rogelj et al. 2018), but collisions with wind turbines can harm wildlife (Smallwood 2013; Zimmerling et al. 2013; Zimmerling and Francis 2016). Mortality at turbines may have population-level consequences for some species, especially migratory bats (Kunz et al. 2007; Frick et al. 2017). Understanding what factors contribute to fatalities can inform wildlife-friendly wind energy operations.

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Fatalities of wildlife at wind turbines vary with several factors, including proximity to key habitats, such as migration corridors, and cut-in speeds, the wind speeds at which turbines begin generating electricity (Cryan and Brown 2007; Arnett et al. 2011; Thompson et al. 2017; Davy et al. 2020). Fatalities also may vary with turbine size (Table 1), which is important because wind turbines are increasing in height to maximize energy capture (Wiser et al. 2011, 2018). Taller turbines presumably increase exposure risk for wildlife flying at higher altitudes (Davy et al. 2017; Wellig et al. 2018; Roemer et al. 2019), and longer rotors sweep a larger area of aerial habitats used by wildlife.

Evidence for the effect of turbine size on bird and bat fatalities is equivocal (Table 1), possibly reflecting the wide variation in locations, affected species, and turbine type and sizes among studies. Variation in the area searched under turbines also affects fatality estimates. Taller turbines propel carcasses further, potentially causing them to land outside standard search areas (Hull and Muir 2010; Huso and Dalthorp 2014; Choi et al. 2020) and biasing estimates of fatalities low at taller turbines compared to shorter turbines with equivalent search areas. Finally, low sample sizes can preclude analyses of species-specific patterns in fatalities in studies based on smaller datasets, but analyzing total fatalities (i.e., with all species or groups of species pooled) may obscure patterns between turbine height and species-specific fatalities (Davy et al. 2020).

Wind turbines in Ontario, Canada, have increased 42% in size, from 120 to 170 m maximum blade height (MBH), between 2006 and 2019 (Fig. 1a and Table S1), a result of increases in hub height (Fig. 1b) and rotor diameter (Fig. 1c). A previous analysis suggested that turbine hub height may contribute to fatalities of some bat species in Ontario, but confidence in the relationship was not strong (Davy et al. 2020). Here, we conducted a detailed investigation of the effects of turbine height on fatalities in two groups of aerial insectivores, bats (nocturnal) and swallows (diurnal), which are experiencing population declines (Nebel et al. 2010; Committee on the Status of Endangered Wildlife in Canada 2013; Smith et al. 2015; Rodhouse et al. 2019). We expanded on previous work by including data from newer, taller turbines and accounting for carcasses falling outside of search areas at taller turbines. We also used MBH as a measure of turbine height because it is more relevant than hub height to measure the altitudes where species may interact with turbine blades.

## Methods

### Data sources

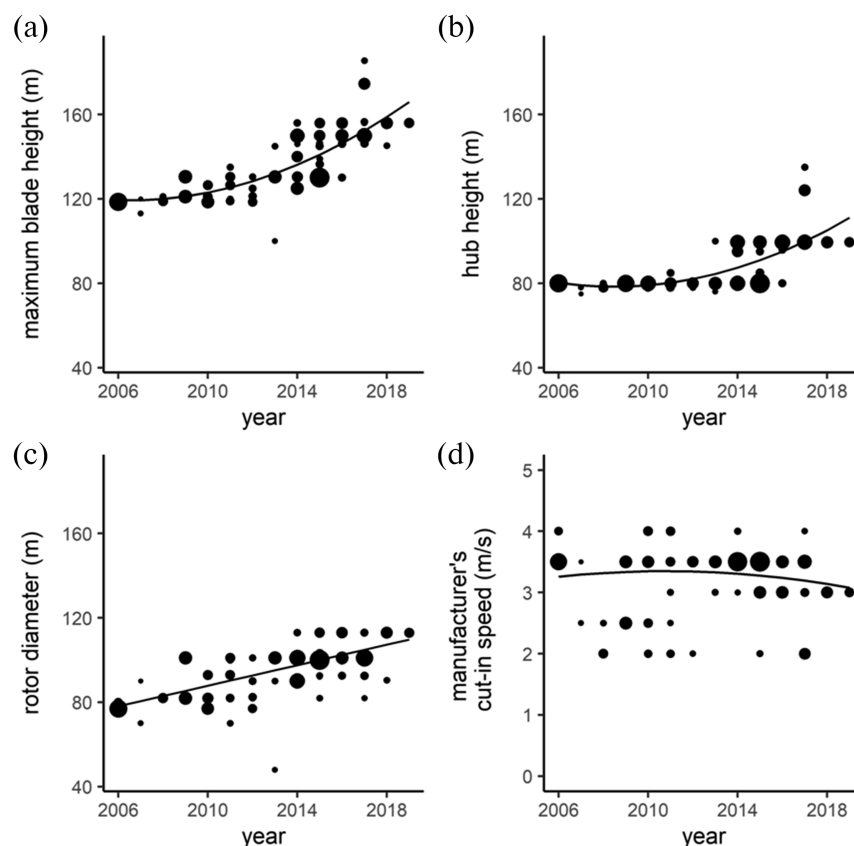
We obtained searcher efficiency, scavenging, and carcass survey data from post-construction monitoring of onshore monopole turbines at wind energy facilities in southern Ontario (south of 48°N) and submitted to the Wind Energy Bird and Bat Monitoring Database (Birds Canada 2020). The database included turbine model names, hub heights, and rotor diameters gathered from facility reports or online (thewindpower.net) and manufacturer-specified cut-in speeds from a repository of wind turbine models (wind-turbine-models.com/turbines).

We used MBH (hub height plus half the rotor diameter) to quantify turbine height in our analyses because hub height and rotor diameter are correlated, and MBH is more ecologically meaningful than hub height alone. Hub height and rotor diameter are moderately correlated in Ontario ( $\rho = 0.49$ ), and including both variables in models could lead to unstable parameter estimates, inflated standard errors, and difficulty assessing the relative importance of the variables (Dormann et al. 2012). Additionally, MBH incorporates hub height and rotor diameter into a single metric representing the true height of a turbine, which has implications for animals flying at different altitudes.

**Table 1.** Studies examining the effects of onshore wind turbine size on number of bat and bird fatalities.

				Turbine size measurement								
				Hub/ nacelle height		MBH (hub height + blade length)		Rotor swept area or rotor diameter		Power capacity		
Type	Taxa	CP and SE correction	Lattice turbines included	Effect	Size (m)	Effect	Size (m)	Effect	Rotor diameter (m)	Effect	Range (MW)	Study
within site	bats	yes	no	+	36 & 50	NA		+	33, 46, & 48	+	0.33–0.75	Johnson et al. (2003), reviewed in Arnett et al. (2008)
within site	bats	yes	no	+	65 & 78	NA		+	46 & 84	+	0.66 & 1.8	Fiedler et al. (2007), reviewed in Arnett et al. (2011)
across sites	birds	yes	yes	O	24–94	NA*		O	15–80	O	<0.1–1.8	Barclay et al. (2007)
	bats	yes	yes	+	24–94	NA*		O	15–80	+	<0.1–1.8	
across sites	birds (raptors)	no	yes	+	21–36	NA		NA		NA		de Lucas et al. (2008)
across sites	bats	yes	NA	+	50–84	NA		NA		NA		Baerwald and Barclay (2009)
across sites	bats	yes	NA	+	24–98	NA		+	NA	NA		Rydell et al. (2010)
across sites	bats	no	NA	+	44–60	NA		O	52–90	NA		Georgiakakis et al. (2012)
across sites	birds	yes	no	+	36–80	NA*		NA*		NA		Loss et al. (2013)
across sites	birds (raptors)	yes	yes	NA		NA		NA		–	<0.1–3.0	Smallwood (2013)
	bats	yes	yes	NA		NA		NA		O	<0.1–3.0	
	birds	yes	yes	NA		NA		NA		O and –	<0.1–3.0	
across sites	birds	SE only	NA	NA		NA		O	23–82	NA		Everaert (2014)
across sites	birds	yes	NA	NA		NA		O	0.9–15	O	<0.1	Minderman et al. (2015)
across sites	bats	yes	no	NA		O	90–135	NA		NA		Zimmerling and Francis (2016)
across sites	bats	yes	no	O	NA	NA*		NA		NA		Thompson et al. (2017)
across sites	bats	yes	no	+ <sup>∞</sup>	83–110	NA		NA		NA		Davy et al. (2020)
within site	birds	yes	NA	NA		+	42–126	+	19–93	+	0.1–2.5	Huso et al. (2021)
	bats	yes	NA	NA		+	42–126	+	19–93	+	0.1–2.5	

Note: NA, not examined or not available in the published study; NA\*, not examined in the study because of correlation with hub height; ∞, not considered informative but in top predictive models; O, neither a positive or negative effect; +, a positive effect; and –, a negative effect. Type of study “within site” is a single wind energy facility with two types of turbine models and “across sites” is comparison of models across wind energy facilities.



**Fig. 1.** Changes in wind turbine (a) maximum blade height, (b) hub height, (c) rotor diameter, and (d) manufacturer's cut-in speed in Ontario, Canada, between 2006–2019. Points represent new turbines added to the landscape in Ontario each year, and the size of the points indicates the total number of turbines with the specified size or speed. Trend lines represent results of linear models with year (and year<sup>2</sup> where applicable) as predictors (Table S1).

## Study species

We modelled the relationship between MBH and fatalities of each bat and swallow species sufficiently represented in the database. These included five species of bat (big brown bat *Eptesicus fuscus* Palisot de Beauvois, 1796; little brown bat *Myotis lucifugus* Le Conte, 1831; eastern red bat *Lasiurus borealis* Müller, 1776; hoary bat *Lasiurus cinereus* Palisot de Beauvois, 1796; and silver-haired bat *Lasionycteris noctivagans* Le Conte, 1831) and four species of swallow (barn swallow *Hirundo rustica* Linnaeus, 1758; tree swallow *Tachycineta bicolor* Vieillot, 1808; cliff swallow *Petrochelidon pyrrhonota* Vieillot, 1817; and purple martin *Progne subis* Linnaeus, 1758).

## Fatality estimation

We estimated the number of individuals of each species killed ( $\hat{M}$ ) at monitored wind turbines using GenEst (Dalthorp et al. 2018) in R (R Core Team 2020). GenEst is a generalized estimator that corrects the number of carcasses found by the probability of detecting a carcass and adjusts for carcasses falling outside of searched areas. Details of the specifications we used to calculate  $\hat{M}$  in GenEst follows.

## Search schedules

We retained data from wind turbines that were consistently monitored (twice per week for 98% of datasets, 2% searched once per week) during spring (May–Jun) and (or) late summer (mid-Jul through end-Sep). We only estimated fatalities for swallows in late summer because few carcasses were found at turbines during spring ( $n = 19$ , 6% of all swallow carcasses found in our data subset).

GenEst requires users to enter search dates for each turbine to estimate mortality, including searches where no carcasses were found. However, only search dates when carcasses were found were available for a subset of facilities and years ( $n = 28$ , 23% of all facility years). To create full search schedules for these facilities and years, we simulated probable search dates for each turbine using rule-based data mining and hierarchical clustering analysis in the *arules* package in R (Hahsler et al. 2005, 2020). In short, we used a priori association rules to group turbines that were often searched on the same days and assigned search dates to the whole group. We then filled gaps in the search schedules of each group so that searches occurred at intervals specified in the database (e.g., every 3 or 4 days for twice-weekly searches) and throughout the survey period (e.g., 15 Jul–30 Sep for late summer).

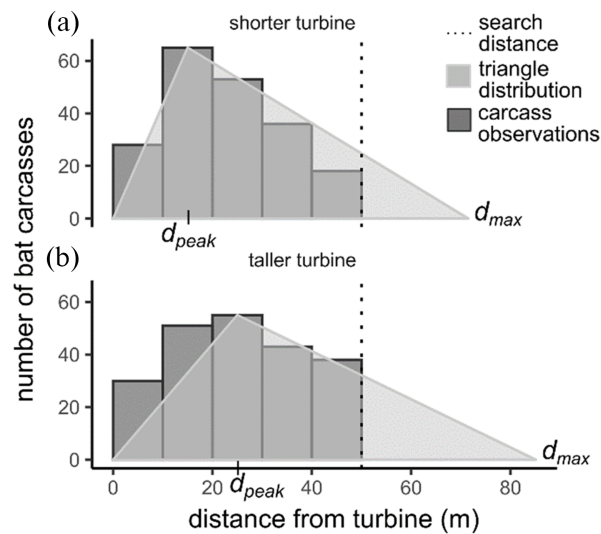
## Detection probability

We used facility- and year-specific searcher efficiency and carcass persistence survey data to produce facility-corrected fatality estimates. Our analysis included facilities and years for which  $\geq 10$  searcher efficiency and  $\geq 10$  carcass persistence surveys were conducted ( $n = 59$  facilities surveyed in at least one year during 2010–2019). Carcasses used in searcher efficiency and carcass persistence surveys were small bats and non-raptor birds. We assumed a constant proportional decline in searcher efficiency ( $k = 0.7$ ) with each subsequent search (Korner-Nievergelt et al. 2012; Simonis et al. 2018), because carcasses missed on the first search may be more difficult to find as a result of decay, cryptic colouration, etc. We could not estimate  $k$  from available data because test carcasses were removed after each searcher efficiency trial. We modelled carcass persistence over time with exponential, Weibull, lognormal, and loglogistic survival models and constant scale and location parameters. Of the four candidate models for each facility and year, we selected the top model with the lowest  $\Delta AIC_c$  value.

## Search area correction

All carcass searches were conducted by walking concentric circles out to a maximum distance of 50 m (as required by government guidelines; Ontario Ministry of Natural Resources and Forestry 2011), regardless of turbine height. Ballistics models estimate that  $\sim 95\%$  of 14 g bat carcasses (and  $\sim 80\%$  of 12 g bird carcasses) fall within 50 m of turbines with MBH = 125 m (i.e., “medium-sized” turbines; Hull & Muir 2010), which were typical sizes of turbines on the landscape at the beginning of our study period (2010, Fig. 1a). However, newer turbines are now much larger ( $\sim 170$  m max turbine blade height in 2019, Fig. 1a), and therefore, a larger proportion of carcasses should land outside of 50 m search areas than for older, shorter turbines.

We used ballistics models (Hull and Muir 2010), field observations of carcasses, and the proportion of the search area covered by searchers to estimate the proportion of carcasses expected to fall in the search area of each turbine ( $dwp$ ). First, for each type of turbine at a facility, we estimated the maximum distance that bat and small bird carcasses should fall from the base of turbines (henceforth, “maximum fall distance”) depending on the blade length (half of rotor diameter) and hub height (i.e., from linear regression equations, Hull and Muir 2010). Next, for the same turbine groups, we counted the number of bat and bird carcasses found every 10 m from the base of turbines out to 50 m. Carcasses tended to peak closer to the turbine base at shorter turbines (Fig. S1) and roughly



**Fig. 2.** Examples of data used to estimate the proportion of carcasses that fall in the search radius ( $p_{\text{search}}$ ) at wind turbines from two facilities in Ontario, Canada: a facility with (a) shorter turbines (MBH = 121 m) and (b) taller turbines (MBH = 150 m). Dark grey bars represent field observations of bat carcasses, and the light grey triangle represents a simple, assumed distribution for carcass fall distances. The triangle distribution is determined by maximum carcass fall distances ( $d_{\text{max}}$ ) estimated by ballistics models and the observed peak in carcass numbers at a given distance from the turbine base ( $d_{\text{peak}}$ ).  $p_{\text{search}}$  is the proportion of the area under the triangle distribution that falls within 50 m from the turbine (i.e., left of the dashed line).

followed a triangular distribution (Fig. 2). We assumed that the triangular distribution extended to the maximum fall distance, with the number of carcasses declining linearly from the peak out to the maximum fall distance. We then estimated the proportion of carcasses that should fall in the 50 m search radius ( $p_{\text{search}}$ ) as the proportion of the total triangular distribution within 50 m of the turbine base (Fig. 2). Carcass distribution data were limited or unavailable for a subset of facilities (23% of facilities for bats and 49% for birds), so we estimated  $p_{\text{search}}$  for the subset using predictive equations from a linear regression between calculated  $p_{\text{search}}$  estimates and taxa type (bird or bat) and MBH (Fig. S2). Finally, we estimated  $dwp$  for each turbine by multiplying  $p_{\text{search}}$  for each facility and turbine type by the average percent of the 50 m search area covered by searchers.

### $\hat{M}$ Estimation

We used search schedule,  $dwp$ , carcass persistence, searcher efficiency, and carcass observation data to calculate  $\hat{M}$  using the `estM` function with 10 000 iterations and the `calcSplits` function to estimate  $\hat{M}$  per turbine and season (spring or late summer). We used the R package *purrr* (Henry and Wickham 2020) to simultaneously estimate  $\hat{M}$  for each facility, year, and species, and we retained median  $\hat{M}$  values for analyses in linear mixed effects models.

### Mixed effects models

We examined the effects of MBH on bat and swallow fatalities using generalized linear mixed effects models (GLMMs) with  $\hat{M}$  per turbine as the response variable. We included species and manufacturer's cut-in speed as predictors, as well as the interactions between species and MBH, and species and cut-in speed. In addition, we included turbine ID and facility ID as random factors to account for repeated measures and for other potential turbine- or facility-specific sources of variation.



We could not include year as a continuous predictor because the number of facilities monitored in some years was low ( $n = 2$  facilities in 2010, 2012, and 2019, respectively). We, therefore, included a random factor of year to estimate the effects of turbine height and cut-in speed while accounting for annual variation in fatalities.

We ran separate models for spring and late summer monitoring seasons (Ontario Ministry of Natural Resources and Forestry 2011). Some turbines in Ontario undergo operational mitigation at night (by increasing turbine cut-in speeds to 5.5 m/s) in late summer to reduce bat fatalities (Ontario Ministry of Natural Resources and Forestry 2011). Therefore, we included mitigation status (turbine undergoing operational mitigation: yes or no) as a predictor in late summer but not in spring models for bats.

We fit GLMMs and zero-inflated GLMMs with Poisson and negative binomial distributions using the R package *glmmTMB* and compared the fits of global models using AIC (Brooks et al. 2017) via the function “AICtab” in the *bbmle* package (Bolker and R Development Core Team 2020). We also used the package *DHARMA* (Hartig 2020) to examine patterns in residuals. We selected the global model with the best fit, indicated by the lowest  $\Delta$ AIC: zero-inflated negative binomial models for bats and swallows in late summer and zero-inflated Poisson models for bats in spring (Table S2). We subsequently used “drop1” function with a chi-square test to drop non-significant terms ( $\alpha = 0.05$ ) from the model. We calculated marginal and conditional  $R^2$  following Nakagawa et al. (2017).

## Results

We examined the effects of MBH and manufacturer’s cut-in speed on fatalities of aerial insectivores (bats and swallows) at 652 turbines at 53 facilities in spring and 811 turbines from 59 facilities in late summer. Turbines consisted of 19 types (make and models), with MBH ranging from 119–186 m ( $139 \pm 13$  m, mean  $\pm$  SD) and manufacturer’s cut-in speeds ranging from 2.0–4.0 m/s ( $3.2 \pm 0.4$  m/s). We were able to include  $>1$  year of monitoring data for 67% of turbines.

### Turbine height

MBH was a significant predictor (Table S3) of the number of bats killed per turbine ( $\hat{M}$ ). This effect differed among species (species by MBH interaction in spring:  $\chi^2 = 14.3$ , d.f. = 4,  $p < 0.01$ ; late summer:  $\chi^2 = 43.3$ , d.f. = 4,  $p < 0.001$ ) though confidence intervals overlapped for fatality estimates at the shortest and tallest turbines (Table 2). The relationship between MBH and bat fatalities was less clear in spring (Fig. 3a) than late summer. In spring, more silver-haired bat, hoary bat, and big brown bat and fewer little brown bat and eastern red bat fatalities were estimated at taller turbines than shorter turbines (Fig. 3a; Table 2). In late summer, more bats were killed at taller turbines than at shorter turbines for all species except little brown bats, which continued to exhibit fewer fatalities at taller turbines (Fig. 3b; Table 2).

Estimated fatalities of swallows in late summer were low, and confidence intervals overlapped for fatality estimates at the shortest and tallest turbines (Fig. 4; Table 2). Fatalities of purple martins and tree swallows were greater at taller turbines than shorter turbines (Fig. 4, species by MBH interaction:  $\chi^2 = 9.4$ , d.f. = 3,  $p = 0.02$ ). Incidence rate ratios from final models (Table S3) showed interspecific differences in the number of fatalities per turbine, with fewer for barn swallow, followed by cliff swallow, purple martin, and tree swallow (Fig. 5).

### Turbine cut-in speed and operational mitigation

Manufacturer’s cut-in speed was not a predictor of bat or swallow fatalities in either season and was dropped from final models (bats spring:  $\chi^2 = 0.007$ ,  $p = 0.93$ ; bats late summer:  $\chi^2 = 0.60$ ,  $p = 0.44$ ;

**Table 2.** Back-transformed model predicted estimates (and 95% confidence intervals) of the number of bats killed per turbine ( $\hat{M}$ ) at the shortest and tallest turbines in our study.

Bats in spring			
	Maximum blade height (m)		
	119 m, shortest turbines	186 m, tallest turbines	
Species	$\hat{M}$ (CI)	$\hat{M}$ (CI)	% change
Eastern red bat	2.5 (1.5–4.2)	0.7 (0.2–2.5)	–72
Big brown bat	2.1 (1.4–3.1)	3.4 (1.6–7.3)	62
Hoary bat	2.0 (1.4–3.0)	2.8 (1.3–6.0)	40
Silver-haired bat	2.1 (1.5–3.1)	3.2 (1.6–6.4)	52
Little brown bat	4.3 (2.6–7.0)	0.5 (0.2–1.9)	–88
<b>Bats in late summer</b>			
Species	$\hat{M}$ (CI)	$\hat{M}$ (CI)	% change
Eastern red bat	3.6 (2.5–5.1)	4.5 (2.5–8.3)	25
Big brown bat	2.7 (1.9–3.9)	4.9 (2.7–9.1)	81
Hoary bat	4.3 (3.0–6.1)	8.6 (4.8–15.3)	100
Silver-haired bat	3.6 (2.5–5.1)	5.9 (3.2–10.8)	64
Little brown bat	2.1 (1.4–3.3)	0.3 (0.1–0.7)	–86
<b>Swallows in late summer</b>			
Species	$\hat{M}$ (CI)	$\hat{M}$ (CI)	% change
Barn swallow	0.1 (0–0.2)	0.1 (0–0.5)	0
Cliff swallow	0.1 (0–0.3)	0.1 (0–0.7)	0
Purple martin	0.1 (0–0.3)	0.4 (0.1–2.1)	300
Tree swallow	0.1 (0–0.3)	1 (0.2–4.6)	900

Note: Estimates were calculated assuming that turbines were not undergoing operational mitigation.

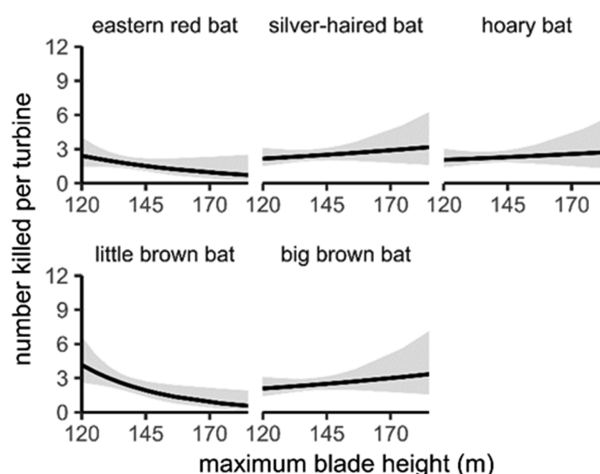
swallows late summer:  $\chi^2 = 0.002$ ,  $p = 0.97$ ). Turbines under operational, nocturnal mitigation killed 33% fewer bats than turbines without cut-in speed adjustments in late summer (back-transformed  $\beta = 0.67$ , 95% CI = 0.56–0.79).

## Discussion

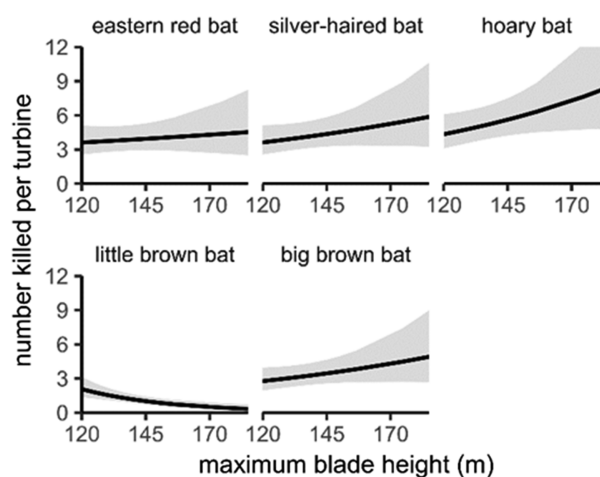
Our results demonstrate an association between wind turbine height and fatality counts of some species of bats and swallows, based on a large dataset of turbines extending 119 to 186 m into the atmosphere. Our findings are consistent with previous studies showing that turbine height explains a portion of bat fatalities at wind energy facilities (Johnson et al. 2003; Fiedler et al. 2007; Barclay et al. 2007; Baerwald and Barclay 2009; Rydell et al. 2010; Georgiakakis et al. 2012; Huso et al. 2021), although other factors, such as the proximity of wind energy facilities to key habitats, migration corridors, and roosts, also play a role (Barclay et al. 2007; Cryan and Brown 2007; Ferreira et al. 2015; Thompson et al. 2017; Davy et al. 2020). Greater uncertainty in the relationship between turbine height and fatalities that we observed in spring could be the result of fewer carcass observations in



## (a) spring



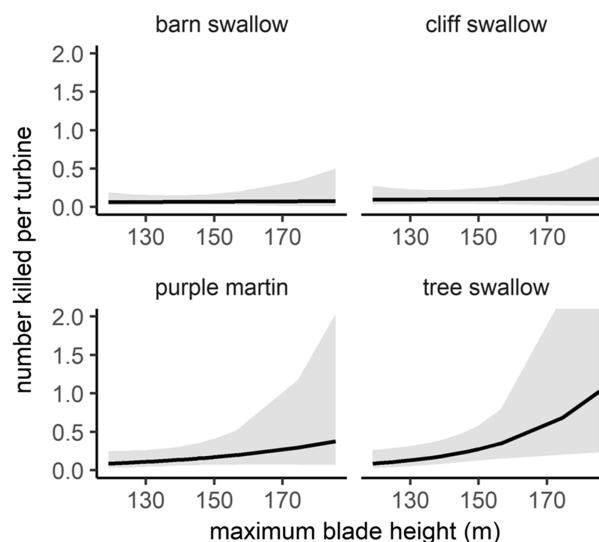
## (b) late summer



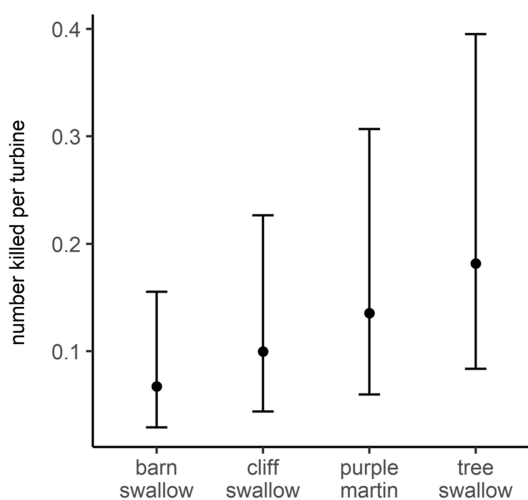
**Fig. 3.** The relationship between maximum blade height of wind turbines and the estimated number of bats killed per turbine ( $\hat{M}$ ) in Ontario, Canada, during (a) spring and (b) late summer. Lines and 95% confidence intervals were predicted from generalized linear mixed effects models.

spring (Birds Canada 2018; Davy et al. 2020) or could indicate seasonal differences in use of the aerosphere.

We estimated fewer bat fatalities at turbines undergoing operational mitigation (i.e., raising cut-in speeds), which agrees with previous findings (Arnett et al. 2011; Davy et al. 2020; Whitby et al. 2021). Despite fewer fatalities at turbines with raised cut-in speeds, cut-in speeds specified by manufacturers were poor predictors of estimated bat and swallow fatalities. Bat activity tends to drop sharply at wind speeds greater than 5–6 m/s (Reynolds 2006; Wellig et al. 2018); therefore, bat activity may not have differed substantially between wind speeds of 2–4 m/s (the range of cut-in speeds specified by manufacturers in our dataset). However, turbine-specific operational cut-in speeds from our study area were not available, so we used manufacturer’s specified cut-in speeds as a proxy for operational cut-in speeds. We recognize that wind farms sometimes allow turbines to “free-wheel”



**Fig. 4.** The relationship between maximum blade height of wind turbines and the estimated number of swallows killed per turbine ( $\hat{M}$ ) in Ontario, Canada. Lines and 95% confidence intervals were predicted from generalized linear mixed effects models.



**Fig. 5.** Back-transformed estimated marginal means (and 95% confidence intervals) of the number of swallows killed per turbine in Ontario, Canada, by species. Means were calculated at the average turbine size in the study area, 139 m maximum blade height.

(i.e., to continue to spin at low wind speeds prior to the onset of electricity generation) and that some facilities may have used different cut-in speeds than that specified by the manufacturer, and both scenarios would obscure the relationship we predicted. Wildlife collide with both free-wheeling turbines and turbines that are generating power (Baerwald et al. 2009; Whitby et al. 2021), so if turbines in our study area were free-wheeling, then manufacturer specified cut-in speed would become a poor proxy for turbine movement at low wind speeds and a poor predictor of fatalities. To address this

potential confounding factor, we recommend that future studies obtain and examine operational cut-in speeds and data on free-wheeling turbines whenever possible.

Population size and flight altitudes of bats are not well understood (Voigt et al. 2018) relative to swallows' population size and flight altitudes (Table S4). Although few swallow carcasses ( $n = 341$ ) were found at turbines compared to bats ( $n = 4850$ ), we speculate that quantitative data about swallow flight altitudes, population size, and fatalities at turbines could provide biologically meaningful interpretations of bat fatality data. Despite their higher relative abundance in Ontario (Partners in Flight 2020; Table S4), barn swallows had the lowest per turbine fatalities compared to other higher-flying swallow species (e.g., tree swallow and purple martin; Dreelin et al. 2018). Additionally, fatalities of purple martins, the least abundant but highest-flying species in Ontario (Table S4), were greater at taller turbines than shorter turbines. We, thus, speculate that differences in fatalities of avian aerial insectivores at turbines are due to differences in flight altitude, not necessarily abundance. Likely, both flight altitude and relative abundance contribute to fatalities, as demonstrated by tree swallow fatality data; tree swallows had the highest number of fatalities per turbine, and they are abundant (Table S4) and also often fly in the rotor swept area of turbines in Ontario (Fig. S3).

If the same relationship between flight altitude and fatalities holds true for bats, then the pattern we observed between turbine height and fatalities of hoary, silver-haired, and big brown bats may reflect these species use of higher altitudes. Collisions of hoary and silver-haired bats with planes support the assertion that they are high-altitude flyers (Peurach 2003; Peurach et al. 2009; McCracken et al. 2021). Perhaps little brown bats and eastern red bats fly at lower altitudes, spending more time below the rotor swept area of newer turbines that tend to be further from the ground than older turbine models, but at this point, this interpretation is speculation that is not yet supported by quantitative data. New, lightweight tracking devices such as GPS units and altimeters can help explicitly test hypotheses about bat flight altitudes (Roeleke et al. 2016; Weller et al. 2016; Conenna et al. 2019; O'Mara et al. 2019, 2021; though see Péron et al. 2020).

Broadly, our results and those of other studies (Johnson et al. 2003; Fiedler et al. 2007; Barclay et al. 2007; Baerwald and Barclay 2009; Rydell et al. 2010; Georgiakakis et al. 2012) indicate a general, ongoing pattern of more bat fatalities at taller turbines. While more bat fatalities at taller turbines is a concern with the increasing size of turbines (Wiser et al. 2011, 2018), fatality rates of bats and birds appear to be relatively consistent per unit of energy produced (GWh) by a turbine (Huso et al. 2021). Therefore, taller turbines that tend to produce more energy may kill more birds and bats than smaller turbines, but the number of animals killed per GWh may remain constant. If we consider how this result can inform efforts to maximize energy production while minimizing wildlife collisions, considering turbine height in the new construction or the repowering of turbines is likely a less effective strategy for reducing wildlife fatalities than efforts such as minimizing turbine blade movement during high-risk periods, for example at night, during migratory periods, and in certain weather conditions for bats (Arnett et al. 2011; Squires et al. 2021) and during the post-breeding and southbound migration period for swallows (Erickson et al. 2014).

Nevertheless, our species-level analyses reveal that turbine height does predict the number of fatalities for some species of bats and birds. Where possible, future analyses of wildlife mortality at wind turbines should explicitly consider interspecific variation in fatalities at wind turbines. Species-specific analyses can inform more accurate risk assessments for each species and may provide insight into aerial habitat selection by each species. Finally, we acknowledge that using mortality as a proxy for exploring flight altitudes is indirect and laden with assumptions. We expect that the increasing availability of altimeters for wildlife tracking will improve interpretation of fatality data at wind turbines and clarify the association between flight altitude and collision risk.

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## Author contributions

AMA, CMD, CBJ, JRZ, EFB: conceptualized and designed the study. AMA: analyzed the data and wrote the first draft of the paper. AMA, CMD, CBJ, JRZ, EFB: contributed significantly to data interpretation, writing, and editing the final version of the manuscript.

## Competing interests

The authors declare there are no competing interests.

## Data availability

Data used for this study are stored in the Wind Energy Bird and Bat Monitoring Database. Reasonable requests to access the database can be made to Birds Canada ([bsc-eoc.org/birdmon/wind/terms.jsp](https://bsc-eoc.org/birdmon/wind/terms.jsp)). Code used in our analyses is available at [osf.io/kd873/?view\\_only=586b03778dea4aa78ec82078696be1f4](https://osf.io/kd873/?view_only=586b03778dea4aa78ec82078696be1f4).

## Supplementary material

The following Supplementary Material is available with the article through the journal website at doi:[10.1139/facets-2022-0105](https://doi.org/10.1139/facets-2022-0105).

Supplementary Material 1

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