Biological Conservation 302 (2025) 110961

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Estimated golden eagle mortality from wind turbines in the western United States

Jay V. Gedir [a,*], Matthew J. Gould [a,1], Brian A. Millsap [a,b,2], Paige E. Howell [b], Guthrie S. Zimmerman [b], Emily R. Bjerre [b], Hillary M. White [b]

[a] *Department of Fish, Wildlife and Conservation Ecology, New Mexico State University, P.O. Box 30003, MSC 4901, Las Cruces, NM 88003, USA*
[b] *U.S. Fish and Wildlife Service, National Raptor Program, Division of Migratory Bird Management, Washington, DC, USA*

ARTICLE INFO

Keywords:
Aquila chrysaetos
Collision risk model
Population-level effects
Prior distribution
Relative abundance
Wind energy development

ABSTRACT

Wind power is increasingly meeting global renewable energy demands; however, more turbines leads to increased bird-turbine collisions, particularly raptors, which can negatively impact populations. We estimated annual turbine mortalities of the federally-protected golden eagle (*Aquila chrysaetos*) in the western United States (2013–2024) with a Bayesian collision risk model (CRM). We used eBird relative abundance data to predict areas where golden eagles are at lower or higher risk of turbine collisions and turbine data from the U.S. Geological Survey U.S. Wind Turbine Database. From 2013 to 2024, estimated turbine hazardous volume in the lower- and higher-risk zone increased by 198 % and 119 %, respectively. We used golden eagle data from wind energy developments in the western United States to create prior-probability distributions for exposure in the lower- ($n = 8$) and higher-risk ($n = 36$) zones and collision probability ($n = 21$). Mean (± SD) risk of golden eagle exposure to turbines (eagle-mins·hr^{-1}·km^{-3}) in the higher-risk zone (1.557 ± 2.265) was >11 times that in the lower-risk zone (0.138 ± 0.162). Annual median [80 % credible interval] golden eagle mortalities predicted from the CRM more than doubled from 110 [28–374] in 2013 to 270 [72–877] in 2024, although estimates had high uncertainty. Anthropogenic mortality is the primary cause of death in adult golden eagles and recent trends indicate their population may be declining. If the current rate of growth of the wind energy industry continues, it could have conservation implications for golden eagle and other raptor populations.

1. Introduction

Global energy demand is increasingly being met from renewable energy sources in an effort to reduce atmospheric carbon dioxide and curb climate change. Wind turbines are one of the leading new sources of power generation worldwide (Dunnett et al., 2020; Lee and Zhao, 2022), and in the United States, the wind energy industry is rapidly growing with the power capacity of wind turbines more than tripling since 2008 (U.S. Department of Energy, 2021). Coincident with this marked increase in the number of turbines on the landscape is increased risk of bird collisions and potential impacts on their populations. Mortality from wind turbine collisions in the United States has been estimated to exceed half a million birds annually, including >80,000 raptors (Smallwood, 2013). Raptors are among the species most strongly affected by wind turbines (primarily through direct mortality) and as such, are particularly vulnerable to expanding wind energy development (Diffendorfer et al., 2021; Madders and Whitfield, 2006; Watson et al., 2018). However, the large-scale population-level impacts of wind energy development remain largely unknown.

Raptor vulnerability to turbine collisions stems from their soaring flight, wide-ranging behavior, generally low displacement by operational wind energy developments, and the limited degree to which they perceive turbine rotors as dangerous (Dahl et al., 2013; Hunt and Watson, 2016; Madders and Whitfield, 2006). Increases in this source of mortality can have negative effects on raptors at a population level, particularly for long-lived, slow-reproducing species (Carrete et al., 2009; Dahl et al., 2012; Watson et al., 2018) and wide-ranging or migratory species (Carrete et al., 2009); even low levels of additional

---

* Corresponding author.
E-mail address: jgedir@nmsu.edu (J.V. Gedir).

[1] Present Address: U.S. Geological Survey, Northern Rocky Mountain Science Center, 2327 University Way, Suite 2, Bozeman, Montana 59,717, USA

[2] Present Address: Department of Fish, Wildlife and Conservation Ecology, New Mexico State University, P.O. Box 30003, MSC 4901, Las Cruces, New Mexico 88003, USA

https://doi.org/10.1016/j.biocon.2024.110961
Received 18 May 2024; Received in revised form 17 December 2024; Accepted 26 December 2024

wind turbine mortality may be significant for long-lived species with low productivity and slow maturation rates (Drewitt and Langston, 2006; Whitfield et al., 2004). The golden eagle (*Aquila chrysaetos*), among the largest of the raptors, is particularly vulnerable to mortality from wind turbines and is a legally protected species in the United States (The Migratory Bird Treaty Act, 16 USC §§703–712; The Bald and Golden Eagle Protection Act, 16 U.S.C. 668-668c). Golden eagles are predatory making them more susceptible to wind turbine collisions than scavengers, because while fixed on prey during hunting, they may not detect turbine rotors (Madders and Whitfield, 2006). A significant amount of golden eagle habitat in the United States overlaps with areas where wind resources are amenable to wind energy development, potentially jeopardizing the viability of local and migratory populations (National Renewable Energy Laboratory, 2017; Tack et al., 2020). Indeed, high levels of golden eagle mortality have been observed at wind energy developments in the western United States (Pagel et al., 2013; Smallwood and Thelander, 2008). The golden eagle population in the western contiguous United States has remained stable (lambda $\approx$ 1) for several decades (2016 population size $\approx$ 32,000 individuals); however, a recent study suggests that anthropogenic sources account for a high proportion (74 %) of adult mortality and that this take is likely unsustainable (Millsap et al., 2013, 2022). For a species with high adult survival and low reproductive rates, like the golden eagle, small increases in mortality rates can lead to population declines; this has been reported in other raptor species (e.g., Eurasian griffon [*Gyps fulvus*], Lekuona and Ursua, 2007; Egyptian vulture [*Neophron percnopterus*], Carrete et al., 2009; white-tailed hawk [*Geranoaetus albicaudatus*], Ledec et al., 2011). Here, we quantified the potential wind turbine mortalities of golden eagles in the western contiguous United States west of the 100th meridian (hereafter western U.S.).

Collision risk models (CRMs) are a useful tool for assessing the potential impacts of wind turbines on birds (see Masden and Cook, 2016) and models have been developed specifically to quantify the risk of raptors colliding with turbines (e.g., Eichhorn et al., 2012; Holmstrom et al., 2011; Murgatroyd et al., 2020). A CRM generally requires site-specific data, including an estimate of the number of birds within a turbine's hazardous space (i.e., to estimate the likely number of collision events) and a calculation of the probability of a collision occurring. However, there is often a lack of site-specific data to inform the CRM, particularly at larger spatial scales. Additionally, many CRMs do not provide estimates of uncertainty around the predicted number of collision events. Knowledge of uncertainty allows managers to make decisions with a more complete understanding of possible risks involved. New et al. (2015) developed a CRM that is structured in a Bayesian framework, whereby site-specific data in the model can be substituted or supplemented with prior-probability distributions created from data collected at wind energy developments. Furthermore, this model was developed using golden eagle data and the Bayesian approach is designed to readily accommodate sources of uncertainty in mortality estimates. Therefore, this serves as an ideal modeling approach for our objective of estimating golden eagle mortalities and quantifying uncertainty at the scale of the western U.S.

An important component of CRMs is exposure of birds to turbines, which is largely correlated with bird density. The western U.S. is the core of the golden eagle range in North America, where >80 % of their North American population occurs (Katzner et al., 2020; Millsap et al., 2013). Within this large geographic area there is substantial heterogeneity in landscape composition and configuration, undoubtedly leading to spatial variation in the distribution of golden eagle relative abundance (Fink et al., 2023). Relative abundance estimates of golden eagles have been mapped using data from a variety of sources, including GPS tracking data (Brown et al., 2017; McCabe et al., 2021), nest sites (Dunk et al., 2019), and data from widespread targeted surveys (Nielson et al., 2016). However, these datasets do not provide adequate temporal (i.e., annual) or spatial coverage to inform exposure of eagles to turbines on a large scale. An alternative is to use relative abundance predictions based on the year-round, semi-structured, and spatially-unrestricted citizen science sampling program eBird (Kelling et al., 2019; Sullivan et al., 2009) to stratify the landscape based on some of the spatial heterogeneity in eagle abundance. Relative abundance predictions generated from eBird checklist data have already been shown to perform well in identifying areas of high importance for eagles at a continental scale (Ruiz-Gutierrez et al., 2021; Stillman et al., 2023).

We took a similar approach as Ruiz-Gutierrez et al. (2021) using eBird relative abundance predictions as the basis for delineating areas of relatively low and high golden eagle wind turbine collision risk in the western U.S. We used these exposure-risk zones and a CRM to predict potential annual golden eagle mortalities from wind turbines in the western U.S. from 2013 to 2024. Our results fill a critical gap in our understanding of the potential effects of wind energy developments on golden eagle population viability at a large scale and in the core of their range. Predictions of golden eagle mortalities, which incorporate uncertainty, will help managers make more informed decisions to balance competing objectives of alternative energy development and eagle population viability.

2. Materials and methods

2.1. Turbine data

The initial release of the U.S. Geological Survey U.S. Wind Turbine Database was in July 2013 and includes the locations and specifications of onshore wind turbines throughout the United States (Diffendorfer et al., 2014). Each release of the database adds newly-installed turbines and undergoes rigorous quality control, including visually verifying turbine locations using high-resolution satellite imagery, removing decommissioned turbines (starting in 2018), removing duplicate turbines, and reclassifying structures incorrectly classified as turbines or residential-scale turbines (Rand et al., 2020). We compiled wind turbine data from years that were available for download (2013, 2014, and 2018–2024; Diffendorfer et al., 2014; Hoen et al., 2018) and used these data to calculate the hazardous volume of each turbine based on individual turbine specifications and annual turbine-specific daylight hours based on turbine locations (see Eq. 2 below). We removed turbines located in Alaska and Hawaii, and in 2013 and 2014, we removed decommissioned turbines, turbine locations that were described as "pad only" or "no turbine", and turbines that were identified as having "no blades" or were non-operational. For turbines where rotor diameters were not reported, but turbine manufacturer and model or capacity were reported, we obtained rotor diameters from "Windturbines database" (https://en.wind-turbine-models.com/turbines). For other turbines where rotor diameters were not reported, in each year's dataset we calculated mean rotor diameter by turbine construction year, identified the year when there was a marked increase in rotor diameter, grouped turbines built in years before and after this cutoff, and imputed the mean rotor diameter by time period. Thus, our assumptions of turbine size, when not provided, are informed by the most similar turbines in terms of construction year and size. We assigned the mean of all reported rotor diameters to those turbines where rotor diameter was not reported and construction year was unknown. We then estimated annual golden eagle mortalities from the total hazardous volume summed from individual turbines.

2.2. Relative abundance

We developed maps of golden eagle exposure to collision risk based on seasonal relative abundance estimates generated by The Cornell University Lab of Ornithology using eBird data (eBird abundance, Version 2021; Fink et al., 2022). The eBird relative abundance represents the average number of golden eagles expected to be seen by an observer during 1 h at the optimal time of day for detecting golden eagles, and who travels ≤1 km during the observation session (for more details on eBird data collection methods, see https://ebird.org/spain/science/status-and-trends/faq#mean-relative-abundance). We used predicted relative abundance rasters at a 2.96- x 2.96-km grid-cell resolution for each of four periods: non-breeding (7 December–7 February); pre-breeding migration (8 February–6 June); breeding (7 June–30 August); and post-breeding migration (31 August–6 December). Using multiple seasons of data allowed us to identify locations with higher predicted relative abundance in any single season and ensures that any important periods of golden eagle use across the annual cycle are captured. We assumed areas of lower and higher golden eagle relative abundance correspond with areas of lower and higher risk of wind turbine mortality, respectively, because of the presumption that there is a predictable relationship between exposure risk (which is related to abundance) and collision probability. Therefore, we mapped the threshold between the lower- and higher-risk zones at the 50th quantile, and thus, the lower 50th quantile corresponds to a lower relative risk of golden eagle mortality and the upper 50th quantile corresponds to a higher relative mortality risk. We excluded cell values of zero when defining the thresholds delineating lower versus higher risk for each seasonal raster. Our assumption was that including zero-value cells in creating those seasonal thresholds would include areas where golden eagles were not at risk for colliding with turbines because they were not predicted to occur in those cells during a given season. We then reclassified grid cells in each seasonal raster as 0 (lower risk; below the 50th quantile) and 1 (higher risk; above the 50th quantile). Finally, we combined seasonal rasters into a single raster, defining a cell as lower risk if all overlapping grid cells were 0 in every season.

2.3. *Collision risk model*

We used a CRM that was developed for estimating avian fatalities at wind energy developments (New et al., 2015). The CRM assumes there is a predictable relationship between pre-construction eagle use (eagle exposure) and subsequent fatalities resulting from collisions with wind turbines, and this relationship is dependent on the hazardous space of the turbines, the time eagles spend in this hazardous space, and the probability of an eagle colliding with a turbine while within the hazardous space (collision probability) (Eq. 1).

$$F = \lambda C \varepsilon \tag{1}$$

Here, F is eagle fatality rate (eagles·year^{-1}), λ is eagle exposure which accounts for the time eagles spend in a turbine's three-dimensional hazardous space as a function of survey effort (eagle-minutes·hour^{-1}·km^{-3}), C is collision probability (eagle collisions·eagle-minute^{-1}), and ε is the expansion factor, which is the combined temporal and spatial exposure risk across all turbines (Eq. 2).

$$\varepsilon = \tau n h \pi r^2 \tag{2}$$

Here, τ is turbine location-specific annual daylight hours (i.e., golden eagles exhibit diurnal behavior), n is the number of turbines, h is turbine hazardous space height (a constant set at 200 m to correlate with the designated height for eagle-use surveys), and r is turbine rotor radius (m; rotor diameter/2; $nh\pi r^2$ = total hazardous volume). We calculated turbine-specific annual daylight hours of golden eagle exposure from daily daylight hours based on the geographic coordinates of the turbine. In addition, we used this location information to adjust the number of hours of exposure to reflect the migratory behavior of golden eagles; for any months when golden eagles are expected to be absent, daylight hours in the model were set to zero. In our models, we treated each turbine independently and turbines were assumed to be operating during all daylight hours.

The CRM parameters are modeled in a Bayesian framework where uncertainty surrounding eagle exposure and collision probability are defined by golden eagle-specific prior-probability distributions (priors) for each parameter. The CRM incorporates priors for eagle exposure and collision probability, and the expansion factor (the only site-specific data included in the CRM for our analyses). We ran the CRM to estimate annual golden eagle mortalities across all onshore wind turbines in the western U.S.

2.4. *Creating prior distributions*

The exposure and collision probability priors that we used in the CRM were created from data collected at wind energy developments throughout the western U.S. We created the exposure priors from golden eagle-use data collected at proposed sites of wind energy developments prior to the construction of turbines. Exposure risk is generally estimated by conducting ≥60-min surveys counting the number of minutes golden eagles are observed flying at a height of ≤200 m within approximately 800-m radius survey plots distributed within the area of the proposed wind energy development. We excluded data from plots with a radius < 800 m and from sites where surveys were not conducted across all seasons. We developed risk zone-specific exposure priors for golden eagles in the western U.S. based on which risk zone the wind energy developments that were used for creating the priors were located (i.e., wind energy development centroids buffered by 3.2 km). If a buffered centroid intersected with a higher-risk zone raster cell, we considered it to be within the higher-exposure risk zone. The collision probability prior is created from golden eagle mortality monitoring data collected once the turbines are operational (see New et al., 2021). The collision probability prior we used in the CRM was taken from New et al. (2021) and was estimated using data from 21 wind energy developments that were not risk-zone specific (Table 1). The mean and variance of the exposure and collision probability priors were calculated from mixture distributions based on these data and were assumed to come from gamma and beta distributions, respectively (New et al., 2015). We conducted all analyses in R 4.1.2 (R Core Team, 2021).

3. Results

3.1. *Turbine data*

As of July 2013, the U.S. Wind Turbine Database contained 25,704 operational onshore wind turbines in the western U.S. with an estimated hazardous volume of 17.4 km^3 (Fig. 1). As of November 2024, there were 32,579 operational onshore turbines in the database in the western U.S. and the hazardous volume was estimated at 47.0 km^3 (Figs. 1 & 2). This represents an increase of 26 % in the total number of turbines and 171 % in hazardous volume from 2013 to 2024. Among the operational turbines where turbine specifications were reported, mean (± SD) turbine sizes increased substantially from 2013 to 2024, with rotor diameters increasing 44 % (2013: 66.0 ± 27.6 m, n = 20,661; 2024: 94.9 ± 25.9 m, n = 29,912), hub heights increasing 22 % (2013: 65.3 ± 21.9 m, n = 20,115; 2024: 79.4 ± 14.2 m, n = 29,527), and total turbine heights increasing 29 % (2013: 98.6 ± 35.0 m, n = 20,135; 2024: 127.2 ± 25.0 m, n = 29,528).

In 2013, turbines in the western U.S. were distributed fairly evenly between the lower- and higher-risk zones (lower 53 %; higher 47 %), and by 2024, about two-thirds (68 %) of the turbines were in the lower-risk zone (Fig. 1). From 2013 to 2024, the number of turbines in the lower-risk zone increased by 63 %, whereas in the higher-risk zone there was a decrease of 14 % (Fig. 1). In contrast, over the same period, estimated hazardous volume of turbines in the lower-risk zone increased by 198 %, and despite the number of turbines in the higher-risk zone decreasing, there was an increase in estimated hazardous volume of 119 % (Fig. 1).

3.2. *Prior distributions*

We used golden eagle-use survey data from 44 wind energy developments in the western U.S. to create the exposure priors for the

Table 1
Golden eagle exposure (eagle-mins·hr^{-1}·km^{-3}; by risk zone) and collision probability (eagle collisions·eagle-min^{-1}) prior distribution parameters used in the collision risk model to estimate golden eagle mortalities in the western U.S.

	Eagle exposure priors				Collision probability priors			
Risk zone			Gamma parameters				Beta parameters	
	n[a]	Mean ± SD	*α*	*β*	*n*[a]	Mean ± SD	*α*	*β*
Lower	8	0.138 ± 0.162	0.7268	5.2507	21	0.006 ± 0.005	1.29	227.6
Higher	36	1.557 ± 2.265	0.4725	0.3034				

[a] number of wind energy developments from which data was used to create prior.

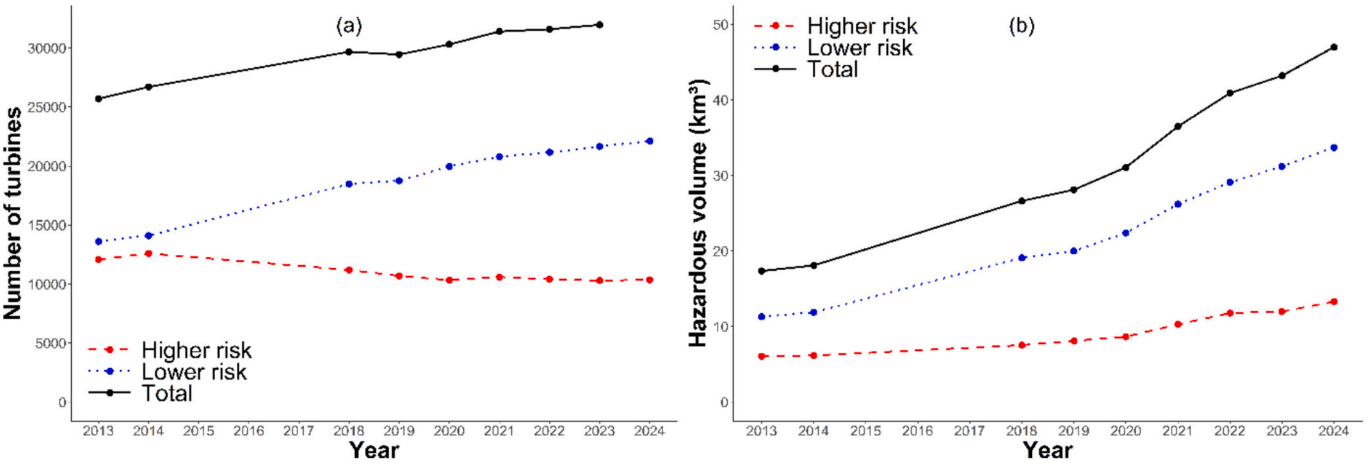

Fig. 1. Total (a) number and (b) hazardous volume (km^3) of wind turbines in the western U.S. (2013–2024) by risk zone and year. Points represent years where data were available.

CRM; 8 developments were in the lower-risk zone and 36 in the higher-risk zone (Fig. 3; Table 1; Supplementary Materials Fig. S1 and Table S1). The exposure prior used to estimate golden eagle mortalities in the lower-risk zone was Gamma(0.7268, 5.2507) with mean (± SD) 0.138 ± 0.162 eagle-minutes·hour^{-1}·km^{-3} (Table 1; Supplementary Materials Fig. S1a). The exposure prior used in the higher-risk zone was Gamma(0.4725, 0.3034) with mean 1.557 ± 2.265 eagle-minutes·hour^{-1}·km^{-3} (Table 1; Supplementary Materials Fig. S1b). The collision probability prior (i.e., not risk-zone specific) used in the CRM was Beta(1.29, 227.6) with mean 0.006 ± 0.005 eagle collisions·eagle-minute^{-1} (Table 1).

3.3. Mortality

Results from the CRMs predict that in 2013 the median number of golden eagle mortalities from wind turbines in the western U.S. was 110 [80 % credible interval: 28–374], and by 2024 estimates more than doubled to 270 [72–877] (Fig. 4). The exposure priors for both the lower- and higher-risk zones are strongly right-skewed (Supplementary Materials Fig. S1) resulting in the estimated golden eagle mortalities having high uncertainty. The annual increase of 8.8 % in predicted median golden eagle mortalities from 2013 to 2020 increased by 49 % annually from 2020 to 2024 (Fig. 4). Modeled estimates of golden eagle mortalities in the higher-risk zone increased at a higher rate and had greater uncertainty than those in the lower-risk zone (Fig. 4). From 2013 to 2024, predicted median golden eagle mortalities in the lower-risk zone increased 197 % (2013: 15 [3–57]; 2024: 44 [8–168]), whereas in the higher-risk zone the increase was 118 % (2013: 62 [7–318]; 2024: 136 [14–695]) (Fig. 4; Supplementary Materials Fig. S2). When increases in predicted mortalities were examined annually, the lower-risk zone showed similar increases from 2013 to 2020 (13.9 %) and 2020–2024 (12.6 %); however, annual increases in estimated mortalities in the higher-risk zone were more than double from 2020 to 2024 (13.4 %) than from 2013 to 2020 (6.0 %) (Fig. 4).

4. Discussion

In this study we examined potential golden eagle mortalities from collisions with wind turbines in the western U.S. from 2013 to 2024. The data indicate that turbine infrastructure (i.e., the total hazardous volume of turbines) has increased significantly over this period. Our CRMs predicted that the annual number of golden eagle mortalities from wind turbines may have more than doubled over eleven years, which could have significant population-level effects. The golden eagle population in the western U.S. has remained stable for several decades (1968–2016; Millsap et al., 2013, 2022), a period which includes the initial influx of wind turbines in the US (c. 1998) followed by slow growth until 2006, but also a period of significant increase in wind power from 2007 to 2016 (Wiser and Bolinger, 2019). It could appear the golden eagle population is resilient to increases in mortality attributed to growth in the wind energy industry; however, recent demographic modeling suggests the population may have been declining at a low rate since 2016, and that additional mortality, unless mitigated for, is not sustainable (Millsap et al., 2022). Indeed, a map based on eBird data produced by the Cornell Lab of Ornithology displaying cumulative trends in estimated golden eagle relative abundance from 2012 to 2022 (https://science.ebird.org/en/status-and-trends/species/goleag/trends-map; Fink et al., 2023) indicates predicted golden eagle relative abundance has declined over much of its western U.S. range and areas where declines occurred align with areas where the most turbines have been constructed (Fig. 2). Furthermore, the increase in turbine hazardous volume in the western U.S. from 2020 to 2024 has been more significant than in the preceding years, and if this trend continues, it would be expected these declines would continue or increase in future.

The primary causes of adult golden eagle mortality in the U.S. come from anthropogenic sources such as shootings, collisions, electrocutions,

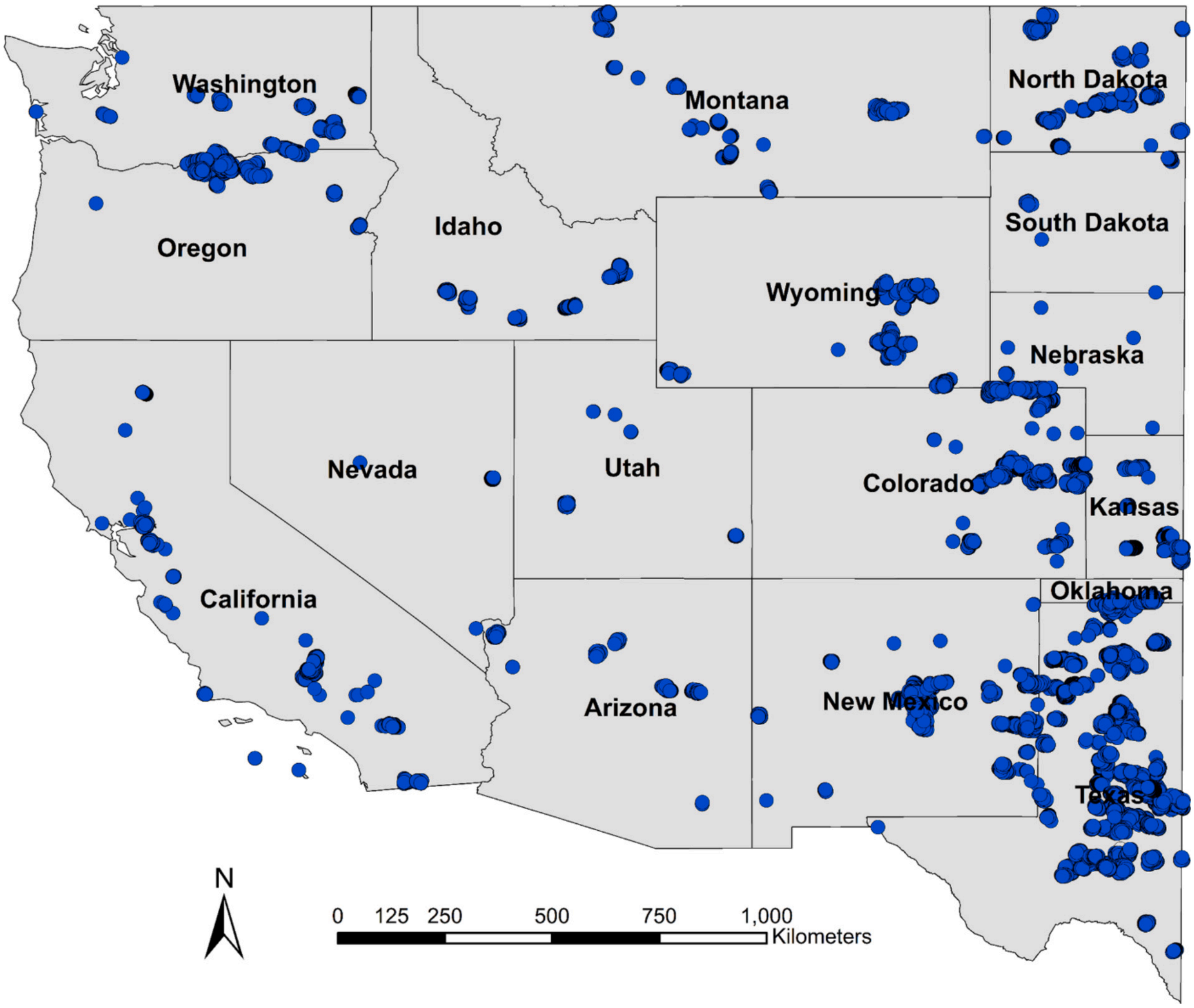

Fig. 2. Location of wind turbines (blue dots; $n = 32{,}579$) in the western U.S. as of November 2024 (Hoen et al., 2018, 2024 update v7_2). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

or poisoning (Hunt, 2002; Russell and Franson, 2014; Hunt et al., 2017; Millsap et al., 2022). Moreover, in areas where wind energy development has become widespread, collisions with wind turbines can account for a significant proportion of anthropogenic mortality in golden eagles (Hunt, 2002; Hunt et al., 2017). Human-caused mortality is generally considered to be additive to natural mortality in golden eagles (Katzner et al., 2017) and other raptor species (Watson et al., 2018), and so has the potential to have a negative impact at a population level. For example, a study of golden eagle survival across the western U.S. (1997–2013) showed that if anthropogenic mortality is assumed to be additive, removing it would increase survival 6–12 % depending on the age class (USFWS, 2016). Furthermore, a subsequent study of the same population extending to 2016 suggested that if anthropogenic mortality was assumed to be entirely additive, it could lead to a decline in the golden eagle population in the western U.S. (Millsap et al., 2022). Other research examining golden eagle survival in California found that when turbine mortality was censored, subadult and floater survival increased from 0.79 to 0.90, and this change was enough to shift the population trajectory from declining to growing (Hunt, 2002). However, in populations of predators that have long lifespans, like the golden eagle, when anthropogenic mortality exceeds natural mortality, the population may demonstrate a reduced density-dependent response resulting in increases in other demographic rates (e.g., Gantchoff et al., 2020). Indeed, Accipitriformes will vary their age at first reproduction in response to competing selective pressures (Millsap et al., 2019). Spatial demographic simulation modeling of a golden eagle population exposed to increasing renewable energy development demonstrated that even small increases in mortality associated with renewable energy infrastructure (e.g., turbine and vehicle collisions, power pole electrocutions) could have negative consequences for population dynamics (Wiens et al., 2017). Therefore, the recent accelerated growth and expected future growth of wind power in the western U.S. could have a significant negative impact on the golden eagle population.

We compared our estimates of mortalities from the CRMs to those estimated by Millsap et al. (2022) from dead recoveries of transmittered golden eagles in the western U.S. The number of golden eagles killed annually from collisions (i.e., including automobile, wind turbine, power line, train, and undetermined collisions) estimated from recovered birds with transmitters deployed between 1997 and 2016 was 611 (median [95 % credible interval]: first year birds 51 [11–143]; after first year birds 560 [322–877]; Millsap et al., 2022). Cause of death could be determined for 126 of the 611 recovered golden eagles of which 16 (12.7 %) were due to collisions, and of those collision mortalities, at least two (12.5 %) were from wind turbines (Millsap et al., 2022). Extrapolating from mortalities where cause was confirmed to total mortalities (i.e., 12.5 % of 611 total collision mortalities) predicts 76 golden eagles killed by wind turbines annually. This estimate includes data up to 2016, and we do not have an estimate of mortalities for 2016; however, our median [80 % credible intervals] estimates of annual golden eagle turbine collisions for 2014 and 2018 are 113 [29–385] and

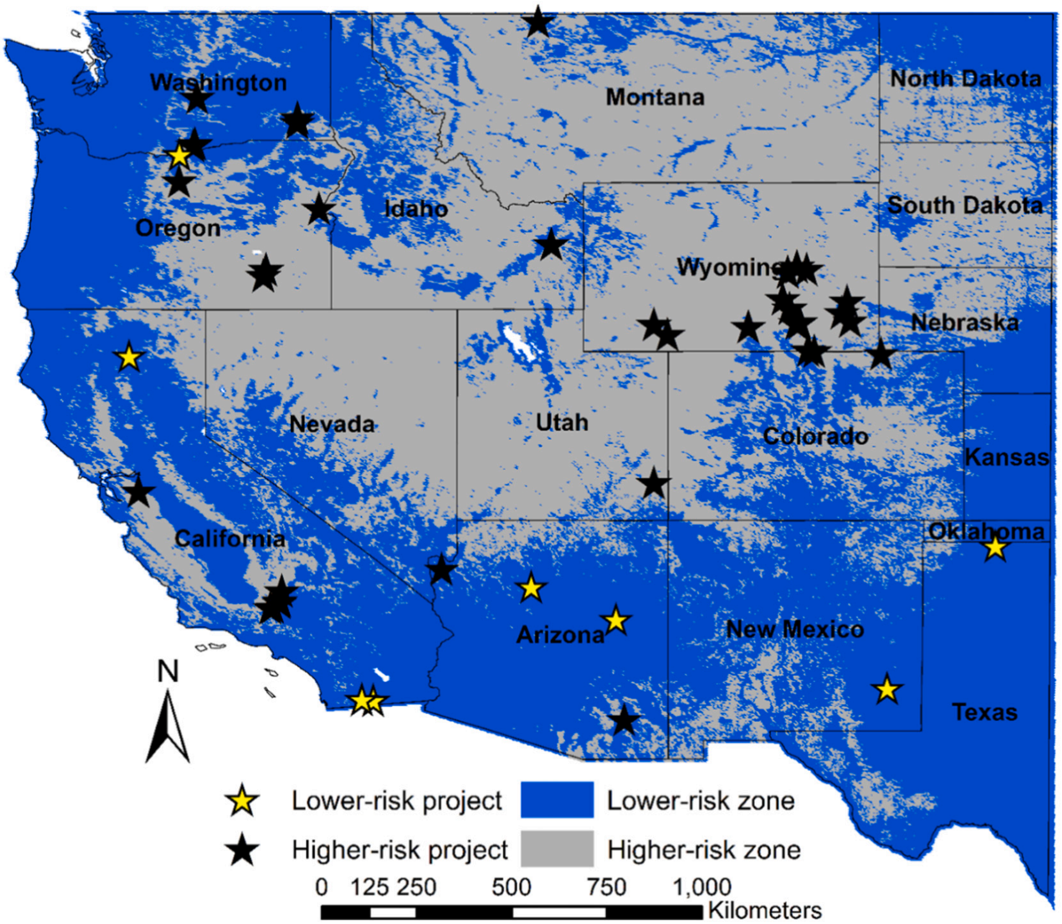

Fig. 3. Golden eagle lower- and higher-risk zones in the western U.S. based on eBird predicted relative abundance at the 50th quantile. White areas indicate where eBird data were inadequate to produce relative abundance predictions. Stars represent locations of wind energy developments that were included in the creation of lower- and higher-risk golden eagle exposure priors.

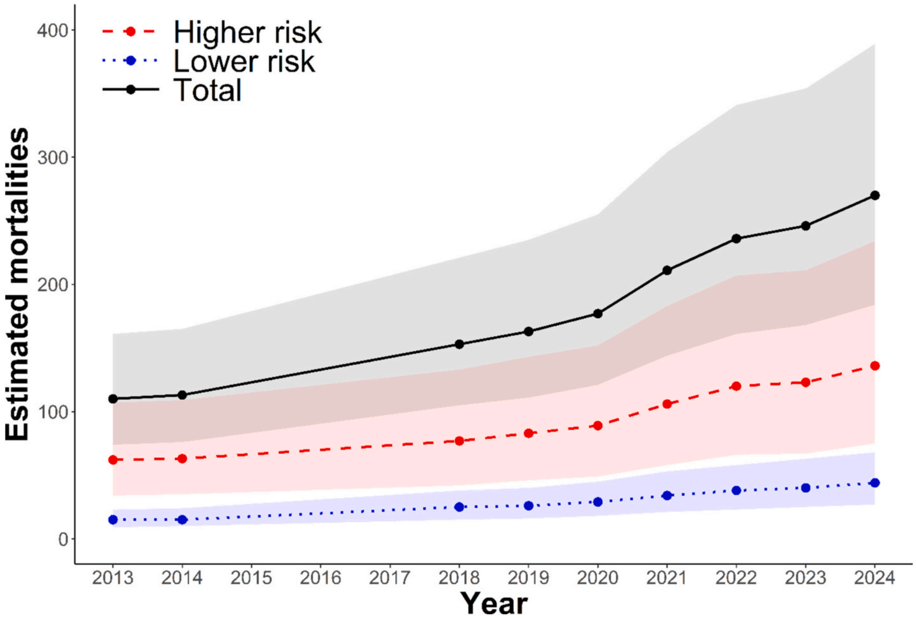

Fig. 4. Estimated median golden eagle mortalities from wind turbines in the western U.S. (2013–2024) by risk zone and year. Points represent years where data were available to estimate mortalities. We present shading of estimates from the 40th to 60th quantiles to enable a clearer visualization of mortality trends and difference in uncertainty between the lower- and higher-risk zone. These are median estimates from the posterior density distributions (see Supplementary Materials Fig. S2), so total mortalities do not equal the sum of lower- and higher-risk mortalities.

153 [41–497], respectively. While based on limited data, this extrapolation suggests results from our analysis are a reasonable estimation of golden eagle mortalities from wind turbines.

Another consideration when determining the impact of turbine mortality on golden eagle population demography, is how age classes may be differentially affected. First year birds in the western U.S. golden eagle population had 75 % mortality from natural causes, whereas 74 % of mortalities in after-first year individuals were human-caused (Millsap et al., 2022). More specifically, golden eagle mortality from wind turbines in a population in California was much higher in subadults and adults (i.e., after-first year birds) than in juveniles (i.e., first year birds), which is likely because juveniles are provisioned by their parents and do not hunt during part of their first year, and turbine collisions often occur when individuals are distracted while pursuing prey (Hunt, 2002; Hunt et al., 2017). Also, among adults, floaters (i.e., non-breeding individuals) had much higher turbine mortality than individuals occupying breeding territories, likely due to the tendency for breeders to stay within or near their territories, and as such, having smaller home ranges (Hunt, 2002; Hunt et al., 2017). Therefore, although our estimates of mortalities are not age-specific, if these patterns are accurate, most mortalities that we predicted are likely to be of subadults or adults (particularly floaters). Elasticity analysis of vital rates of golden eagles in the western U.S. identified adult survival as the most important demographic parameter relative to population growth (Millsap et al., 2022), and consequently, this higher turbine mortality in adults could have detrimental effects at a population level.

There are several advantages to the structure of the CRM used in this study to estimate golden eagle mortalities and incorporating turbine specification and location data from the U.S. Wind Turbine Database. Although the CRM was designed to be used to estimate mortalities at individual wind energy developments incorporating site-specific eagle-use and fatality data, this approach is not realistic at the spatial scale of the western U.S. Hence, the Bayesian framework allowed the models to be informed with priors for exposure risk that were developed for areas identified as lower or higher risk of golden eagle turbine collisions specific to the western U.S., which yielded more relevant estimates of mortalities at this large spatial scale than if nationwide priors had been used that were not risk-zone specific. Furthermore, the data used to create the prior distributions capturing the variability among wind energy developments facilitated estimates of exposure risk and collision probability that can be applied to any region or at any spatial scale, and these priors can be updated in an adaptive management framework as more data become available. Additionally, the Bayesian approach allows estimation of uncertainty in mortalities which provides insights into the potential risks involved when managers are making decisions, essential for guiding species conservation and management strategies. Although our estimates of mortalities generally had high uncertainty, this uncertainty should decrease as more data are included. Secondly, relating hazardous volume to exposure risk and collision probability instead of turbine numbers, allows accounting for the repowering of turbines. For example, considering the growth of the wind energy industry from 2013 to 2024, it is surprising to see a decrease of 14 % in the number of turbines in the higher-risk zone despite a 119 % increase in hazardous volume; this is primarily due to a combination of the greater size of newly-constructed turbines and the repowering of old turbines with larger turbines. Advances in turbine technology allow for partial repowering with larger rotors without the need to replace towers (Brøndsted et al., 2023), which makes for an efficient, less expensive way to increase power output, but further increases hazardous volume and golden eagle exposure to collision risk. Thirdly, using eBird relative abundance predictions generated from eBird checklist data from important periods of the golden eagle annual cycle allowed us to identify areas where golden eagles are at lower or higher risk of turbine exposure in any single season. This is essential to capture any period of higher use for migratory species like the golden eagle that have important spatio-temporal population dynamics (Johnston et al., 2020), and provides the necessary temporal and spatial coverage to suitably inform golden eagle exposure to turbines at the large scale of the western U.S. Our estimates of exposure risk calculated from data collected at wind energy developments in the lower- and higher-risk zones (Fig. 3) suggested that golden eagle exposure to turbines in the higher-risk zone was potentially more than eleven times that of the lower-risk zone (Table 1), and as such, estimating risk zone-specific golden eagle mortalities provides relevant overall estimates of mortalities in the western U.S. This suggests that building a fraction of the number of turbines in the higher-risk zone compared to the lower-risk zone could result in a similar number of golden eagle mortalities (see Figs. 1 & 4), and so highlights the importance of careful consideration when proposing locations for construction of new wind energy developments.

Following many years of apparent stability for the golden eagle population in the western U.S., recent trends indicate it may now be declining (Millsap et al., 2013, 2022). Given that the primary cause of mortality in adult golden eagles is from anthropogenic sources (including mortalities from wind turbine collisions), the rapidly growing wind energy industry is cause for concern, which highlights the conflict that exists between the desire, perhaps necessity, to pursue channels for renewable energy production and the consequences on wildlife conservation. This increasing wind energy development is occurring globally (Dunnett et al., 2020; Lee and Zhao, 2022) and negative population-level effects are becoming apparent in many raptor species around the world (Watson et al., 2018). The potential impact of increasing wind power on threatened and protected species emphasizes the importance of monitoring construction of wind energy developments and quantifying subsequent turbine mortalities to better conserve and manage these species that are at risk of decline. The accelerated increase in wind power from 2020 to 2024 suggests that wind will remain an important source of renewable energy in future and will continue its significant growth. When projecting impacts on raptor populations into the future, it is essential to also consider how climate change could affect species ranges and distributions, as well as wind turbine buildout scenarios in response to potential shifts in areas that are amenable to effective wind power. Further research should assess the conservation implications for golden eagle and other raptor populations in response to a rapidly growing wind energy industry and whether it is plausible for other sources of mortality to be sufficiently reduced to mitigate potential increases in wind turbine mortality.

CRediT authorship contribution statement

Jay V. Gedir: Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Matthew J. Gould:** Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Brian A. Millsap:** Writing – review & editing, Methodology, Formal analysis, Conceptualization. **Paige E. Howell:** Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Guthrie S. Zimmerman:** Writing – review & editing, Methodology, Formal analysis, Conceptualization. **Emily R. Bjerre:** Writing – review & editing, Methodology, Formal analysis, Data curation, Conceptualization. **Hillary M. White:** Writing – review & editing, Methodology, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

Financial and logistical support was provided by the USFWS (Grant G23AC00258). We thank L. New and M. Otto for their assistance

developing the mixture distributions to create the priors. The findings and conclusions in this article are those of the authors and do not necessarily represent the views of the USFWS.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2024.110961.

Data availability

The data are publically available

References

Brøndsted, P., Nijssen, R., Goutianos, S. (Eds.), 2023. Advances in Wind Turbine Blade Design and Materials, 2nd edition. Woodhead Publishing, Kidlington, UK.

Brown, J.L., Bedrosian, B., Bell, D.A., Braham, M.A., Cooper, J., Crandall, R.H., DiDonato, J., Domenech, R., Duerr, A.E., Katzner, T.E., Lanzone, M.J., LaPlante, D. W., McIntyre, C.L., Miller, T.A., Murphy, R.K., Shreading, A., Slater, S.J., Smith, J.P., Smith, B.W.,…Woodbridge, B., 2017. Patterns of spatial distribution of golden eagles across North America: how do they fit into existing landscape-scale mapping systems? J. Raptor Res. 51, 197–215.

Carrete, M., Sánchez-Zapata, J.A., Benítez, J.R., Lobón, M., Donázar, J.A., 2009. Large scale risk-assessment of wind-farms on population viability of a globally endangered long-lived raptor. Biol. Conserv. 142, 2954–2961.

Dahl, E.L., Bevanger, K., Nygård, T., Røskaft, E., Stokke, B.G., 2012. Reduced breeding success in white-tailed eagles at Smøla windfarm, western Norway, is caused by mortality and displacement. Biol. Conserv. 145, 79–85.

Dahl, E.L., May, R., Hoel, P.L., Bevanger, K., Pedersen, H.C., Røskaft, E., Stokke, B.G., 2013. White-tailed eagles (*Haliaeetus albicilla*) at the Smøla wind-power plant, Central Norway, lack behavioral flight responses to wind turbines. Wildl. Soc. Bull. 37, 66–74.

Diffendorfer, J.E., Compton, R., Kramer, L.A., Ancona, Z.H., Norton, D., 2014. Onshore industrial wind turbine locations for the U.S. through July 2013. In: U.S. Geological Survey data release. https://doi.org/10.5066/F7X06561.

Diffendorfer, J.E., Stanton, J.C., Beston, J.A., Thogmartin, W.E., Loss, S.R., Katzner, T.E., Johnson, D.H., Erickson, R.A., Merrill, M.D., Corum, M.D., 2021. Demographic and potential biological removal models identify raptor species sensitive to current and future wind energy. Ecosphere 12. https://onlinelibrary.wiley.com/doi/10.1002/ecs2.3531.

Drewitt, A.L., Langston, R.H.W., 2006. Assessing the impacts of wind farms on birds. Ibis 148, 29–42.

Dunk, J.R., Woodbridge, B., Lickfett, T.M., Bedrosian, G., Noon, B.R., LaPlante, D.W., Brown, J.L., Tack, J.D., 2019. Modeling spatial variation in density of golden eagle nest sites in the western U.S. PLoS One 14, e0223143.

Dunnett, S., Sorichetta, A., Taylor, G., Eigenbrod, F., 2020. Harmonised global datasets of wind and solar farm locations and power. Sci. Data 7. https://doi.org/10.1038/s41597-020-0469-8.

Eichhorn, M., Johst, K., Seppelt, R., Drechsler, M., 2012. Model-based estimation of collision risks of predatory birds with wind turbines. Ecol. Soc. 17. https://doi.org/10.5751/ES-04594-170201.

Fink, D., Auer, T., Johnston, A., Strimas-Mackey, M., Ligocki, S., Robinson, O., Hochachka, W., Jaromczyk, L., Rodewald, A., Wood, C., Davies, I., Spencer, A., 2022. eBird status and trends, data version: 2021; released: 2022. Cornell Lab of Ornithology, Ithaca, New York. https://doi.org/10.2173/ebirdst.2021.

Fink, D., Auer, T., Johnston, A., Strimas-Mackey, M., Ligocki, S., Robinson, O., Hochachka, W., Jaromczyk, L., Crowley, C., Dunham, K., Stillman, A., Davies, I., Rodewald, A., Ruiz-Gutierrez, V., Wood, C., 2023. eBird status and trends, data version: 2022; released: 2023. Cornell Lab of Ornithology, Ithaca, New York. https://doi.org/10.2173/ebirdst.2022.

Gantchoff, M.G., Hill, J.E., Kellner, K.F., Fowler, N.L., Petroelje, T.R., Conlee, L., Beyer Jr., D.E., Belant, J.L., 2020. Mortality of a large wide-ranging mammal largely caused by anthropogenic activities. Sci. Rep. 10, 8498.

Hoen, B.D., Diffendorfer, J.E., Rand, J.T., Kramer, L.A., Garrity, C.P., Hunt, H.E., 2018. U.S. wind turbine database (July 2013), ver. 1.2 (march 2014), ver. 1.0.2 (march 2018), ver. 1.3.2 (January 2019), ver. 2.3 (January 2020), ver. 3.3 (January 2021), ver. 4.3 (January 2022), ver. 5.3 (January 2023), and ver. 7.2 (November 2024). https://eerscmap.usgs.gov/uswtdb/data/.

Holmstrom, L.A., Hamer, T.E., Colclazier, E.M., Denis, N., Verschuy, J.P., Ruché, D., 2011. Assessing avian-wind turbine collision risk: an approach angle dependent model. Wind Eng. 35, 289–312.

Hunt, G.W., 2002. Golden Eagles in a Perilous Landscape: Predicting the Effects of Mitigation for Wind Turbine Blade-Strike Mortality. California Energy Commission, Sacramento, CA.

Hunt, G.W., Watson, J.W., 2016. Addressing the factors that juxtapose raptors and wind turbines. J. Raptor Res. 50, 92–96.

Hunt, G.W., Wiens, J.D., Law, P.R., Fuller, M.R., Hunt, T.L., Driscoll, D.E., Jackman, R.E., 2017. Quantifying the demographic cost of human-related mortality to a raptor population. PLoS One 12, e0172232.

Johnston, A., Auer, T., Fink, D., Strimas-Mackey, M., Iliff, M., Rosenberg, K.V., Brown, S., Lanctot, R., Rodewald, A.D., Kelling, S., 2020. Comparing abundance distributions and range maps in spatial conservation planning for migratory species. Ecol. Appl. 30, e02058.

Katzner, T.E., Nelson, D.M., Braham, M.A., Doyle, J.M., Fernandez, N.B., Duerr, A.E., Bloom, P.H., Fitzpatrick, M.C., Miller, T.A., Culver, R.C.E., Braswell, L., DeWoody, J., 2017. Golden eagle fatalities and the continental-scale consequences of local wind-energy generation. Conserv. Biol. 31, 406–415.

Katzner, T.E., Kochert, M.N., Steenhof, K., McIntyre, C.L., Craig, E.H., Miller, T.A., 2020. Golden eagle (*Aquila chrysaetos*), version 2.0. In: Rodewald, P.G., Keeney, B.K. (Eds.), Birds of the world. https://birdsoftheworld.org/bow/species/goleag/cur/introduction.

Kelling, S., Johnston, A., Bonn, A., Fink, D., Ruiz-Gutierrez, V., Bonney, R., Fernandez, M., Hochachka, W.M., Julliard, R., Kraemer, R., Guralnick, R., 2019. Using semistructured surveys to improve citizen science data for monitoring biodiversity. BioScience 69, 170–179.

Ledec, G.C., Rapp, K.W., Aiello, R.G., 2011. Greening the Wind: Environmental and Social Considerations for Wind Power Development. World Bank, Washington, DC.

Lee, J., Zhao, F., 2022. GWEC Global Wind Report 2022. Global Wind Energy Council, Brussels.

Lekuona, J.M., Ursua, C., 2007. Avian mortality in wind power plants of Navarra (northern Spain). In: de Lucas, M., Janss, G.F.E., Ferrer, M. (Eds.), Birds and Wind Farms: Risk Assessment and Mitigation. Quercus/Libreria Linneo, Madrid, pp. 177–192.

Madders, M., Whitfield, D.P., 2006. Upland raptors and the assessment of wind farm impacts. Ibis 148, 43–56.

Masden, E.A., Cook, A.S.C.P., 2016. Avian collision risk models for wind energy impact assessments. Environ. Impact Assess. Rev. 56, 43–49.

McCabe, J.D., Clare, J.D., Miller, T.A., Katzner, T.E., Cooper, J., Somershoe, S., Hanni, D., Kelly, C.A., Sargent, R., Soehren, E.C., Threadgill, C., Maddox, M., Stober, J., Martell, M., Salo, T., Berry, A., Lanzone, M.J., Braham, M.A., McClure, C. J.W., 2021. Resource selection functions based on hierarchical generalized additive models provide new insights into individual animal variation and species distribution. Ecography 44, 1756–1768.

Millsap, B.A., Zimmerman, G.S., Sauer, J.R., Nielson, R.M., Otto, M., Bjerre, E.R., Murphy, R., 2013. Golden eagle population trends in the western U.S.: 1968–2010. J. Wildl. Manag. 77, 1436–1448.

Millsap, B.A., Madden, K., Murphy, R.K., Brennan, M., Pagel, J.E., Campbell, D., Roemer, G.W., 2019. Demographic consequences of sexual differences in age at first breeding in Cooper's hawks (*Accipiter cooperii*). Auk. https://doi.org/10.1093/auk/ukz032.

Millsap, B.A., Zimmerman, G.S., Kendall, W.L., Barnes, J.G., Braham, M.A., Bedrosian, B. E., Bell, D.A., Bloom, P.H., Crandall, R.H., Domenech, R., Driscoll, D., Duerr, A.E., Gerhardt, R., Gibbs, S.E.J., Harmata, A.R., Jacobson, K., Katzner, T.E., Knight, R.N., Lockhart, J.M., Watson, J.W., 2022. Age-specific survival rates, causes of death, and allowable take of golden eagles in the western U.S. Ecol. Appl. 32. https://doi.org/10.1002/eap.2544.

Murgatroyd, M., Bouten, W., Amar, A., 2020. A predictive model for improving placement of wind turbines to minimize collision risk potential for a large soaring raptor. J. Appl. Ecol. 58, 857–868.

National Renewable Energy Laboratory, 2017. U.S. wind resource maps. http://www.nrel.gov/gis/wind.html.

New, L., Bjerre, E.R., Millsap, B.A., Otto, M.C., Runge, M.C., 2015. A collision risk model to predict avian fatalities at wind facilities: an example using golden eagles, *Aquila chrysaetos*. PLoS ONE 10, e0130978.

New, L., Simonis, J.L., Otto, M.C., Bjerre, E.R., Runge, M.C., Millsap, B.A., 2021. Adaptive management to improve eagle conservation at terrestrial wind facilities. Conserv. Sci. Pract. 3. https://doi.org/10.1111/csp2.449.

Nielson, R.M., Murphy, R.K., Millsap, B.A., Howe, W.H., Gardner, G., 2016. Modeling late-summer distribution of golden eagles (*Aquila chrysaetos*) in the western U.S. PLoS One 11, e0159271.

Pagel, J.E., Kritz, K.J., Millsap, B.A., Murphy, R.K., Kershner, E.L., Covington, S., 2013. Bald eagle and golden eagle mortalities at wind energy facilities in the contiguous U. S. J. Raptor Res. 47, 311–315.

R Core Team, 2021. R: A language and environment for statistical computing. https://www.R-project.org/.

Rand, J.T., Kramer, L.A., Garrity, C.P., Hoen, B.D., Diffendorfer, J.E., Hunt, H.E., Spears, M., 2020. A continuously updated, geospatially rectified database of utility-scale wind turbines in the United States. Sci. Data 7, 15. https://doi.org/10.1038/s41597-020-0353-6.

Ruiz-Gutierrez, V., Bjerre, E.R., Otto, M.C., Zimmerman, G.S., Millsap, B.A., Fink, D., Stuber, E.F., Strimas-Mackey, M., Robinson, O.J., 2021. A pathway for citizen science data to inform policy: a case study using eBird data for defining low-risk collision areas for wind energy development. J. Appl. Ecol. 58, 1104–1111.

Russell, R.E., Franson, J.C., 2014. Causes of mortality in eagles submitted to the National Wildlife Health Center. Wildl. Soc. Bull. 38, 697–704.

Smallwood, K.S., 2013. Comparing bird and bat fatality-rate estimates among north American wind-energy projects. Wildl. Soc. Bull. 37, 19–33.

Smallwood, K.S., Thelander, C., 2008. Bird mortality in the Altamont pass wind resource area, California. J. Wildl. Manag. 72, 215–223.

Stillman, A.N., Howell, P.E., Zimmerman, G.S., Bjerre, E.R., Millsap, B.A., Robinson, O.J., Fink, D., Stuber, E.F., Ruiz-Gutierrez, V., 2023. Leveraging the strengths of citizen science and structured surveys to achieve scalable inference on population size. J. Appl. Ecol. 60, 2389–2399.

Sullivan, B.L., Wood, C.L., Iliff, M.J., Bonney, R.E., Fink, D., Kelling, S., 2009. eBird: a citizen-based bird observation network in the biological sciences. Biol. Conserv. 142, 2282–2292.

Tack, J.D., Noon, B.R., Bowen, Z.H., Fedy, B.C., 2020. Ecosystem processes, land cover, climate, and human settlement shape dynamic distributions for golden eagle across the western US. Anim. Conserv. 23, 72–82.

US Department of Energy, 2021. Distributed Wind Market Report, 2021 edition. Office of Energy Efficiency and Renewable Energy, Wind Energy Technologies Office, Washington, DC.

US Fish and Wildlife Service (USFWS), 2016. Bald and Golden Eagles: Population Demographics and Estimation of Sustainable Take in the U.S., 2016 Update. Division of Migratory Bird Management, Washington, DC.

Watson, R.T., Kolar, P.S., Ferrer, M., Nygård, T., Johnston, N., Hunt, W.G., Smit-Robinson, H.A., Farmer, C.J., Huso, M., Katzner, T.E., 2018. Raptor interactions with wind energy: case studies from around the world. J. Raptor Res. 52, 1–18.

Whitfield, D.P., Fielding, A.H., Mcleod, D.R.A., Haworth, P.F., 2004. Modelling the effects of persecution on the population dynamics of golden eagles in Scotland. Biol. Conserv. 119, 319–333.

Wiens, J.D., Schumaker, N.H., Inman, R.D., Esque, T.C., Longshore, K.M., Nussear, K.E., 2017. Spatial Demographic Models to Inform Conservation Planning of Golden Eagles in Renewable Energy Landscapes.

Wiser, R., Bolinger, M., 2019. 2018 Wind Technologies Market Report. Lawrence Berkeley National Laboratory, Berkeley, CA. https://escholarship.org/uc/item/2tc1819q.