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Population-Level Risk to Hoary Bats Amid Continued Wind Energy Development

Assessing Fatality Reduction Targets Under Broad Uncertainty

2020 TECHNICAL REPORT

Population-Level Risk to Hoary Bats Amid Continued Wind Energy Development

Assessing Fatality Reduction Targets Under Broad Uncertainty

3002017671
Final Report, March 2020

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Acknowledgments

Abstract

Across the United States and Canada, hoary bats (*Lasiurus cinereus*) are the most common species among reported bat fatalities at wind farms. Although large gaps in our knowledge about hoary bat population size and dynamics makes the impact of wind turbine fatality difficult to assess, expert opinion places the population parameters in a range in which the risk of decline or extinction may be high. A key question for the wind energy industry, regulators, and conservation groups is whether fatality minimization strategies such as operational curtailment or the use of bat deterrent devices could manage population-level risks, particularly considering continued growth in installed capacity.

Approach
The risk of hoary bat population decline or extinction in the presence of wind turbine mortality was projected from 2012 through 2050 using a simple population model. Future levels of turbine mortality were a function of expected capacity buildout in the United States and Canada. High- and low-buildout scenarios were explored. Non-probabilistic scenarios of population growth rate and abundance addressed the large uncertainties around hoary bat population status.

Results
Risks to hoary bats and the level of fatality reduction necessary to minimize them were highly dependent on population size. In the absence of fatality reduction, declines of at least 50% occurred at all initial population sizes tested and in some scenarios were likely to occur prior to 2020. However, extinction risk without fatality reduction was present only in populations that started with fewer than 5 million hoary bats, even in cases in which the population was assumed to be declining independently of wind mortality.

Adopting fatality reductions in 2020 with efficacies within the range observed from curtailment and deterrent studies was sufficient to manage extinction risk in populations of at least about 2 million. However, this reduction could not be implemented in time to prevent significant declines in populations of fewer than about 3 million hoary bats.

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Applications, Value, and Use

Understanding risks to hoary bats posed by wind turbine mortality can help in the development of strategies to ensure that listing is unnecessary. Though existing uncertainty about population status is large enough to produce a variety of forecasts, the model itself provides a tool to integrate new information from sources such as post-construction monitoring, genetic analyses, or acoustic surveys.

Keywords

Acoustic deterrents
Hoary bats
Lasiurus cinereus
Operational minimization
Population model
Smart curtailment
TIMR
Wind and wildlife

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Deliverable Number: 3002017671

Product Type: Technical Report

Product Title: Population-Level Risk to Hoary Bats Amid Continued Wind Energy Development: Assessing Fatality Reduction Targets Under Broad Uncertainty

PRIMARY AUDIENCE: Environmental managers at electric utilities and wind energy companies

SECONDARY AUDIENCE: Interested researchers and agency staff at organizations interested in wind/wildlife interaction management issues

KEY RESEARCH QUESTION

Wind energy development is an important part of the response to climate change, but it is not without environmental impacts—including turbine-related fatalities of birds and bats. Hoary bats are the most common bat fatality reported at wind farms. Expert elicitation of factors related to population status, combined with mathematical modeling, has recently suggested the potential for wind turbine fatalities to substantially elevate population-level risks for hoary bats. The current study assessed these risks considering anticipated wind energy capacity additions through 2050 and sought the level of fatality reduction necessary to minimize declines or the risk of extinction.

RESEARCH OVERVIEW

Future hoary bat fatalities and abundance were projected by simulation over a range of initial abundance and population growth rates. The risks of decline and extinction were assessed in the absence of wind fatality, with wind fatality, and with varying levels of fatality reduction reflecting minimization strategies. The level of fatality reduction necessary to manage extinction risk was then compared with existing studies of operational curtailment and ultrasonic deterrents to ascertain the potential for existing technologies to meet conservation needs.

KEY FINDINGS

- Hoary bat population declines of 50% or more were likely without fatality reduction regardless of initial abundance. Such declines may have already occurred (Section 5).
- Managing extinction risk in the smallest population simulated in the study required nearly 90% fatality reduction, while extinction risk was negligible if the hoary bat population was initially greater than about 5 million (Section 5).
- Existing fatality reduction strategies—including curtailment, smart curtailment, and deterrents—may be sufficient to manage extinction risk if the hoary bat population is 2 million or more (Section 6).
- Population size remains the top research priority, but greater understanding of demography, behavior, and sex-specific patterns of fatality would have management relevance.

WHY THIS MATTERS

Although economic and technological considerations will play key roles in the management of bat fatalities at wind farms, it is important that management also have a firm biological basis. This study does not deliver a single, actionable statement of guidance on fatality reductions but strives to provide context useful in prioritizing further research and understanding the potential magnitude of risks.

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EXECUTIVE SUMMARY

HOW TO APPLY RESULTS

This report presents the use of a model to outline the potential for risk and minimization needs by exploring a range of parameter space. More specific guidance requires a more precise estimate of hoary bat population size. Although a direct assessment of population size is likely not feasible, several avenues are being investigated for finding recent trends in abundance. An indirect estimate of population size could be obtained from such trend information using the population model developed in this study.

LEARNING AND ENGAGEMENT OPPORTUNITIES

- This report addresses an area of active research with participants from a wide spectrum of organizations including the wind energy industry, academia, government researchers, regulators, and conservation organizations.
- EPRI is available to develop collaborative research projects and to inform electric power industry stakeholders.
- Additional detail about expert opinions of hoary bat population size and growth can be found in the Frick et al. (2017) study referenced in this report.
- A summary of current bat/wind interaction research and future priorities can be found in the 2018 Bats and Wind Energy Cooperative Workshop Proceedings.
- Recent research findings are summarized in the Proceedings of the 2018 NWCC Wind Wildlife Research Meeting.

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PROGRAMS: Endangered and Protected Species, P195; Environmental Aspects of Wind, P192

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Section 1: Introduction

Wind energy is commonly seen as a key element among actions to mitigate continued climate change. As such, there is broad societal support for wind energy development. However, no known method of energy production is without impacts. Interactions between wind turbines and wildlife are one of the main environmental issues faced by the wind industry. Perhaps because of the youth of wind energy as a commercial enterprise and the collective hope around its success, the industry has been the focus of advanced quantitative frameworks for assessing direct impacts and managing cumulative take. For on-shore wind farms in the United States, a handful of species account for most regulatory focus because of their protected status, including golden eagles (USFWS 2016) and Indiana bats (USFWS 2019).

Wind farm interaction with wildlife in the United States and Canada includes bat fatalities. Amid the rapid expansion of wind energy since the beginning of the millennium, concern has grown that these fatalities could have population-level impacts for some species (Kunz et al. 2007). The risk of such impacts to hoary bats (*Lasiurus cinereus*) now appears significant (Frick et al. 2017).

Evidence for the direct impact of wind farms on bats is typically sourced from post-construction survey reports. Efforts to obtain an accurate estimate of the total number of bat fatalities have been hindered by non-systematic reporting as well as variable effort and methodology among surveys (Huso and Dalthorp 2014). Recent analysis using the most complete, albeit still limited and non-systematic, collection of fatality data available (n = 202 studies) yielded a median of 2.66 bat fatalities per megawatt (MW) per year over all geographic regions and bat species in the United States (AWWI 2018). Though this analysis was not intended as a projection of total national or continental fatality, it attempts to illustrate the unevenness of fatality rates among facilities. Based on data from this report, 75% of studies reported fewer than five bats per MW per year, while a smaller number of facilities—mostly in the Midwest and Northeast—experienced higher fatality rates. As a result, mean fatality is higher than the median. An estimate based on data from 2000–2011 and adjusted for regional levels of installed capacity put the mean fatality rate at just under five bats per MW per year (Arnett and Baerwald 2013). The 2018 AWWIC[1] analysis also found a fatality rate just under five bats per MW (depicted in Figure 13 on page 33 of AWWI 2008).

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[1] https://awwi.org/wp-content/uploads/2020/01/Fact-Sheet-AWWIC-2020.pdf

Not all bat species are represented equally among reported fatalities. Cave-dwelling bats make up a relatively small proportion of all carcasses. Indiana bats, which are listed as endangered, have only 13 documented fatalities at wind farms (USFWS 2019). By comparison, White-Nose Syndrome, a fungal disease that affects bats hibernating in caves, has caused severe population declines in multiple species of hibernating bats (Frick et al. 2010; Langwig et al. 2012; Frick et al. 2015) and prompted the listing of northern long-eared bats as federally threatened (80 CFR 17974, 2015). Three species of bats that have been highly impacted (little brown bats, tricolored bats, and northern long-eared bats) are now listed as endangered in Canada and have been listed in several states as well. Impacts of other sources of mortality on small, remnant populations of these species throughout eastern North America is a current concern for managing these populations. White-Nose Syndrome is expected to pose a far greater risk to the persistence and recovery of Indiana bats than will wind farm mortality in the short term (Shoemaker et al. 2012).

Unlike cave-dwelling bats, migratory tree bats are killed in large numbers. Three tree bat species make up approximately 72% of carcasses reported in post-construction surveys and represent most fatalities at wind farms except at sites in the southwest (AWWI 2018).

The Most Common Fatality

Of all species in the United States and Canada, the hoary bat (see Figure 1-1) is the most abundant among reported fatalities and is found at the highest proportion of facilities. Arnett and Baerwald (2013) reported that 38% of fatalities were hoary bats. The AWWIC 2018 analysis reported a lower percentage of hoary bats among carcasses, 32%, but noted that its data set included additional facilities and a greater representation of the southwestern United States. Another recent analysis with a larger sample of post-construction studies found that hoary bats were 35% of reported fatalities (Thompson et al. 2017). The more recent numbers, though based on raw carcass counts not adjusted for searcher efficiency and scavenging (Dalthorp et al. 2018), support the 2013 assessment that hoary bats make up a substantial proportion of all wind-related bat fatalities. To date, only the 2013 study provides an estimate of total fatality and a prevalence of hoary bats based on the same data set.

Figure 1-1
An adult hoary bat, Lasiurus cinereus. *Photo credit: USDA Forest Service.*

Mind the Knowledge Gaps

The hoary bat is a mostly solitary tree-roosting bat. Despite its large size, with a wingspan over a foot, it is difficult to locate and study because of its migratory behavior and the fact that individuals do not aggregate in hibernacula, as cave-dwelling bats do. Basic features of hoary bat biology are still being discovered, such as hibernation-like behavior during cooler months (Weller et al. 2016), echolocation signals that may be difficult for acoustic surveys to detect (Corcoran and Weller 2018), and migratory patterns that lead to interaction with wind turbines (Baerwald et al. 2014).

Fundamental ecological parameters of the hoary bat population are unknown. Primary among these is the current size of the population (BWEC 2018), but gaps also include any trend in abundance and the potential for population growth. A lack of definite knowledge about hoary bat population status and biology has been the prominent feature of attempts to assess the potential impact of wind energy development on its viability (Kunz et al. 2007; Frick et al. 2017).

The Frick et al. (2017) study, which this report will cite often, includes an expert elicitation of the size and growth rate of the hoary bat population in the United States and Canada. In the absence of direct empirical measurements, the results of this elicitation offer the best existing assessment of population status. Using a demographic model informed by the elicitation, the study demonstrated the

potential for wind turbine fatality to strongly impact hoary bat population viability.

Study Goals

This study aimed to assess the level of risk to hoary bat populations posed by wind energy development through 2050. The study employed mathematical modeling to project possible future states of the population. This approach closely followed that of Frick et al. (2017). The current study extended the 2017 investigation by asking two questions:

1. How will continued buildout of wind energy capacity affect the risk of hoary bat population decline or extinction?

2. How much fatality reduction is necessary to manage population-level impacts?

Approach

To answer these key questions, this investigation did the following:

1. Developed a simple model for hoary bat population dynamics and wind turbine mortality and validated the model against the Frick et al. study

2. Developed two scenarios of wind capacity buildout in the United States and Canada through 2050

3. Selected a model for scaling fatalities with installed capacity

4. Combined these elements in a stochastic modeling framework that projected both the uncertainty about mean fatality rates as well as the variability of population growth to generate probabilistic outcomes

5. Addressed a spectrum of possible abundance, the most significant knowledge gap concerning hoary bat population status

Section 2: Reproducing the Frick et al. Model

The mathematical model used in the 2017 study was relatively simple. It was a scalar model in which abundance was described as a single measure addressing all individuals in the population (male and female, young and old). Annual changes in abundance were driven by a single population growth rate term that combined net survival and reproduction as well as a separate term representing wind-related mortality. One year of population change was computed as

$$N_{t+1} = N_t \lambda_t (1 - m)$$
<div align="right">*Eq. 2-1*</div>

where N_t is hoary bat abundance in year t, λ_t is baseline population growth rate in year t, and m is a constant wind mortality rate. Abundance was limited to 10 times its initial value, a simple way to match the reality that populations cannot become infinitely large.

The model was stochastic; that is, the population growth rate was sampled at random from a lognormal distribution every year rather than being held constant. This stochasticity represented environmental variation and made the trajectories produced by an individual run of the model nearly unique. Many replicates (10,000) of the model were run using the same mean parameter values to produce a statistical distribution of projected population size over time, a process known as *first-order Monte Carlo simulation*.

Is a More Complex Model Justified?

The construction of models under conditions of large epistemic uncertainty is subjective. An argument for simplicity is justified by a lack of knowledge about the details of a more complex model. On the other hand, a simple model is itself an assertion about population processes that are poorly understood.

We considered three ways in which the 2017 model could be made more complex—age structure, spatial structure, and compensatory dynamics—and evaluated whether they were justified from the perspective of risk analysis. Our conclusions were as follows:

1. Age structure would not add information to the model given our current state of knowledge.

2. Spatial structure deserves attention and will be addressed in a future study.

3. Compensatory dynamics have the potential to reduce population risks but cannot be characterized from empirical evidence. Appendix A presents an assessment in which compensatory dynamics arise from hypothetical density-dependent population growth.

A discussion of our consideration of each complicating feature follows.

Age Structure

Only a brief analysis was necessary to reject the utility of introducing age-specific abundance and vital rates. The driving force behind this rejection was the lack of knowledge about age-specific wind mortality.

We developed a generic model dividing the population into first-year and adult individuals to check the potential for altered responses to wind fatality. Two key assumptions were that 1) wind mortality affects all ages and 2) any change in wind mortality would have the same proportional effect on mortality for all age classes. The first assumption is supported by the timing of hoary bat reproduction. Pups are born in the summer and are volant prior to fall migration, which is commonly the peak period for hoary bat fatalities at wind farms. The second assumption is justified by the absence of any knowledge on age-specific risk.

Under these two assumptions, the addition of age structure to the model would not introduce any new information with respect to how wind mortality affects the population growth rate of hoary bats. The long-term population growth rate in an age-structured model is given by the eigenvalue of its transition matrix. Under the previous simplifying assumptions, the eigenvalue is simply proportional to survival at wind farms. In other words, the two-stage model reduces to Equation 2-1 in the absence of evidence for age-specific impacts of wind mortality.

Spatial Structure

The hoary bat population can be considered panmictic in the continental United States and Canada. Individuals appear to undertake large migratory movements (Cryan et al. 2014b), and genetic studies generally concur that there is no evidence for geographically distinct populations (Russell et al. 2015; Sovic et al. 2016). Informal discussion with experts suggested that one could not safely assume that an individual observed in Alberta would not be affected by a wind farm in Texas.

However, wind-related fatality rates have strong geographic structure—therefore, the geographic pattern of future buildout will likely affect our projections of mortality. This question will be addressed in a future study.

Compensatory Dynamics

The model used in our main analysis, particularly when the population is assumed to be stable, is a conservative assessment tool. Even small amounts of

additional mortality will set the population on a path toward eventual extinction. This sensitivity is offset, however, by the limited time horizon of the analysis; only acute impacts will affect risk so quickly.

The hoary bat population's sensitivity to additional mortality may be buffered by a compensatory increase in survival or reproduction. A common framework for examining such compensation is density-dependent population growth, in which the population is regulated around an equilibrium size or density.

Appendix A presents an assessment of risk and fatality reduction targets under density-dependent growth. We have no empirical evidence with which to formulate or parameterize the model. As with population size, maximum population growth rate could be set high enough to offset or trivialize wind-related mortality. Using the 2017 expert elicitation as a guide to maximum growth rate, compensatory dynamics decreased risk but did not fundamentally alter the main conclusions of our study.

Validation

To ensure that the current study was comparable to that of the 2017 investigation, we undertook a reproduction of the previous model's results. W. F. Frick supplied a table of numerical results corresponding to the 10 levels of abundance and 10 levels of population growth rate explored in the 2017 study (see Table 2-1).

Our reproduction of the model yielded nearly identical extinction risks (Table 2-2) and highlighted the same critical population growth rates at which the probability of extinction fell below 1% (right-most entry in each row of Table 2-1 and Table 2-2).

Table 2-1
Extinction risk from the 2017 model, supplied by W. F. Frick. Zeros omitted for clarity.

Abundance (millions)	Population Growth Rate									
	0.94	0.964	0.989	1.014	1.04	1.067	1.094	1.122	1.151	1.18
1	1.00	1.00	1.00	1.00	1.00	0.89	0.10	0.00		
2	1.00	1.00	0.85	0.07	0.00					
3	1.00	0.82	0.06	0.00						
4	0.98	0.32	0.00							
5	0.88	0.08	0.00							
6	0.69	0.03	0.00							
7	0.52	0.01	0.00							
8	0.38	0.00								
9	0.27	0.00								
10	0.20	0.00								

Table 2-2
Extinction risk in our reproduction of the 2017 model. Zeros omitted for clarity.

Abundance (millions)	Population Growth Rate									
	0.94	0.964	0.989	1.014	1.04	1.067	1.094	1.122	1.151	1.18
1	1.00	1.00	1.00	1.00	1.00	0.91	0.11	0.00		
2	1.00	1.00	0.86	0.07	0.00					
3	1.00	0.83	0.05	0.00						
4	0.98	0.32	0.00							
5	0.88	0.09	0.00							
6	0.72	0.03	0.00							
7	0.54	0.01	0.00							
8	0.40	0.00								
9	0.29	0.00								
10	0.20	0.00								

◄ 2-4 ►

Notes on Reproducing the Model

Several technical clarifications from W. F. Frick aided in this reproduction and could be useful to other researchers attempting to build on that work:

1. The distribution of population growth rates over time had a standard deviation of 0.1. The paper reported this value as the variance.

2. The standard error described the underlying normal distribution used to generate lognormal variability.

3. All population projection was performed with log-transformed population growth rates. Therefore, the reported values for lambda represent the median rather than mean values for each simulation.

4. The main body of the paper refers to the median of expert opinions on the most likely population size as 2.5 million. This is a slight deviation from the value reported in the appendix, which was 2.25 million.

The reproduced model was used to explore the relationship among abundance, population growth rate, and extinction risk at greater parameter resolution than the 2017 study. We examined 37 levels of abundance (1 to 10 million by 0.25 million) and 100 levels of population growth rate. The minimum population growth rate necessary to hold the probability of extinction below 1% declined smoothly approximately as a function of the inverse of initial abundance (see Figure 2-1).
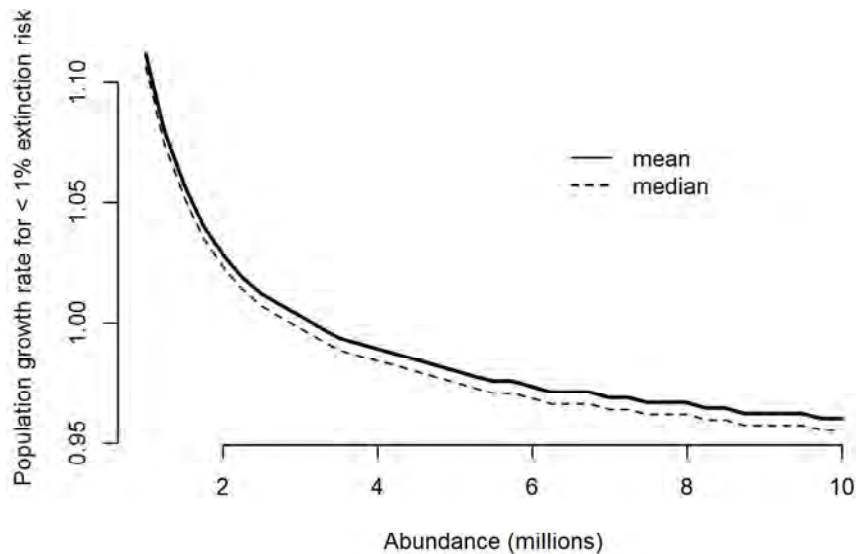
Figure 2-1
The relationship among hoary bat abundance, population growth rate, and extinction risk after 100 years at 2014 levels of installed capacity

Section 3: Projected Wind Energy Buildout

The purpose of the current study was to ascertain the level of fatality minimization required to offset hoary bat population-level responses to existing installed capacity and anticipated growth in that capacity over time. We established two scenarios of wind energy buildout in the United States and Canada, each through 2050. Although neither scenario was extreme, one targeted the higher end of projected capacity additions while the other focused on lower growth. The high-buildout scenario considered national-scale development targets reflecting, to some extent, societal aspiration for maximized renewable energy development. The low-growth scenario considered opposing market forces. Though there are many market forecast models indicating varying levels and tempos of buildout, using a pair of scenarios is sufficient to illustrate how minimization needs might be affected by changes in the future rate of wind energy development.

High-Buildout Scenario

The high-buildout scenario, which served as the focal scenario in this study, used two government reports to determine future wind capacity. For the United States, buildout was based on the Wind Vision report (USDOE 2015), which lays out a goal of achieving 318 GW of installed land-based capacity by 2050 (see Figure 3-1). For Canada, projected buildout was based on the reference case of the Canada's Energy Future market assessment (NEB 2017), which puts total Canadian installed capacity at just over 26 GW by 2040. Though a market assessment, the reference case was influenced by government renewable energy goals. A newer assessment became available at the time of writing (NEB 2018) and projected slightly less wind energy capacity by 2040. This difference would not substantively affect our findings. Canadian projections were extended to 2050 by assuming that the capacity addition in 2040 continued annually over the following decade, leading to a final capacity of 29 GW.
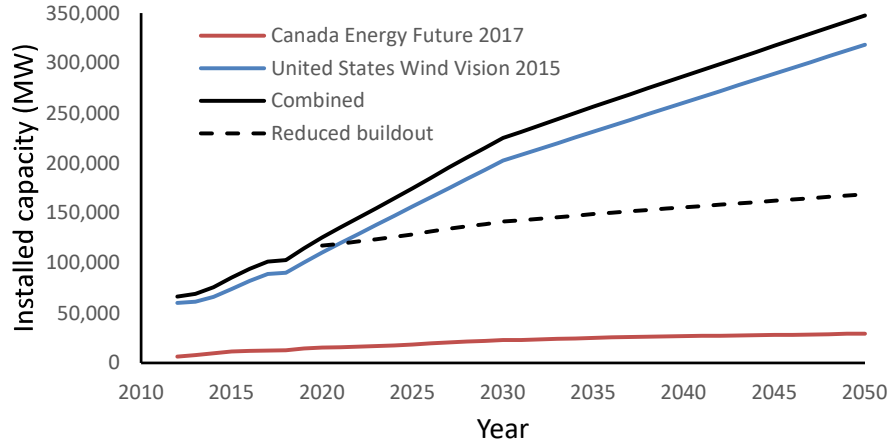
Figure 3-1
Wind energy capacity scenarios in the United States and Canada through 2050

The sum of projected U.S. and Canadian installed capacity (Figure 3-1) was used to compute total hoary bat fatality through 2050. Regional differences in fatality rate or projected capacity additions were not considered in this study.

Low Buildout Scenario

Some market assessment scenarios anticipate that buildout in the United States will fall far short of the levels established in the 2015 Wind Vision report. Such projections typically consider the expiration of wind energy incentives as well as low natural gas prices. For reference, the U.S. Energy Information Administration (USEIA) forecasts about 140 GW of land-based wind energy installed capacity by 2050, with development slowing substantially after the anticipated expiration of tax credits in 2020 (USEIA 2018). For the low-buildout scenario, we matched final projected U.S. capacity to the USEIA projection by reducing capacity additions by 82% after 2020, relative to the Wind Vision scenario. We did not alter the Canadian buildout projection. The resulting total capacity projection through 2050 reached 169 GW, compared with the high-buildout scenario's 348 GW (Figure 3-1).

Section 4: Projecting Future Mortality

Mortality vs. Fatality

Importantly, the estimate of population-level **mortality** associated with a particular rate of **fatalities** per MW depends on how many hoary bats there are. *Fatalities* are the number of bat deaths estimated to be caused by wind turbines each year. *Mortality* is the per capita risk that a bat will be killed. It is this per capita rate that links wind turbines and hoary bat population dynamics.

To get mortality from fatality requires an estimate of population size. In this study, the population in question was the total number of bats in the United States and Canada. Frick et al. (2017) assumed this was one well-mixed population, citing genetic evidence suggesting that there are no strong geographic barriers dividing the continental population up into more insular units in North America (Russell et al. 2015). As such, the experts consulted in the Frick et al. elicitation developed a range of population size estimates they considered low, likely, and high (see Figure 4-1). In the absence of data on hoary bat abundance, the expert elicitation served as a basis for assessing current and future hoary bat mortality due to wind.

Hoary bat population size (2014)
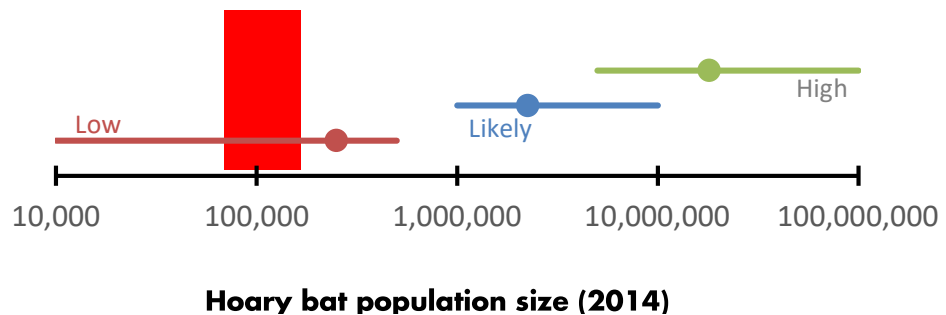
Figure 4-1
Expert-elicited range of possible hoary bat abundance in the United States and Canada, based on Frick et al. (2017). Circles indicate medians. The width of the red column indicates the 95% confidence interval for total bat fatalities at wind farms in 2012, from Arnett and Baerwald (2013). Note that abundance is on a log scale.

Scaling Mortality with Buildout

It is not currently possible to predict the number of fatalities that will occur over the course of a year at any given facility. However, the most common assumption is that the risk of collisions is proportional to the total rotor-swept area of a wind farm. This assumption is implicit when using a fatality rate per MW because rated capacity is approximately proportional to rotor-swept area. An advantage of the MW-based rate is that, at least theoretically, it allows the comparison or extrapolation of data from turbines of disparate designs and capacities.

Frick et al. (2017) used an estimate of total hoary bat fatality across the United States and Canada developed in Arnett and Baerwald (2013). This study found an average fatality rate for all bats using post-construction survey reports, which covered only a fraction of installed capacity and only reported 1 to 4 years of fatality at each facility over the 11-year span of data examined. The authors used the data to estimate regional fatality rates and then extrapolated total fatality using 2012 regional installed capacity levels (Figure 4-1). It was assumed that fatalities are simply proportional to capacity, so that regional total fatality was the product of mean fatality rate and installed capacity. Frick et al. (2017) followed this logic, applying the per-MW rate averaged across the United States and Canada to 2014 installed capacity for an updated estimate of total fatality.

In the current study, the assumption that fatalities are simply proportional to installed capacity was problematic. So much future buildout was expected under the Wind Vision scenario that simple proportionality projected an improbably large number of annual fatalities. If the population was assumed to be small enough, the projected number of fatalities could exceed the total number of bats. This was not a problem in the 2017 study because capacity was fixed at 2014 levels.

The simplest solution to the excessive fatality problem would be to put a ceiling on the number of fatalities at 100% of the population size. However, this still seems biologically unreasonable given the relatively small percentage of land area covered by wind farms.

A biologically reasonable approach to wind mortality projections is to retain the assumption that hoary bat fatalities are proportional to MW but apply that proportionality to a time scale much smaller than a year. If N is the hoary bat population size and C is installed capacity, then the number of fatalities in a short time interval, not a full year, is aNC, where a is an interaction term and is much smaller than the annual fatality rate. The per capita change in hoary bat population size over a short time interval is then

$$\frac{dN}{Ndt} = -aC \qquad\qquad Eq.\ 4\text{-}1$$

Annual mortality can be found by integrating Equation 4-1 over one year:

$$mortality = 1 - e^{-aC} \qquad\qquad Eq.\ 4\text{-}2$$

where e is the base of the natural logarithm.

One way to understand the right side of Equation 4-2 is as the probability of a hoary bat randomly encountering and being killed by a turbine. If there are enough turbines, an individual hoary bat is likely to encounter more than one of them over the course of a year. However, if these encounters are fatal, only the first will occur. Assuming that mortality is proportional to installed capacity overestimates the number of fatalities by allowing the same bat to be killed multiple times. Equation 4-2 is the probability of a bat encountering and being killed by **at least one** turbine over the course of a year, assuming that encounters occur at random.

Another way to understand Equation 4-2 is that it accounts for the relatively continuous removal of bats from the population. Applying a fatality rate derived from annual estimates to a population that is decreasing quickly will result in an overestimate of projected mortality. Accounting for the within-year decline in abundance helps avoid that problem.

Equation 4-2 is not novel. For instance, it is part of the Nicholson-Bailey predator-prey model (Murdoch et al. 1985), a fundamental model in ecology. More generally, it is the discretized solution to mass action models, which are common across diverse disciplines from chemistry and physics to ecology and economics.

The practical difference between the integrated mortality model (Equation 4-2) and assuming that mortality is simply proportional to installed capacity depends on how much conditions differ from the year in which the model is trained. If the value of a is set using 2012 conditions of hoary bat fatality and population size, the expected fatality in 2014 does not differ meaningfully from that estimated using the assumption of simple proportional risk, as in Frick et al. (2017). But over a future of increasing capacity, the integrated fatality model projects lower annual mortality. The difference is most pronounced at small population size (see Figure 4-2).
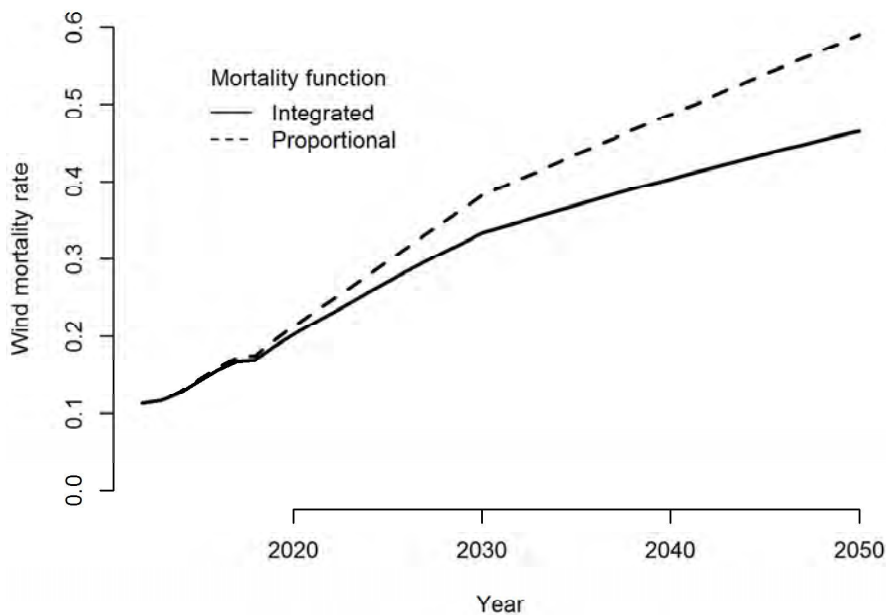
Figure 4-2
Projected mean annual hoary bat mortality due to wind turbines through 2050 in
the high-buildout scenario using two mortality functions and assuming a 2012
population of 1 million bats

The integrated model approaches but never exceeds 100% mortality. Because of
this nonlinearity, mortality becomes less sensitive to the addition or subtraction
of capacity as installed capacity increases. However, this same insensitivity applies
to fatality reduction. If the fatality rate is reduced by 30%, a population that
started with 1 million individuals in 2012 would experience only a 25% reduction
in mortality in 2050. A 60% reduction in fatality rate would yield only a 46%
reduction in 2050 mortality (see Figure 4-3).
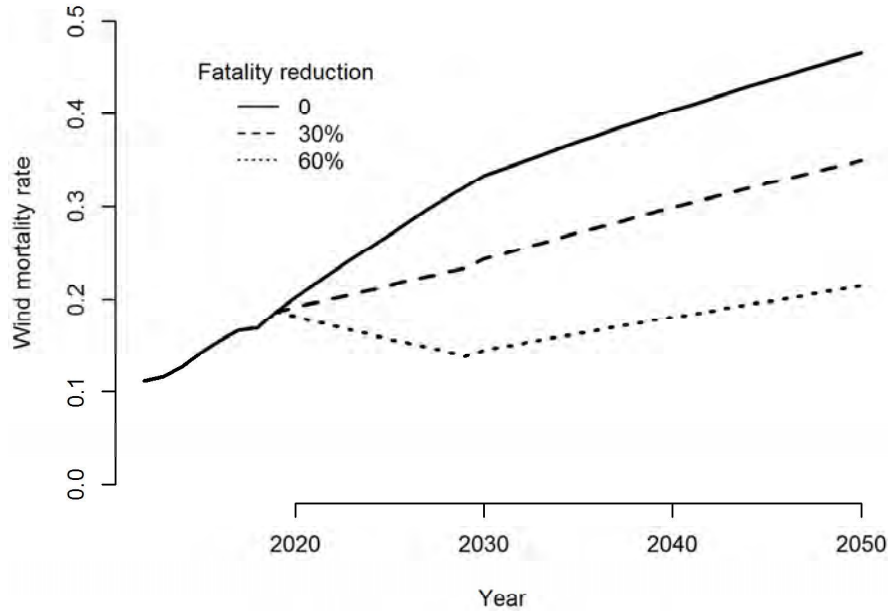
Figure 4-3
The response of wind mortality to reductions in the rate of fatalities per MW

Estimating Mortality

Using the integrated model of mortality (Equation 4-2), future mortality was scaled to installed capacity by first solving for the interaction term, *a*, under 2012 conditions. If F_{2012} is the number of fatalities in 2012 and N_{2012} is the initial population size, the wind mortality in 2012, m_{2012}, is F_{2012}/N_{2012}. Taking the installed capacity in that year to be C_{2012}, Equation 4-2 can be manipulated to solve for *a*:

$$a = \frac{ln\,(1 - m_{2012})}{C_{2012}} \qquad\qquad Eq.\ 4\text{-}3$$

Equation 4-3 was used to estimate *a* for each initial abundance level examined. This parameter value was then inserted into Equation 4-2 to determine mortality over a scenario in which installed capacity, *C*, increased over time.

Adoption

The projections of mortality with fatality reduction (Figure 4-3) assumed that whatever technology or strategy was being used would be adopted in all new facilities starting in 2020. However, existing facilities would take up to 10 years to fully implement fatality reduction measures. As a result, full adoption was delayed until 2029 (see Figure 4-4).
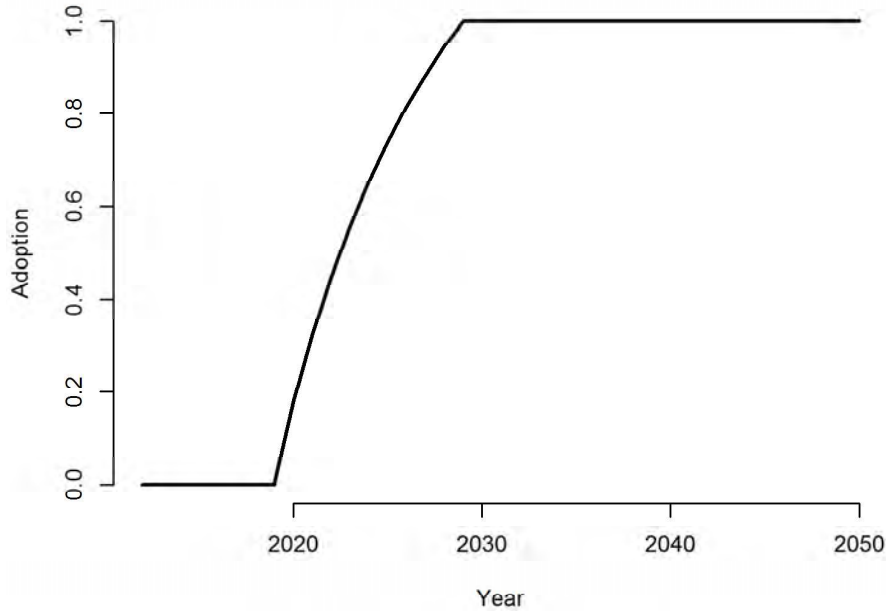
Figure 4-4
The proportion of installed capacity adopting hypothetical fatality reduction measures over time

Adoption of fatality reduction measures (Figure 4-4) was incorporated into the projection of future wind mortality by modifying Equation 4-3. Fatality reduction, γ, occurred only at the proportion of installed capacity adopting in year t, represented as p_t in Equation 4-4:

$$a_t = \frac{ln\,[1-m_{2012}(1-\gamma p_t)]}{c_{2012}} \qquad Eq.\ 4\text{-}4$$

Mortality Uncertainty and Variability

Two forms of uncertainty were included in model projections. The first was variability. *Variability* is the year-to-year variation in environmental conditions that affect the vital rates of the population. As in Frick et al. (2017), variability was expressed through lognormal variation in the population growth rate, λ, where the distribution of $ln(\lambda)$ was normal with a standard deviation of 0.1.

Wind Mortality Variability

Unlike the 2017 study, where mortality was held constant at the 2014 level, wind mortality varied from year to year, drawn from an independent lognormal distribution with the underlying normal mean for each year computed from the natural log of Equation 4-2 and standard deviation set assuming a coefficient of variation of 0.1. Random deviates for wind mortality were generated carefully to avoid or minimize the need to truncate extreme values at 0 or 1. Following Akcakaya and Root (2013), random deviates with a mean of m were chosen when

mean mortality was less than 50%. In cases in which mean wind mortality exceeded 50%, a random deviate was drawn from a lognormal distribution with a mean of $1 - m$ (which is wind survival).

Wind Fatality Uncertainty

The second form of uncertainty in each model scenario was epistemic, reflecting the lack of precise knowledge about total fatality in 2012. As with variability in wind mortality, fatality uncertainty was not part of the 2017 study. Total wind fatality in 2012 was drawn from the normal distribution corresponding to the mean and 95% confidence interval for 2012 fatalities given in Arnett and Baerwald (2013). As a result, 2012 fatality for hoary bats had a mean of 112,494 and a standard deviation of 19,358. This distribution should not be confused with that of fatality rates among individual wind farms, which is highly skewed (AWWI 2018). Rather, its symmetric variation reflects measurement error in the estimate of the mean sum of fatality across all sites per year. Every run of the simulation began by selecting one random deviate from this normal distribution to be used as the "true" total fatality. This value was then used to compute a_t for all years using Equation 4-4 and, in turn, the wind mortality means for all years. Variability was then computed as random deviates from these means.

The inclusion of probabilistic uncertainty in 2012 fatality is an example of second-order Monte Carlo simulation. Uncertainty in other parameters—including population growth rate and abundance—were simulated by exploring parameter space, a non-probabilistic approach to uncertainty analysis.

Section 5: Population-Level Risks over Time

When examining a large range of parameter space, it is convenient to reduce each scenario's outcome to a single measure, such as the risk of extinction in the final year of the simulation. However, it is also useful to look at a reduced set of parameter combinations to investigate how risk changes over time. Though less comprehensive, this type of analysis helps determine how uncertainties may affect the timing of impacts to the population and the degree to which such impacts can be delayed by minimization.

To assess how risk to the hoary bat population changes over time, dynamics were projected over a period corresponding to 2012 through 2050. Simulations examined a range of initial population sizes encompassing the minimum (1 million), median (2.25 million), and maximum (10 million) of the range of expert opinions on "most likely" abundance reported in Frick et al. (2017; see Figure 4-1). We refer to these as *low*, *median*, and *high abundance*. Unlike the 2017 study, which used these levels to represent population size in 2014, the current study used them as abundance at the start of 2012.

Population Growth and Variability

In all simulations, the baseline median population growth rate was assumed to be 1; that is, on average, population size was expected to remain unchanged over the simulation period in the absence of wind mortality. In reality, it is likely that there are ecological factors that regulate the size of the hoary bat population at or near an equilibrium. It can be imagined that the population is limited by habitat, prey, predators, disease, or territoriality. However, data on these aspects of hoary bat ecology are scant or absent. The population model therefore omitted any stabilizing or limiting mechanisms, except a ceiling at 10 times the initial abundance to prevent unrealistic population explosions (as in Frick et al. 2017). This approach produces conservative estimates of risk (Ginzburg et al. 1990).

Simulations included random annual variation in population growth rate, representing year-to-year differences in environmental conditions for hoary bat survival and reproduction across the United States and Canada. Each replicate used an estimate of fatality drawn at random from a normal distribution representing uncertainty in total 2012 fatality. These two sources of uncertainty caused individual simulation runs to follow unique trajectories and resulted in a distribution of final population sizes (for example, Figure 5-1). To generate robust distributions of future population size, 10,000 replicates were run for every scenario of population size, wind mortality, and fatality reduction.
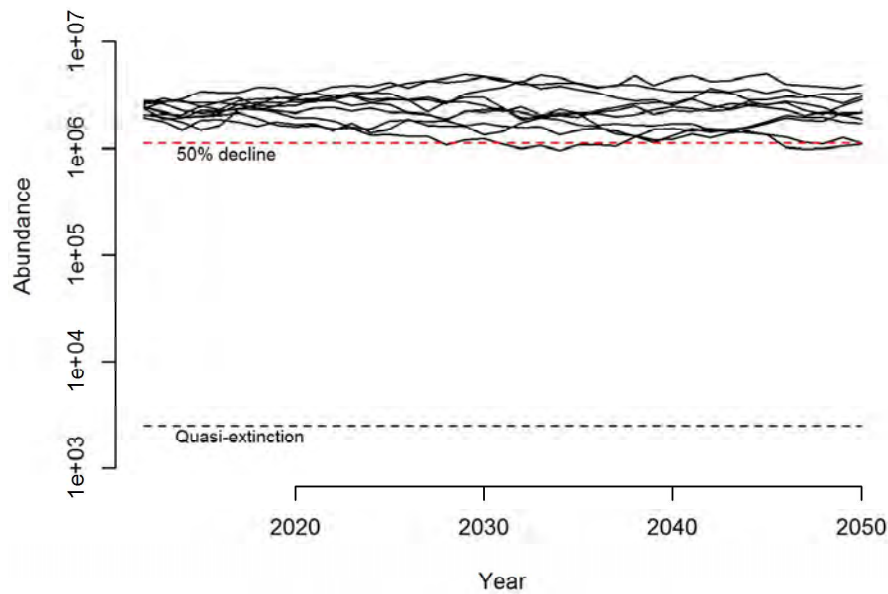
Figure 5-1
Ten sample trajectories of hoary bat population dynamics in the absence of wind mortality. Note log scale on y-axis.

Measuring Risk

Risk over time was summarized using two metrics. The first was *decline risk*, defined as the proportion of simulations that declined to 50% of their initial abundance. The second was *extinction risk*, defined as the proportion of simulations that fell below a quasi-extinction threshold of 2500 individuals in every year. These measures mirrored those used in Frick et al. (2017) but with a reduced time horizon. The 2017 study scored decline risk after 50 years and extinction risk after 100 years, whereas the current study focused on risk through 2050 and therefore had a time horizon of only 39 years.

All examination of the change in risk over time was performed using the high-buildout scenario of wind buildout and fatality reduction adoption outlined in Section 4.

Population Decline Risk over Time

Baseline Population Decline Risk Without Wind Mortality

The baseline model was constructed specifically so that mean abundance did not change over time in the absence of wind-related mortality. However, stochastic environmental variation of growth rate caused by the variance in abundance increased linearly with time. Therefore, selecting too high a variability could lead to a high baseline risk of decline or extinction.

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American Wind Wildlife Institute (2020)
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In benchmarking simulations without wind-related mortality, no model trajectories from any of the three initial abundances went extinct, and relatively few replicates showed appreciable decline. The 10 sample trajectories in Figure 5-1 illustrate the small variability in the forecast abundance under baseline conditions as well as the absence of a mean population trend.

Approximately 13% of the 10,000 replicates declined by 50% or more by 2050, regardless of initial abundance (see Figure 5-2). This baseline information is useful for the interpretation of decline risk in impacted scenarios because it emphasizes that some level of risk exists even in the absence of wind energy development.
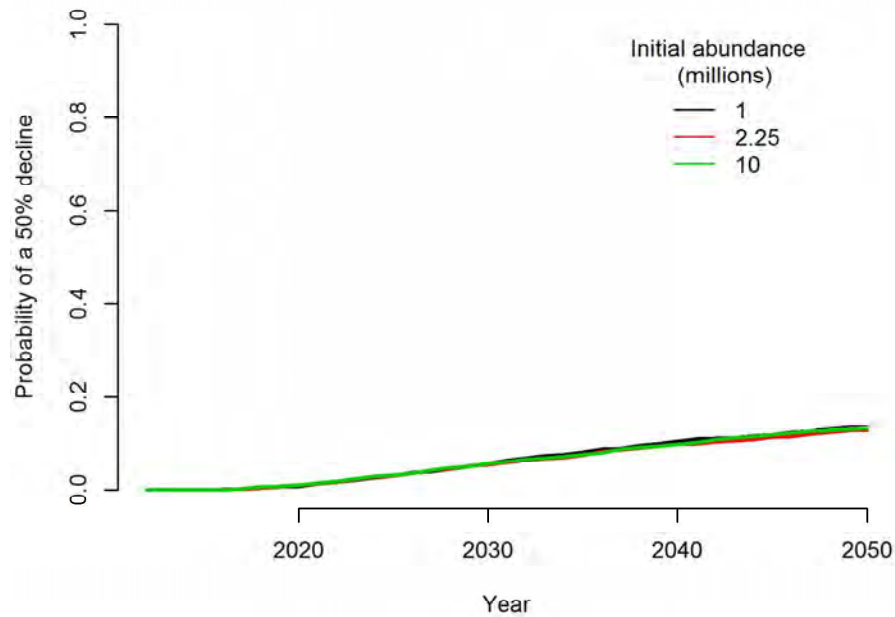
Figure 5-2
Hoary bat population decline risk through 2050 in the absence of wind turbine fatality

Population Decline Risk with Full Wind Mortality

With the addition of wind mortality computed using the high-buildout scenario, simulated population trajectories declined dramatically. Figure 5-3 illustrates that many replicates went extinct or were approaching extinction by 2050. Given the log scale of the abundance axis in Figure 5-3, a constant rate of population decline would appear as a straight line with negative slope. Instead, the trajectories curve downward, indicating an accelerating rate of decline that reflects increasing mortality with continued buildout.
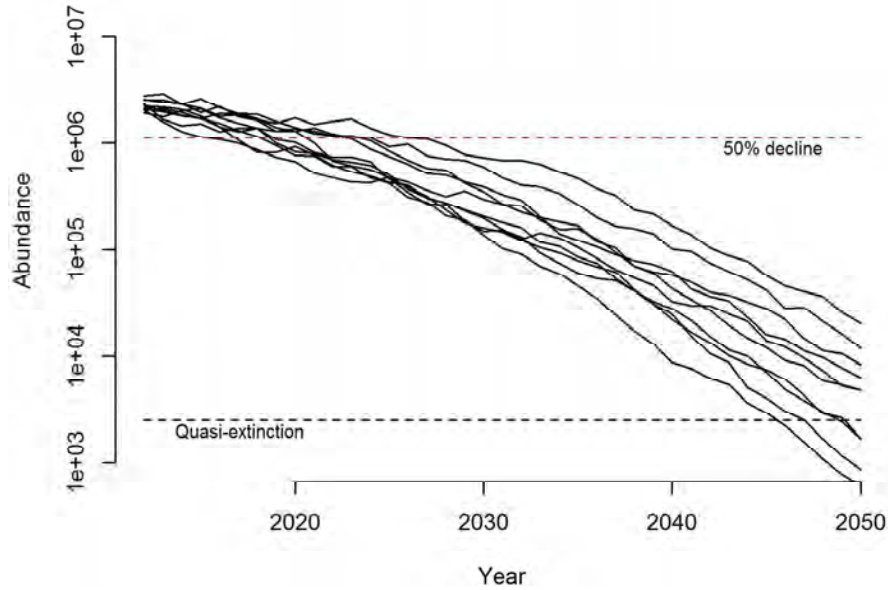
Figure 5-3
Ten sample trajectories of a median-sized population without any reduction in wind mortality. Note log scale on y-axis.

The probability of a 50% decline increased rapidly over time at the low and median abundance levels. Decline risk first exceeded 99% in 2021 at low abundance and 2030 at median abundance. At high abundance, which started with 10 million hoary bats, 87% of simulations projected a decline of 50% or more by 2050 (see Figure 5-4A), an increase of 74 percentage points over the baseline scenario without wind mortality. The median year of decline occurred in the first decade of the simulation for low and median abundance levels. The median year of decline at high initial abundance was 2036 (see Table 5-1).

Figure 5-4
Decline risk by 2050 for three initial hoary bat population sizes with (A) full mortality projected under the high-buildout scenario and (B) 50% fatality reduction

Table 5-1
Median year of decline or extinction with or without a 50% reduction in wind fatality

	Median Year			
	50% Population Decline		Extinction	
Initial Abundance	No Minimization	50% Fatality Reduction	No Minimization	50% Fatality Reduction
1 million	2016	2016	2033	2041
2.25 million	2021	2021	>2050	>2050
10 million	2036	2045	>2050	>2050

Population Decline Risk with 50% Fatality Reduction

Introducing a 50% reduction in fatality to the high-buildout scenario only reduced decline risk if the population was initially large enough (see Figure 5-4B). Trajectories all declined, but the rate of decline was noticeably slower than under full wind mortality (Figure 5-5).
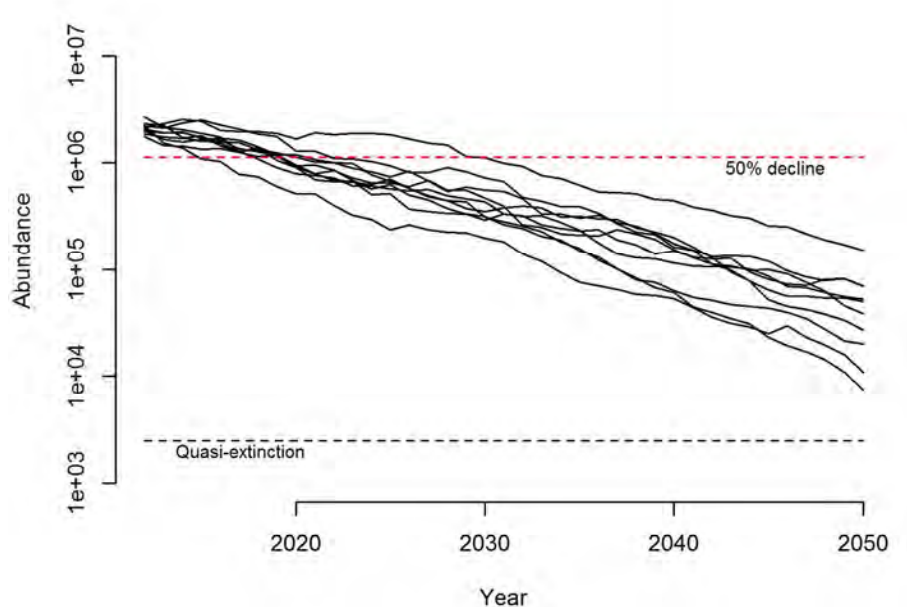
Figure 5-5
Ten sample trajectories of median-sized populations with 50% wind fatality reduction. Note log scale on y-axis.

Fatality reductions did not begin until 2020, according to the adoption curve in Figure 4-4, which was the ninth year of the simulation. This was too late to affect the median year of a 50% decline at the low and median abundance levels. However, at the high abundance level, the median year of decline was delayed by nine years (Table 5-1). At low and median initial abundance, the probability of decline first exceeded 99% in 2022 and 2038, respectively. At high abundance, decline risk in 2050 was 58%, a 16 percentage point decrease in risk compared with full wind fatality but still 45 percentage points higher than baseline decline risk in the absence of wind farms.

Extinction Risk over Time

Extinction Risk with Full Wind Mortality

In the absence of wind mortality, the model projected no risk of extinction by 2050. Introducing wind mortality using the high-buildout scenario, extinction risk was present only at the low and median abundance levels (see Figure 5-6A). Only the low abundance level had a median year of extinction prior to 2050 (Table 5-1), and its probability of extinction first exceeded 99% in 2043. At median initial abundance, 44% of replicate simulations went extinct by 2050.

Extinction Risk with 50% Fatality Reduction

With 50% fatality reduction, only the low abundance level displayed a risk of extinction by 2050 (see Figure 5-6B). The median year of extinction was delayed eight years to 2041 (Table 5-1). The probability of extinction by 2050 was 94%. Use of a longer time horizon would reveal that extinction risks are delayed rather than eliminated because the declining trend illustrated in Figure 5-5 would continue beyond 2050.
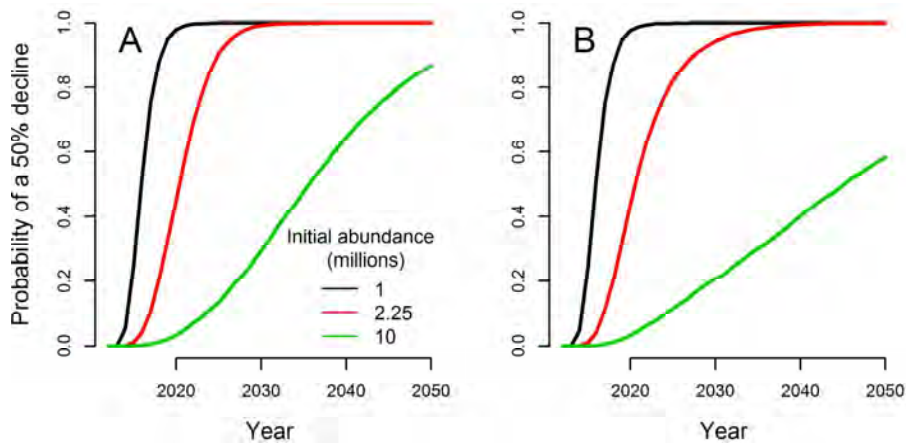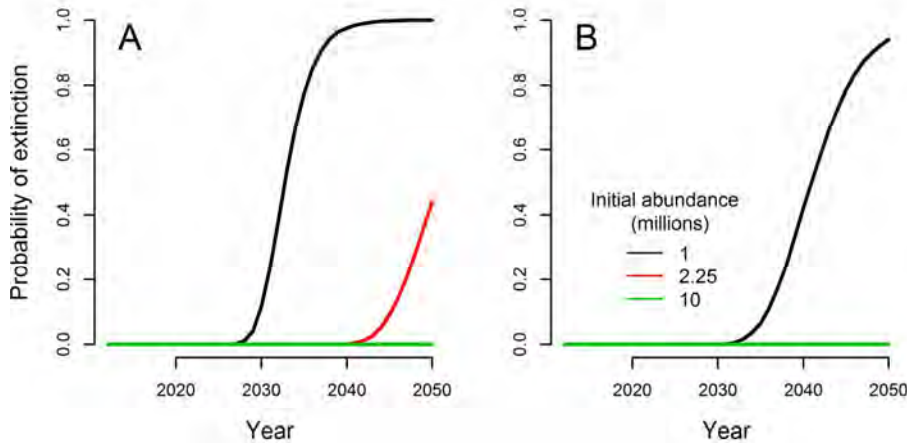

Figure 5-6
Extinction risk over time for three initial hoary bat population sizes with (A) full wind mortality under the high-buildout scenario and (B) with 50% fatality reduction

Section 6: Minimization Targets to Manage Population-Level Risk

Regardless of the strategy or technology used to reduce bat fatalities at wind farms, it is necessary to evaluate whether such reduction will be sufficient to manage population-level risks. The hoary bat population model was used to assess the level of fatality reduction needed to minimize the risk of population decline or extinction across a range of possible population sizes and growth rates.

The lack of knowledge about population status was addressed by simulating three initial mean population trends in the absence of wind farms: decreasing at 5% per year, stable, or increasing at 5% per year. Each trend was examined in combination with wind turbine fatality across initial population sizes ranging from 1 million to 10 million in increments of 0.25 million.

Within this parameter space, we examined how the risk of extinction or decline responded to reductions in wind turbine fatality. To find the fatality reduction necessary to hold the probability of extinction by 2050 under 1%, the proportion of fatality reduction was raised by increments of 0.01 from 0 up to 1 for each combination of population trend and size until fewer than 100 in 10,000 replicate simulations fell below the quasi-extinction threshold of 2,500 bats. Minimization targets for managing population decline were characterized by finding the reduction level at which the probability of a 50% decline fell below 50% for each combination of population trend and size.

Population Decline Risk

Management of decline risk required more fatality reduction than did extinction risk. For populations assumed to be declining or stable in the absence of wind farms and initially containing fewer than 3.5 million individuals, the impact of wind turbine fatality from 2012 through 2019 was such that no level of minimization beginning in 2020 prevented a 50% decline in the majority of simulations (see Table 6-1). This was true whether future buildout was high (Figure 6-1) or low (Figure 6-2) because historical buildout did not differ between the two scenarios.

Table 6-1
Wind turbine fatality reduction targets to manage the risk of hoary bat decline or extinction in the United States and Canada for a range of population sizes and trends

Risk	Abundance (Millions)	Population Trend					
		High Buildout			Low Buildout		
		Down[a]	Stable	Up[b]	Down[a]	Stable	Up[b]
Decline	1	-	-	0.97	-	-	0.93
Decline	2.25	-	-	0.71	-	-	0.47
Decline	3.5	-	0.98	0.45	-	0.94	0.02
Decline	10	-	0.61	0	-	0.29	0
Extinction	1	1.00	0.88	0.77	0.98	0.76	0.56
Extinction	2.25	0.64	0.38	0.14	0.36	0	0
Extinction	3.5	0.26	0	0	0	0	0
Extinction	10	0	0	0	0	0	0

[a] Population declining by 5% per year in the absence of wind turbine fatalities.
[b] Population increasing by 5% per year in the absence of wind turbine fatalities.
- indicates that decline or extinction cannot be managed by turbine fatality reduction.
0 indicates fatality reduction target <0.01.
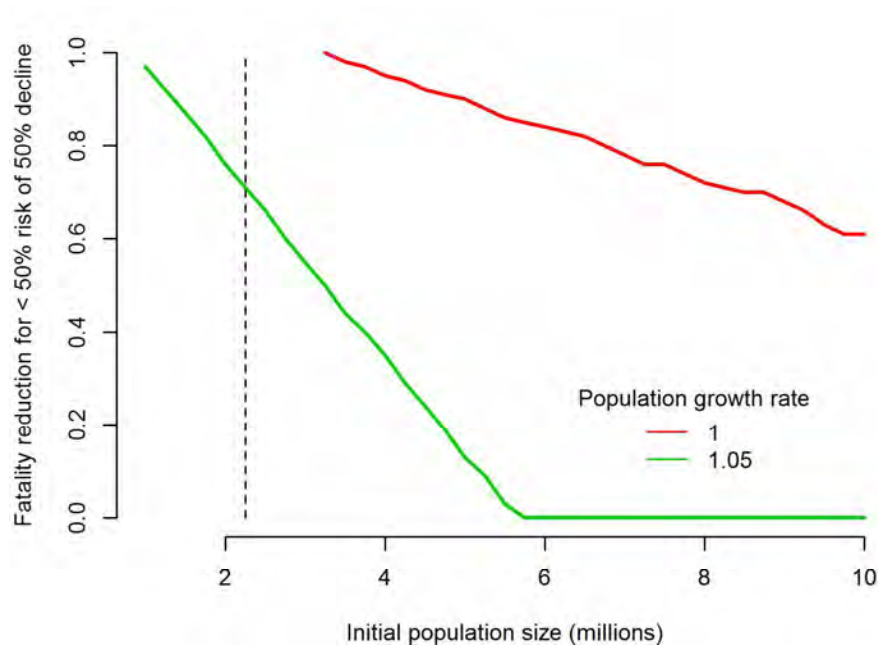
Figure 6-1
Fatality reduction necessary to hold the probability of a 50% decline below 50% across a range of initial population sizes. Vertical dashed line marks the median expert opinion for hoary bat abundance.

◄ 6-2 ►
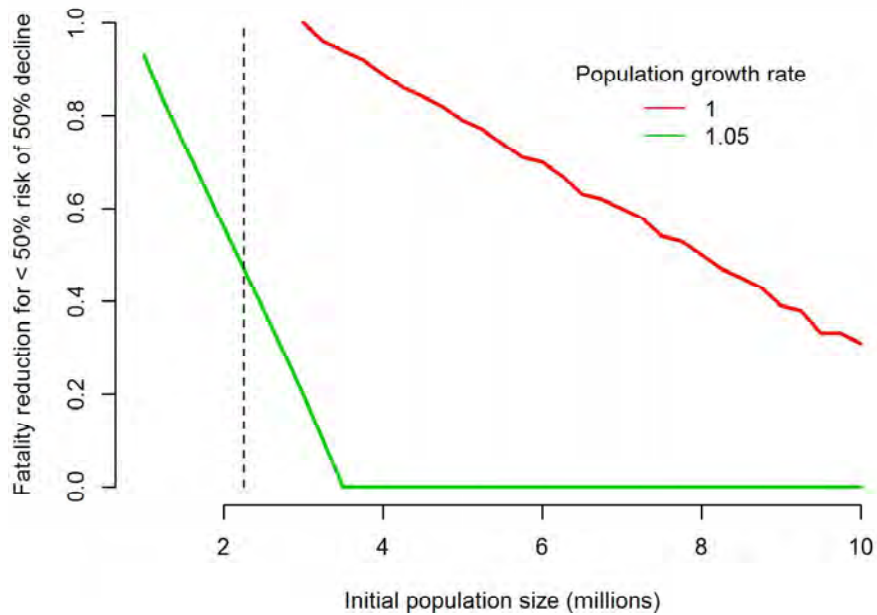
Figure 6-2
Fatality reduction necessary to manage decline risk under the low buildout scenario

Decline risk was manageable in stable populations of 3.5 million or more hoary bats. At the low end of this range, fatality reduction targets were similar for high and low buildout, both near 100%. In a population of 10 million, holding decline risk below 50% required more than 60% fatality reduction in the high-buildout scenario. The low-buildout scenario, which had about half as much installed capacity in 2050, required about half as much fatality reduction for a population of 10 million (Table 6-1).

Decline risk in increasing populations was manageable at all population sizes investigated. Nearly 100% fatality reduction was necessary to prevent the majority of simulations from declining by 50% if population size was only 1 million. However, no minimization was required in populations above 6 million in the high-buildout scenario (Figure 6-1) and 4 million in the low-buildout scenario (Figure 6-2).

Extinction Risk

To manage extinction risk, fatality reduction was necessary only in populations smaller than about 4.5 million individuals. Even at this level, extinction risk existed only in populations that were declining for reasons other than wind turbine fatality amid a high rate of capacity additions (see Figure 6-3 and Figure 6-4).
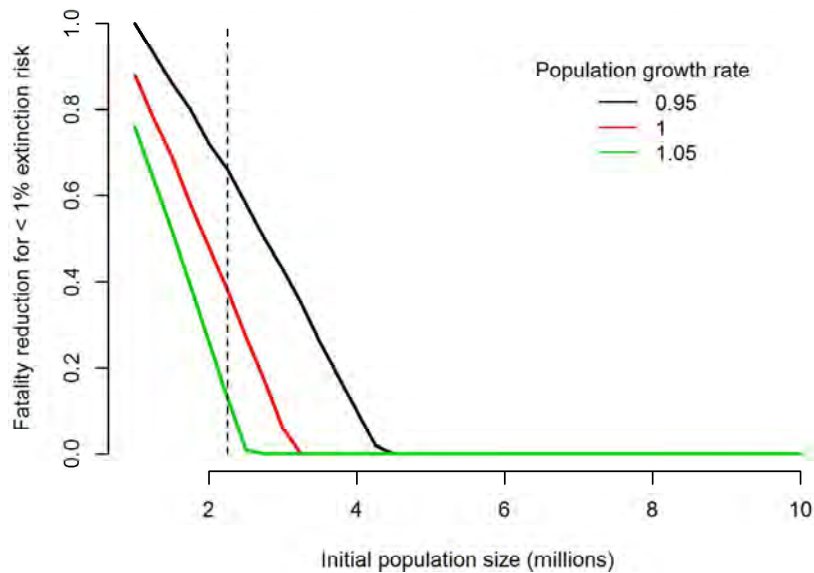
Figure 6-3
Fatality reduction necessary to manage extinction risk across a range of initial hoary bat population sizes under high buildout. Vertical dashed line marks the median expert opinion for hoary bat abundance.
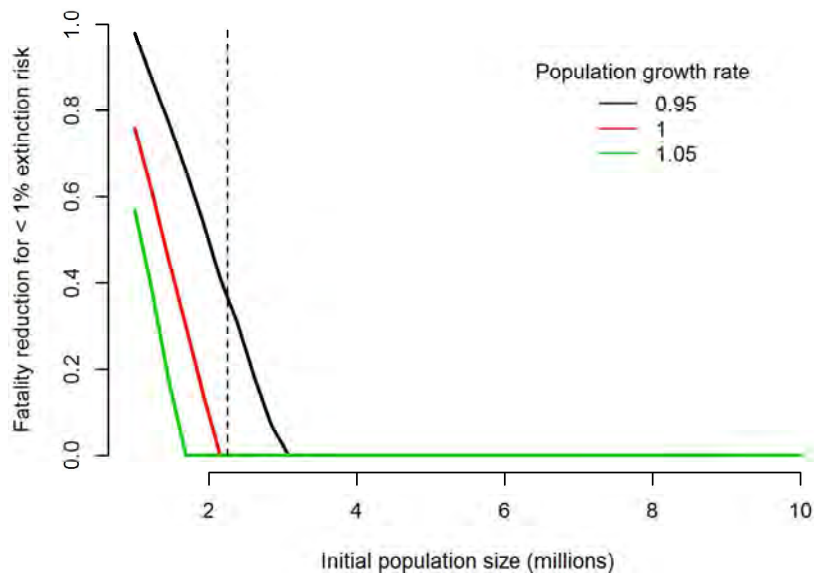
Figure 6-4
Fatality reduction necessary to manage extinction risk by 2050 under the low-buildout scenario

Extinction risk—and the fatality reduction required to manage it—depended critically on population size, growth rate, and buildout. At the median expert opinion level of population size, 2.25 million, a 38% fatality reduction was needed to manage extinction risk in the high-buildout scenario. That minimization target doubled and halved in the declining and increasing scenarios, respectively, while no risk of extinction existed in the low-buildout scenario except in a declining population (Table 6-1).

Section 7: Discussion

Several strategies appear to have potential for reducing bat fatality rates at wind farms. Broadly, these strategies fall into two approaches. The first, operational curtailment, feathers turbine blades under conditions of high fatality risk. Blanket operational curtailment, in which turbine blades are feathered below a set wind velocity (the cut-in speed), has reduced bat fatalities in experimental applications (e.g., Arnett et al. 2011; Good et al. 2018; Schirmacher et al. 2018). The amount by which fatalities are reduced generally increases with the cut-in speed used (Arnett et al. 2013b), though not all species are affected equally. The precise benefit of curtailment to hoary bats is difficult to ascertain from published fatality monitoring reports because fatality estimates are not broken out by species. However, the change in number of carcasses found under different curtailment conditions, though lacking important adjustments for effort and detection probability, may be an approximate indicator. For instance, a 3.5 m/s cut-in speed reduced total estimated bat fatality by 36% in a 2011 study at the Fowler Ridge wind farm (Good et al. 2012). The total number of carcasses decreased by 37%, but this decrease was mainly in the number of eastern red bats. The number of hoary bat carcasses found did not decrease. Cut-in speeds of 4.5 and 5.5 m/s decreased estimated total bat fatality by 59% and 75%, respectively, and reduced the total number of carcasses found by 60% and 76%, respectively. The number of hoary bat carcasses discovered declined by 60% at both cut-in speeds. An 85% reduction in carcasses was observed at cut-in speeds of 5–6.5 m/s in separate studies at the Fowler Ridge (Good et al. 2013) and Pinnacle (Hein et al. 2014) wind farms.

Further optimizations to operational curtailment, known collectively as *smart curtailment*, aim to reduce the production losses of curtailment by using sensors or statistical models to minimize production losses due to curtailment while maximizing fatality reduction. The turbine-integrated mortality reduction (TIMR) system for sensor-based automated curtailment is an example of an active smart curtailment system. TIMR showed an 81% reduction in hoary bat carcasses in an initial test (Hayes et al. 2019). Smart curtailment also includes the more static approach of using predictive models to limit curtailment to periods of high risk, making higher cut-in speeds more economically viable. Preliminary statistical models predicting total bat fatality based on temperature and wind speed data suggest that moderate fatality reduction (~50%) could be accomplished with minimal energy production loss (Peterson 2018). Such statistical approaches can be tuned to specific species if sufficient fatality data are available.

Preliminary results of recent meta-analysis of curtailment studies found a 70% reduction in hoary bat fatality rate on average (Whitby and Schirmacher, *unpublished*).

The second category of strategies for reducing bat fatalities involves the use of deterrents. The appeal of deterrents is that they decouple bat conservation efforts from the production of energy at wind facilities. These devices typically use ultrasonic noise to confuse or repel echolocating bats. Tests of deterrents have deployed a variety of technologies as research and development continues, yielding varied performance. For instance, a prototype ultrasonic deterrent system produced a 20–50% reduction in hoary bat fatality (Arnett et al. 2013a). Research on the effect of the technology used and the placement of deterrents may improve their performance. A more recent deterrent test in Texas (Weaver 2019) observed a 78% (95% confidence interval [CI]: 62–95%) reduction in hoary bat fatality, on par with the performance of smart curtailment.

How Much Fatality Reduction Is Needed to Manage Risk?

The current state of knowledge about the hoary bat population is insufficient to provide concise guidance on minimization targets. We note that our results encompass nearly all possible answers with respect to extinction risk. If the hoary bat population is 10 million (or larger), no fatality reduction is required to avoid extinction by 2050. If the population is 1 million (or smaller), preventing extinction would require an 80–100% reduction in fatalities.

Knowledge gaps are not the only challenge to identifying a minimization target. Even if an estimate of hoary bat population size becomes available, it will have error around it—often expressed in terms of a confidence interval. The range of minimization targets generated in our model showed strong dependence on population, as indicated by the steepness of the curves in Figure 6-3 and Figure 6-4. That steepness has both positive and negative implications for hoary bat conservation. The positive implication is that risk declines steeply to zero at population sizes well short of the maximum likely abundance. The negative implication is that error in any eventual population size estimate could have a large effect on the conservation efficacy of a given degree of minimization.

Reframing the Question

The amount of fatality reduction required to manage population-level risk will depend on the hoary bat population size and population growth rate—neither of which is known. To find out how much fatality reduction is required, one must choose a point from the range of population sizes indicated in Figure 6-3 and Figure 6-4. At 2.25 million hoary bats, for instance (the median of expert opinions on population size; Frick et al. 2017), an initially stable population would require 40% fatality reduction to manage extinction risk. This guidance must be interpreted considering the large uncertainties left unaddressed by the model, such as the actual population growth rate. Moreover, there is little to guide one in the selection of a particular population size. There is no evidence supporting the median abundance over other values.

◄ 7-2 ►

In its current form, the model answers several related questions.

Q: Will hoary bats go extinct by 2050?

A: The model suggests a risk of extinction by 2050 if the population had fewer than 4.5 million individuals in 2012. However, the projected timing of events in the model should be interpreted with caution.

Statements about absolute timing in this report are not precise but are also not purely relative. It is significant that population trajectories (even those buffered by compensatory dynamics; see Appendix A) declined to the quasi-extinction threshold over 39 years of capacity buildout in some scenarios. We see this result as evidence that such a risk exists.

In general, the probability of decline or extinction increases in small populations. Our model provided a conservative estimate of 4.5 million as the threshold 2012 abundance below which the population was at risk of extinction (Figure 7-1). This threshold is conservative in that it reflects risk in a population that was already declining by 5% per year in the absence of wind turbine fatalities. Though conservative, this result should still be treated with caution because the model is a generic description of population growth and lacks detail specific to hoary bat biology. It is also important to note that significant declines occurred at all initial abundances and would lead to extinction over longer time horizons. In a less conservative assessment (see Appendix A), extinction occurred prior by 2050 only in populations under 2 million hoary bats, though that model also projected that a population of about 4 million would be necessary to avoid eventual quasi-extinction (Appendix A, Figure A-3B).
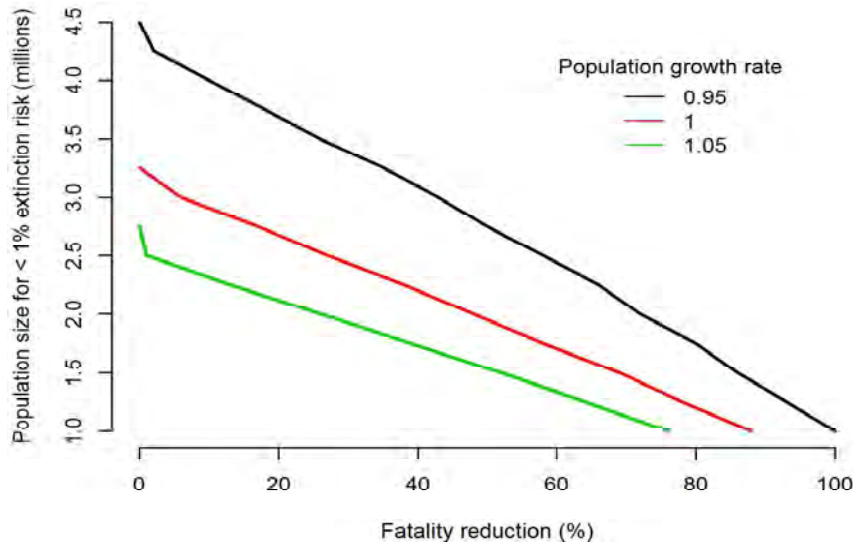
Figure 7-1
The population size at which a given level of fatality reduction would be sufficient to manage hoary bat extinction risk in the high-buildout scenario. Curves represent decreasing (0.95), stable (1), and increasing (1.05) initial population trends.

Q: Can fatality reduction prevent hoary bat extinction by 2050?

A: Minimization studies have demonstrated the technical feasibility of fatality reduction sufficient to prevent extinction over most of the range of possible hoary bat abundance.

A practical way of considering the model results is to tip Figure 6-3 on its side, making fatality reduction the x-axis (see Figure 7-1). This representation emphasizes the model's ability to act as a guide for maintaining operational relevance of empirical research on hoary bat knowledge gaps. Figure 7-1 provides the threshold population size for risk management given a particular minimization level (which could be selected based on technical and economic feasibility). Presented with a minimization strategy that can reduce fatalities by 70%, for instance, Figure 7-1 shows that the population would need to comprise at least about 2.25 million hoary bats if there is concern that factors other than wind mortality are causing a decline in abundance. Identifying this threshold instantly gives empirical researchers greater statistical power. The threshold is a null hypothesis against which to test the single-tailed question of whether the hoary bat population is large enough to persist.

Taking a conservative approach to setting population size thresholds for extinction risk management, we can relate existing curtailment and deterrent studies to thresholds for a declining population under high buildout (Figure 7-1, black curve). Operational minimization studies using various cut-in speeds or a smart curtailment system, such as TIMR, suggest the ability to manage extinction risk for population sizes down to 1.75 million hoary bats (the lower extent of the "Curtailment" bar in Figure 7-2 at 5–6.5 m/s cut-in). Existing studies of deterrents suggest that they could be sufficient to manage extinction risk in populations of at least 1.9–3.75 million hoary bats. Both strategies require additional research to establish their reliability not only over time, but also over geographic regions, local environments, and turbine models.
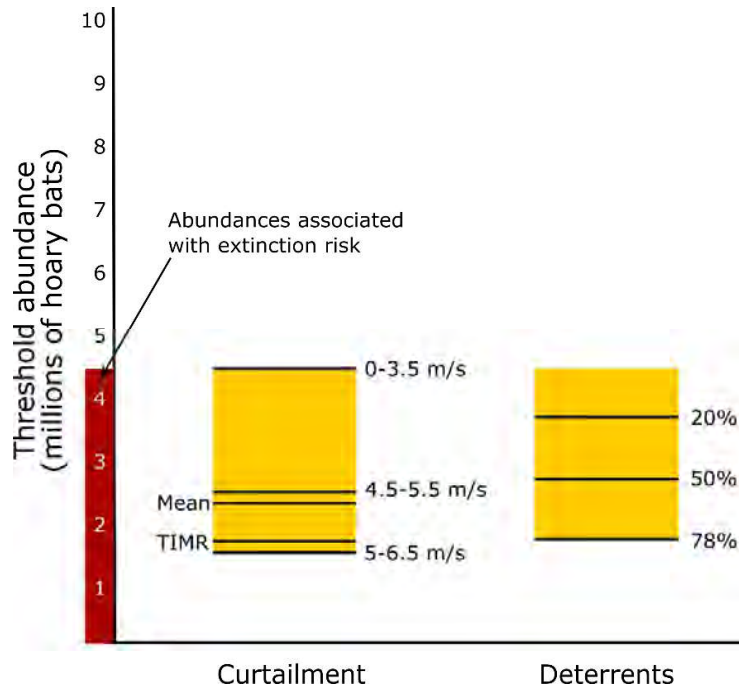
Figure 7-2
Threshold hoary bat population sizes to minimize extinction risk by 2050 over the range of hoary bat fatality reductions that curtailment and deterrent studies suggest are possible.

Population sizes that exhibit extinction risk are highlighted in red. Horizontal lines and labels refer to specific minimization results detailed in the text. "Mean" indicates the average 70% reduction in hoary bat fatality found in a preliminary meta-analysis by BCI (Whitby and Schirmacher, unpublished).

Our study did not address the effects of the rate of adoption or other factors that might change the pace at which fatalities are reduced. Intuitively, this pace could impact how much curtailment is necessary in the long term to manage population-level risks. If the adoption rate could be improved, for instance, through a two-phase plan in which wind farms initially meet a lower minimization target, the faster delivery of fatality reduction could offset its lower magnitude.

Q: What is (or was) the critical population size for hoary bats?

A: We suggest that two threshold population sizes are of interest. Rounding to the nearest million, the first threshold is 5 million, below which fatality reduction is necessary to manage extinction risk by 2050. The second is 2 million, at which point the fatality reduction required to avoid extinction surpasses the maximum fatality reduction documented so far.

Alternative thresholds could be supported by referencing decline risk rather than extinction. Focusing on the stable population case—below 6 million bats—no documented fatality reduction would be sufficient to prevent decline. Below an

initial abundance of 3 million, a 50% decline is likely to have already occurred. A model with compensatory dynamics projected similar declines for small populations (Appendix A, Figure A-2A). If the hoary bat population has the capacity to compensate for wind-related mortality, trends could be reversed by fatality reduction. As a result, decline risk could be managed with a fatality reduction of about 85% (Appendix A, Figure A-3A), only slightly higher than the efficacy observed in the first test of the TIMR responsive curtailment system (Hayes et al. 2019).

Importantly, these thresholds refer to the size of the population in 2012. Across much of the parameter space we investigated, the population would be smaller now, sometimes by a margin of 50%. Therefore, the choice of a current-day threshold involves consideration of factors such as population trend and environmental variability that introduce additional uncertainty in our results.

Q: What did this model tell us that we didn't already know?

A: An important and robust result of our study is that significant declines may already have occurred if the population was initially small. Both the stable density-independent model and a model with compensatory dynamics showed median years of decline prior to 2020 for populations of 1 million and prior to 2025 for populations of 2.25 million (Appendix A, Figure A-2).

The basic lessons of our study are the same as those found in Frick et al. (2017). However, our focus on historical and projected buildout and delayed adoption of minimization measures puts into sharper focus that wind energy development may have already affected the hoary bat population in ways that make the need for future protection more likely.

Q: Did any optimistic results for bat populations come out of this study?

A: Yes. The most obvious optimistic result is that fatalities may be a small enough proportion of the hoary bat population for extinction risk to be low even under a high-buildout scenario. It is also optimistic to note that technically feasible fatality reductions are sufficient to manage extinction risk over much of the range of likely hoary bat abundance.

Research Priorities

Mathematical models help identify and prioritize knowledge gaps and structure the design of empirical research. The following is a list of empirical research efforts in order of the degree to which their results are likely to improve the precision with which population-level impacts of wind mortality on hoary bats can be assessed. This ordering considers both the importance of the information and the feasibility of obtaining conclusive results.

Spatial Structure

Although the hoary bat population is thought to be well-mixed over the United States and Canada (Cryan et al. 2014b; Russell et al. 2015; Sovic et al. 2016), there is strong spatial structure to wind-related fatalities. The rate of fatalities per MW varies regionally (AWWI 2018), as does the current and future distribution of wind farms. Our projection of future mortality assumed that buildout was similar in all regions. Accounting for regional variation in projected buildout would allow us greater insight into future mortality levels and therefore a clearer picture of the minimization required to manage hoary bat population risks.

This research question is listed first because it is the most feasible. Answering it will not address the main source of uncertainty in our assessment.

Abundance

Importance

A better understanding of hoary bat abundance would have the greatest impact on the precision with which population-level impacts of wind farm fatality can be assessed. In this respect, the most dramatic modeling results come from an examination of extinction risk. Decline risk was less sensitive to abundance. An initially stable population was at high risk of declining by more than 50% prior to 2050 regardless of abundance. The opportunity to mitigate declines existed only for larger population sizes, and even then the fatality reductions necessary were large (>60%).

Specific Need

Research should focus on developing evidence for a minimum plausible abundance. A central estimate of population size is not necessary to begin developing a minimization target for hoary bat fatality at wind farms. Extinction risk decreases with increasing population size. At any level of minimization that is practical given technological and economic constraints, there is a minimum population size for which that level is sufficient to meet risk management targets.

Focusing on obtaining a lower bound for abundance makes the problem "one-tailed," meaning that researchers need to rule out only lower population sizes. For example, if it appears that at least 75,000 hoary bats were killed by turbines in the United States and Canada in 2012 (the lower 95% confidence limit in Arnett and Baerwald 2013) and that fatalities have continued to occur since then, it can be surmised that there must be substantially more than 75,000 hoary bats.

Approaches

Estimating total hoary bat abundance in the United States and Canada is difficult. The species is widespread but at the same time so cryptic that there is no firm understanding of its population density in any one place or how that density varies over the landscape. In addition, the migratory behavior of the

population makes it difficult to extrapolate from point counts or acoustic detections to a larger spatial scale because population density is not geographically static through the year.

Efforts to better characterize hoary bat abundance have been dominated by genetic studies (Korstian et al. 2014; Russell et al. 2015; Pylant et al. 2016; Sovic et al. 2016). Several additional approaches, including the use of tagging (Weller et al. 2016), stable isotopes (Baerwald et al. 2014; Cryan et al. 2014b; Pylant et al. 2016), and acoustic monitoring, have added insight into movement and limits to or changes in distribution. The first record of a hoary bat in Alaska was recently made through acoustic detection (Blejwas et al. 2014). Projects to coordinate or aggregate acoustic monitoring data, such as BatAMP (batamp.databasin.org) and NABat (nabatmonitoring.org) may be of use in estimating population size or trends over time. The NABat program has developed a sophisticated sampling design and analytical framework to detect trends in species occupancy that may be used as proxies for trends in abundance (Rodhouse et al. 2015).

Existing genetic estimates of effective population size (a metric that commonly underestimates true abundance in long-established populations), do not yet establish a consensus. Korstian et al. (2015) found estimates of current effective population size with confidence intervals that included both zero and infinity but suggested that the nature of their results indicated very large population size. A surprisingly narrow confidence interval for effective population size, 730,000–880,000, was found in a study using a far larger number of loci from bats collected at a single wind farm (Sovic et al. 2015). In contrast to these two studies, an estimate based on 15 loci in samples from the central Appalachian Mountains found a much smaller effective population size of 2,400–90,000 individuals (Pylant et al. 2016). Stable isotope evidence from the same study suggested that this estimate reflected a regional rather than national or continental population. Future genetic studies could be improved by using large numbers of loci and sampling over a large geographic area. With sampling from across generations of bats, genetic studies can also be aimed at providing population trend information.

Demography

Importance

Demography typically refers to information on the survival and reproduction of a population. When combined, these components can be used to calculate population growth rate. A great deal of work in applied and theoretical population biology has gone into assessing demographic rates; how they change with age, environment, and population density; and how these changes affect population growth, stability, and persistence.

In our study, all the details of baseline demography were encapsulated in a single parameter: the population growth rate. We investigated a range of population growth rates from 5% decline to 5% increase. Although this range was smaller

than that of growth rates derived from expert opinions on hoary bat survival and fecundity or of empirical measurements of other bat species (Frick et al. 2017), it addresses baseline scenarios of both alarming decline and robust growth.

The relative value of researching demography (as opposed to population size) depended on the risk metric being used. Decline risk was more sensitive to baseline demography than to population size. However, this finding is not surprising. For instance, a population with a baseline declining trend would have a high risk of decline regardless of population size (or fatality reduction). When looking at extinction risk, population size seemed to be the more important factor to understand. Above a critical population size, the probability of extinction by 2050 was non-existent regardless of baseline demography. However, within the range of abundance over which extinction risk can be managed with documented levels of fatality reduction (Figure 7-1), demography made a large difference in the minimization required.

There are reasons to suspect that hoary bats would be declining even in the absence of turbine fatality. Large-scale changes in western forests over the past two decades—resulting from the combined action of insect pests and wildfires— may have reduced or degraded available habitat. Prey availability may also be a concern. A recent study estimated a global decline in insect biomass at a rate of 2.5% per year (Sánchez-Bayo and Wyckhuys 2019). If hoary bats are prey-limited, their rate of abundance could by impacted by a change in available insect biomass. It is important to note that the documented trend in insects is the net change over many species; some species are increasing. In addition, the response of the hoary bat population to a decrease in prey availability is likely to be nonlinear; that is, any corresponding decline could be larger than, similar to, or greater than that of its prey and could be delayed in time.

Specific Need

A better understanding of age-specific survival and fecundity might provide insight into the current, baseline, and potential growth rate of the hoary bat population. However, it has been noted that the hoary bat is possibly the most difficult terrestrial mammal to study and that the marking studies necessary to measure survival have largely been abandoned (Cryan et al. 2014b).

An estimate of trend alone is sufficient to assess the conservation status of hoary bats.

If measurements of demographic components cannot be obtained, substantial information can still be gleaned from a direct estimate of population trend. Importantly, our simulations suggested that populations at high risk would have declined substantially since 2012. Therefore, an estimate of trend alone is sufficient to assess the conservation status of hoary bats, though it would not necessarily indicate whether turbine fatalities drive the trend.

Approaches

Three approaches are viable for estimating a trend in hoary bat abundance. The first is to use a genetic analysis that incorporates samples from across multiple generations of bats. Though commonly thought of as a way to assess population

structure and size, genetic analysis that encompasses multiple generations can also give insight into trends in abundance by tracking the fate of rare alleles over time (Beaumont 2003). Hoary bats, with their purported lack of spatial structure, may be particularly amenable to genetic characterization of population trend (Chikhi et al. 2010).

The second approach is to analyze trends in occupancy using data in the NABat acoustic monitoring program (Rodhouse et al. 2012). United States Geological Survey (USGS) scientists have begun applying NABat monitoring data to hoary bat trend assessment using data from the northeastern United States (Rodhouse et al. 2019). Trends in other regions will likely become available as more data are added to the NABat repository.

Post-construction fatality data could be used to develop a trend in hoary bat abundance.

A third approach to estimating the trend in hoary bat abundance is to use data collected at wind farms. Post-construction fatality surveys have been conducted since the late 20th century. Changes and inconsistencies in monitoring practices could be addressed either through reanalysis of raw historical monitoring data using a modern fatality estimator such as GenEst (Dalthorp et al. 2018) or correction of reported fatality estimates using a statistical model (sensu Thompson et al. 2017). An ongoing study using fatality data at reference sites is using occupancy analysis to estimate the hoary bat population trend east of the Rocky Mountains since 2011 (Rabie 2019). Using post-construction monitoring data could deliver a longer time series.

Other Priorities

Behavior

Improved understanding of behaviors could be used to modify the existing population model and thereby affect risk assessments. Two behaviors in particular could affect model structure; the first is hoary bat attraction to wind turbines (Cryan et al. 2014a). If turbines are attractive at a sufficient distance, the number of fatalities could fall more slowly than the total size of the hoary bat population, violating the current model assumption that fatalities are a simple function of population size. Attraction will increase the risks of decline and extinction. Another important class of behaviors is territoriality related to reproduction, including male aggression during mating season and female defense of roosting territories while rearing young. Defensive interactions to maintain territories are energetically expensive and could carry risks such as increased predation or collision rates. Such behaviors are of interest because they would presumably intensify with increasing population density, potentially contributing to population regulation. If wind energy development reduced population density, such behaviors would abate and could lead to an improvement in survival or reproductive success. This improvement would in turn lessen further population-level impacts of turbine fatalities. Therefore, territorial behavior may reduce the risks of decline and extinction.

Sex-Specific Migration Patterns

Our model makes the simplifying assumption that fatality rates are equal for male and female hoary bats. Fatalities have historically been biased toward males, though this may reflect error in sex identification (Nelson et al. 2018). Stable isotope analysis suggests that males and females separate outside of the mating season (Cryan et al. 2014b), leading to potentially strong geographic differences in sex ratio during migration, when most fatalities occur. Theoretically, male fatality has less impact on population growth rate than that of females because males can mate multiple females per year. Therefore, fatality in regions with male-dominated migration could present lower risk to the continental population.

Although gathering more information on seasonal movements in general might help illuminate regions of high traffic during migration—and therefore the geographic distribution of risk at the time of year when fatalities are most common—such information may be difficult to obtain. Further stable isotope studies could give greater insight, but the geographic resolution of such analyses is low. Tracking of individuals by GPS would be informative but relies on capturing live individuals and is therefore difficult to scale up to a national study.

A relatively inexpensive approach might be to use carcasses to map seasonal patterns of sex ratios across the country. Combined with improved sex identification, carcasses from post-construction monitoring and other sources could help to map seasonal patterns of sex ratios across the country and in turn inform hypotheses for net migratory routes. Understanding sex-specific distributions and migration routes would allow us to revise risk projections considering state or regional forecasts for wind energy development.

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Appendix A: Density-Dependent Assessment

The main body of our risk assessments employs a density-independent model of population growth. Though justified by the lack of evidence for more complex dynamics, such a model is also conservative with respect to risk. Particularly in scenarios in which we assume a mean population growth rate of 1 or less in the absence of wind farm fatalities, the density-independent model makes the strong assertion that any source of additional fatality will cause decline and eventual extinction—though this sensitivity is buffered by the short time horizon of the assessment.

There is a strong argument from first principles that most or all populations do not grow in a purely density-independent way: populations do not become infinitely large. Therefore, something must limit their growth as they become more abundant. The corollary, in typical ecological models, is that populations are **regulated** near an equilibrium abundance by mechanisms that increase their growth rate when rare and decrease it when overpopulated. This phenomenon gives rise to compensatory dynamics so that the impact of a new source of mortality is muted by a counteracting increase in population growth rate. In turn, compensation reduces the risks introduced by factors such as wind-related fatalities of hoary bats.

The Evidence for Density Dependence

Broad empirical evidence for density dependence exists, but it is not universal. A statement such as "The vast majority of populations exhibit stabilizing mechanisms" is not supported from data. Moreover, if there were few species without the capacity for compensatory dynamics, they would likely be the ones in decline. Therefore, the assumption of density dependence is not a good starting point for risk analyses of poorly understood populations.

Analyses of a large database of ecological time series have found evidence of density dependence in about 40–60% of data sets (Brook and Bradshaw 2006; Knape and de Valpine 2012a). However, density independence is often supported both for species of conservation concern (e.g., grizzly bears, Brook and Bradshaw 2006) and for healthy populations (e.g. red kangaroos, Knape and de Valpine 2012b). In these analyses, our central model (no density dependence, no trend) is considered the null hypothesis for how populations change over time. It produces time series that, over the typical length of most data sets, look very similar to

density-dependent data. This fact leads to two difficult statistical problems. First, naïve assessments (such as used in Brook and Bradshaw's analysis) have a false positive rate of 50%; that is, they dramatically over-diagnose density dependence. Second, more sophisticated assessments have a false negative rate of 80%; that is, they dramatically under-diagnose density dependence (Knape and de Valpine 2012a). The conclusion is that, although density dependence is supported from first-principles reasoning, its role in regulating populations in the real world is generally humble enough to be difficult to assess.

Do Regulated Populations Decline?

There is a widespread notion that the equilibrium abundance of a regulated population is insensitive to factors that might reduce maximum population growth rate. But this conclusion is the result of confusion caused by semantics (Ginzburg 1992; Mallet 2012). Perhaps the most familiar form of density-dependent model is logistic growth. The instantaneous population growth rate is commonly written in the notation

$$\frac{dN}{Ndt} = r\left(1 - \frac{N}{K}\right)$$
Eq. A-1

where r is the maximum growth rate and K is the carrying capacity. It is this second parameter that causes confusion. No matter the value of r, the population ceases to change size when $N = K$. The notion of carrying capacity has given K the interpretation of being a characteristic of the environment indifferent to the plight of species. As such, sources of additional mortality are often assumed to reduce r only and leave K intact. Under this illusion, wind farm fatalities (or any other stress on a population) might decrease the rate of recovery from an impactful event, but the population will always strive toward its original level.

In fact, K is itself a function, not a parameter. It is no doubt a function of the environment, but also of demographic rates. When MacArthur introduced K (1962), it was to show that selection can act on equilibrium abundance—explicitly asserting that the ability to survive or reproduce should affect K. In the original formulation of the logistic equation (Verhulst 1845), growth was the result of two processes. In population growth terms:

$$\frac{dN}{Ndt} = r - cN$$
Eq. A-2

where c is the rate at which growth decreases with population density or size. If rhetoric is helpful, c can be thought of as a friction term, or per capita rate of intraspecific interference or competition, or the rate at which access to resources, shelter, and so on is diluted by population size. In Verhulst's formula, c is independent of r. Comparing Equations A-1 and A-2, carrying capacity is not independent of r. Rather,

$$K = \frac{r}{c}$$
Eq. A-3

For the equilibrium abundance to be unaffected by elevated mortality, there would have to be a perfectly compensating change in *c*.

There are situations in which Equation A-1 is a respectable representation of the world. If there is, say, a fixed number of nest holes of equal quality that determines *K* and the population will always fill them all as long as it has the capacity to grow, it may well be a good approximation to assume that equilibrium abundance is impervious to elevated mortality. This is true for territorial animals, such as golden eagles, up to a point (Monzón and Friedenberg 2018).

However, Equation A-2 is generally more parsimonious. Only two demographic rates determine changes in abundance in a closed population: births and deaths. An equilibrium population size exists if and only if births exceed deaths at low abundance and deaths exceed births at high abundance. Equilibrium is achieved at the abundance where the two rates are equal, such as represented by the closed circle in Figure A-1. Whether it is births or deaths that change with abundance, an increase in the death rate (Figure A-1, dotted line) will cause births and deaths to equilibrate at a lower population size (Figure A-1, open circle).
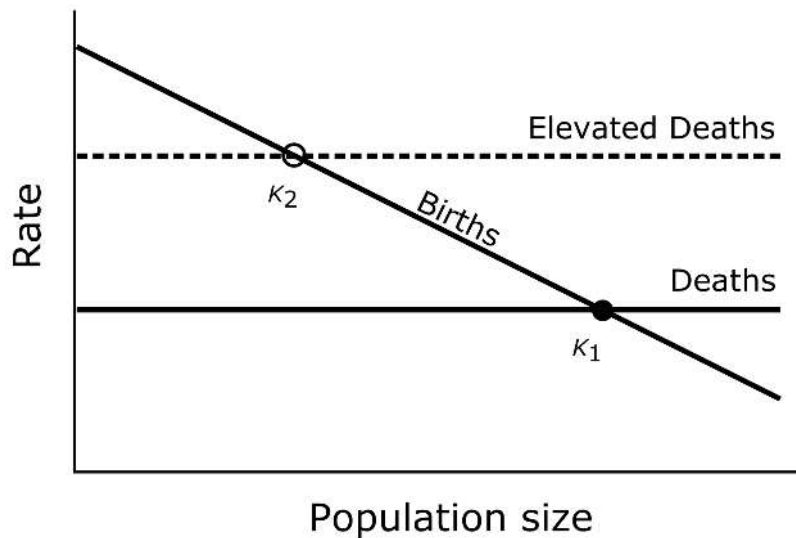

Figure A-1
A simple model of equilibrium with density-dependent birth and constant death rates. Under some initial level of death rate, the population will come to an equilibrium size that results in a compensating level of births, indicated by the closed circle at K_1. At larger population sizes, deaths will exceed births and the population will decline. At lower population sizes, births will exceed deaths and the population will increase. All else being equal, an elevation of the death rate results in a lower equilibrium density, such as that indicated by the open circle at K_2.

A Density-Dependent Model for Hoary Bats

For a species such as hoary bats—with unknown abundance, low perceived population density, and a near absence of controlled longitudinal monitoring data—it is difficult to assert any particular narrative for the mechanisms of population regulation. In service of exploring risk in the presence of compensatory dynamics, we developed a density-dependent model of hoary bat population growth that included sensitivity to sources of additional mortality. Our conceptual model consisted of a limited supply of high-quality host trees for rearing pups. We assumed this mechanism to induce contest competition affecting recruitment of young during the summer. We assumed that fatalities at wind farms occurred after reproduction but before the census. As a result, wind farm fatalities decrease census population size below the baseline carrying capacity. The mathematical form of the model employed Beverton-Holt density dependence with additional mortality:

$$\lambda_t = \lambda_{max} \left[1 + \frac{N_t}{K}(\lambda_{max} - 1) \right]^{-1} (1 - m_t) \qquad \text{Eq. A-4}$$

where λ_t is realized population growth rate at time t, λ_{max} is the maximum population growth rate, N is abundance, K is the carrying capacity in the absence of wind farms, and m is mortality associated with wind farms. In the absence of additional mortality, a stable equilibrium abundance, $N^* = K$, exists if the population can grow ($\lambda_{max} > 1$) and K is positive. If $\lambda_{max} = 1$, Equation A-4 becomes density independent.

As constructed, equilibrium abundance declines linearly as m_t increases, so that

$$N^* = K \left(1 - m \frac{\lambda_{max}}{\lambda_{max}-1} \right) \qquad \text{Eq. A-5}$$

for $\lambda_{max} > 1$. Reducing wind-related mortality increases population growth rate and equilibrium abundance. However, N^* remains below the original equilibrium as long as some wind farm mortality exists.

The risk of decline or extinction will be lower under density-dependent growth, as will the level of fatality reduction necessary to meet risk management goals. The extent of this difference will depend on parameterization, with higher values of λ_{max} providing more resilience.

A criticism of density-dependent assessment is that all populations impacted at any level are heading for extinction. The introduction of density dependence ameliorates this issue somewhat but suggests that greater importance should be given to the ultimate equilibrium abundance because the mechanisms stabilizing the population are being given significant credence. It can be seen that Equation A-5 implies an equilibrium abundance less than 0 for some combinations of mortality and maximum growth rate. The mortality level, m^*, that prevents equilibrium from falling below a threshold, Q, is

$$m^* \leq \frac{\lambda-1}{\lambda} \left(1 - \frac{Q}{K} \right) \qquad \text{Eq. A-6}$$

Finally, the threshold mortality level indicated by Equation A-6 can be related back to the target reduction in fatality rate, γ, by

$$\gamma \geq 1 - \frac{K}{F_{2012}}\left[1 - (1 - m^*)^{\frac{c_{2012}}{c_{2050}}}\right] \qquad \text{Eq. A-7}$$

where F_{2012} is the number of hoary bat fatalities in 2012 and the fraction in the exponent is the ratio of installed capacity in 2050 and 2012.

Methods

For consistency with the density-independent assessments, density-dependent abundance projections were executed with lognormal temporal variability in expected growth rate and wind mortality, so that in both sets of analyses,

$$\lambda_t = \lambda_{max}\left[1 + \frac{N_t}{K}(\lambda_{max} - 1)\right]^{-1}(1 - m_t\varepsilon_{1t})\varepsilon_{2t} \qquad \text{Eq. A-8}$$

where ε_{1t} and ε_{2t} are uncorrelated random lognormal deviates, each with mean 1 and a 10% coefficient of variation. The density-independent model used in the main analysis is equivalent to Equation A-8 with $\lambda_{max} = 1$.

Simulations addressed both density independence and density-dependent growth. The density-dependent analysis used the highest value of population growth rate found in the 2017 elicitation, 1.18, as the maximum population growth rate. We assumed that expert opinions on vital rates used to calculate growth rates were independent of their assessment of abundance. Further, we assumed that the mental model experts held of total abundance was indicative of initial carrying capacity, K. Initial abundance was set to K. Assessments of risk over time or the fatality reduction required to manage risk were performed using 10,000 replicates for each value of abundance and fatality reduction.

In addition, we found the target reduction in fatality rate necessary to hold equilibrium abundance above a 50% decline or the quasi-extinction threshold of 2,500 hoary bats in the long term using Equations A-6 and A-7. We used mean estimated fatalities in 2012, ignoring uncertainty, and the installed capacity projection of the high-buildout scenario. Parameter values and sources can be found in the main analysis.

Results

Decline and Extinction Risk over Time

As expected, density dependence reduced the risk of both a 50% decline and extinction compared with the density-independent model (Figure A-2). However, results were qualitatively similar at low initial abundance, where both models suggested that a significant decline would already have occurred (Figure A-2A) and therefore could not be remediated by adopting fatality reduction measures (Figure A-2B). Both models also projected a chance of extinction at baseline and reduced fatality rates (Figure A-2C, D). At median and high initial

abundance, fatality reduction delayed or eliminated decline and extinction risk under density dependence.
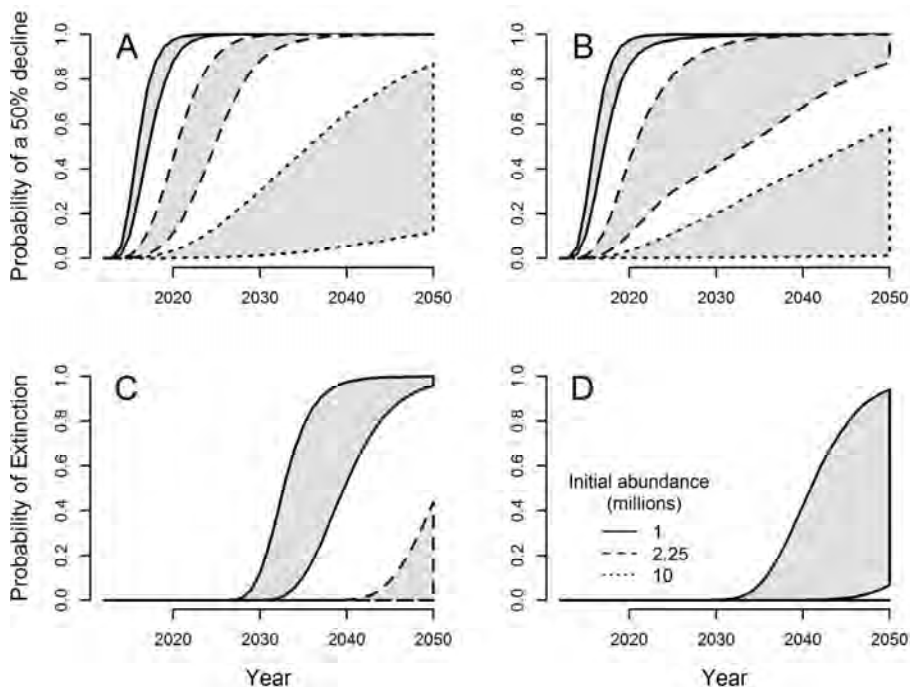

Figure A-2
*The probability of decline or extinction over time for hoary bats interacting with wind farms in the United States and Canada. Grey areas denote intervals of risk at three initial abundances. Each is bounded by the maximally sensitive model on the left and the more resilient model on the right. The intervals represent fundamental uncertainty about demographic processes. **A.** Decline risk without reduction of wind-related fatality rate. **B.** Decline risk with fatality rate reduced 50%. **C.** Extinction risk without reduction of fatality rate. **D.** Extinction risk with fatality rate reduced 50%.*

Fatality Reduction

The reduction in fatality rate (fatalities per MW) necessary to manage the risk of decline or extinction by 2050 was predictably lower under density dependence than in the density-independent model (Figure A-3). Decline risk was manageable at all initial abundances investigated, requiring at most a roughly 85% reduction in the fatality rate (Figure A-3A). The probability of extinction was less than 1% for initial abundances greater than or equal to the median of the expert elicitation (Figure A-3B). A 60% reduction in fatality rate was sufficient to manage the risk of decline by 2050 in populations with initial abundance of 3 million or more hoary bats. That level of fatality reduction also managed extinction risk for all initial abundances investigated.
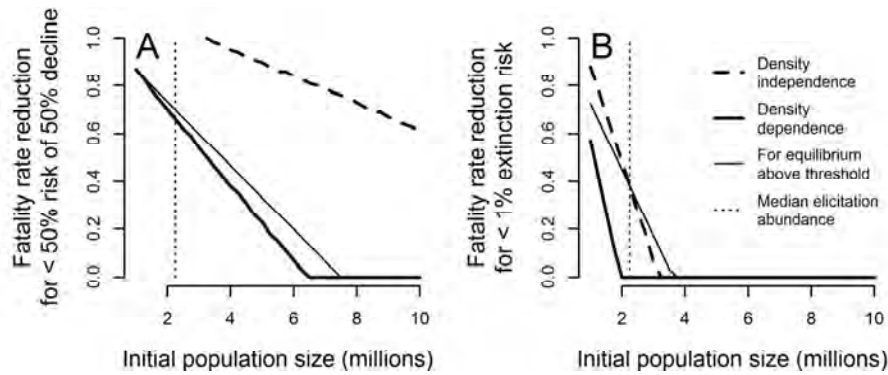
Figure A-3
*The level of fatality rate reduction needed to manage the risk of **A.** decline or **B.** extinction under density-independent and density-dependent population growth*

Managing decline risk beyond 2050 required fatality reduction over a greater portion of the range of initial abundance investigated. Although the long-term risk of decline and extinction is 100% in the density-independent model, the density-dependent model has a deterministic equilibrium. Achieving an equilibrium abundance above 50% of initial abundance required only slightly more fatality reduction than necessary to manage decline risk (Figure A-3A). Maintaining equilibrium abundance above the quasi-extinction threshold of 2,500 hoary bats required fatality reduction levels similar to those for risk management under density independence (Figure A-3B).

Discussion

Using a less conservative model that assumed hoary bat population size is regulated near an equilibrium led to decreased risk of decline or extinction by 2050 compared with a trendless density-independent model. Under the more stable model, decline risk was theoretically manageable even in the smallest population size assessed, and extinction risk was present only below the median expert opinion of abundance.

However, the stabilizing effect of population regulation was not a panacea; several qualitative results of the more conservative assessment were reconfirmed. The size of the hoary bat population was still the primary knowledge gap obscuring guidance on how much fatality reduction is necessary. Furthermore, maintaining equilibrium abundance above the quasi-extinction level required fatality reduction similar to that suggested by the density-independent analysis.

The existence of more than one model for hoary bat population dynamics is an example of model uncertainty, which is pervasive in forecasting the behavior of natural systems. When multiple models are supported by data, model uncertainty is often collapsed by averaging their predictions. However, when model uncertainty arises from a diversity of perspectives in the absence of data, averaging makes less sense. Taking the envelope of their predictions, as in Figure A-2, better communicates the state of knowledge (or lack thereof).

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