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Preliminary Methodology to Assess the National and Regional Impact of U.S. Wind Energy Development on Birds and Bats

Scientific Investigations Report 2015–5066

U.S. Department of the Interior
U.S. Geological Survey

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Conversion Factors

Multiply	By	To obtain
	Length	
meter (m)	3.281	foot (ft)
	Area	
square kilometer (km^2)	0.3861	square mile (mi^2)
	Power	
megawatt (MW)	1×10^6	joules per second (J/s)
gigawatt (GW)	1×10^9	joules per second (J/s)
	Energy	
megawatthour (MWh)	3.6×10^9	joule (J)

Abbreviations, Acronyms, and Symbols

a	age at first reproduction
BBS	North American Breeding Bird Survey
c	annual chance an individual will die from a collision with a wind turbine
$CV(N)$	coefficient of variation of population size
DOI	U.S. Department of the Interior
EIA	U.S. Energy Information Administration
EIS	environmental impact statement
F	recovery factor
FRI	fatality-risk index
FT	proportion of annual fatalities caused by turbines
GAMMS	Guidelines for Assessing Marine Mammal Stocks
GW	gigawatt
h	number of habitats used by a species
IEA	International Energy Agency
IRI	indirect-risk index
IUCN	International Union for Conservation of Nature
K	carrying capacity
m	maternity
MARSS	multivariate autoregressive state-space (model)
MMPA	Marine Mammal Protection Act
MW	megawatt
MWh	megawatthour
N	population size
N_{min}	lower bound on an estimate of population size
N_t	population size at year t
n	annual fatalities from wind turbines
NA	not applicable
NABat	North American Bat Monitoring Program
NEPA	National Environmental Policy Act
NREL	National Renewable Energy Laboratory
p	percentage of a species' range that overlaps with the locations of wind turbines
PBR	potential biological removal

R^2	coefficient of determination
r_{max}	maximum annual population growth rate under optimal conditions
RR	risk ratio
s	adult survival
SGCN	species of greatest conservation need
SWAP	State wildlife action plan
U.S.C.	U.S. Code
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey
WEIAM	Wind Energy Impacts Assessment Methodology (USGS project)
λ	population growth rate
λ_b	growth rate of the baseline population, with no fatalities from wind energy facilities
λ_w	growth rate of a population, with fatalities from wind energy facilities

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By J.E. Diffendorfer,[1] J.A. Beston,[1] M.D. Merrill,[1] J.C. Stanton,[1] M.D. Corum,[1] S.R. Loss,[2] W.E. Thogmartin,[1] D.H. Johnson,[1] R.A. Erickson,[1] and K.W. Heist[3]

Abstract

The U.S. Geological Survey has developed a methodology to assess the impacts of wind energy development on wildlife; it is a probabilistic, quantitative assessment methodology that can communicate to decision makers and the public the magnitude of these effects on species populations. The methodology is currently applicable to birds and bats, focuses primarily on the effects of collisions, and can be applied to any species that breeds in, migrates through, or otherwise uses any part of the United States. The methodology is intended to assess species at the national scale and is fundamentally different from existing methods focusing on impacts at individual facilities.

Publicly available fatality information, population estimates, species range maps, turbine location data, biological characteristics, and generic population models are used to generate both a ranked list of species based on relative risk as well as quantitative measures of the magnitude of the effect on species' population trend and size. Three metrics are combined to determine direct and indirect relative risk to populations. A generic population model is used to estimate the expected change in population trend and includes additive mortality from collisions with wind turbines. Lastly, the methodology uses observed fatalities and an estimate of potential biological removal to assess the risk of a decline in population size. Data for six bird species have been processed through the entire methodology as a test case, and the results are presented in this report.

Components of the methodology are based on simplifying assumptions and require information that, for many species, may be sparse or unreliable. These assumptions are presented in the report and should be carefully considered when using output from the methodology. In addition, this methodology can be used to recommend species for more intensive demographic modeling or highlight those species that may not require any additional protection because effects of wind energy development on their populations are projected to be small.

1.0. Introduction

Recent growth in wind energy generation has led to concerns over the effect of this development on wildlife in the United States. Investigations of impacts to volant species (winged species capable of flying) are conducted at many wind energy facilities, yet there remains a paucity of knowledge regarding the effects on species at the national and regional level (Arnett and others, 2008; Katzner and others, 2013). The U.S. Geological Survey (USGS) is using research, monitoring data, and modeling to develop a probabilistic and quantitative methodology to assess both the current and future population-level consequences of wind energy development on those species of birds and bats that are present in the United States during any part of their life cycle. The methodology is national in scope because the population assessed is defined as all the individuals of a species that occur in the United States, not a subset of them in a particular region or State. The methodology specifically addresses whether the fatalities caused by collisions with wind turbines can be sustained by a species in the long term. It also produces a ranked list of species' potential responses to the habitat loss associated with wind energy facilities. The methodology also uses projections of new wind energy in 2025 to make model projections about future effects of wind energy development on birds and bats. The USGS developed this methodology in response to renewable energy initiatives of the U.S. Department of the Interior (DOI), starting with the New Energy Frontier initiative (U.S. Department of the Interior and U.S. Department of Agriculture, 2011) and continuing with the Powering Our Future initiative (U.S. Department of the Interior, 2014).

USGS scientists studying minerals and energy have decades of experience in producing transparent, thoroughly vetted methods to assess (1) undiscovered resources of conventional and continuous oil and gas, coal, gas hydrates, and

[1]U.S. Geological Survey.

[2]Oklahoma State University, Department of Natural Resource Ecology and Management.

[3]University of Minnesota, Department of Fisheries, Wildlife, and Conservation Biology.

minerals and (2) potential geologic carbon dioxide storage resources. Geographers, ecologists, and statisticians of the USGS conduct a wide array of applied research on species population biology and spatial distributions, with an emphasis on responses to stressors at multiple scales. This assessment methodology is a product of the interdisciplinary cooperation of these scientists, and devising a methodology is the first step toward completing a full assessment. The objectives of this report are to describe the methodology, its central question and key assumptions, and how it would be implemented. A case study of six bird species is provided as an example of implementation and output.

The methodology was developed over a 3-year period. Initially, the authors participated in meetings with stakeholders, including a large kickoff meeting and targeted meetings with industry representatives, conservation organizations, and individual Federal agencies involved in energy and wildlife issues related to wind energy development. From these meetings, three key issues became apparent that influenced the goals for the methodology. First, the stakeholders felt it was most important to understand the population-level consequences of wind energy development on species. Second, they wanted a rapid method to prioritize species in terms of their risk from wind energy generation. Third, they were concerned about both the direct effects of wind energy development caused by collisions and the indirect effects caused by habitat loss and behavioral avoidance of wind turbines. Once the general goals of the methodology were defined, the components of the methodology were developed, tested using data from actual species, and discussed with species experts, all of which led to refinement and the current methodology.

Wildlife populations can be described by their size and trend, and thus “population-level consequences” can be defined as changes in a population’s size or trend. The determinants of a population’s size and trend are numerous and varied and likely differ both across species and across different areas or time periods for the same species. Information on these determinants is unavailable for most species, and overly complex models that require assumptions about much of the input may not provide better population analyses than simplified models (Morris and Doak, 2002).

This methodology uses different approaches to measure potential population-level consequences from wind energy facilities. All methods focus on flexibility and applicability to multiple species without attempting to describe the nuances of complex population dynamics. This focus may undermine the accuracy of some estimates, but the estimates will be useful for determining which species are likely to experience population-level consequences from wind energy development and for addressing whether the fatalities caused by collisions with wind turbines can be sustained by a species. The methodology is not intended to supplant the more detailed analyses required to make decisions under the Endangered Species Act (16 U.S.C. § 1531 et seq.), the Bald and Golden Eagle Protection Act (16 U.S.C. § 668 et seq.), or the Migratory Bird Treaty Act (16 U.S.C. §§ 703–712).

1.1. State of Wind Energy Development

Wind-powered electricity generation has increased significantly over the last decade to 167 million megawatthours (MWh) in 2013, which represents a cumulative installed capacity of 62.3 gigawatts (GW) in the United States by September 2014 (American Wind Energy Association, 2014). Wind energy generation currently represents 31.4 percent of U.S. electricity from renewable sources and 4.1 percent of total net electricity generation (U.S. Energy Information Administration, 2014b, table 7.2a). Wind energy is growing at a rapid pace and has overtaken all but conventional hydroelectric generation for renewable energy sources (fig. 1). Projections for U.S. wind energy generation by 2025, from reports by the U.S. Energy Information Administration (2014a) and the International Energy Agency (2012), suggest that installed capacity could nearly double to a range from 80 to 114 GW (see section 3.0 of this report).

1.2. Research on the Impact of Wind Energy Development on Wildlife

Academic and government scientists, including those from the USGS (Phillips, 2011; Ellison, 2012), have conducted research on the effects on wildlife from wind energy development. Electricity is generated from wind-driven turbines, and the physical collision of the turbine blades with volant species, such as birds and bats, causes injury and death. Fatalities from collisions are considered “direct effects” throughout this report. This report includes the potential for barotrauma (damage to body tissue caused by the difference in air pressure around the turbine blade) in the potential for collision damage (Baerwald and others, 2008; Grodsky and others, 2011; Rollins and others, 2012). The construction of wind energy facilities, the road networks required to service them, and the energy grid necessary to transport electricity to consumers can also affect wildlife through habitat loss and alteration; moreover, some species avoid areas near turbines (Stevens and others, 2013; Winder and others, in press). Habitat loss and alteration, as well as avoidance behavior, are considered indirect effects in this report. This assessment methodology primarily addresses direct effects, but the prioritization approach (section 2.4) includes an output ranking of species’ potential responses to indirect effects.

Review papers such as those by W.P. Erickson and others (2001, 2014), National Research Council, Committee on Environmental Impacts of Wind-Energy Projects (2007), Arnett and others (2008), and Strickland and others (2011) provide thorough summaries of existing research on the effects of wind energy facilities on wildlife and introduce the large and active research community working on this issue in the United States. Works by international authors and agencies or groups show that the effects of these facilities also are of concern outside the United States; see studies by the United Kingdom Joint Nature Conservation Committee (Crockford, 1992),

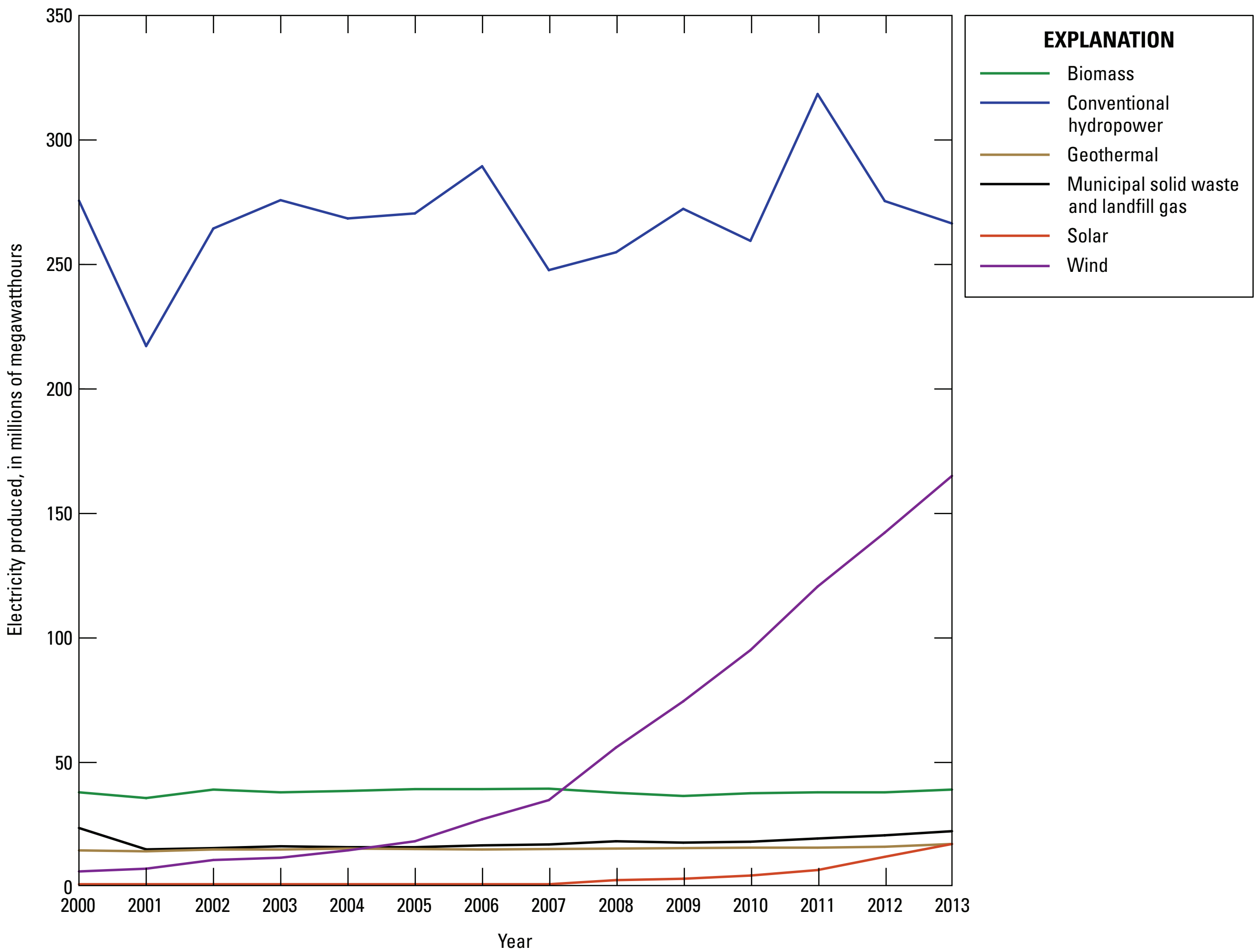


Figure 1. Graph showing U.S. electricity production from renewable energy sources, in millions of megawatthours, from 2000 through 2013. Wind energy use in this time interval increased more than the use of other renewable sources. Data from U.S. Energy Information Administration (2014b, table 7.2a).

Scottish Natural Heritage (Gill and others, 1996), Council of Europe (Langston and Pullan, 2004), Canadian Wildlife Service (Kingsley and Whittam, 2005), Rydell and others (2010), and Voigt and others (2012).

The scientific methods for assessing the effects of wind energy facilities on the population size and trend (the increasing, decreasing, or stable pattern of the population through time) of species over regional or larger areas are still developing. Conducting an assessment on species over large areas can have considerable temporal and logistical constraints. Each individual species can be the subject of a highly complex multiyear effort; examples include research on whooping cranes (Pearse and Selbo, 2012; Butler and others, 2013) and golden eagles (U.S. Fish and Wildlife Service, 2011a, b; Pagel and others, 2013; Johnston and others, 2014; Watson and others, 2014). Likewise, each wind energy facility can be monitored and assessed for years, but such intensive efforts, although useful for a few species of greatest concern, are impractical for studying all species and all locations.

A few attempts have been made to estimate the population-level consequences on bird and bat populations of fatalities caused by collisions with wind turbines. Although the result is not a direct estimate of population-level effects, W.P. Erickson and others (2014) divided an estimate of the annual fatalities per year by a population estimate for 20 bird species calculated by the Partners in Flight Science Committee (2013). Carrete and others (2009) developed a spatially explicit population model for Egyptian vultures and investigated the role of increased fatalities from wind turbines on population trends. Bellebaum and others (2013) modeled annual fatalities of 3.1 percent of the population for red kites in Germany and used the potential biological removal (PBR) method (see section 2.4) to estimate a PBR value of 4 percent, indicating the species may be near an unsustainable level of fatality. In another study of red kites, Schaub (2012) used an individual-based computer simulation of individual kites to study how the location of turbines might affect kite population trends.

When estimating effects at the population level is not possible, prioritizing species by using qualitative approaches can assist decision making that is related to wind energy generation and wildlife. In several studies, researchers developed methods for prioritizing birds relative to their potential risk from wind turbines. These studies targeted marine birds that could be affected by offshore wind energy development. Garthe and Hüppop (2004) used nine factors (such as flight maneuverability, adult survival, and current conservation status) to develop a wind facility sensitivity index. Desholm (2009) ranked 38 marine bird species in terms of their relative abundance and the elasticity of population growth to adult survival. Bright and others (2008) developed a basic sensitivity ranking (high versus medium) for birds in Scotland by using variables associated with sensitivity to wind facilities obtained from a literature review and combined these values with data from other sources to develop an overall risk map for wind energy facilities in Scotland.

1.3. The Assessment Methodology

The common principles underpinning the USGS methodologies currently in use for assessing undiscovered oil and gas and geologic carbon dioxide storage resources have been adopted in this methodology for assessing the effects of wind energy development on wildlife (Schmoker and Klett, 2005; Brennan and others, 2010; Charpentier and Cook, 2011). Methodologies for assessing resources (geological, hydrological, biological, and so on) normally include two components. First is the methodology's step-by-step procedure for estimating and using input parameters to calculate an output specific to the resource question. Second is the methodology's implementation plan, a structured approach that focuses on maintaining consistency and ensuring that the steps of the methodology are followed with peer oversight. The USGS Wind Energy Impacts Assessment Methodology (WEIAM) project team is currently tasked only with the completion of the first component, the authoring of the steps and calculations of the methodology. Although the implementation procedure is still in the initial stages of development, it is anticipated that it will follow established USGS procedures. According to procedure, a core group of scientists (generally the creators of the methodology plus other experts, as needed) serve as the assessment panel. An assessor (most likely a conservation ecologist, biologist, or biostatistician) will give multiple presentations to the panel during the course of an assessment. These presentations (1) establish the need to assess a particular species and (2) allow the panel to review the input data that the assessor plans to use in the various assessment components of the methodology (discussed further in section 2). The panel then implements the models and calculations of the methodology to produce outputs.

1.3.1. Principles

The USGS methodology described in this report is designed to be quantitative, with results reported in a probabilistic manner, and to be completely transparent in process and structure. Similar concepts in conservation biology and applied ecological modeling also have been used in ecological risk assessment (Athreya and Karlin, 1971; Boyce, 1992). Quantitative results are the most useful output for an assessment. Numerical results can be analyzed more readily than qualitative results, generally provide a clearer statement of finding, and can best inform decisions. A quantitative result is only as valuable as the certainty associated with that value; therefore, the methodology described here, like others produced by the USGS, is probabilistic, producing low and high bounds around a most likely result. Higher levels of uncertainty result in a greater spread between upper and lower bounds.

A transparent process helps maintain the consistent application of the method and allows researchers external to the USGS to conduct assessments using the same methods and vet the results. An additional value of this transparency is the ease of improvement through iterative upgrades to the methodology. Methodologies often are revised and modified over time as knowledge and technology improve. For example, the USGS has published numerous revisions to its assessment methodology for continuous (unconventional) undiscovered oil and gas. These changes were adopted as significant increases in production of continuous resources and additional data revealed limitations in previous methods. Thus, as studies of the effects of wind energy facilities on birds and bats continue, we anticipate changes in knowledge that will affect the methodology. Because this is the first version of a methodology to assess the effects of wind energy development on wildlife at broad scales, it could certainly be improved through future advances in research and understanding. Ongoing research by the WEIAM project team that may lead to improvements of the methodology is included in appendix 1.

Finally, in a complex multipart methodology such as the one described here, the various components should be structured and linked to each other to create a cohesive methodology that achieves the designers' goal. For example, the results for each component should be both internally consistent and consistent across components. Furthermore, individual components should produce unbiased results in the face of incomplete or biased input data. Lastly, because science activities related to the management of species often intersect with policy, structuring the methodology to avoid subjective policy decisions is necessary. When avoidance is not possible, the methodology must clearly demarcate when policy issues affect the structure of the methodology and how these nonscientific decisions will be included in a functioning assessment.

1.3.2. Assessment Unit

In general terms, the assessment unit describes the individual entity that the methodology assesses; it is the core of the assessment process. The results from an assessment are calculated by using the methodology and are presented at the assessment-unit level. The methodology described here works with U.S. populations of individual species and will apply to any species that is present in the United States during any part of its life cycle. For birds and bats, species designations follow those in the "American Ornithologists' Union Checklist of North and Middle American Birds" (American Ornithologists' Union, 2014) and "Mammal Species of the World" (Wilson and Reeder, 2005), respectively.

The spatial scale and population assessed will change from species to species, though the methodology will focus on the population of individuals that intersect with the United States. For example, a nonmigratory species distributed in the southwestern United States and Mexico will be assessed using that portion of the population that resides in the United States. For migratory species, the portion of the total population that may be present in the United States at any time during its life cycle will be assessed. Thus, if all individuals of a neotropical migrant bird species may be found in the United States during a stopover while flying between their more northern breeding habitat and more southern overwintering grounds, this entire population will be assessed because all of the individuals move through the United States annually and are possibly exposed to wind turbines in the United States. In other words, the assessment considers the impacts of all the wind turbines that exist in a species' range within the United States, including Alaska and Hawaii. Smaller regions within a species' range that may reflect demographic or genetic subunits could be assessed with the same methodology if interest, need, and data were available. However, the methodology is not designed to estimate site- or facility-specific impacts of wind energy development on a local population.

1.3.3. Output and Potential Uses

The assessment methodology produces outputs to address two main objectives. First, it produces ranked lists of species based on potential direct risk (from collisions) and indirect risk (from habitat change) from wind energy development. This prioritization meets the objective to quickly filter bird and bat species with respect to their risk by ranking them into qualitative relative risk categories.

Second, for those species prioritized as high *direct* risk, the methodology produces two outputs that address the objective of quantifying population-level consequences from collision fatality. The first is an estimate of the expected change in population trend with the addition of turbine collision mortality based on a demographic population model. The second is an estimate of the risk ratio, which quantifies whether the fatalities from collision with wind turbines would reduce the population below a target size. Thus, the demographic model approaches population-level consequences by estimating changes in trend, while the risk ratio addresses population-level consequences resulting in changes in population size.

1.3.4. Intended Audience

USGS assessment results, and the methodologies that produce them, are created with multiple end users in mind. Due to the national scale and numerous assessment units involved, USGS national assessments are typically of most use to decision makers, government agencies, and U.S. citizens who need a broad, generalized understanding of a particular resource. That understanding is based on the summation of individual quantitative results for each assessment unit, but these results should not be confused with site-specific or single-facility impact assessments. Assessments such as environmental impact statements (EIS) that are completed to comply with the National Environmental Policy Act (NEPA) (42 U.S.C. § 4321 et seq.) are commonly site specific. The method described here also is not related to the siting guidelines established by the U.S. Fish and Wildlife Service (USFWS) for reducing potential wildlife impacts at specific wind energy facilities (U.S. Fish and Wildlife Service, 2012).

2.0. Methodology

2.1. Methodology Overview

The methodology will be implemented through a formal assessment process based on those currently in use by the energy and mineral programs at the USGS. A generalized flowchart of the methodology is presented in figure 2; detailed flowcharts of the components of the process are provided in the applicable sections of this report. For each species, an assessor will gather data (step 1 in figure 2) and develop a species description that (1) summarizes any studies related to the species' responses to wind turbines, including studies related to the species' response to habitat loss, fragmentation, and roads; (2) includes the sources and reasoning for the model parameter estimates and their distributions, the observed fatality estimates at turbines, any time series of abundance data, and estimates of population size; (3) synthesizes information about the species to define the population being assessed and its spatial scale; and (4) includes a species distribution map. This information will be presented in both oral and written form to a peer-review assessment panel and modified as warranted.

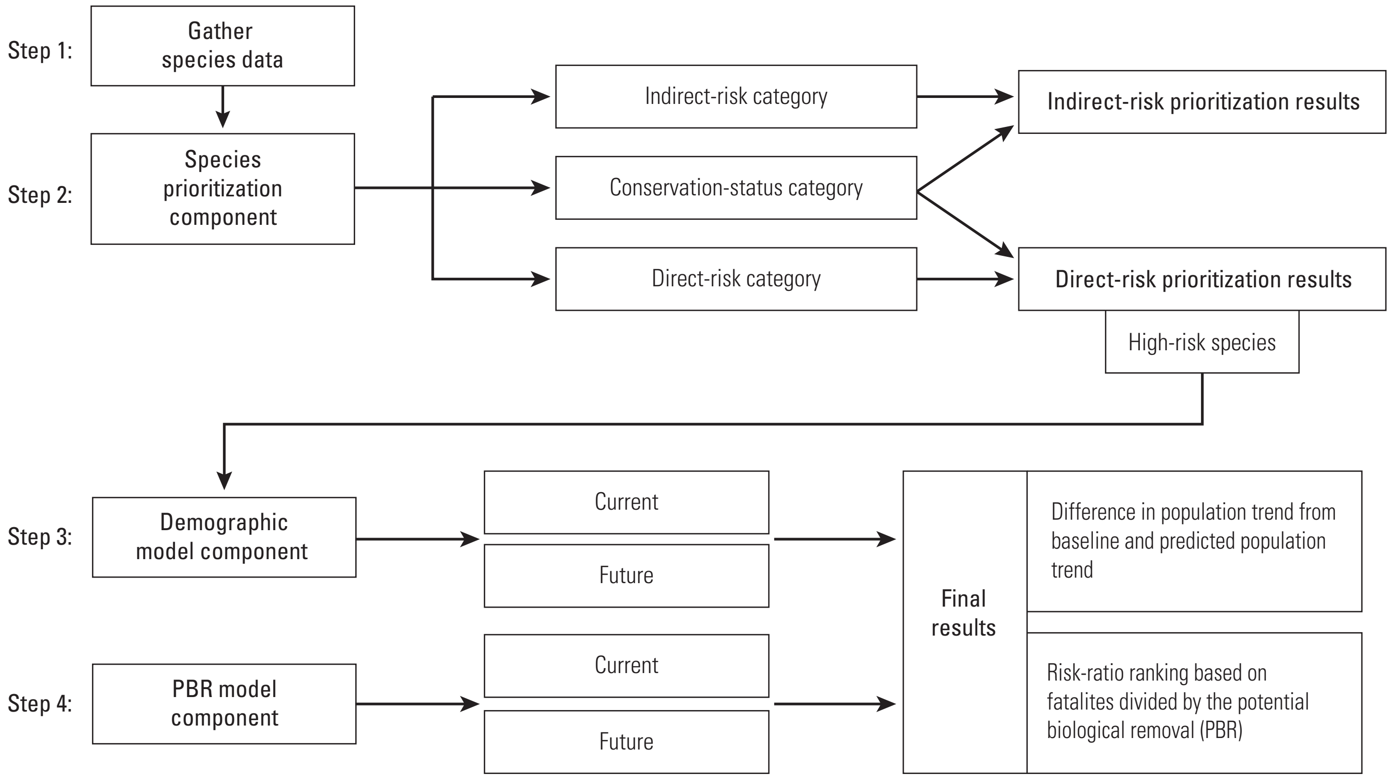


Figure 2. Flowchart showing the generalized steps in the assessment methodology. "Current" refers to methodology steps based on installed capacity and number of turbines in 2014. "Future" refers to steps that use projections of installed capacity and number of turbines for 2025.

Once data are collected, species will be prioritized with respect to direct risk and indirect risk from wind facilities (step 2 in figure 2). Direct-risk prioritization will be based on two metrics that estimate a species' potential risk from the fatalities caused by collisions with turbines: the proportion of annual fatalities due to turbines (FT) (section 2.2.3.1) and the fatality-risk index (FRI) (section 2.2.3.2). The values of FT and FRI will be combined with a species' conservation status to rank species on the basis of their potential risk from collisions with turbines (table 1). Species classified into a high-risk category during an external review of the risk rankings (explained below in section 2.2.3) will then be further assessed by using the demographic model and risk ratio. Indirect-risk prioritization will be based on the indirect-risk index (IRI), which will measure the potential consequences of behavioral avoidance, habitat loss, and degradation associated with wind energy facilities (see section 2.2.4). The IRI value will be combined with a species' conservation status to rank species on the basis of their potential risk from habitat modification (table 1).

For prioritization, most efforts will be spent amassing species-level information and developing distribution or range maps from available sources such as eBird (Cornell Lab of Ornithology, 2015) or the Global Biodiversity Information Facility Secretariat (2015). This work also could include a formal process for eliciting expert opinion for those species lacking data (Runge and others, 2011; Martin and others, 2012). Prioritization will be conducted for all bird and bat species that occur in the United States during any part of their life cycle.

Once prioritization is complete, the species ranked highly for direct risk will be assessed by using the demographic model (step 3 in figure 2) and the risk ratio (step 4 in figure 2). The demographic model estimates a change in population trend caused by fatalities from collision with wind turbines. All estimates are probabilistic and include measures of uncertainty. For each species, estimating the population trend may require estimating demographic rates from raw data, performing a meta-analysis from existing studies, or again eliciting expert opinion. The risk ratio estimates how close the estimated annual fatalities from wind turbines are to the potential biological removal (PBR), which estimates total fatalities that can occur before the population will decline below a target population size. Prioritization will be done relatively rapidly across many species simultaneously; however the demographic model and PBR-based risk ratio will require more effort and will rely on the panel process described above.

Late in the demographic-model and the PBR-based risk-ratio processes (steps 3 and 4 in figure 2), the respective models are repeated using estimates of projected wind energy development, described in section 3, for the highly ranked species. Thus, prioritization is only done using current levels of wind energy development, and then the highly ranked species are assessed using the demographic model and the risk ratio

Table 1. Ranking system for species analysis in the prioritization component of the assessment methodology based on combinations of conservation status and direct or indirect risk.

Rank	Direct- or indirect-risk category	Conservation-status category
1	High	High
2	High	Medium
3	Medium	High
4	High	Low
5	Medium	Medium
6	Low	High
7	Medium	Low
8	Low	Medium
9	Low	Low

for the current levels and two projected levels of wind energy development. Species will not be reprioritized under projected levels of wind energy development because their future conservation status cannot be reliably predicted.

The final output of the assessment includes the following information for each species: (1) its relative risk of response to habitat loss or degradation (indirect-risk prioritization) and (2) its relative risk of population-level consequences from wind energy development (direct-risk prioritization). In the direct-risk category, for species having a high prioritization rank, the final output also includes (1) the estimated change (with uncertainty) in population trend (demographic model) and (2) the risk ratio, which is based on the estimated annual fatalities and the PBR.

2.2. Species Prioritization

The first component of the assessment methodology, species prioritization, is designed to rapidly characterize a large number of bird and bat species in terms of their relative risk from wind energy facilities. International conservation organizations, Federal and State agencies, and others involved in resource management commonly prioritize species by using variables such as current population size, population trend, and known threats. The use of prioritization as a tool in species management has been studied by conservation biologists; although uncertainty exists in the prioritization process (Burgman and others, 1999), risk-based ranking approaches can correctly estimate extinction risk with a 70- to 80-percent success rate (Keith and others, 2004).

USGS investigators developed the prioritization component partially on the basis of two earlier studies. Desholm (2009) combined information about exposure (the proportion of a species' total population that moved through the area of a wind energy facility) and the ability of the species to respond to added mortality from wind energy facilities (the elasticity of adult survival), whereas W.P. Erickson and others (2014) examined the mortality rates of small passerines at wind energy facilities by dividing estimates of annual fatalities by continent-wide population estimates. The approach described here expands on these ideas by considering impacts beyond single facilities and including more indicators of a species' response to wind energy.

2.2.1. Implementation

The species prioritization approach uses two general types of information to assess the relative risk of population-level consequences to a species from wind energy facilities: (1) a species' current conservation status and (2) estimates of direct risk and indirect risk. A species' conservation status is likely correlated with its ability to withstand added mortality from wind turbines or habitat loss. For example, imperiled species with higher conservation status should, in general, be more vulnerable to any additional adverse impacts than species with lower conservation status (Andelman and others, 2004). Estimates of direct risk and indirect risk quantify some of the potential impacts of wind energy facilities on a species. The combination of a species' current conservation status and its level of direct or indirect risk determines its ranking relative to other species.

Each species is assigned three risk metrics that combine to determine its rank for direct- and indirect-risk prioritizations (fig. 3): (1) relative risk based on the species' conservation status (section 2.2.2), (2) relative risk from collision fatalities, or "direct risk" (section 2.2.3), and (3) relative risk from habitat modification, or "indirect risk" (section 2.2.4). Metrics 1 and 2 are combined using a qualitative approach to assign a direct-risk ranking between 1 and 9 (table 1), and metrics 1 and 3 are combined in the same way to assign an indirect-risk ranking (table 1).

The ranking system shown in table 1 currently emphasizes direct or indirect risks over conservation status when ranking species. For example, high direct or indirect risk in combination with medium conservation status is ranked higher (rank 2) than its inverse (rank 3). Given the goals of the assessment, this ranking system was established to emphasize the possible effects of wind energy facilities more than a species' current conservation status and is an initial effort that will require review by experts, decision makers, and regulators prior to an implemented assessment.

To rank species, breakpoints that place species into high, medium, or low categories must be set for the metrics that are used to describe conservation status and direct or indirect risk. Because these types of breakpoints are somewhat arbitrary and frequently include both scientific and policy-related considerations, the methodology instead estimates an average rank by randomly sampling breakpoints from across the range of observed values for each metric.

For each set of random breakpoints, species are assigned a numerical rank (1–9) by combining their maximum values for conservation status and either direct or indirect risk; this rank is then averaged across all sets of randomly selected

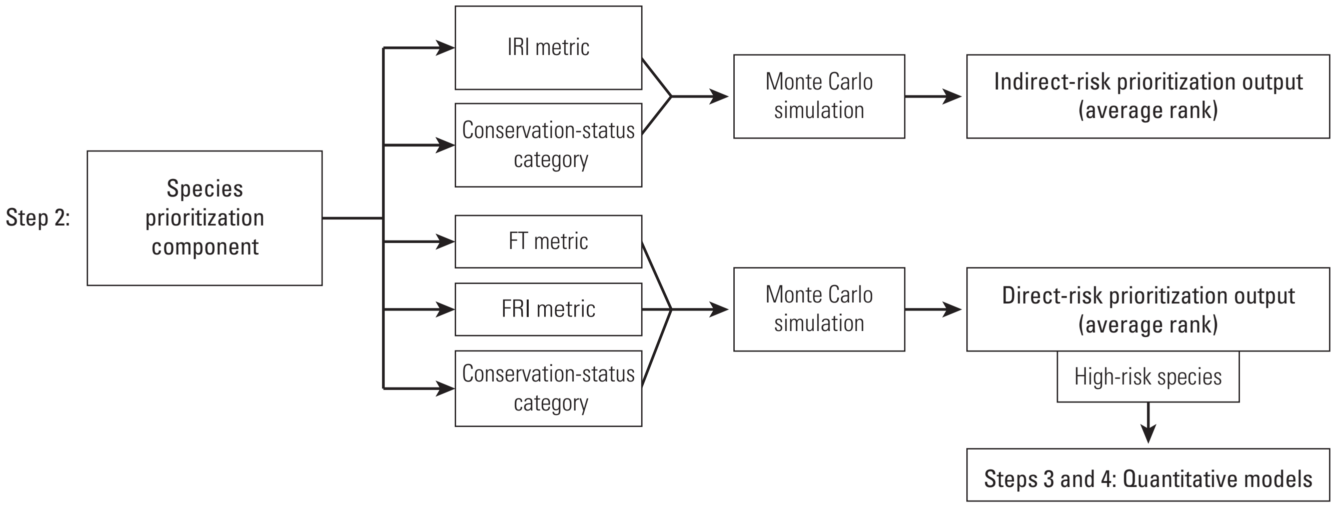


Figure 3. Flowchart of the species prioritization component of the assessment methodology. FRI, fatality-risk index; FT, proportion of annual fatalities due to turbines; IRI, indirect-risk index.

breakpoints. For direct risk, the highest risk category for either the proportion of fatalities due to turbines (FT) or the fatality-risk index (FRI) determines the direct-risk category ranking, and the indirect-risk category is determined by the indirect-risk index (IRI). The final outputs from prioritization are two ranked lists of species: (1) a direct-risk prioritization list with an average rank value for the combination of direct risk and conservation status and (2) an indirect-risk prioritization list with an average rank value for the combination of indirect risk and conservation status. A species with limited data may appear on only one of the lists until more information is available.

A key decision point would occur after prioritization. Given a list of averaged direct-risk prioritization scores for each species, a subset of these must be selected for assessment using the demographic model and risk ratio. This step is ultimately subjective, as how far down the average rankings one chooses to go is not specifically dictated by this methodology. Ultimately, the USGS would not make this decision. Instead, the USGS would convene a stakeholder meeting consisting of members of Federal and State agencies responsible for implementing laws and policies related to wind energy development and wildlife issues to review the prioritization process and its outputs and to determine the cutoff for species that should be further assessed.

2.2.2. Current Conservation Status

The conservation status of many species in the United States has been classified by a variety of organizations for different purposes and with different standards of assessment. This methodology uses the conservation status measured at the State level by the proportion of States that listed the species as "Species of Greatest Conservation Need" in a State wildlife action plan (SWAP; Beach and others, 2011). Other potential sources of conservation status information include (1) the Red List of the International Union for Conservation of Nature (2012, 2014), (2) listing by the Endangered Species Act of 1973, as amended (16 U.S.C. §1531 et seq.), and (3) the list, "Birds of Conservation Concern 2008" (U.S. Fish and Wildlife Service, 2008).

2.2.3. Direct Risk

2.2.3.1. Proportion of Fatalities Due to Turbines

The most obvious impact of wind energy generation on wildlife is usually fatalities of birds and bats from collisions with turbine blades. Various studies have quantified fatalities at individual wind energy facilities (Smallwood and Karas, 2009; Kitano and Shiraki, 2013), and researchers have projected fatality rates to larger scales (Loss and others, 2013; Smallwood, 2013). W.P. Erickson and others (2014) divided species-specific fatality estimates by population size to compare mortality rates from turbines across a suite of small passerines. This approach is unsuitable for comparisons across more varied species because of differences in natural history. Instead, the proportion of annual fatalities due to turbines (FT) compares species-specific fatality estimates from wind energy facilities to annual fatalities from all sources.

To calculate FT, the estimate of the number of individuals killed by wind turbines annually (n) is divided by the product of population size (N) and the adult mortality rate (calculated as 1 minus adult survival, s):

$$\mathrm{FT} = \frac{n}{(1-s)N} \tag{1}$$

In general, long-lived species with low mortality rates are more likely to experience additive mortality from anthropogenic sources than short-lived species with inherently high mortality (Péron and others, 2013). Therefore, species with a higher FT value (that is, number of animals killed at wind energy facilities divided by number of animals that die from any cause) are at higher risk of population-level consequences from wind energy facilities. Risk assessment using FT may be biased toward higher values for those species that strongly compensate for turbine mortality with a reduction in mortality from other sources or an increase in reproduction. Thus, FT is a conservative measure of risk.

The number of individuals killed by wind turbines each year, n, could be directly estimated from fatality studies if these data were available. Currently, direct estimates of fatality from turbine collisions across a species' range are not available for any species. Alternatively, one can coarsely estimate n by multiplying the total avian fatalities from wind turbines by the proportion of observed fatalities attributed to each species. These types of estimates have been done for birds (Zimmerling and others, 2013; W.P. Erickson and others, 2014), but not for bats, although such estimates for bats are possible. In making these estimates, the researcher assumes that (1) the wind energy facilities chosen for sampling to estimate fatalities are representative of all wind facilities in the United States and (2) the carcasses of species killed by turbine collisions have similar levels of detectability. Both of these assumptions are clearly violated by the available studies, yet no alternative approaches have been developed. Estimating species-specific fatality rates with this approach can produce considerable bias in the assessment output, perhaps leading to unreliable results. Output from the methodology will be improved if species-specific fatality rates become available.

Total population size, N, is estimated by a number of organizations for birds, but is very difficult to assess for most species of bats. The Partners in Flight program (Partners in Flight Science Committee, 2013), the Midwinter Waterfowl Survey (Sharp and others, 2002), and the Waterbird Conservation for the Americas program (Kushlan and others, 2002) generate estimates of population size for some, but not all, bird species in the United States. Bat population sizes are difficult to estimate because of their nocturnal behavior, small size, and similar appearances across species, which make them difficult to identify by sight. A few cave-roosting species of bats in the United States are monitored at a sufficient number of caves to produce rough estimates of population size (Thogmartin and others, 2012), but bats currently are not systematically monitored in the United States. The North American Bat Monitoring Program (NABat) seeks to address this data deficiency (U.S. Geological Survey, 2015; Loeb and others, in press). When estimates of population size are not available, rough bounds based on expert judgment may be required (Russell and others, 2014).

Survival has been estimated for birds by a variety of sources and individual studies. These estimates were published in species-specific demographic modeling papers by the Institute for Bird Populations (Michel and others, 2006) and in Birds of North America species accounts (Poole, 2005). Data for bats are sparse, but most bats are long lived and estimates of survival exist for some species (Barclay and Harder, 2003; McCracken, 2003; O'Shea and others, 2003; Frick and others, 2007; Thogmartin and others, 2012). When survival estimates are unavailable for a species, it may be possible to substitute estimates from closely related species. Note that the survival estimate used when calculating adult mortality should include fatalities from wind energy facilities. If not, FT becomes a ratio of wind-facility-related fatality to non-wind-facility-related fatality, rather than the proportion of wind-facility-related fatality to all fatalities. If a survival rate does not include fatalities from wind energy facilities, then the estimated fatalities from wind should be added to $(1-s)N$ in the denominator of equation 1.

2.2.3.2. Fatality-Risk Index

Annual fatalities from wind turbines are not always available and may be poorly estimated for some species. To account for the lack of data, the methodology uses the fatality-risk index (FRI), which is based on the assumption that species with slow life histories and high exposure to wind turbines are more likely to be at risk of population effects from fatalities caused by collisions. A similar assumption was made in a prioritization approach designed for use at individual wind farms (Desholm, 2009). To calculate the FRI, the percentage of the population impacted (p) is divided by an index of life-history speed (the ratio of maternity [m] to age at first reproduction [a]):

$$\mathrm{FRI} = \frac{p}{m/a} \tag{2}$$

For birds, data for these values are in the same sources as data for survival, whereas for bats, they are more difficult to obtain. For birds, maternity could be taken directly from the literature or calculated as the product of nest success rate, average clutch size, hatchability rate, average number of clutches per year, and a presumed 1:1 sex ratio of offspring.

The ratio of m/a was developed to account for the known relation between life-history traits and the elasticity of population trend as related to these demographic parameters (Oli, 2004; Stahl and Oli, 2006). For example, the population trend for a species with a ratio of m/a greater than 0.75 is most sensitive to changes in reproductive parameters, whereas the population trend of a species with a ratio of m/a less than 0.25 is most sensitive to survival. Because wind turbine collisions affect survival, m/a should measure the potential sensitivity of species to turbine collisions. Ultimately, m/a is a measure of life-history speed; some species have a low ratio because they have only a few offspring (small m) and delay reproduction (large a), whereas other species have a high ratio because they produce many offspring (large m) and reproduce earlier in their life (often a equals 1). Larger values of the FRI indicate a higher risk, which can be a result of either the presence of many wind turbines in a species' range or the slow life history of the species.

Calculating the proportion of a species' range that overlaps with turbine locations requires both a range map for a species and a map of the known locations of turbines. A range map is a geospatial data layer representing the spatial distribution of a species and may show variation in abundance across space or simply the presence or absence of the species. Turbine locations are available from a national turbine dataset developed by researchers working on this project (Diffendorfer and others, 2014). For both birds and bats, migration makes defining and mapping the range of a species complex because abundance changes across space during different times of the year, and for bats, fatalities increase during migration (Cryan, 2011; Ellison, 2012). Furthermore, some species may move through particular geographic regions intensively during migration (Miller and others, 2014).

For birds, breeding-season ranges and distributions of many species have been mapped with the North American Breeding Bird Survey (BBS) data (Sauer and others, 2015). Maps of overwintering locations could potentially be developed by using data from the Christmas Bird Count (National Audubon Society, 2015). Maps of year-round distribution, including breeding, migratory, and overwintering areas, could perhaps be developed by using eBird, a citizen-science-based repository of bird locations (Cornell Lab of Ornithology, 2015). Ideally, the resulting maps would be used to integrate the proportion of the population near wind energy facilities over an entire year as an estimate of p. Regardless of the final data source(s) used to make distribution maps, the same mapping method must be applied to all species during prioritization to ensure that relative rankings are comparable across species. The BBS abundance maps include an estimate of relative abundance in each grid cell (21.475 square kilometers). The relative abundance in each grid cell in the United States (some species' ranges included Canada) can be multiplied by the area of each grid cell, and then standardized to sum to 1 across all grid cells in the United States. Doing so creates an estimate of the proportion of the species' breeding population in each grid cell across the United States; see an example in figure 4. The cumulative proportion of the population in grid cells containing one or more turbines is one estimate of p. This approach may overestimate p because isolated turbines may affect only a small portion of the population that actually is located in a grid cell. Conversely, if a grid cell is used as a migratory pathway or wintering habitat, then p may be underestimated when based on data from only breeding birds.

Scientists in North America do not yet have a centralized monitoring program for bats (see NABat at U.S. Geological Survey, 2015, and Loeb and others, in press) or a repository for distribution maps. These species may require distribution modeling or range-map development as part of estimating p, as may some bird species. In addition, because bat fatalities are the highest during migration (Arnett and others, 2008; Cryan, 2011), seasonal range maps might be most useful for modeling p in bats. Appendix 1 describes additional research that the USGS is performing to improve estimates of p through an improved understanding of how animals use airspace.

2.2.4. Indirect-Risk Index

Species that rarely collide with wind turbines can still suffer population consequences due to disturbance, displacement, and habitat fragmentation and loss (Leddy and others, 1999; Langston and Pullan, 2004; Percival, 2005; Fox and others, 2006). Scientists consider indirect effects to be a potentially serious consequence of wind energy. For example, Kuvlesky and others (2007, p. 2490) stated:

> European conservationists generally consider the habitat loss associated with wind farm developments to be a greater threat to bird populations than are collision fatalities.

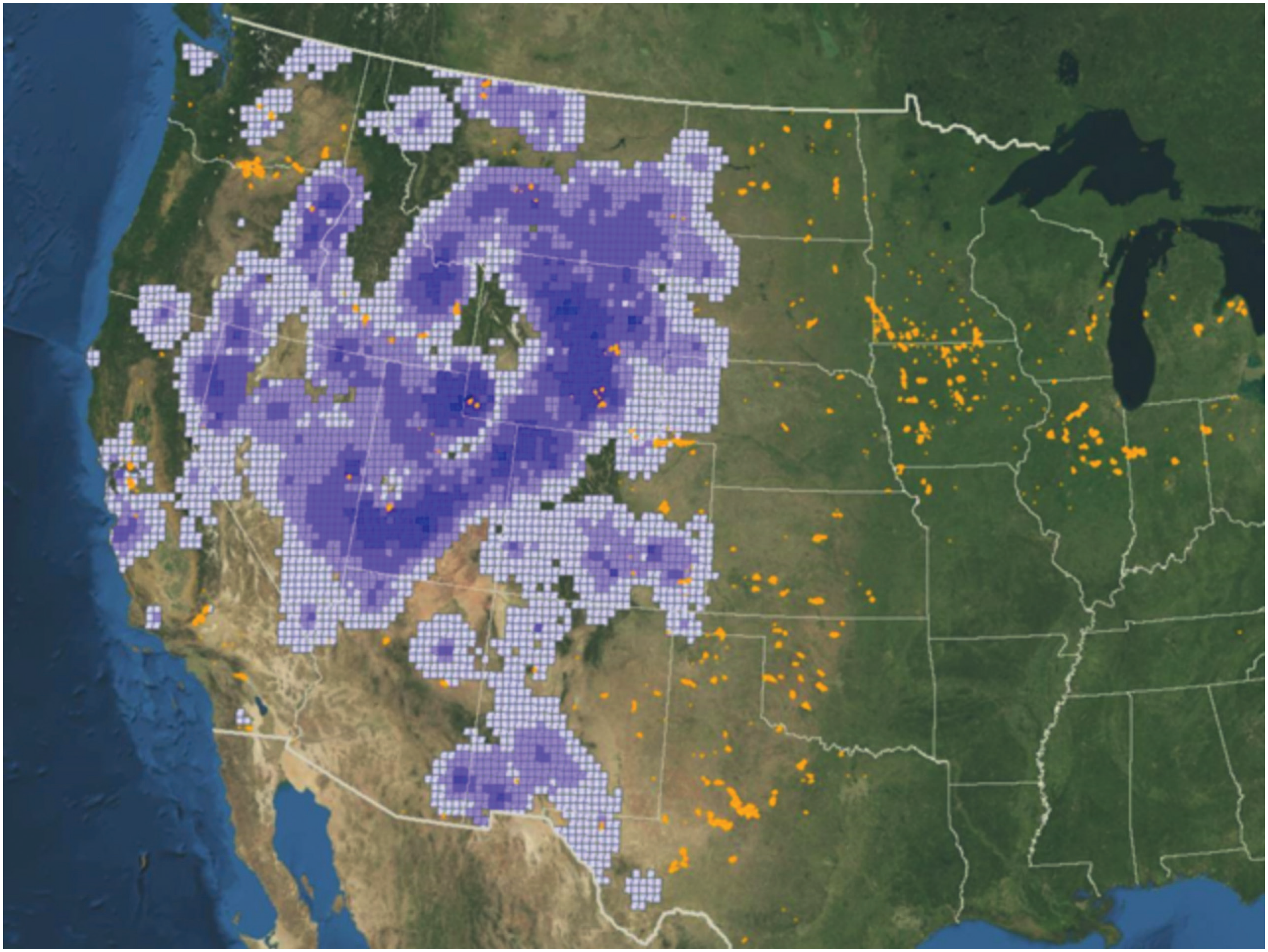
This sentiment was repeated by Katzner and others (2013). A number of studies suggest that some species avoid turbines at variable distances (Leddy and others, 1999; Stevens and others, 2013). Furthermore, wind energy facilities include roads and transmission lines, both of which can affect species (Forman and Alexander, 1998; Coffin, 2007).

Although the population-level consequences caused by indirect effects have not been quantified for most species, information about the natural history of species may help categorize risk. In general, highly specialized species are more sensitive to changes in habitat than generalist species (Swihart and others, 2003; Munday, 2004); in one study of the indirect effects of turbines on wintering birds, only one of the most specialized species considered (Le Conte's sparrow; *Ammodramus leconteii*) appeared to be displaced by turbines (Stevens and others, 2013). The indirect-risk index (IRI) was developed to take advantage of the expectation that species that use fewer habitats will be more sensitive to the indirect effects of wind energy facilities and is calculated as

$$\mathrm{IRI} = \frac{p}{h} \tag{3}$$

where p, as above, is the percentage of a species' range that overlaps with the turbines and h is the number of habitats used by a species. Information on the number of habitats considered suitable for each species is available from the IUCN's species database (International Union for Conservation of Nature, 2014). Higher values of IRI indicate a higher risk because a large proportion of the population is exposed to turbines or the species occupies fewer habitats.

Because of stakeholders' concerns, the WEIAM project team tried to include a measure of species response to indirect effects from wind energy development; however, the team found that a generalizable and quantitative approach that linked indirect effects to changes in population status was not feasible. In light of that realization, the IRI is estimated and included as part of the indirect-risk prioritization, but it is not used beyond this point in the methodology. Species that rank highly for indirect risk may require further scrutiny and research to understand if indirect effects from wind energy facilities do, in fact, occur.



Base imagery from Esri

Figure 4. Map of the Central and Western United States showing the distribution of golden eagles (purple grids) relative to turbine locations (gold dots). Darker colors in the grid cells represent a high normalized relative abundance. Golden eagle data are from the North American Breeding Bird Survey (Sauer and others, 2015). Turbine locations are from Diffendorfer and others (2014).

2.3. Demographic Model

2.3.1. Background

Demographic models represent a well-developed set of mathematical tools in both population biology and ecological risk assessment. They can be used to understand the interactions between management actions and population trends (Crouse and others, 1987), compare projected population dynamics under different scenarios (Rose and Cowan, 2003), and investigate the factors and processes contributing to observed population trends (Koons and others, 2005; Thogmartin and others, 2013). Methods and tools for implementing demographic models are well established, including commercial software programs and open-source packages (for example, R.A. Erickson and others, 2014). Furthermore, the behavior of demographic models is well understood, and scientists have developed a framework for using them in risk assessments and population viability analyses (Burgman and others, 1993; Beissinger and McCullough, 2002; Morris and Doak, 2002).

The general goal of the modeling used in this assessment is to understand if, and by how much, observed levels of fatalities from wind turbines across a species' range in the United States affect that species' population trend given the uncertainties associated with the number of fatalities, the processes driving the demography of the species, and the observed population trend (when available). Although population trend is not the only variable associated with the status of a population, it does indicate whether a population is growing or declining and if it is correlated with estimates of extinction risk (O'Grady and others, 2004). Developing models of biological phenomena must balance generality, precision, and realism because an ecological model cannot simultaneously include high levels of all of these characteristics (Levins, 1966). For this USGS assessment, a modeling approach should be general enough to apply to many species, including those lacking time series of abundance estimates, yet have sufficient realism and precision to produce results capable of indicating risk to the overall population from wind energy development. However, appendix 1 describes an example of a more complex model developed for Indiana bats that the USGS can use to develop a tailored assessment for *Myotis* bat species in general.

After considering a number of alternative approaches (see appendix 2), the WEIAM team developed a method based on simple, generalized population models to compare population trends with and without the addition of fatalities caused by turbines. This component is termed the "demographic model" throughout this report (fig. 5). The demographic model includes two steps: (1) estimation of population growth rate in the absence of wind energy facilities and (2) estimation of the change in population growth rate with the addition of fatalities from wind energy facilities.

2.3.2. Estimating Population Growth Rate

The population growth rate is the proportional change in population size from one year to the next. Annual growth rates and associated measures of uncertainty can be calculated with a time series of population size estimates (or a time series of population size indices). Monitoring programs, such as the BBS, routinely estimate population trends using such time series, and the current assessment methodology can use a similar approach to directly estimate population trends for species susceptible to direct impacts from wind energy production.

In the absence of empirical estimates of population growth rate, the dominant eigenvalue of a matrix population model is an alternative estimate of population growth rate (Caswell, 2000). Matrix models include assumptions that shape their applicability to particular ecological problems. These assumptions are well described by a number of authors (Ebert, 1998; Caswell, 2000) and include the assumption that animals can be classified by age, density independence, a stable age distribution, and constant vital rates. For this component of the methodology, a stage-structured, postbreeding matrix model, with an annual time step, is used to describe the female segment of a population (Caswell, 2000; Morris and Doak, 2002). The age of first reproduction determines the size (number of rows and columns) of the matrix. For example, species that begin breeding within the first year of life would have a matrix with 2 rows and 2 columns, whereas a matrix of 4 rows by 4 columns would model species that begin breeding in the third year of life. The values in the first row determine recruitment (new individuals entering the population), and the subsequent rows describe annual survival after birth or hatching and each year thereafter. The second-to-last column represents juveniles that survive and mature to reproductive age and then reproduce before the next year.

$$\begin{bmatrix} 0 & 0 & \cdots & s_j m & s_a m \\ s_j & 0 & \cdots & 0 & 0 \\ 0 & s_j & \cdots & 0 & 0 \\ \vdots & \vdots & \ddots & \vdots & \vdots \\ 0 & 0 & \cdots & s_j & s_a \end{bmatrix}$$

Parameters include survival (s, the chance a female survives from one year to the next) and maternity (m, the number of female offspring per female per year). Thus, fecundity is defined as the female offspring born to females surviving since the previous census ($s_j \times m$ or $s_a \times m$, for juvenile and adult females, respectively). Estimates of survival and maternity are those used in the prioritization component (section 2.2). To parameterize these distributions, means and variances are taken from (in decreasing order of quality) the literature or available data, professional opinion, data on surrogate species, or an assumed value based upon theoretical expectations. For

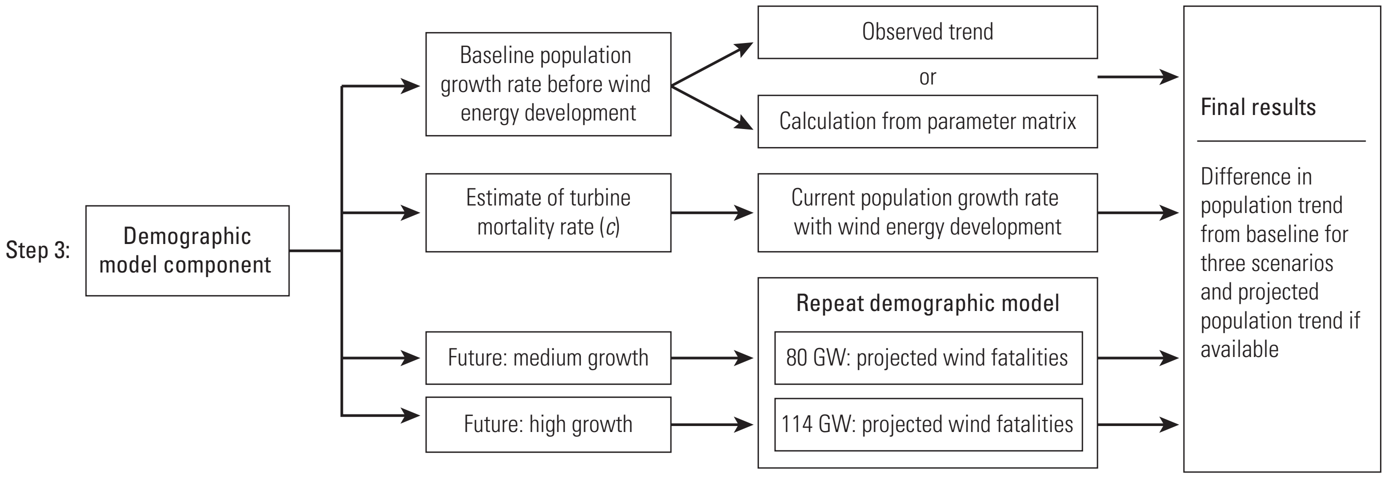


Figure 5. Flowchart of the demographic model component of the assessment methodology. GW, gigawatts.

most species, when data on differential breeding success by age or life stage are not available, a single maternity value will be assumed for all reproductive stages.

For both empirical estimates and matrix-based estimates of population growth rate, uncertainty or bias in the estimates of input parameters can introduce bias in the estimated trend. Care must be taken to ensure that estimates are representative of the population occurring in the United States and that they do not include populations already affected by wind energy development. In addition, the estimated population growth rate must be accompanied by some descriptor of the associated uncertainty, which can be estimated directly from time series or propagated through a matrix model by sampling survival from a beta distribution and maternity from a lognormal distribution.

2.3.3. Estimating Change in Population Growth Rate

To estimate the change in population growth caused by collisions with wind turbines, estimates of fatalities must be made and then used to estimate a reduction in the existing population trend. The approach uses the turbine mortality rate (*c*), which is the annual chance an individual will die from a collision with a wind turbine. If wind fatalities are assumed to affect all species' stage and age classes at the same rate, then *c* is the same across all individuals, and an unstructured population model can be used to determine how those fatalities affect the population growth rate.

In the absence of density dependence, populations grow exponentially, and the population size of a species is equal to its population size the previous year multiplied by its population growth rate, λ. Mathematically, population dynamics can be written as $N_{t+1} = \lambda N_t$, where N_t is the population size in year *t*. With the addition of fatalities from wind, $N_{t+1} = \lambda N_t - c\lambda N_t$, where $c\lambda N_t$ is the number of individuals killed by wind turbines between *t* and $t+1$. With minor mathematical rearrangement, $N_{t+1} = (1-c)\lambda N_t$ and the growth rate of a population experiencing a turbine mortality rate of *c* is equal to the population growth rate without fatalities caused by wind turbines multiplied by $(1-c)$. This population growth rate can be written $\lambda_w = (1-c)\lambda_b$, where λ_w is the growth rate of a population including the turbine mortality rate and λ_b is the growth rate of the baseline population, with no fatalities from wind energy facilities. It follows that the change in population growth rate is equal to $c\lambda_b$. To estimate the change in population growth rate, the turbine mortality rate must first be estimated.

Current and projected future (see section 3.0) turbine-collision mortality rate estimates can be calculated by dividing the number of individuals of each species estimated to be killed in turbine collisions each year by the estimated population size. The estimated number of individuals killed by wind turbines each year, *n*, is described in section 2.2.3.1. For birds, the range of uncertainty in the denominator of the turbine-collision mortality estimate FT (equation 1 above) spans an order of magnitude, stemming from the range used by the Partners in Flight Science Committee (2013) for total U.S. population sizes for each bird species (Rich and others, 2004; Confer and others, 2008). For bats, uncertainty on population size estimates will be extremely large, mainly because population sizes have not been estimated for most species.

To estimate the effects of collisions with wind turbines, random samples of population growth rate without fatalities from turbines are drawn from a normal distribution with mean and standard deviation from estimates in section 2.3.2, and random samples of turbine mortality rate are drawn from a triangular distribution, where the mode is *n* divided by the

Partners in Flight Science Committee's (2013) point estimate of population size and the end points from the order of magnitude uncertainty bounds. The change in population growth rate and resulting expected population growth rate with wind are then calculated for each set of samples in section 2.2.3 above, generating distributions for both the change and the expected growth rate. The methodology then estimates the probability that the population trend is <1 for both the original growth rate and the growth rate with the effects of wind turbines considered. The change between the observed and predicted probability that the population trend is <1 is reported as output. This output, the change in the probability that the population trend is <1, represents an estimate of the added risk that fatalities from wind turbines pose to a species' population trend.

In the demographic model component, collision fatalities are assumed to be additive to natural mortality, and awareness of this assumption is critical to interpreting the results. In reality, many species are likely to compensate for wind mortality by means of reductions in other sources of mortality or increased reproduction, and thus the estimates of change in trend are most likely overestimates. Thus, the results from this component should be considered estimates of the maximum possible change in population trend caused by the fatalities. The role of additive versus compensatory mortality on population dynamics is not well understood (Burnham and Anderson, 1984; Nichols and others, 1984); therefore, developing stochastic simulation models that included density dependence and compensation was not feasible in the generalized approach (appendix 2). The potential biological removal component, described next, is an alternative approach that attempts to include the role of density dependence while addressing collision fatalities.

2.4. Potential Biological Removal and Risk Ratio

2.4.1. Background

Potential biological removal (PBR) estimates indicate the total number of animals that could be killed as marine bycatch before a population would decline below a population size deemed sustainable, often considered half of a species' carrying capacity (Wade, 1998). Since its inception, using the PBR has become a standard approach for managing human-induced deaths of marine mammal species (Taylor and others, 2000), and scientists have studied various details of the approach, such as how it is used to predict risk (Wade, 1998; Milner-Gulland and Akçakaya, 2001), how parameters can be estimated (Niel and Lebreton, 2005; Dillingham and Fletcher, 2008), and how management decisions and assumptions of risk influence PBR assessments (Lonergan, 2011; Moore and others, 2013). In addition, the PBR and variations of it have been applied to bird species in relation to deaths caused by long-line fishing (Richard and Abraham, 2013), hunting (Runge and others, 2009), and wind turbines (Bellebaum and others, 2013). The PBR is calculated as

$$\mathrm{PBR} = F\frac{r_{max}}{2}N_{min} \tag{4}$$

where r_{max} is the maximum annual population growth rate under optimal conditions, N_{min} is a lower bound on an estimate of the population size, and F is a recovery factor, set by decision makers. In the PBR, $r_{max}/2$ represents the rate of take (number of individuals harvested, or killed) maximizing the net productivity of a population when logistic growth is assumed (Wade, 1998). The parameter F is essentially a safety factor set by decision makers (1) to adjust the value of the PBR to increase the rate of recovery of populations that are too small or (2) to account for uncertainties in the data used to calculate the PBR and to ensure that the PBR is not set too high. Under the Marine Mammal Protection Act (MMPA; 16 U.S.C. § 1362(20)), values for F can fall between 0.1 and 1.

In practice, estimated values of the PBR are compared to observed fatality rates to gage the level of risk for a species. If fatalities are well below the PBR, then a population should remain above an acceptable level and be at low risk. In a study by Richard and Abraham (2013), the ratio of fatalities to the PBR was calculated by using Monte Carlo approaches and was called the "relative risk"; this ratio is called the risk ratio (RR) in the USGS methodology. When the value for the RR is much less than 1, fatalities are much lower than the PBR and a species is at low risk. As RR values increase, risk increases, and at values at or above 1, the PBR is either met or exceeded by fatalities and the species is at high risk.

Currently, both the National Marine Fisheries Service and the U.S. Fish and Wildlife Service calculate PBRs for marine mammal stocks, including cetaceans, pinnipeds, sea otters (*Enhydra lutris*), polar bears (*Ursus maritimus*), and West Indian or American manatees (*Trichecus manatus*). They follow the regularly updated "Guidelines for Assessing Marine Mammal Stocks" (GAMMS; National Marine Fisheries Service, 2015) to perform these assessments. This methodology adopts these guidelines because of their historical and current use by the Federal Government and the levels of research and effort used to regularly update them. For each species with sufficient data, the PBR is calculated by using a Monte Carlo approach that includes the uncertainty associated with r_{max}.

The maximum rate of population growth under optimal conditions, r_{max}, is difficult to estimate because, for most species, populations are rarely observed existing in optimal conditions. Estimates of r_{max} are available for a few species, such as bacteria in petri dishes, species expanding into new areas, or those recovering from a population crash or overharvest (Gedamke and others, 2009). A number of approaches exist to estimate r_{max} (Slade and others, 1998; Millar and Meyer, 2000; Niel and Lebreton, 2005; Gedamke and others, 2007), and the assessment could use any of these depending on the data available for a species. For example, if a bird species has temporal data showing its recovery after reaching a small population size, the methods in Gedamke and others (2007) or Millar and Meyer (2000) might be useful. Runge and others (2009) used the method of Slade and others (1998) to estimate r_{max} for black vultures (*Coragyps atratus*). Runge and others (2009) assigned probability distributions to input variables and used Monte Carlo simulations to propagate that uncertainty when estimating r_{max}. Ultimately, the final decision on the approach used to estimate r_{max} for a species will be a key undertaking of the assessor and assessment panel. For marine mammals, GAMMS recommends using default values previously developed for groups of species (for example, pinnipeds or cetaceans) in the absence of other compelling data. It may be possible to develop default values for groups of birds and bats on the basis of body size and other life history parameters, but the authors of this report are not aware of such values being estimated.

As with r_{max}, N_{min} can be calculated by using a variety of approaches that are often dependent upon the data available. In the Marine Mammal Protection Act (16 U.S.C. § 1362(27), as amended in 2004), N_{min} is defined as follows:

> (27) The term “minimum population estimate” means an estimate of the number of animals in a stock that—
>
> (A) is based on the best available scientific information on abundance, incorporating the precision and variability associated with such information; and,
>
> (B) provides reasonable assurance that the stock size is equal to or greater than the estimate.

Taylor (1993) and Wade (1998) used simulations to study how setting the lower limit of population size (on the basis of a percentile from the distribution around an average abundance) affected how a PBR could be used to meet conservation criteria. Wade (1998) showed that setting N_{min} to the 20th percentile (the lower bound of a log-normal 60-percent confidence limit), resulted in 95 percent of simulation runs remaining at or above the target population size and the recovery of populations in simulations where population sizes were started below the target population size. The target population size was defined as the population size that resulted in the “optimum sustainable population,” which is defined in the Marine Mammal Protection Act as—

> (9) … the number of animals which will result in the maximum productivity of the population or the species, keeping in mind the carrying capacity of the habitat and the health of the ecosystem of which they form a constituent element.

The 20th percentile approach is now recommended by GAMMS and is formalized into a specific equation for calculating N_{min}:

$$N_{min} = N/e^{\left(0.842\left(\ln\left(1+CV(N)^2\right)\right)^{1/2}\right)} \tag{5}$$

where N is an unbiased abundance estimate and $CV(N)$ is the coefficient of variation of the abundance estimate. When possible, the assessors will use this method when estimating N_{min}.

The term F was originally described as a recovery factor (Wade, 1998), but it more broadly represents management goals and has biological implications (Runge and others, 2009). For example, values of F between 0 and 2 result in PBR values that, if met, will achieve a sustainable population. The resulting sustainable population size will vary with F. Assuming linear density dependence of the population growth rate, at $F = 0$ and 1, the population will equilibrate at carrying capacity (K) and $K/2$, respectively. As F increases towards 2, fatalities approach r_{max}, and at $F = 2$, the PBR is $r_{max} \times N_{min}$ and theoretically the population size should decline to 0. Thus, values of F near 2, but less than 2, would result in very small equilibrium population sizes, well below the population’s carrying capacity.

The relation between F and the equilibrium population size can be used to set F for particular policy objectives. Under the MMPA, F is restricted to be between 0.1 and 1. This range in F is based primarily on extensive simulation studies of marine mammal populations (Barlow and others, 1995; Wade, 1998) and the management objectives defined in the MMPA. The results of the GAMMS simulation analyses suggest a value for F of 0.1 for stocks considered endangered; 0.5 for stocks considered depleted, threatened, or unknown; and 1.0 for stocks known to be at their target population size, stocks that are increasing, or stocks that are not decreasing and are harvested only by subsistence hunting.

This methodology will follow the GAMMS suggestions for F with modifications for birds and bats. Endangered species are assigned a value for F of 0.1 to reflect a management objective of recovery. Threatened species, species showing population declines, or species with an unknown population status are assigned a value of 0.5. Species that are not listed, exhibit stable or growing populations, or are not considered overabundant are assigned a value of 1. Finally, overabundant species with management goals directed towards reducing population size are assigned a value of 1.5.

2.4.2. Implementation

To assess the risk of population decline for a species, the assessment will calculate the risk ratio (RR), which is simply the annual estimated fatalities divided by the PBR (fig. 6). The risk ratio is the proportion of the PBR accounted for by

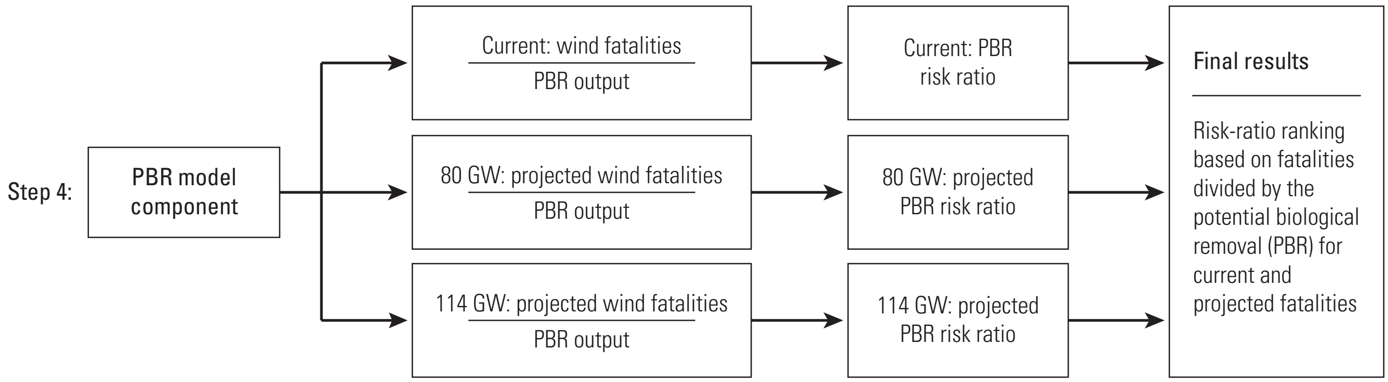


Figure 6. Flowchart of the PBR model component of the assessment methodology. GW, gigawatts; PBR, potential biological removal.

fatalities at wind energy facilities. As this value increases from 0, the risk goes up; at a value of 1, fatalities are equal to the PBR. When the RR is greater than 1, fatalities are greater than the PBR, and the population is expected to decline. Uncertainty in the RR will be estimated by sampling across the uncertainty in the PBR and the uncertainty in estimated fatality to calculate a mean RR and an accompanying confidence interval.

As with the FT metric in the prioritization component (section 2.2.3.1), the observed number of fatalities is currently calculated by using an approach that does not correct for differences in species detection rates and can cause significant biases in the estimated number of fatalities. The use of these current estimates could lead to unreliable assessment results at the national or regional level.

The methodology includes the RR to allow the identification of species at different levels of risk. However, the broader implications of the RR and PBR for the overall status of a species are critical to consider. A species with a low RR may still be imperiled because fatalities from other sources push the species above the PBR, or because other processes not measured by the PBR, such as habitat loss, are affecting a species. Thus, the RR measures only the impact of fatalities from wind turbines in the United States on a species, not the effects of other sources of fatality or other effects of wind turbines.

During an implemented assessment, a number of activities would occur to better improve estimates of the PBR. First, assessors would meet with members of regulatory agencies to better define management objectives and possibly revise the default values of F. Second, as r_{max} is estimated for more species and the range of r_{max} values is better understood, simulation studies similar to those performed during development of the GAMMS recommendations could be performed to determine values of F and N_{min} that would meet established management objectives. Third, a meta-analysis of r_{max} estimates could be conducted to perhaps estimate defendable default values of r_{max} by taxon or life-history type. Fourth, the nature and shape of density dependence in birds and bats would require study. Most PBR applications assume linear density dependence, and the methodology currently follows this assumption. Additional research on birds and bats could help verify or refine this assumption.

3.0. Future Projections

The methodology uses projections of newly installed capacity of wind energy facilities to extrapolate the impacts of future wind development on wildlife. Both the demographic model and the risk ratio are rerun with the updated fatality estimates based on the projections. A key assumption is that fatalities will increase as additional turbines are installed. This may not be true if new facilities can be developed in a manner that avoids and minimizes impacts to wildlife.

A number of organizations have projected future levels of installed wind capacity for the United States. These projections produce a single estimate of new capacity for the entire Nation. An exception is a study by the National Renewable Energy Laboratory (NREL; Hand and others, 2012), which estimated new wind capacity for each State under different levels of national renewable energy production. The NREL researchers investigated varying levels of new capacity and used an economic-demand-based model to predict installed capacity of wind energy by State through time.

Projecting future levels of annual fatalities from wind energy development involves taking existing fatality rates (deaths per megawatt per year) and adjusting them on the basis of predicted levels of installed wind turbine capacity in megawatts (MW). The most basic approach would simply multiply existing fatalities by the percent increase in installed capacity (for instance, if 100 animals die per year and installed capacity will double, then 200 animals are projected to die per year).

The approach in this methodology differs between birds and bats because there is more statistical modeling of bird fatality than bat fatality. Estimates of overall bat fatality exist and suggest that fatalities may be influenced by a number of factors (Baerwald and Barclay, 2009; Arnett and Baerwald, 2013; Hayes, 2013; Smallwood, 2013; Huso and Dalthorp, 2014). However, statistical models of these factors do not exist and cannot be used for projecting. Bat fatality data are being compiled to allow such modeling (see appendix 1), and these models may be available by the time an assessment is implemented. For now, only raw extrapolations based on national increases in wind energy production can be performed for bats, and extrapolations will be referred to from here on.

For birds, the approach uses an existing statistical model of fatalities (Loss and others, 2013). The model requires an estimate of the number of turbines and their heights in each of four regions of the contiguous United States (California, East, Great Plains, and West excluding California, as in figure 1 of Loss and others, 2013). To calculate these estimates, the approach first partitioned the projected new capacity among the regions and added this to the existing capacity in each region. Then each regional capacity was converted into a prediction of the number of turbines by dividing the regional capacity by an estimate of the average size (in megawatts of capacity) of each turbine. Finally, because turbine height is correlated with turbine capacity, this relation was used to predict the average turbine height in each region. These values were then used in the statistical model to project future fatalities.

Thirty-five projections of future wind energy capacity in the United States in 2025 were chosen to estimate medium and high build-out scenarios, which are estimates of the amount of new wind energy facilities installed in the United States. Of the 35 projections, 3 were developed by private companies, 2 by the International Energy Agency (IEA), and 30 by the U.S. Energy Information Administration (EIA); details for the projections were provided by the U.S. Energy Information Administration (2014a) and the International Energy Agency (2012). The build-out scenarios developed by the EIA included factors such as energy demand, regulatory limits on greenhouse-gas emissions, and changes in the amount of electricity produced by nuclear power. The medium scenario was defined as the average installed capacity across all 35 projections (80 GW), whereas the high scenario represented the value at the 95th percentile of the distribution of projections (114 GW). The projected installed capacities (80 and 114 GW) were national estimates that were partitioned across the regions of the United States (California, East, Great Plains, and West excluding California) used in the statistical models developed by Loss and others (2013). To do so, the projected additional national capacities (80 − 61 GW = 19 GW or 114 − 61 GW = 53 GW) were multiplied by the predicted proportions of capacity in each region in the Loss and others (2013) models and then added to the existing installed capacity in each region (fig. 7).

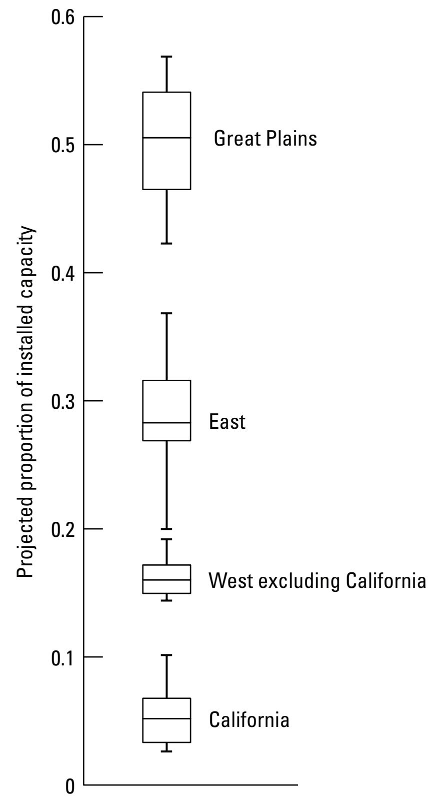


Figure 7. Boxplot showing projected proportions of wind energy capacity based on 33 projections for 2025 by the National Renewable Energy Laboratory (Hand and others, 2012), partitioned by U.S. region (from Loss and others, 2013). The box contains the middle 50 percent of values, with a line at the median. Whiskers mark the maximum and minimum values. Average values were used when projecting future wind capacity in each region.

The proportional estimates came from predictions of installed capacity in each State from 33 simulations of wind energy growth modeled by the NREL (Hand and others, 2012). Predictions for each State were summed within a region, then the proportion of the total installed capacity was estimated for each region. These proportions were then averaged across the 33 simulations.

Once national projections of additional installed capacity were allocated to each region, the regional capacity values (in gigawatts) were converted to the number of turbines expected to be installed in each region. Current information on the average capacity of turbines (in megawatts) and its trend through time was used to convert regional capacity (in gigawatts) into the number of turbines. A linear increase in average capacity of turbines was assumed on the basis of the rate of increase in average capacity of active turbines installed in each region since 2005 (fig. 8). The projected future regional capacity was then divided by the estimated capacity per turbine to determine the number of installed turbines in each region.

Because increasing turbine height correlates with increases in fatalities (Loss and others, 2013), we also modeled the projected height of future turbines. Turbine height is related to turbine capacity (fig. 9). A power function was fitted to the heights and capacities of turbines that were active in each region in 2014. This relation was used to estimate average turbine height based on average capacity (in megawatts) in each scenario.

The predicted number of turbines in each region and their associated heights for each wind energy projection scenario were input into the statistical model developed by Loss and others (2013) to estimate the number of annual avian fatalities expected at onshore windpower facilities in the United States in 2025 (table 2). Although it would be possible to develop more sophisticated methods relying on distributions of turbine

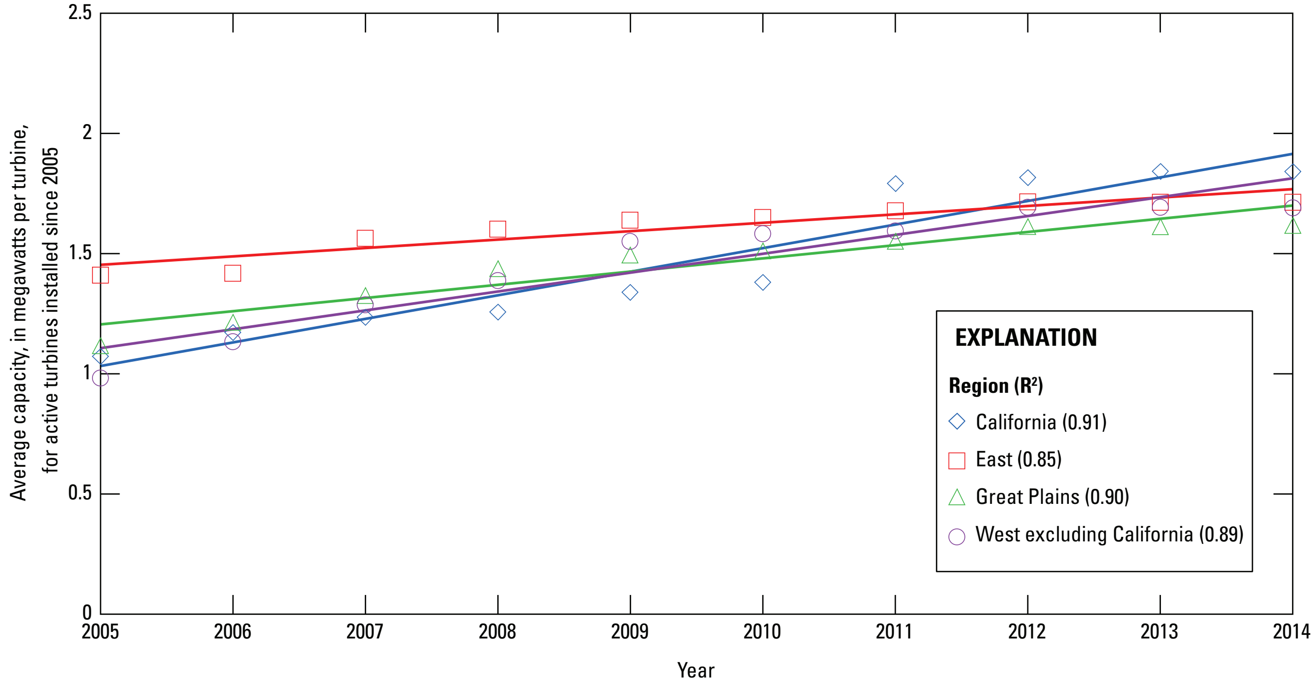


Figure 8. Graph showing the average turbine capacity of active turbines installed since 2005 in the four regions of Loss and others (2013) in the contiguous United States. The lines represent a linear regression run for each region. R^2 is the coefficient of determination and indicates how well a model fits the data.

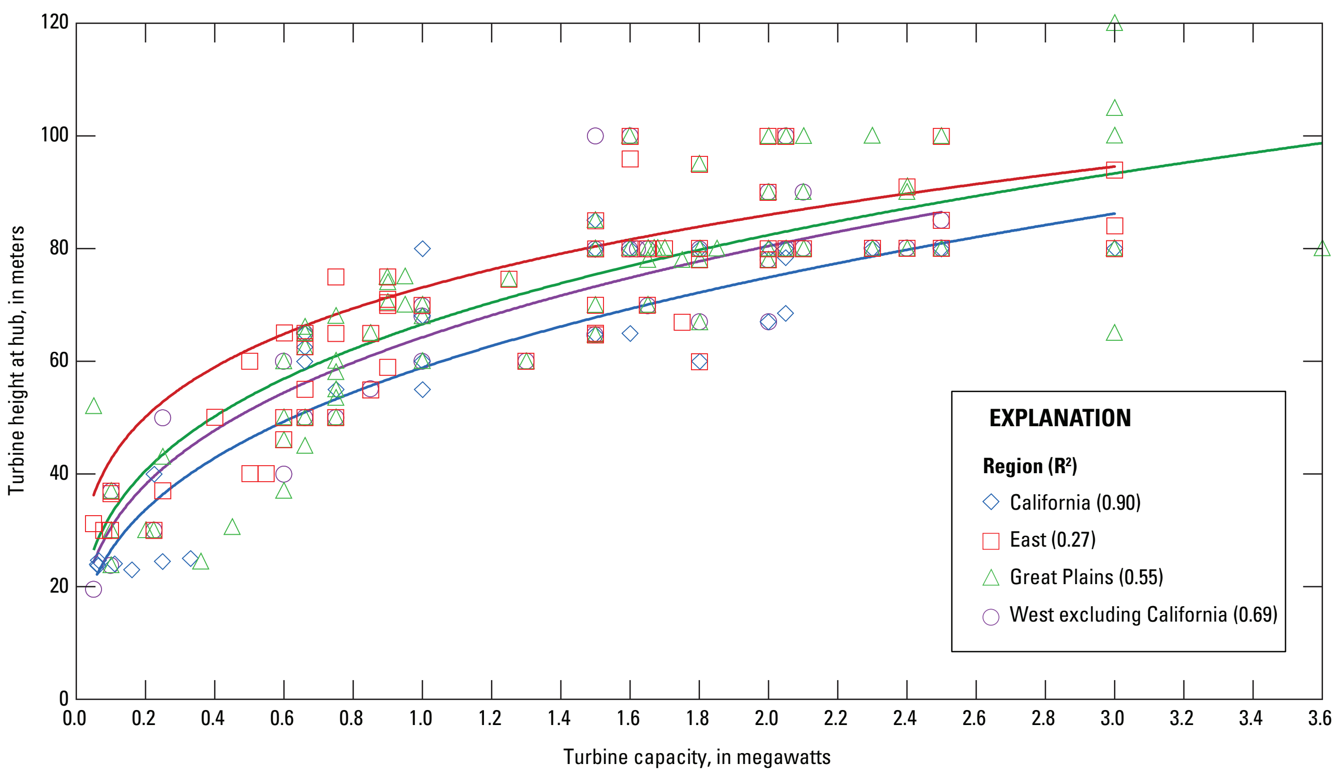


Figure 9. Graph showing the relation between turbine height (hub height) and capacity in the four regions of Loss and others (2013) in the contiguous United States for turbines active in 2014. Curved lines represent power functions (linear regression on a log-log scale) relating turbine capacity to hub height in each region. R^2 is the coefficient of determination and indicates how well a model fits the data.

Table 2. Projected number of annual avian fatalities expected at onshore windpower facilities in the United States in 2014 and in 2025.

[GW, gigawatt; max, maximum; min, minimum; NA, not applicable]

Year (installed-capacity scenario)	Mean annual avian fatalities (min–max)	Increase in fatalities (percent)
2014 (62.3-GW capacity)	182,000 (123,000–240,000)	NA
2025 (80-GW capacity)	233,000 (115,000–323,000)	28.0
2025 (114-GW capacity)	333,000 (206,000–460,000)	83.0

capacities and heights, assumptions about the shapes of these distributions would be speculative. Because estimates produced using mean heights and capacities for current turbines differed by less than 3 percent from those produced using numbers of turbines of different heights and capacities, the added detail is unlikely to greatly improve the estimates of fatalities.

4.0. Overall Methodology Development and Validation

As noted in section 1.3, the methodology has a number of elements that should be checked during the implementation of an assessment. Furthermore, specific elements of the methodology will need to be refined if an assessment is implemented. These refinements include guidelines for (1) estimating demographic parameters when data are sparse, (2) developing species distribution maps by a common method, (3) estimating species-specific fatalities, and (4) selecting species after prioritization.

While developing the methodology, specific approaches were implemented for the case study species described below, but some of the approaches used may not be optimal as data on more species are used in the methodology. For example, the components of the methodology produce consistent results across the case study of six species, but as more species are prioritized and their data are used in the demographic model and risk ratio components, consistency across the components should be checked. In addition, the results of the demographic and PBR components could be compared with the outcomes of more detailed research as they become available. Customized population models are being developed for several species that would also be assessed with this methodology, including the golden eagle and whooping crane (Butler and others, 2013), and these models could be used to check that the simpler models used in the methodology produced qualitatively consistent results. If consistency declines, modifications may be necessary, depending on the reasons.

The methodology would also be much improved, and would produce more reliable results, if four key issues could be resolved. First, robust, species-specific fatality estimates do not currently exist. The current approach used to estimate species-specific fatality rates (section 2.3) may produce unknown biases because it does not account for species detectability. Second, population-size estimates are poor for most species, particularly bats. The uncertainty around the turbine mortality rate could be decreased considerably with better population-size information. Third, range maps describing a species' relative abundance across space and through time are not currently available for most species. Because these maps would be used to compare species, a consistent method to produce them on the basis of widely available location data (for example, from eBird; Cornell Lab of Ornithology, 2015) would be an invaluable contribution to the assessment. Fourth, the methodology would be improved if scientists could develop a greater understanding of the role of additive versus compensatory mortality in birds and bats. The methodology currently uses the assumption that all of the fatalities from wind energy facilities are additive. This assumption is likely violated for many species, but assuming that fatalities from wind energy facilities are compensatory also essentially means that one assumes that the fatalities from wind energy facilities have no effect on a species. Instead, the methodology evokes a precautionary assumption, and this should be considered when interpreting the results of the methodology.

Finally, during an official assessment, the methodology would require input from decision makers at three points in the process. These inputs are unavoidable and required for those steps in the methodology that are primarily policy, not scientific, decisions. First, the ranking systems in table 1 are subjective and currently based on the authors' perception of a need to consider the effects of wind energy development ahead of conservation status. This ranking system requires external review and adjustment prior to an assessment. Second, a decision must be made about what species should be further analyzed after prioritization; that is, decide how far down the list of average ranks of relative risk the assessment should proceed because species below the designated stopping point will not be included in the demographic model or risk-ratio analyses. Third, the methodology currently uses the guidelines associated with marine mammals as default values for F when calculating the PBR. The parameter F is associated with levels of acceptable risk and should be set by regulators, not the USGS. Thus, the USGS would require input from regulatory agencies to determine whether the existing values of F are appropriate.

5.0. Test Case Results

The results described here should not be considered definitive statements of risk associated with wind energy development. The input data were gleaned from a variety of sources and not thoroughly vetted as they would be during a formal assessment. Because of this, species names are not reported. The results presented here give readers an example of the output an assessment would produce, but they may not correctly characterize the impact of wind energy development.

5.1. Introduction

Data for six avian species were processed through the complete assessment methodology to demonstrate the required input parameters and the methodology outputs. These species were selected because data were readily available and because analysis of these species may produce a diverse range of potential results in the assessment. The following section presents test case results from the prioritization method, the demographic model, and the risk ratio. The results are shown for three scenarios defined by installed capacity as follows: (1) current (2014) wind energy development (62.3 GW), (2) moderate wind energy development for the year 2025 (80 GW), and (3) high wind energy development for the year 2025 (114 GW). See section 3 for details on the methods used to develop the scenarios.

5.2. Data Sources

Published research, data from resource-management agencies, and reference works were used to develop the parameter values. Conservation-status information was found at Web sites and in reports of the IUCN (International Union for Conservation of Nature, 2014), the USFWS (U.S. Fish and Wildlife Service, 2014), and individual State wildlife action plans (Association of Fish and Wildlife Agencies, 2015). Population size (N) for all species came from the Partners in Flight Science Committee (2013), and 30 percent of the population size estimate was used as a rough approximation of N_{min}. Either direct estimates or data used to calculate demographic parameters (survival, maternity, r_{max}, and age at first reproduction) were found in Poole (2005) and Michel and others (2006). When possible, maternity (m) was calculated as the product of nest success, clutch size, hatchability, clutches per year, and presumed equal sex ratios at hatching. If data on hatchability or the number of clutches were lacking, the parameters for them were set to 0.9 and 1, respectively. The age of first reproduction (a) for these species was presumed to be the first breeding season after fledging unless evidence suggesting otherwise was available.

The percentage of a species' range that overlaps with the locations of wind turbines (p) was calculated as described in section 2.4, using North American Breeding Bird Survey relative abundance maps (Sauer and others, 2015) and the USGS wind turbine location data (Diffendorfer and others, 2014). The numbers of fatalities (n) were estimated from turbine mortality data compiled from multiple sources by Loss and others (2013) following the approach described in section 2.2.3.1.

5.3. Prioritization Results

Input data, calculated conservation status, and turbine-risk metrics for the prioritization component are listed in tables 3 and 4. Although the data sources described here may be the same as those used in an actual assessment, a vetting process by an assessment panel would ensure that the data were appropriately selected and used. The vetting process was not included in this test case.

Of the six species investigated by using the prioritization method, three had average direct-risk ranks of less than 5, and two had average indirect-risk ranks of less than 5 (table 4). Raptor 1 ranked the highest of all species for average direct risk because it had the highest FT and FRI. The corvid and raptor 2 ranked high on direct risk because of their high FRI values. Both songbirds ranked low on direct risk despite an elevated conservation status because few die as the result of collisions with turbines, and their "fast" life histories resulted in low FRI values. Species that use fewer habitats and that have conservation status (both songbirds) scored higher on the average indirect risk. The three demonstration species with direct-risk ranks of less than 5 were evaluated by using the demographic model and potential biological removal.

5.4. Demographic Model Results

Following the high-priority ranking from the prioritization step, the corvid and both raptors were analyzed with the demographic model. The input parameters that could be used for demographic modeling are provided in table 5. The analysis was conducted following the process described in section 2.3.

The key results of the demographic model are presented in table 6. Because all three species had adequate time series to calculate an observed trend, the matrix model was not used. Note that the trend was calculated by using data up to 1990, prior to large numbers of turbines being installed, but this endpoint may not be appropriate in an actual assessment given potential changes in population status since 1990 due to factors other than wind energy development. The change in the population trend and the resulting projected trend with additional mortality from wind energy development are reported as means with 90-percent confidence intervals for each of the three scenarios. Finally, the increase in the percentage of population growth rate values that were less than 1 is reported along with the percentage of population growth rate values less than 1 with wind-facility-related mortalities for each scenario.

The corvid showed very small responses to fatalities from wind turbines, even in the wind-energy-development scenarios involving a high installed capacity. The corvid ranked

Table 3. Input data for calculated conservation status and direct- and indirect-risk metrics for the prioritization component of the test case.

[Species are anonymized to reduce the chance of inferences being developed prior to full vetting of the input data and the methodology. No., number; SGCN, species of greatest conservation need]

Species	No. of States that consider species as an SGCN	No. of States in which species occurs	No. of turbine collision fatalities for species, annually (*n*)	Population size of species in United States, in millions (*N*)	Adult survival (*s*)	Percentage of species range overlapping with locations of wind turbines (*p*)	Maternity (*m*)	Age at first reproduction (*a*)	No. of habitats (*h*)
Corvid	2	49	260	17	0.865	4.04	0.60	2	7
Raptor 1	14	49	7,594	1.7	0.490	5.42	1.29	1	10
Raptor 2	34	50	519	0.5	0.700	4.47	1.02	1	8
Raptor 3	1	49	4,933	5	0.770	4.80	0.51	1	6
Songbird 1	24	48	130	0.8	0.529	3.74	1.33	1	2
Songbird 2	29	46	1,363	11	0.610	2.89	0.99	1	3

Table 4. Output values of conservation status and direct- and indirect-risk metrics and the resulting average rank for risk from collision fatalities and risk from habitat change (lower values indicate higher priority) from the prioritization approach.

[IUCN, International Union for Conservation of Nature; NA, not applicable; percent SGCN, percentage of States where species is present that consider it a species of greatest conservation need (SGCN); USFWS, U.S. Fish and Wildlife Service]

Species	Conservation status			Percentage of fatalities in United States caused by turbines (FT)	Fatality-risk index (FRI)	Indirect-risk index (IRI)	Average rank for direct risk	Average rank for indirect risk
	Percent SGCN	USFWS designation	IUCN designation					
Corvid	4.08	NA	Least concern	0.0113	13.518	0.578	3.84	7.54
Raptor 1	28.57	NA	Least concern	0.8751	10.627	0.542	2.93	6.87
Raptor 2	68.00	NA	Least concern	0.3462	4.393	0.558	4.14	5.48
Raptor 3	2.04	NA	Least concern	0.4280	8.334	0.800	5.41	7.04
Songbird 1	50.00	Bird of conservation concern	Least concern	0.0345	2.173	1.869	6.72	2.25
Songbird 2	63.04	Bird of conservation concern	Least concern	0.0318	2.934	0.964	5.99	4.45

Table 5. Parameter inputs for the matrix model for three prioritized species in the test case investigation.

[Nest success is the proportion of nests that hatch at least 1 egg. Hatchability is the proportion of eggs that hatch in successful nests. No., number; NA, not available]

Species	No. of turbine fatalities	Population size of species in United States	Juvenile survival (s_j)	Adult survival (s_a)	Turbine mortality rate (c)	Age at first reproduction (years)	Clutch size	Nest success	Hatchability	Clutches per year	Maternity (m)
Corvid	260	1.7×10^7	0.63	0.86	0.002	2	4	0.33	0.91	1	0.60
Raptor 1	7,594	1.7×10^6	0.34	0.49	0.447	1	4.6	NA	NA	1	1.29
Raptor 2	519	5.0×10^5	0.41	0.70	0.104	1	4.4	0.51	NA	1	1.02

Table 6. Results of the demographic model for 2014 and projected levels of wind development.

[The observed population trend, projected decrease in population trend, and expected population trend are reported with 90-percent confidence intervals. Installed capacity in gigawatts (GW) for each scenario is as follows: current (2014) scenario, 62.3 GW; medium-capacity scenario for 2025, 80 GW; high-capacity scenario for 2025, 114 GW. w/o, without; λ, population growth rate; <, less than]

Observed population trend w/o wind energy development, 1966–1990	Scenario	Projected decrease in population trend	Expected population trend with wind energy development	Projected increase in percentage λ<1	Expected percentage λ<1
		Corvid			
1.005 (0.983–1.027)	Current	7×10^{-5} (2×10^{-5}–2×10^{-4})	1.005 (0.983–1.026)	0.2	35.1
	Medium	1×10^{-4} (2×10^{-5}–2×10^{-4})	1.005 (0.983–1.026)	0.3	35.2
	High	1×10^{-4} (3×10^{-5}–3×10^{-4})	1.005 (0.984–1.026)	0.4	35.3
		Raptor 1			
0.989 (0.950–1.028)	Current	0.016 (0.003–0.035)	0.973 (0.931–1.014)	18.2	85.7
	Medium	0.022 (0.004–0.047)	0.967 (0.923–1.011)	21.1	88.6
	High	0.031 (0.007–0.067)	0.958 (0.907–1.006)	25.0	92.5
		Raptor 2			
0.995 (0.912–1.077)	Current	0.004 (0.001–0.008)	0.991 (0.909–1.073)	3.1	56.9
	Medium	0.005 (0.001–0.012)	0.990 (0.907–1.072)	4.1	57.9
	High	0.007 (0.002–0.016)	0.988 (0.905–1.070)	6.0	59.8

highly during prioritization because its life-history parameters produced a high FRI. However, relative to its population size, a small fraction of individuals were projected to be killed (table 5), resulting in small estimates of added mortality and minimal changes in population trend. The projected declines in population trend for the corvid ranged from 0.00002 to 0.0003 across current and projected levels of wind energy generation (table 6).

Both raptors had long-term declines in population size with observed mean population trends less than 1. Current levels of wind energy facilities had a larger negative effect on raptor 1 (mean decline=0.016) than raptor 2 (mean decline=0.004). For raptor 1, with current levels of wind energy facilities, the mean observed population trend was projected to decrease from 0.989 to 0.973, while raptor 2 declined from a mean of 0.995 to 0.991. As expected, declines in population trend increased with higher levels of wind energy development. For raptor 1, the model also projected large increases in the percentage of population growth rates less than 1 between the trends without and with wind energy development.

5.5. Potential Biological Removal and Risk Ratio

As with the demographic model, the three high-ranking species from prioritization were analyzed by using potential biological removal (PBR) and risk ratios. The three species showed considerable differences in both the PBR and the risk ratio (table 7).

The corvid had the largest PBR across all values of *F*, followed by raptor 1, then raptor 2 (fig. 10). To calculate risk ratios, an *F* value of 0.5 was selected because all three species showed declines or uncertain trends in the last 11 years of available BBS data (2002 through 2012; Sauer and others, 2015).

With *F* at 0.5, raptor 1 had a high risk ratio and the confidence interval overlapped 1 at current levels of wind energy development (fig. 11). Under both projections of future wind energy development, the best estimates of the risk ratio were at or over 1. Raptor 2's risk ratio at current levels of wind energy development was moderate, and the upper confidence interval almost overlapped 1. The risk ratio increased with high levels of projected wind energy development. Finally, the corvid risk ratio was near 0, indicating low risk for this species.

This demonstration of the methodology indicates that of the six species originally considered, three were prioritized for a more in-depth investigation. Of those three, raptor 1 indicated a higher risk than the other species both in potential declines in population trend and in the risk ratio. Raptor 2 indicated moderate levels of potential decreases in trend and a moderate to high risk ratio. The corvid was prioritized on the basis of its high FRI score, but because its fatalities were projected to be low relative to its population size, it showed negligible effects from wind turbines.

5.6. Implications of the Test Case

The USGS approach outlined here provides a quantifiable and replicable means of determining the relative risk and subsequent impact of adverse effects of wind energy facilities on volant wildlife. Output from the approach (risk indexes and estimates of the change in population trend for current and future projected wind energy development) could be provided for all high-priority bird and bat species for which sufficient data have been gathered. This approach relies on the development of new sources of information, such as species-specific fatality data and a suite of basic population parameters for the less studied bat species affected by wind energy generation. This approach should be used iteratively, updating risk as new information regarding species exposure and wind energy capacity changes. An example of a potential summary table that includes the major final outputs for the test case species is presented in table 8.

Table 7. Parameter inputs and outputs for potential biological removal and risk ratios at current levels of wind energy development for three prioritized species.

[Values in parentheses represent the 95-percent confidence interval. *F*, recovery factor; N_{min}, minimum population size; PBR, potential biological removal; r_{max}, maximum annual population growth rate; risk ratio, turbine fatalities divided by the PBR]

Species	No. of turbine fatalities	*F*	r_{max}	N_{min}	PBR	Risk ratio
Corvid	260	0.5	0.07 (0.06–0.15)	5,100,000	89,250 (73,950–91,250)	0.003 (0.001–0.004)
Raptor 1	7,594	0.5	0.08 (0.03–0.10)	510,000	10,124 (3,824–12,750)	0.750 (0.463–2.055)
Raptor 2	519	0.5	0.030 (0.014–0.06)	150,000	1,106 (525–2,250)	0.469 (0.180–0.988)

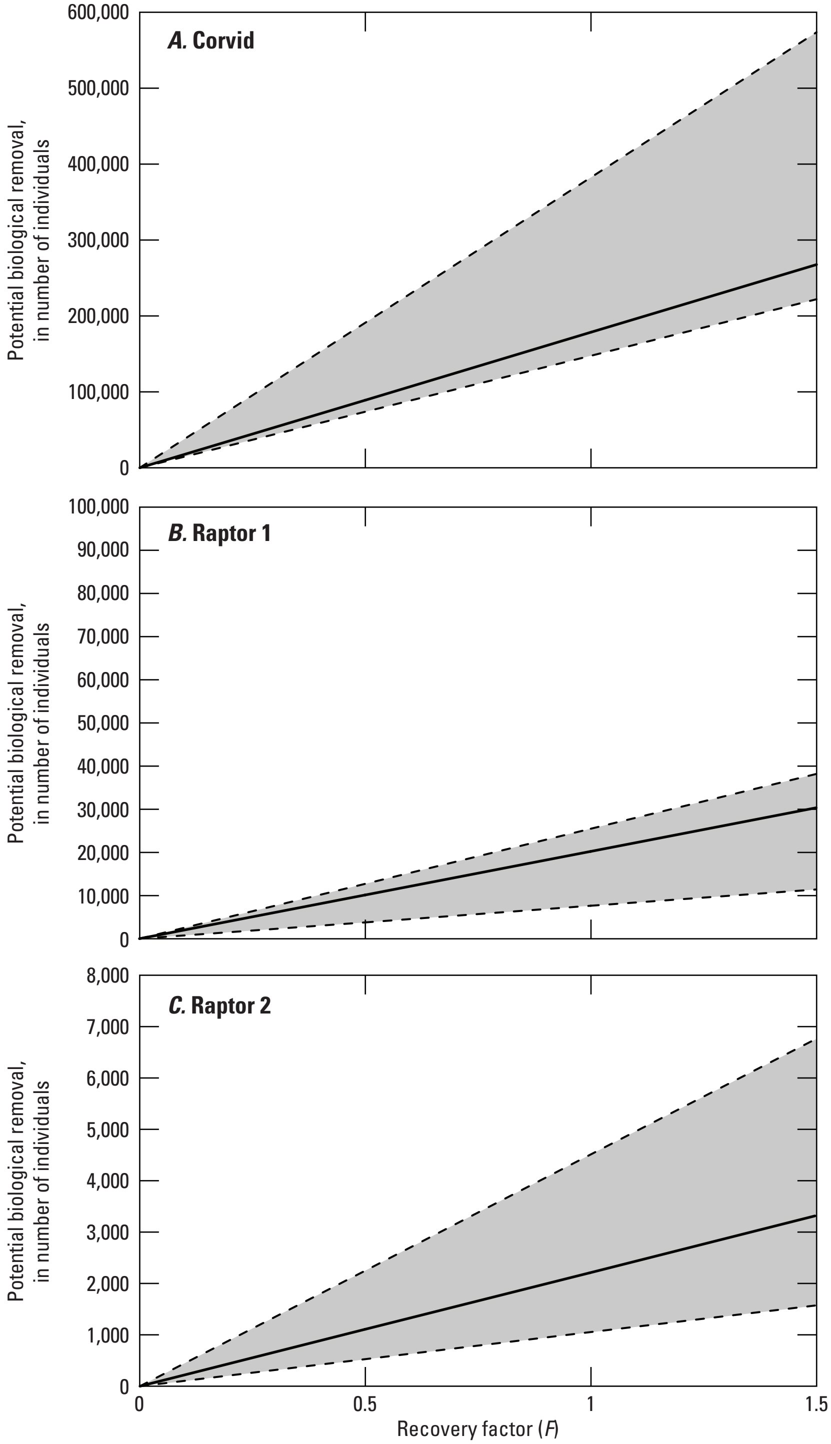


Figure 10. Graphs showing the relations between the estimated potential biological removal (PBR) and recovery factor (F) for three species at current (2014) levels of wind energy development. The solid black line represents the best estimate of the PBR; dashed lines represent the lower and upper confidence limits. *A*, Corvid; *B*, Raptor 1; *C*, Raptor 2.

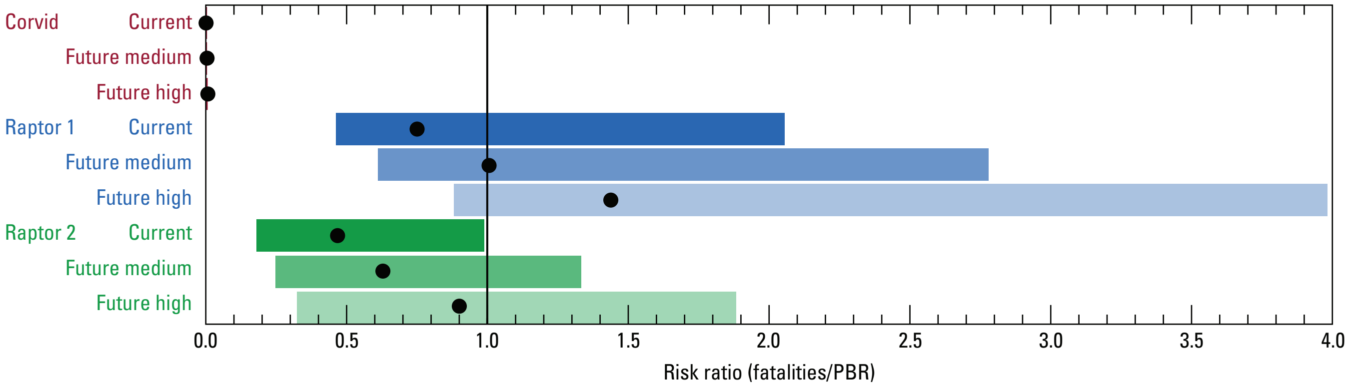


Figure 11. Graph showing the ranges of relative risk ratios when the recovery factor (*F*) equals 0.5 for three species at current (2014) levels of installed capacity and for both medium- and high-capacity scenarios for 2025. The black circles represent the best estimate, whereas the colored bars represent the range spanned by the upper and lower confidence intervals. The projected scenarios are as follows, by installed capacity in gigawatts (GW): current (2014), 62.3 GW; future medium (2025), 80 GW; future high (2025), 114 GW. PBR, potential biological removal.

Table 8. Summary table of the main output values for all components of the methodology: prioritization, the demographic model, and potential biological removal.

[Installed capacity in gigawatts (GW) for each scenario is as follows: current (2014) scenario, 62.3 GW; medium-capacity scenario for 2025, 80 GW; high-capacity scenario for 2025, 114 GW. Risk-ratio values are presented as an average, and values in parentheses represent the 95-percent confidence interval; risk ratios greater than 1 indicate a decrease in population size. NA, not applicable; PBR, potential biological removal; λ, population growth rate; --, species were not highly prioritized and, therefore, no quantitative model outputs are available]

Species	Scenario	Species prioritization component		Demographic model component	PBR model component
		Average rank for direct risk	Average rank for indirect risk	Projected increase in percentage $\lambda<1$	Risk ratio
Corvid	Current	3.84	7.54	0.2	0.003 (0.001–0.004)
	Medium	NA	NA	0.3	0.004 (0.001–0.005)
	High	NA	NA	0.4	0.006 (0.002–0.007)
Raptor 1	Current	2.93	6.87	18.2	0.750 (0.463–2.055)
	Medium	NA	NA	21.1	1.006 (0.611–2.780)
	High	NA	NA	25.0	1.438 (0.882–3.983)
Raptor 2	Current	4.14	5.48	3.1	0.469 (0.180–0.988)
	Medium	NA	NA	4.1	0.629 (0.248–1.333)
	High	NA	NA	6.0	0.900 (0.325–1.884)
Raptor 3	Current	5.41	7.04	--	--
Songbird 1	Current	6.72	2.25	--	--
Songbird 2	Current	5.99	4.45	--	--

6.0. Conclusions

The USGS Wind Energy Impacts Assessment Methodology (WEIAM) project team created an assessment methodology to provide both qualitative and quantitative metrics related to the effects on birds and bats from wind energy development at the national scale and species level. This work built on a variety of quantitative approaches focused on ecological risk assessment to provide a regional to national perspective that will inform decision makers, industry, and the public. The assessment methodology described in the preceding sections is an initial attempt to quantify the significance of the effects of wind energy development on species population trends. Uncertainty is captured in the input ranges for the model parameters and reflected in the probabilistic assessment output. The test case provided is only an example of the method and is not an official assessment result. As the science of impact research matures, this method may be updated and improved to reflect new knowledge in this rapidly growing field of study. A national assessment of impacts to species is not currently part of this project; however, the creation of a methodology and process for assessment is a fundamental first step in any USGS assessment project.

The model described here and the resulting output should be considered only for a scientific assessment, not for a method to develop management strategies related to wind energy development for a particular species. The model produces an estimate, with uncertainty, of the effects on population trend or a risk ratio from observed fatalities at wind turbines. Decision makers prioritizing species for avoidance, minimization, and mitigation actions might use this information, and they might also include information about the feasibility, probability of success, and cost.

This approach should not replace detailed, species-specific studies or population models of those species garnering high levels of attention. Furthermore, the approach is not designed to estimate the total capacity levels of wind energy that could be installed across the Nation before species show population declines. Instead, the approach could help inform decisions related to—

1. Identification of those species at low risk from wind energy development.

2. Identification of those species that may be at risk from wind energy development.

3. Quantification of the expected decline in population trend for identified high-risk species from current and future levels of wind energy development.

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Glossary

additive mortality Mortality (often from humans) that occurs in addition to natural mortality. Additive mortality adds to the number of deaths that would have occurred naturally.

adult survival (s) Probability that an adult animal survives from one year to the next.

age at first reproduction (a) Age at which females become reproductively mature.

barotrauma Damage to body tissue caused by the difference in air pressure around the turbine blade.

capacity The rated maximum amount of energy a turbine is capable of generating.

compensatory mortality Mortality (often from humans) that somehow is compensated for by changes (reductions) in natural mortality. Compensatory mortality does not add to the number of deaths that would have occurred from natural causes.

current conservation status The methodology uses conservation status information from State wildlife action plans, where some species are considered “species of greatest conservation need”; other sources of information on conservation status include (1) the Red List of the International Union for Conservation of Nature (2014), (2) listing by the Endangered Species Act of 1973, as amended (16 U.S.C. §1531 et seq.), or (3) the list, “Birds of Conservation Concern 2008” (U.S. Fish and Wildlife Service, 2008).

demographic model A mathematical model used to generate quantitative and probabilistic estimates of impacts to species populations associated with the additional fatalities caused by collisions with wind turbines.

fatalities from wind turbines (n) The estimated number of fatalities per year caused by collisions with wind turbines. To date, n has been estimated by multiplying an estimate of total fatalities across all species by the proportion of the observed fatalities accounted for by a species.

fatality-risk index $\mathrm{FRI} = p/(m/a)$ where p is the percentage of a species’ range that overlaps with the locations of wind turbines, m is maternity (number of female offspring per adult female per year), and a is the age at which females become reproductively mature.

habitats (h) Number of habitats used by a species.

juvenile survival Chance of juveniles surviving from one year to the next.

maternity (m) Number of female offspring per adult female per year.

overlap with wind turbines (p) The proportion of a species’ range that overlaps with the locations of wind turbines.

population size (N) The total population size of the species that moves through the United States at some point in its life cycle.

population trend The increasing, decreasing, or stable pattern of the population of a species through time.

potential biological removal (PBR) An estimate of the total fatalities that can occur before the population of a species will decline below a target population size.

range map Geographic distribution of a given species.

species of greatest conservation need (SGCN) Species considered to have an elevated conservation status in State wildlife action plans.

species prioritization Qualitative approach to rapidly screen a large number of species and rank them in terms of the relative risk of a population-level consequence from wind energy development.

State wildlife action plan (SWAP) Proactive plan, developed as a comprehensive wildlife conservation strategy, that helps to conserve nongame wildlife and vital natural areas before they become too rare and costly to protect.

time series A group of data collected sequentially, usually at fixed intervals of time.

volant species Winged species capable of flying.

Appendixes 1 and 2

Appendix 1. Ongoing Research to Improve Future Methodologies

Introduction

Members of the U.S. Geological Survey's Wind Energy Impacts Assessment Methodology (WEIAM) project team investigated a number of approaches designed either to directly estimate the effects of wind energy development on wildlife or to understand key information gaps related to the issue of wind energy development's effects on wildlife. In addition to those initial investigations, research is ongoing and may help to improve future iterations of this methodology, as summarized in the following sections.

Flight Intensity as a Tool in Wildlife Impact Research

The use of airspace by flying animals is the nexus between wind energy facilities and the fatalities they cause. If a wind energy facility is located where few animals fly, then few fatalities are likely to result, whereas if a facility is located where flight activity by animals is intense, then there may be a much higher risk of fatalities. Accordingly, knowledge of airspace use is fundamental for assessing risks of wind energy development to wildlife. Identifying areas where flight intensity is high and where wind energy development has occurred or is likely to occur will greatly facilitate risk assessment. In terms of the methodology, increased understanding of airspace use relative to wind turbines would allow a more accurate estimate of population growth rate (λ) in the species prioritization metrics and would increase the accuracy of forecasts by predicting the proportion of a species' range that overlaps with the locations of wind turbines (p).

This research focuses on assessing the intensity of flight activity by animals and uses new types of information developed over the last decade. For example, the use of radiotelemetry unleashed a flood of information about how animals actually move (Duerr and others, 2012; Katzner and others, 2012; Lanzone and others, 2012; Miller and others, 2014). Fixed and mobile radar facilities provide information about mass movements of birds (Diehl and Larkin, 2005), as well as bats and insects (Kunz and others, 2008). New technologies such as acoustic monitors provide information on airspace use by bats (Britzke and others, 2013) and migrating birds (Blumstein and others, 2011), stable-isotope analysis of animal tissues sheds light on migration pathways of many species (Zimmo and others, 2012), and photosensitive geolocators can record information on approximate latitude and longitude traversed by animals (Stutchbury and others, 2009). Genetic markers, thermal cameras, Internet tools such as eBird (Cornell Lab of Ornithology, 2015), and cooperative monitoring projects such as the Nocturnal Flight Call Activity Index ("oldbird" at Old Bird, Inc., 2015) also provide information about movements of birds.

This research is evaluating the potential of the tools described above and data they generate for determining the intensity of low-elevation flight of birds, bats, and other flying animals. Information derived from these tools could be used to facilitate the assessment of the effects of wind energy facilities on wildlife. For example, the direct- and indirect-risk metrics used in the species prioritization component of the assessment would be improved with more accurate estimates of the proportion of species' ranges that overlap with the locations of wind turbines.

To evaluate the capabilities and limitations of each technology, our approach uses a matrix formulation that identifies strengths and weaknesses of the various technologies. No consolidated effort has been made to assess airspace use by animals, although numerous studies have considered isolated aspects of it. The new paradigm of considering airspace as habitat will facilitate fresh thinking about the broader issue and help to improve the methodology.

Indiana Bat and Generalized Bat Population Model

The Indiana bat (*Myotis sodalis*) is listed as "endangered" under the Endangered Species Act (16 U.S.C. § 1531 et seq.), and the little brown bat (*Myotis lucifugus*) is under consideration for listing due to population losses from white-nose syndrome. Currently, a population model and a graphical user interface have been developed for use by U.S. Geological Survey (USGS) scientists (Thogmartin and others, 2012; Erickson, Thogmartin, and Szymanski, 2014). This model has served as the basis for a theoretical framework providing insight into the effects of wind energy facilities on migratory patterns and spatial dynamics of bats (Erickson, Thogmartin, Russell, and others, 2014). WEIAM project scientists are working to parameterize and apply this model to bat populations by using data from the U.S. Fish and Wildlife Service. In addition to the effects of wind energy facilities, the concurrent effects of white-nose syndrome are included in the model, which can be expanded to assess the effects of climate change and other stressors on migratory bat species.

The current assessment methodology uses a matrix model for both birds and bats. Developing a highly customized model for Indiana bats and then generalizing it to a broader number of bat species benefits the methodology in two ways. First, the methodology can be partially validated by comparing its results to those from the more complex model. Second, if a generalized demographic model can be created for bats, then the methodology could use two types of generalized models (one for bats and one for birds) that better match the life history and ecological differences between these two groups.

Investigating Wind Energy Development Effects with Breeding Bird Survey Information

Observations of bird fatality events due to collisions with wind turbines are well documented. However, these incidental fatality events may or may not scale up to increasing the risk of significant decline or extinction at the population level.

WEIAM project scientists are developing a method that uses data from the North American Breeding Bird Survey (BBS; Sauer and others, 2015) for detecting effects of wind energy development at the population level. This method could assess whether wind energy development has had a detectable effect on the population trends of monitored bird populations through time. The method could also assess the likelihood of wind energy development's contributions to future imperilment of avian species in the United States. The method has the potential to place the monitoring data collected through the BBS observation routes spatially and temporally within the context of wind energy development as it has been documented in the USGS wind turbine map and database (Diffendorfer and others, 2014).

The analysis removes observation error from BBS abundance indexes to estimate underlying population trends and variability through linear multivariate autoregressive state-space (MARSS) models. Spatial and temporal covariates related to the location and timing of wind energy development can be incorporated into the models to estimate current effects on observed trends at multiple spatial scales.

Unlike the current methodology, this approach is an empirically based attempt to directly measure the effects of wind energy development on bird species at the population level. The approach can also be generalized to assess non-avian species when similar monitoring data are available. This approach represents a strong addition to the methodology that relies on population models and could be included as an additional component if the results warrant its use. In addition, the population parameter estimates from the MARSS models may improve the observed population trend estimates used in the demographic model components. These analyses can assist in the parameterization of detailed demographic models that can explore specific scenarios for wind energy development. In addition, this approach is useful for making short- and long-term projections of risk to bird populations at both the regional and national scale for the purposes of management and priority ranking.

Improved Bat Fatality Data

Though national estimates of bat fatalities exist (Arnett and others, 2008; Arnett and Baerwald, 2013; Hayes, 2013; Smallwood, 2013; Huso and Dalthorp, 2014), no systematic and national-scale analyses have been conducted to identify correlates of bat fatality rates or to use fatality correlates to estimate national fatality. In addition, despite substantial variation in study designs among studies conducted at different wind energy facilities and in different years, little research has investigated how varying the approaches to study design, data collection, and statistical estimation at local sites influences large-scale fatality estimates.

Project researchers have reviewed roughly 120 original studies that have investigated collision fatality at individual U.S. wind energy facilities. These data may be used to conduct the following research: (1) to estimate bat fatalities at U.S. wind energy facilities (a) to identify significant predictors of fatality-rate variation and (b) to use these predictors to estimate national bat fatality (for example, following general approaches in Loss and others, 2013) and (2) to assess how varying the study design and data collection approaches (in fatality surveys, scavenger removal trials, and searcher detection rate trials) influences national estimates of bird and bat fatality.

Results from such analyses could be used to enhance the methodology or its implemented assessment in two ways. First, the resulting statistical models could be used to estimate future fatality under scenarios of increased installations of wind turbines. Second, depending on the results of the analyses regarding bias, it may be possible to estimate species-specific rates of fatality for bats, which would allow more precise estimates from multiple components in the general demographic model component of the assessment methodology.

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Appendix 2. Alternative Modeling Approaches Considered

Stochastic Population Model

The U.S. Geological Survey (USGS) initially developed a stochastic matrix population model that included juvenile and adult survival, as well as age-specific maternity. The model was used to project population size forward through time. To model the effects of wind energy facilities, mortality was incrementally increased until the population trends met a quasi-extinction breakpoint (a percentage of runs showing a given percent decline in 10 years). This level of mortality was then compared to the observed fatality rates at turbines to assess risk. The model was developed with ranges of uncertainty around each parameter and then was run repeatedly, sampling across the parameter ranges in each model run.

The USGS elected not to use this approach for a number of reasons. First, given parameter uncertainty, a large proportion of runs either declined rapidly or increased exponentially. In both of these cases, added mortality from wind energy facilities had negligible effects on population trends so the approach was a fairly coarse tool for focusing on potentially small population-level consequences from wind energy facilities. Second, deciding upon the appropriate level of stochasticity to include for each species was extremely difficult. For many species, parameter uncertainty included both process and sampling error, which biased variance, and therefore risk, upward. Third, the approach required the setting of quasi-extinction breakpoints to determine how frequently a given turbine mortality rate met criteria of decline, as well as determining what proportion of runs needed to be declining before the turbine mortality rate was considered to cause population declines. Setting these arbitrary breakpoints, as noted above in section 2, adds an additional layer of complexity to the assessment.

Complex, Highly Individualized Models

If time and resources were available, the USGS could develop models tailored to each species of interest that would be based on the current state of knowledge of that species and the available data. These models might include complex spatial structure, simulate individuals across space and time, and use time series of population size to help estimate values for model input (Conroy and others, 1995). The Indiana bat model (appendix 1) served as a case study for this alternative. Although highly tailored models would enhance the assessment and could be run for species of high conservation concern, resource limitations prevent this from being practical for a larger number of species.

Empirically Based Population Viability Analyses

When spatially replicated time series of species abundance exist, a variety of approaches can use the information included in them to forecast future population dynamics. These approaches include multivariate autoregressive state-space (MARSS) models, corrupted stochastic exponential growth with Gaussian errors (CSEG) (Holmes and others, 2007), and other approaches. These approaches vary considerably in methodology, with some simply estimating a population trend and others estimating demographic rates associated with a trend. Overall, these approaches are valid and useful given the goals of the USGS Wind Energy Impacts Assessment Methodology (WEIAM) project. The USGS did not use them mainly because they require time series with an adequate signal-to-noise ratio to detect a trend, which would require highly precise estimates of abundance or relatively long time series. Though time series exist for many bird species in the North American Breeding Bird Survey (BBS; Sauer and others, 2015), nocturnal and migratory birds are not included and most bat species do not have time series, which limits the generality of these methods. Future iterations of the assessment methodology may allow for multiple assessment approaches that all produce the same output but use different algorithms. If so, then this class of models could be used for those species with time series of abundance data.

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