Species Status Assessment Report
for the
Northern long-eared bat
(*Myotis septentrionalis*)

Version 1.2

Photo by: Jill Utrup, USFWS

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[1] This SSA was developed in tandem with SSA analyses and reports for the tricolored bat (TCB; *Perimyotis subflavus*) and little brown bat (LBB; *Myotis lucifugus*).

EXECUTIVE SUMMARY

This report summarizes the results of a species status assessment (SSA) that assessed the northern long-eared bat's (NLEB; *Myotis septentrionalis*) viability over time. Although this SSA is its own separate report, it was developed in tandem with SSA analyses and reports for the tricolored bat (*Perimyotis subflavus*) and the little brown bat (*Myotis lucifugus*).

NLEB, a wide-ranging bat species, found in 37 states and 8 provinces in North America, typically overwinters in caves or mines and spends the remainder of the year in forested habitats. As its name suggests, NLEB is distinguished by its long ears, particularly as compared to other bats in genus *Myotis*.

In conducting our status assessment, we first considered what NLEB needs to ensure viability. We then considered factors that are currently influencing those viability needs or expected to in the future. Based on the species' viability needs and current influences on those needs, we evaluated NLEB's current condition. Lastly, we projected plausible future scenarios for NLEB based on its current condition and expected future influences on viability.

For survival and reproduction at the individual level, the NLEB requires access to food and water resources when not hibernating, along with suitable habitat throughout its annual life cycle. During the spring, summer and fall seasons, NLEB requires suitable foraging, roosting, traveling (between summer and winter habitat) and swarming habitat with appropriate conditions for maternity colony members; during the winter, NLEB requires habitat with suitable conditions for prolonged bouts of torpor. For NLEB populations to be healthy, they require a population size and growth rate sufficient to withstand natural environmental fluctuations, habitat of sufficient quantity and quality to support all life stages, gene flow among populations, and a matrix of interconnected habitats that support spring migration, summer maternity colony formation, fall swarming, and winter hibernation.

At the species level, NLEB viability requires having a sufficient number and distribution of healthy populations to ensure NLEB can withstand annual environmental and demographic variation (resiliency), catastrophes (redundancy), and novel or extraordinary changes in its environment (representation). Resiliency is best measured by the number, distribution, and health of populations across the species' range. Redundancy can be measured through the duplication and distribution of resilient populations across the species' range relative to potential catastrophic events. Representation can be measured by the number and distribution of healthy populations across areas of unique adaptive diversity. For NLEB, we identified five representation units (RPUs): Eastern Hardwoods, Southeast, Midwest, Subarctic, and East Coast.

Although there are countless stressors affecting NLEB, the primary factor influencing the viability of the NLEB is white-nose syndrome (WNS), a disease of bats caused by a fungal pathogen. Other primary factors that influence NLEB's viability include: wind energy mortality, effects from climate change, and habitat loss.

- WNS has been the foremost stressor on NLEB for more than a decade. The fungus that causes the disease, *Pseudogymnoascus destructans* (*Pd*), invades the skin of bats and

infection leads to increases in the frequency and duration of arousals during hibernation and eventual depletion of fat reserves needed to survive winter, and often results in mortality. WNS has caused estimated NLEB population declines of 97–100% across 79% of the species' range.

- Wind energy-related mortality of NLEB, is also proving to be a consequential stressor at local and regional levels, especially in combination with impacts from WNS. Most bat mortality at wind energy projects is caused by direct collisions with moving turbine blades. Wind energy mortality may occur over 49% of the NLEB range.
- Climate change variables, such as changes in temperature and precipitation, may influence NLEB resource needs, such as suitable roosting habitat for all seasons, foraging habitat, and prey availability. Although there may be some benefit to NLEB from a changing climate, overall negative impacts are anticipated, especially at local levels.
- Habitat loss may include loss of suitable roosting or foraging habitat, resulting in longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality. Loss of or modification of winter roosts (i.e., making hibernaculum no longer suitable) can result in impacts to individuals or at the population level.

In evaluating current and future conditions of the NLEB, we used the best available data. Winter hibernacula counts provide the most consistent, long-term, reliable trend data, and provide the most direct measure of WNS impacts, even for species such as NLEB that may be undercounted (due to their proclivity to roost in crevices). Although the availability and quality of summer data vary temporally and spatially, this data offered additional support (to winter data results) in evaluating population trends since *Pd* arrival. We relied upon the data derived from North American Bat Monitoring Program (NABat) analyses for all available winter (NABat 2021) and summer data (NABat 2020).

Available evidence, including both winter and summer data, indicates NLEB abundance has and will continue to decline substantially over the next 10 years under current demographic conditions. Winter abundance (from known hibernacula) has declined rangewide (49%) and across most RPUs (0–90%). In addition, the number of extant winter colonies declined rangewide (81%) and across all RPUs (40–88%). By 2030, rangewide abundance declines by 95% and the spatial extent declines by 75%. There has also been a noticeable shift towards smaller colony sizes, with a 96–100% decline in the number of large hibernacula (≥100 individuals). Declining trends in abundance and occurrence are also evident across much of NLEB's summer range. Rangewide summer occupancy declined by 80% from 2010–2019. Data collected from mobile acoustic transects found a 79% decline in rangewide relative abundance from 2009–2019 and summer mist-net captures declined by 43–77% compared to pre-WNS capture rates. To assess NLEB's future viability, we determined how WNS occurrence and wind energy capacity is likely to change into the future. We described two scenarios that bound our uncertainty on WNS spread and wind energy capacity. The first scenario included WNS spread under the Hefley et al. (2020, entire) model and lower wind energy capacity (low impact scenario) and the second scenario included WNS spread under Wiens et al. (2022, pp. 215–248) model and higher wind energy capacity (high impact scenario).

Using these scenarios, we projected the species' abundance and distribution. Under these future scenarios, NLEB declines worsen precipitously. Rangewide abundance declines 95% by 2030 and 99% by 2040. The number of extant winter colonies decline to only 9 (99% decline) by 2030 and 0 by 2050. Colony size also declines, with the number of large hibernacula (≥100 bats) declining 89% between 2020 and 2030. NLEB's winter spatial extent also declines by 75% by 2030 and by 100% by 2060. There are no areas within the species range where similar declines were not observed, with all RPUs experiencing declines in abundance, number of extant winter colonies, and spatial extant. We also qualitatively considered impacts from climate change, habitat loss, and conservation efforts. We expect that these impacts will result in further reduction in the species' resiliency, representation, and redundancy.

Unquestionably, WNS is the primary driver (or influence) that has led to the species' current condition and is predicted to continue to be the primary influence into the future. As is the case for all species status assessments, we do not have perfect information (see Appendix 1) on NLEB's occurrence, but the best available data suggest that bats at unknown hibernacula will undergo similar declines observed at known winter colonies. Wind energy related mortality is projected to be a more impactful influence in the future as annual mortality is projected to increase between 202 and 2,926 individuals by 2050 under the future low and high build-out scenarios, respectively. Although there may be some offsetting of effects under current climate conditions, increasing negative impacts are anticipated in the future. Increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase, leading to increased NLEB mortality and reduced reproductive success. Although we consider habitat loss pervasive across the NLEB range, impacts to NLEB and its habitat are often realized at the individual or colony level. Also, loss of hibernation sites (or modifications such that the site is no longer suitable) can result in impacts to winter colonies.

In conclusion, multiple data types and analyses indicate downward trends in NLEB population abundance and distribution over the last 14 years and consequently, we found no evidence to suggest that this downward trend will change in the future. NLEB abundance (winter and summer), number of occupied hibernacula, spatial extent, probability of persistence, and summer habitat occupancy across the range and within all RPUs are decreasing. Since the arrival of WNS, NLEB abundance steeply declined. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, NLEB's ability to recover from these low abundances is limited given their low reproduction output (1 pup per year). Therefore, NLEB's resiliency is greatly compromised in its current condition and is projected to decline under future scenarios. Additionally, because NLEB's abundance and spatial extent are projected to decline dramatically, NLEB will also become more vulnerable to catastrophic events. NLEB's representation has also been reduced. The steep and continued declines in abundance have likely led to reductions in genetic diversity, and thereby reduced NLEB adaptive capacity. Further, the projected widespread reduction in the distribution of hibernacula will lead to losses in the diversity of environments and climatic conditions occupied, which will impede natural selection and further limit NLEB's ability to adapt. Moreover, at its current low abundance, loss of genetic diversity via genetic drift will likely accelerate. Consequently, limiting natural selection process and decreasing genetic diversity will further lessen NLEB's ability to adapt to novel changes (currently ongoing as well as future changes) and exacerbate declines due to continued exposure to WNS, mortality from wind turbines, and impacts associated with habitat loss and climate change. Thus, even without further WNS spread and additional wind energy development, NLEB’s viability is likely to rapidly decline over the next 10 years. Further, given the projected low abundance and the few number and restricted distribution of winter colonies, NLEB’s currently impaired ability to withstand stochasticity, catastrophic events, and novel changes will worsen under the range of plausible future scenarios.

Contents

ABBREVIATION AND ACRONYMS:

%Sp – Percent Species Composition
AC – Adaptive Capacity
AEO – Annual Energy Outlook
AWEA – American Wind Energy Association
AWWI – American Wind Wildlife Institute
Bfat – Bat Fatality
BWEC – Bat Wind Energy Association
C – Celsius
CanWEA – Canadian Wind Energy Association
CC – Climate Change
CE – Catastrophic Event
CER – Canadian Energy Regulator
CI – Confidence Interval
CONUS – Continental United States
CWTD – Canada Wind Turbine Database
DFW – Department of Fish and Wildlife
ESA – Endangered Species Act
F – Fahrenheit
GRTS – Generalized Random-Tessellation Stratified
Hibs – Hibernacula
IUCN – International Union for Conservation of Nature
km – Kilometers
LBB – Little brown bat (*Myotis lucifugus*)
MAST – Mean Annual Surface Temperature
mi – Miles
MLRC – Multi-Resolution Land Characteristics
MW – Megawatts
MYLU – *Myotis lucifugus*
MYSE – *Myotis septentrionalis*
N – Abundance
NLCD – National Land Cover Database
NLEB – Northern long-eared bat (*Myotis septentrionalis*)
NPS – National Park Service
NREL – National Renewable Energy Laboratory
Pd – Pseudogymnoascus destructans
PESU – *Perimyotis subflavus*
pPg – Probability of Population Growth
RPA – Resources Planning Act
RPU – Representation Unit
SSA – Species Status Assessment
TCB – Tricolored bat (*Perimyotis subflavus*)
USDOE – U.S. Department of Energy
USEIA – U.S. Energy Information Administration
USFS – U.S. Forest Service

USFWS – U.S. Fish and Wildlife Service
USGS – U.S. Geological Survey
USWTDB – U.S. Wind Turbine Database
WNS – White-Nose Syndrome
YOA – Year of Arrival
YSA – Years since Arrival
λ (Lambda) – Population Growth Rate
λavg – Average Population Growth Rate
λtot – Total Population Growth Rate

CHAPTER 1 – INTRODUCTION

Background

This report summarizes the results of a species status assessment (SSA) conducted for northern long-eared bat (NLEB, *Myotis septentrionalis*). It delivers the best available scientific and commercial information available on the NLEB in a transparent and defensible peer reviewed report for immediate and future Endangered Species Act (ESA) related decisions. Therefore, while the report is not a decisional document, it does serve as a synthesis of the best available information on the biological status, helpful in promoting the current and future conservation of the species. For this reason, after reviewing this document relative to all relevant laws, regulations, and policies, the U.S. Fish and Wildlife Service (USFWS) plans to utilize the results of this report to make and publish a listing determination in the *Federal Register*.

This chapter describes the analytical framework and methods used to assess NLEB's viability over time. Chapter 2 summarizes the ecological requirements for survival and reproduction at the individual, population, and species levels. Chapter 3 summarizes the historical condition of NLEB. Chapter 4 describes the key drivers that led to NLEB's current condition and the anticipated plausible change in the primary drivers (referred to as influences) over time. Chapter 5 summarizes the current condition assuming no change in influences. Chapter 6 describes the species' future conditions given the plausible projections of the key influences. Lastly, Chapter 7 synthesizes the above analyses and describes how the consequent change in the number, health, and distribution of populations influence NLEB viability over time as well as the sources of uncertainty and the implications of this uncertainty. Appendices 1–5 provide further information on uncertainty and sensitivity, supplemental methodology information, supplemental results, supplemental threat background information, and supplemental data.

Analytical Framework

Viability is the ability of a species to maintain populations in the wild over time. To assess viability, we use the conservation biology principles of resiliency, redundancy, and representation (Shaffer and Stein 2000, pp. 308–311). Meaning, to sustain populations over time, a species must have a sufficient number of populations distributed throughout its geographic range to withstand:

(1) environmental and demographic stochasticity and disturbances (Resiliency),
(2) catastrophes (Redundancy), and
(3) novel changes in its biological and physical environment (Representation).

Viability is a measure of the likelihood of sustaining populations over time. A species with a high degree of resiliency, representation, and redundancy (the 3Rs) is generally better able to adapt to future changes and to tolerate catastrophes, environmental stochasticity, and stressors, and thus, typically has high viability.

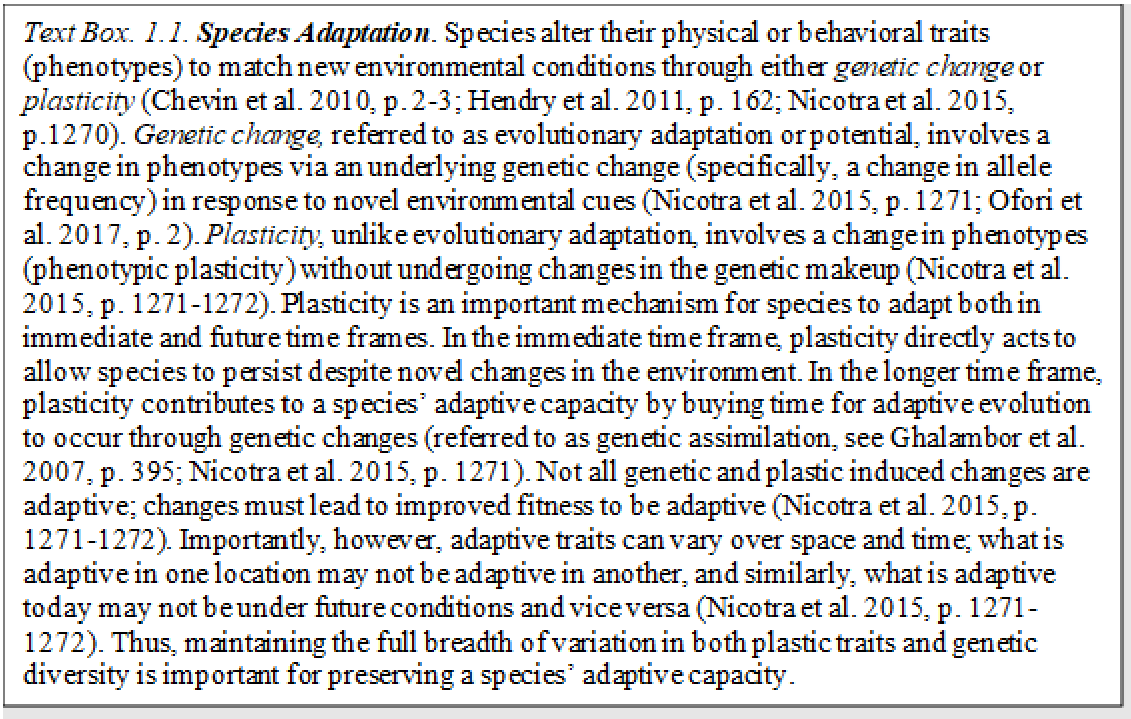
Resiliency is the ability of a species to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions such as temperature, rainfall), periodic disturbances within the normal range of variation (e.g., fire, floods, storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity) (Redford et al. 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

Resiliency is multi-faceted. First, it requires having healthy populations demographically (robust survival, reproductive, and growth rates), genetically (large effective population size, high heterozygosity, and gene flow between populations), and physically (good body condition). Second, resiliency also requires having healthy populations distributed across heterogeneous environmental conditions (referred to as spatial heterogeneity; this includes factors such as temperature, precipitation, elevation, and aspect). Spatial heterogeneity is particularly important for species prone to spatial synchrony (regionally correlated fluctuations among populations). Populations can fluctuate in synchrony over broad geographical areas (Kindvall 1996, pp. 207, 212; Oliver et al. 2010, pp. 480–482) because environmental stochasticity can operate at regional scales (Hanski and Gilpin 1997, p. 372). Spatial heterogeneity induces asynchronous fluctuations among populations, thereby guarding against concurrent population declines. Lastly, resiliency often requires connectivity among populations to maintain robust population-level heterozygosity via gene flow among populations and to foster demographic rescue following population decline or extinction due to stochastic events.

Redundancy is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health (Mangal and Tier 1993, p. 1083). For all species, a minimal level of redundancy is essential for long-term viability (Shaffer and Stein 2000, pp. 307, 309–310; Groves et al. 2002, p. 506). Reducing the risk of extinction due to a single or series of catastrophic events requires having multiple populations widely distributed across the species' range, with connectivity among groups of locally adapted populations to facilitate demographic rescue following population decline or extinction. Redundancy provides a margin of safety to reduce the risk of losing substantial portions of genetic diversity or the entire species to a single or series of catastrophic events.

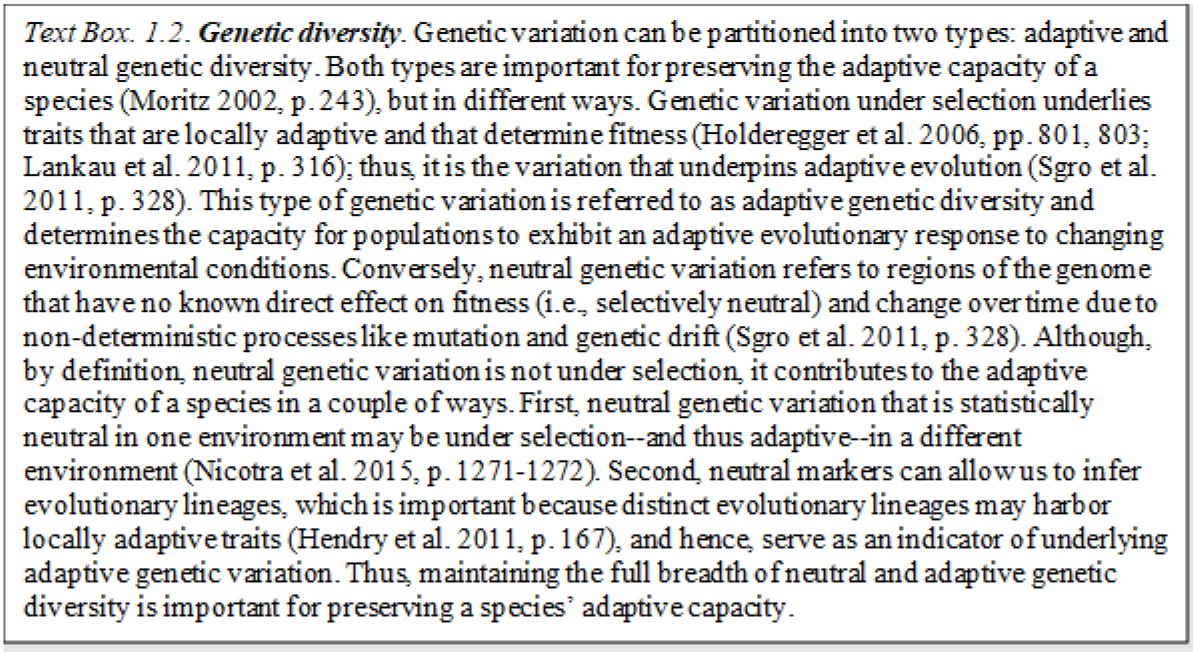
Representation is the ability of a species to adapt to both near-term and long-term novel or extraordinary changes in the conditions of its environment, both physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (novel pathogens, competitors, predators, etc.). This ability to adapt to changing and novel conditions-- referred to as adaptive capacity--is essential for viability as environmental conditions are continuously changing (Nicotra et al. 2015, p. 1269). Species adapt to novel changes in their environment by either 1) moving to new, suitable environments or 2) by altering (via plasticity or genetic change) their physical or behavioral traits (phenotypes) to match the new environmental conditions (Nicotra et al. 2015, p. 1270; Beever et al. 2016, p. 132). Maintaining a species' *ability to disperse* and colonize new environments fosters adaptive capacity by allowing species to move from areas of unsuitable conditions to regions with more favorable conditions. It also fosters adaptive capacity by increasing genetic diversity via gene flow, which is, as discussed below, important for evolutionary adaptation (Hendry et al. 2011, p. 173; Ofori et al. 2017, p. 1). Thus, maintaining natural levels of connectivity among populations is important for preserving a species' adaptive capacity (Nicotra et al. 2015, p. 1272).

Maintaining a species' *ability to adapt* to novel and extraordinary conditions requires preserving the breadth of genetic variation. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either genetic change or plasticity (see Text Box 1.1). For adaptation to occur, whether through plasticity or evolutionary adaptation, there must be genetic variation upon which selection can act (Hendry et al. 2011, pp. 164–165; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 326). Without genetic variation, the species cannot adapt and is more prone to extinction (Spielman et al. 2004, p. 15263; also see Text Box 1.1).



Text Box. 1.1. **Species Adaptation**. Species alter their physical or behavioral traits (phenotypes) to match new environmental conditions through either *genetic change* or *plasticity* (Chevin et al. 2010, p. 2-3; Hendry et al. 2011, p. 162; Nicotra et al. 2015, p.1270). *Genetic change*, referred to as evolutionary adaptation or potential, involves a change in phenotypes via an underlying genetic change (specifically, a change in allele frequency) in response to novel environmental cues (Nicotra et al. 2015, p. 1271; Ofori et al. 2017, p. 2). *Plasticity*, unlike evolutionary adaptation, involves a change in phenotypes (phenotypic plasticity) without undergoing changes in the genetic makeup (Nicotra et al. 2015, p. 1271-1272). Plasticity is an important mechanism for species to adapt both in immediate and future time frames. In the immediate time frame, plasticity directly acts to allow species to persist despite novel changes in the environment. In the longer time frame, plasticity contributes to a species' adaptive capacity by buying time for adaptive evolution to occur through genetic changes (referred to as genetic assimilation, see Ghalambor et al. 2007, p. 395; Nicotra et al. 2015, p. 1271). Not all genetic and plastic induced changes are adaptive; changes must lead to improved fitness to be adaptive (Nicotra et al. 2015, p. 1271-1272). Importantly, however, adaptive traits can vary over space and time; what is adaptive in one location may not be adaptive in another, and similarly, what is adaptive today may not be under future conditions and vice versa (Nicotra et al. 2015, p. 1271-1272). Thus, maintaining the full breadth of variation in both plastic traits and genetic diversity is important for preserving a species' adaptive capacity.

Genetic variation that is adaptive is difficult to identify for a species and represents a significant challenge even when there is genetic information available. To denote variation as 'adaptive' we need to identify which loci are under selection, which traits those loci control, how those traits relate to fitness, and what the species' evolutionary response to selection on those traits will be over time (Hendry et al. 2011, p. 162–163; Lankau et al. 2011, p. 316; Teplitsky et al. 2014, p. 190). Although new genomic techniques are making it easier to obtain this type of information (see Funk et al. 2019), it is lacking for most species. Fortunately, there are several proxies that collectively can serve as indicators of potentially underlying adaptive genetic variation. One of the easiest proxies to measure is variation in biological traits (also described as phenotypic variation). Phenotypic variation, which on its own can be a mechanism for adapting to novel changes, can be due to underlying adaptive genetic variation (Crandall et al. 2000, p. 291; Forsman 2014, p. 304; Nicotra et al. 2015, p. 3). A second proxy for adaptive genetic variation is neutral genetic variation, which is usually the type of genetic data first reported in species-specific genetic studies (see Text Box 1.2). A third, and more distant, proxy for adaptive genetic variation is disjunct or peripheral populations (Ruckelhaus et al. 2002, p. 322). These populations can be exposed to the extremes in habitat/ecological/climate conditions and thus harbor unique and potentially adaptive traits. Similarly, populations that occur across steep environmental gradients can be indicators of underlying adaptive genetic diversity because local adaptation is driven by environmental conditions, which are continually changing at different rates and scales (Sgro et al. 2011, pp. 330, 333).



Text Box. 1.2. ***Genetic diversity***. Genetic variation can be partitioned into two types: adaptive and neutral genetic diversity. Both types are important for preserving the adaptive capacity of a species (Moritz 2002, p. 243), but in different ways. Genetic variation under selection underlies traits that are locally adaptive and that determine fitness (Holderegger et al. 2006, pp. 801, 803; Lankau et al. 2011, p. 316); thus, it is the variation that underpins adaptive evolution (Sgro et al. 2011, p. 328). This type of genetic variation is referred to as adaptive genetic diversity and determines the capacity for populations to exhibit an adaptive evolutionary response to changing environmental conditions. Conversely, neutral genetic variation refers to regions of the genome that have no known direct effect on fitness (i.e., selectively neutral) and change over time due to non-deterministic processes like mutation and genetic drift (Sgro et al. 2011, p. 328). Although, by definition, neutral genetic variation is not under selection, it contributes to the adaptive capacity of a species in a couple of ways. First, neutral genetic variation that is statistically neutral in one environment may be under selection--and thus adaptive--in a different environment (Nicotra et al. 2015, p. 1271-1272). Second, neutral markers can allow us to infer evolutionary lineages, which is important because distinct evolutionary lineages may harbor locally adaptive traits (Hendry et al. 2011, p. 167), and hence, serve as an indicator of underlying adaptive genetic variation. Thus, maintaining the full breadth of neutral and adaptive genetic diversity is important for preserving a species' adaptive capacity.

Lastly, preserving a species' adaptive capacity requires maintaining the processes that allow for evolution to occur; namely, natural selection and gene flow (Crandall et al. 2000, pp. 290–291; Zackay 2007, p. 1; Sgro et al. 2011, p. 327). Natural selection is the process by which heritable traits can become more (selected for) or less (not selected for) common in a population via differential survival or reproduction (Hendry et al. 2011, p. 169). To preserve natural selection as a functional evolutionary force, it is necessary to maintain populations across an array of environments (Shaffer and Stein 2000, p. 308; Hoffmann and Sgro 2011, p. 484; Lankau et al. 2011, p. 320; Sgro et al. 2011, p. 332). Gene flow serves as an evolutionary process by introducing new alleles (variant forms of genes) into a population, thereby, increasing the gene pool size (genetic diversity). Maintaining the natural network of genetic connections between populations will foster and preserve the effectiveness of gene flow as an evolutionary process (Crandall et al. 2000, p. 293). Preserving genetic connections among populations along with maintaining large effective population sizes will minimize the loss of genetic variation due to genetic drift (Crandall et al. 2000, p. 293). Maintaining large population abundance also fosters adaptive capacity as the rate of evolutionary adaptation is faster in populations with high diversity, which is correlated with population size (Ofori et al. 2017, p.2).

General Methods

Below we describe our methods for assessing NLEB viability over time. Our approach entailed: 1) describing the historical condition (abundance, health, and distribution of populations prior to 2020), 2) describing the current condition (abundance, health, and distribution of populations in 2020), 3) identifying the primary influences leading to the species' current condition and projecting the future states (scope and magnitude) of these influences, 4) projecting the number, health, and distribution of populations given the current and future states of the influences, and 5) assessing the implications of the projected changes in the number, health, and distribution of populations for the species' viability and extinction risk under both current and future conditions (Figure 1.1). We briefly explain these steps below and provide further details in Appendix 2. Because of the difficulty of delineating populations, we used winter colonies (hibernacula) to track the change in number, health, and distribution of populations over time. Henceforth, the terms populations, winter colonies, and hibernacula are used interchangeably.

As is the case for all species status assessments, we do not have perfect information. Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios and making reasonable assumptions based on the best available data. The key uncertainties and how we addressed these uncertainties are described in Appendix 1.

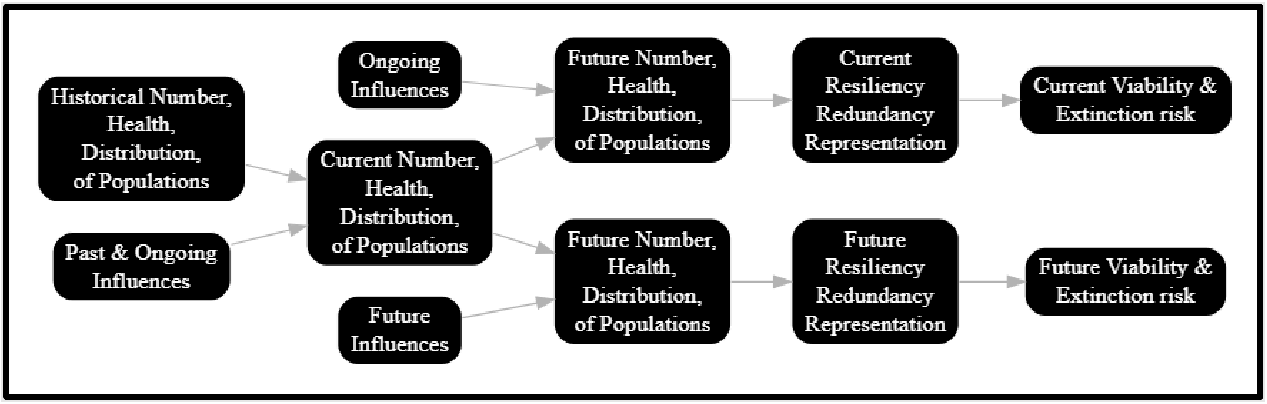


Figure 1.1. Simplified conceptual diagram depicting the analytical framework for assessing bat viability over time given current and future conditions.

Step 1. Historical Abundance, Health, and Distribution

We reached out to partners (Tribal, Federal, State and other) across the range to garner all relevant and available data. The majority of these data were collected by State agencies and are now maintained in the North American Bat Monitoring Program (NABat) database, unless otherwise requested by the data contributor or data was not in a format compatible with NABat. Using this information, we compiled a list of all known hibernacula and associated yearly winter counts (NABat 2021). Winter counts are conducted as internal surveys of caves, mines, tunnels, culverts and other accessible subterranean habitats. Winter counts are conducted in mid to late winter when bats are expected to be less likely to move between hibernacula and prior to spring emergence. Colony counts in hibernacula provide the best estimate of species abundance consistently available for NLEB. Colony count data represent the largest amount (geographic and in amount of survey) of abundance data throughout the range of the species. Because not all hibernacula are known and accessible, we assume that hibernacula for which data are available are representative of all known and unknown hibernacula for the species. Additionally, to provide a non-model approach, we calculated historical abundances by summing the observed counts within each year. To account for missing data, we applied the last observed count. We refer to this third approach as "constant interpolation."

We measured population health as abundance within hibernacula (*N*) and population trend (*λ*). To estimate historical *N* and *λ*, we relied upon analyses completed by Wiens et al. (2022. pp. 231–233). Using a linear mixed effects model (henceforth, status and trends model), Wiens et al. (2022) estimated the yearly population abundance (*N*) from 1990 to 2020. From these yearly abundances, *λ* was estimated over time for each hibernaculum. For sites with insufficient data points, *λ* values were applied from the nearest neighbor (see Appendix 2). To capture uncertainty in the year of arrival of *Pseudogymnoascus destructans* (*Pd*), we calculated yearly abundance trajectories under two different *Pd*-occurrence models (Wiens et al. 2022, pp. 226–229 and Hefley et al. 2020, entire). Additionally, to provide a non-model approach, we calculated historical abundances by summing the observed counts within each year. To account for missing data, we applied the last observed count. We refer to this third approach as “constant interpolation.”

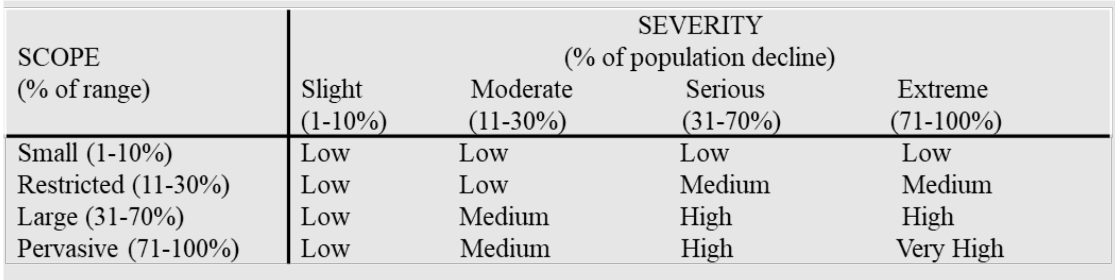
Step 2. Describe Current Abundance, Health, and Distribution

To estimate current conditions, we relied upon analyses completed by Wiens et al. (2022, p. 215–251) as described above. Additionally, because bats occupying a given hibernaculum disperse to many different locations on the summer landscape and because colony estimates are not available for all hibernacula, we also relied upon the results from USGS-led analyses of available summer capture records and acoustic records to garner insights on population trends at regional scales (see *Summer Data Analyses* subsection below).

Step 3. Identify the Primary Drivers (Influences)

We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) drivers of population numbers. We identified white-nose syndrome (WNS), wind related mortality, habitat loss, and climate change as the primary drivers in NLEB abundance.

We qualitatively assessed the scope, severity, and impact of the four stressors using an approach adapted from Master et al. (2012, pp. 28–35) to allow a comparison between influences. For each influence, we assigned a scope, severity, and impact level for both current and future states. The criteria used to assign levels are shown in Figure 1.2.



SCOPE (% of range)	SEVERITY (% of population decline)			
	Slight (1-10%)	Moderate (11-30%)	Serious (31-70%)	Extreme (71-100%)
Small (1-10%)	Low	Low	Low	Low
Restricted (11-30%)	Low	Low	Medium	Medium
Large (31-70%)	Low	Medium	High	High
Pervasive (71-100%)	Low	Medium	High	Very High

Figure 1.2. Comparative threat assessment criteria and definitions (adapted from Master et al. 2012).

For WNS and wind related impacts, we quantitatively modeled the current and future severity of these stressors. We used an existing demographic population model (BatTool, Erickson et al. 2014) to estimate the impacts (severity) from WNS and wind related mortality (described below).

To assess the impact of WNS and wind related mortality into the future, we used published data, expert knowledge, and professional judgment to form plausible future scenarios. To capture the uncertainty in our future state projections, we identified plausible upper and lower bound changes for each influence. The lower and upper bounds for each influence were then combined to create composite plausible "lower" and "upper" impact scenarios. The future scenarios are described in Chapter 4.

To calculate the impact of WNS, Wiens et al. (2022, pp. 231–247) derived the yearly effects of WNS, referred to as "WNS impacts schedule" from winter counts at sites upon WNS arrival (see Appendix 2 for further detail). Based on current information, we do not foresee a scenario in which *Pd* is eradicated from sites, and thus, we expect the fungus will continue to cause disease in populations even as some individuals exhibit resistance or tolerance to it. Thus, we set the duration of impacts to 40 years (i.e., the time throughout which WNS will affect survival in the population). However, to understand the sensitivity of the results to the duration of disease dynamic and to fully capture the uncertainty, we also incorporated a shorter disease dynamic duration. Based on current data (i.e., data from caves documented with WNS in 2008 continue to show continued impacts of disease through 2021, 14-years), 15 years is the shortest duration WNS would affect a population after *Pd* arrives. Thus, our lower impact scenario assumes a 15-year impact duration (i.e., no further WNS impacts after year 15 since *Pd* arrival) and high impact scenario assumes a 40-year impact duration (i.e., the last and least severe WNS disease stage carries through to 2060) (see Appendix 5 for further detail).

To calculate the impact from wind related mortality, we estimated species-specific wind fatality rates as:

$$\text{NLEB}\ \textit{per MW fat rate} = \textit{Bfat} * \%\textit{Sp}$$

Where *Bfat* is the all-bat fatality rate per megawatt (MW) and *%Sp* is the species-specific percent composition of fatalities reported (see Appendix 2 for further details of how *Bfat* and *%Sp* were calculated).

Step 4. Project the Number, Health, and Distribution of Populations Under Current and Future Influences

To project future abundance and trend given current and future state conditions for WNS and wind, we used the population model, BatTool (updated with NLEB-specific demographic values). In sum, the BatTool projects hibernaculum abundance over time given starting abundance (*N*), trend (*λ*), environmental stochasticity, WNS stage, annual WNS impacts schedule, and annual wind mortality as specified by the wind capacity scenarios. Starting abundance (*N*) and trend (*λ*) were derived from Step 2 above. We projected abundance through 2060 to capture the colony response to the 2050 wind energy build-out. Given the species' generation time is 5–7 years, 10 years is sufficient to discern the impacts of the annual mortality levels associated with the 2050 wind capacity build-out.

Using these projected abundance estimates, we calculated various hibernaculum-level and Representation Unit (RPU) metrics to describe the species' historical, current, and future condition (number, health, and distribution of populations) given current and future influences. The results are summarized in chapters 3, 4, and 6. RPUs are further described in Chapter 2.

Step 5. Assess the Current and Future Viability

We evaluated how the change in the number, health, and distribution of populations from historical to present to future influences NLEB's ability to withstand stochastic events, catastrophes, and novel changes in its environment, i.e., the 3Rs over time. Specifically, we used the change in the abundance and distribution of winter colonies over time--to evaluate NLEB's resiliency to stochasticity, disturbances, and stressors. To assess redundancy, we qualitatively assessed how the current and projected abundance and distribution of colonies affect the risk of catastrophic losses due to extreme weather eventsand epizootics.. To assess NLEB's ability to adapt to novel changes in its physical and biological environment, we characterized NLEB adaptability relative to 12 recognized core adaptive capacity attributes (Thurman et al. 2020, entire) and assessed the likelihood of maintaining colonies across the breadth of adaptive diversity given geographic-specific influences and vulnerability to catastrophic events (Appendix 2).

Summary of NABat Data Sources

Our analyses relied on existing information and upon the data and analyses conducted by NABat. Wiens et al. (2022, entire) provided estimates of past, current, and future abundance based on available winter count data (NABat 2021; accessed February 10, 2021). Deeley and Ford (2022, entire), Stratton and Irvine (2022, entire), and Whitby et al. (2022, entire), provided estimates of population trend since *Pd* arrival based on available summer data (NABat 2020; accessed November 18, 2020). Udell et al. (2022, entire) estimated hibernaculum-specific wind energy mortality estimates. How we used these data are briefly described in Table 1.1, with more detail in Appendix 2. A conceptual model of the BatTool is provided in Figure 1.3. Using Wiens et al. (2022, entire) data, we calculated summary statistics at rangewide and RPU scales over time. For ease of reading, we do not cite the source of the data within the text of Chapters 3–7. In several cases, contributed data could not be utilized in these range-wide analyses due to incompatibility with the database structure of NABat or infeasibility of transferring data files, e.g., New York State Department of Environmental Conservation acoustic data. In these cases, we reviewed any data summaries and analyses provided by the contributing partner and assessed them alongside analyses from NABat.

Table 1.1. NABat analyses used in the SSA analyses. Steps refer to the 5 steps of our analytical approach.

Citation	**Data/Analyses**	**Step in Analytical Process**	**Chapter**
Cheng et al. 2021	Impacts of WNS	Step 3: past WNS impacts	Chapter 4
Cheng et al. 2022	Winter colony count analysis	Step 3: past WNS impacts	Chapter 4
Deeley and Ford 2022	Rangewide analysis of summer capture rates from 1999–2019	Step 2 - Current conditions	Chapter 5
Stratton and Irvine 2022	Rangewide change in occupancy from 2010 – 2019 based on summer acoustic & mist-net data	Step 2 - Current conditions Step 3 – Characterize impact of wind	Chapter 5 Chapter 4
Whitby et al. 2022	Rangewide analysis of relative abundance based on summer mobile acoustic data from 2009 – 2019	Step 2 - Current conditions Step 3 – Characterize impact of wind	Chapter 5 Chapter 4
Udell et al. 2022	Estimated wind related bat mortality & allocation to known hibernacula	Step 3. Define future scenarios for wind energy mortality	Chapter 4
Wiens et al. 2022 pp. 231–247	Status & trends linear effects model using winter colony count data	Steps 1 & 2 Historical & current abundance (*N*) and population trend (λ) over time Step 3 past WNS impacts, construct WNS impacts schedule	Chapter 3 Chapters 4, 5
Hefley et al. 2020	*Pd*-occurrence model 2	Steps 1 & 2 – feeds into status & trends model; Step 3 – define future low impact scenario for *Pd*-spread	NA Chapter 4
Wiens et al. 2022, pp. 226–229	*Pd*-occurrence model 1	Steps 1 & 2 – feeds into status & trends model; Step 3 – define future high impact scenario for *Pd*-spread	NA Chapter 4
Wiens et al. 2022, pp. 236–247	Future projections of N via BatTool	Step 4. Project abundance over time	Chapters 5, 6

Figure 1.3. A conceptual diagram showing where the NABat data sources are used in our analytical process.

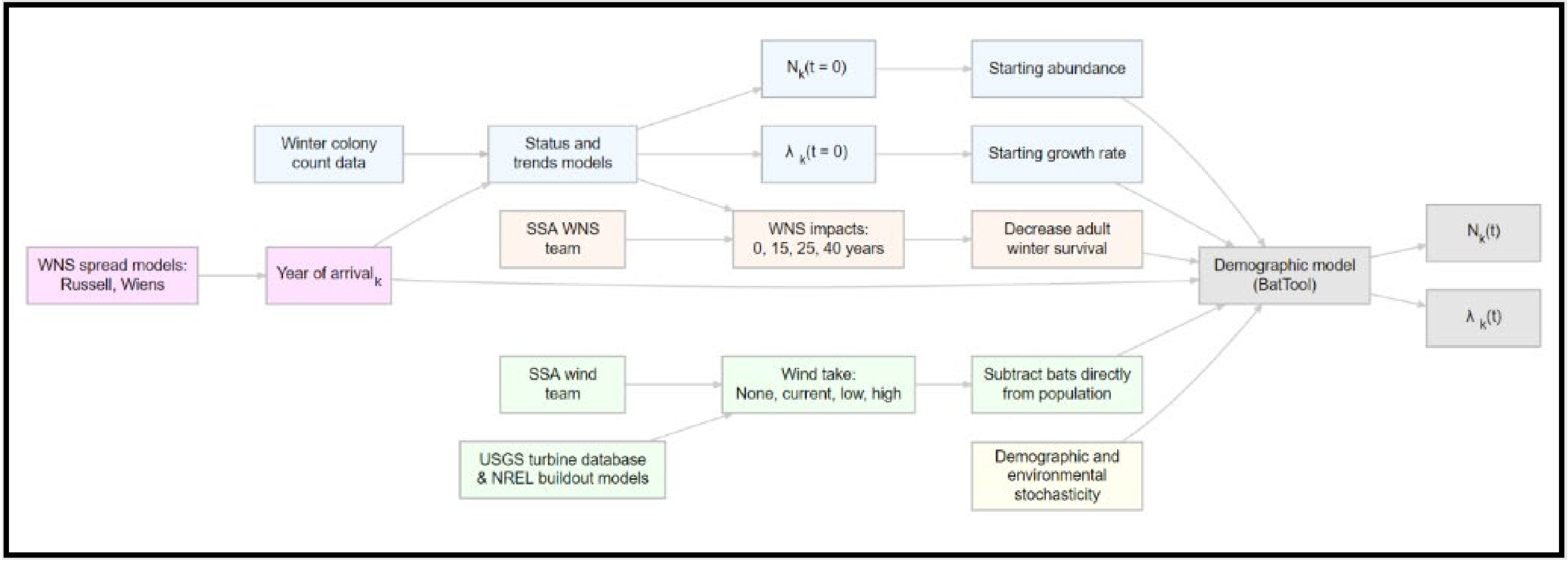


Figure 1.3. BatTool conceptual model. Top (blue boxes): raw data (winter colony) feeds into the status and trends model, which outputs current colony size (N) and population trend (λ) values to input into the BatTool. Middle (pink boxes): 2 Pd occurrence models give Pd year of arrival, which is used in both the status and trends model and BatTool. Middle (peach boxes): SSA core team derived WNS annual impacts schedule, which feeds into the BatTool as decreases in adult winter survival. Bottom (green boxes): SSA core team calculated species-specific bat fatality per MW and USGS projected allocation of this mortality are used to project colony specific mortality over time, which feeds into the BatTool as direct loss of adult females. Far right boxes (gray boxes): projected abundance (N) over time is the output, which is used to calculate colony and RPU level statistics, e.g., λ, number of extant sites, etc.

CHAPTER 2 – SPECIES ECOLOGY AND NEEDS

Taxonomy and Genetics

NLEB belongs to the order *Chiroptera*, family *Vespertilionidae*, subfamily *Vespertilioninae*, genus *Myotis*, and subgenus *Myotis* (Caceres and Barclay 2000, p. 1). The NLEB was first considered a subspecies of Keen's long-eared myotis (*Myotis keenii*) (Fitch and Schump 1979, p. 1), but was recognized as a distinct species by van Zyll de Jong in 1979 (1979, p. 993), based on geographic separation and difference in morphology (as cited in Nagorsen and Brigham 1993, p. 87; Caceres and Pybus 1997 p. 1; Whitaker and Hamilton 1998, p. 99;Caceres and Barclay 2000, p. 1; Simmons 2005, p. 516; Whitaker and Mumford 2009, p. 207), and more recently genetically by Platt et al. (2018, p. 239). The NLEB is currently considered a monotypic species, with no subspecies described for this species (van Zyll de Jong 1985, p. 94; Nagorsen and Brigham 1993, p. 90;; Caceres and Barclay 2000, p. 1; Whitaker and Mumford 2009, p. 214; USFWS 2015, p. 17975).

Although there have been few wide-ranging genetic studies on this species, information collected to date indicates the species to be panmictic (random mating within a population). Johnson et al. (2014, entire) assessed nuclear genetic diversity at one site in New York and several sites in West Virginia, and found little evidence of population structure in NLEBs at watershed or regional scales. In addition, studies conducted in Ohio, Nova Scotia and Quebec, Canada, and Kentucky showed variation in NLEB haplotypes at local levels; however, these studies also indicated relatively low levels of overall genetic differentiation between groups and high levels of diversity overall (Arnold 2007, p. 157, Johnson et al. 2015, p. 12; Olivera-Hyde et al. 2020, p.729).

This species has been recognized by different common names, such as: Keen's bat (Whitaker and Hamilton 1998, p. 99), northern myotis (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207), and the northern bat (Foster and Kurta 1999, p. 660). For purposes of this SSA, we recognize it as a listable entity under the ESA (USFWS 2015, p. 17975).

Species Description

NLEB's adult body weight averages 5 to 8 grams (g) (0.2 to 0.3 ounces), with females tending to be slightly larger than males (Caceres and Pybus 1997, p. 3). Average body length ranges from 77 to 95 millimeters (mm) (3.0 to 3.7 inches [in]), tail length between 35 and 42 mm (1.3 to 1.6 in), forearm length between 34 and 38 mm (1.3 to 1.5 in), and wingspread between 228 and 258 mm (8.9 to 10.2 in) (Barbour and Davis 1969, p. 76; Caceres and Barclay 2000, p. 1). Pelage (fur) colors include medium to dark brown on its back; dark brown, but not black, ears and wing membranes; and tawny to pale-brown fur on the ventral side (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207). As indicated by its common name, the NLEB is distinguished from other *Myotis* species by its relatively long ears (average 17 mm (0.7 in); Whitaker and Mumford 2009, p. 207) that, when laid forward, extend beyond the nose up to 5 mm (0.2 in; Caceres and Barclay 2000, p. 1; Figure 2.1). The tragus (projection of skin in front of the external ear) is long (average 9 mm [0.4 in]; Whitaker and Mumford 2009, p. 207), pointed, and symmetrical (Nagorsen and Brigham 1993, p. 87; Whitaker and Mumford 2009, p. 207). There is an occasional tendency for the NLEB to exhibit a slight keel on the calcar (spur of cartilage arising from inner side of ankle; Nagorsen and Brigham 1993, p. 87). This can add some uncertainty in distinguishing NLEBs from other sympatric *Myotis* species (Lacki 2013, in litt.). Within its range, the NLEB can be confused with the little brown bat (*Myotis lucifugus*) or the western long-eared myotis (*Myotis evotis*). The NLEB can be distinguished from the little brown bat by its longer ears, tapered and symmetrical tragus, slightly longer tail, and less glossy pelage (Caceres and Barclay 2000, p. 1; Kurta 2013, in litt.). The NLEB can be distinguished from the western long-eared myotis by its darker pelage and paler membranes (Caceres and Barclay 2000, p. 1).

Figure 2.1. Hibernating NLEB. Photo credit: Al Hicks, New York Department of Environmental Conservation (retired).

Species Distribution

NLEB's range includes much of the eastern and north-central U.S., and all Canadian provinces west to the southern Yukon Territory and eastern British Columbia (Nagorsen and Brigham 1993, p. 89; Caceres and Pybus 1997, p. 1; Environment Yukon 2011, p. 10) (Figure 2.2[2]). In the U.S., the species' range reaches from Maine west to Montana, south to eastern Kansas, eastern Oklahoma, Arkansas, and east to South Carolina (Whitaker and Hamilton 1998, p. 99; Caceres and Barclay 2000, p. 2; Simmons 2005, p. 516; Amelon and Burhans 2006, pp. 71–72). The species' range includes all or portions of the following 37 states and the District of Columbia: Alabama, Arkansas, Connecticut, Delaware, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, Montana, Nebraska, New Hampshire, New Jersey, New York, North Carolina, North Dakota, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, South Dakota, Tennessee, Vermont, Virginia, West Virginia, Wisconsin, and Wyoming.

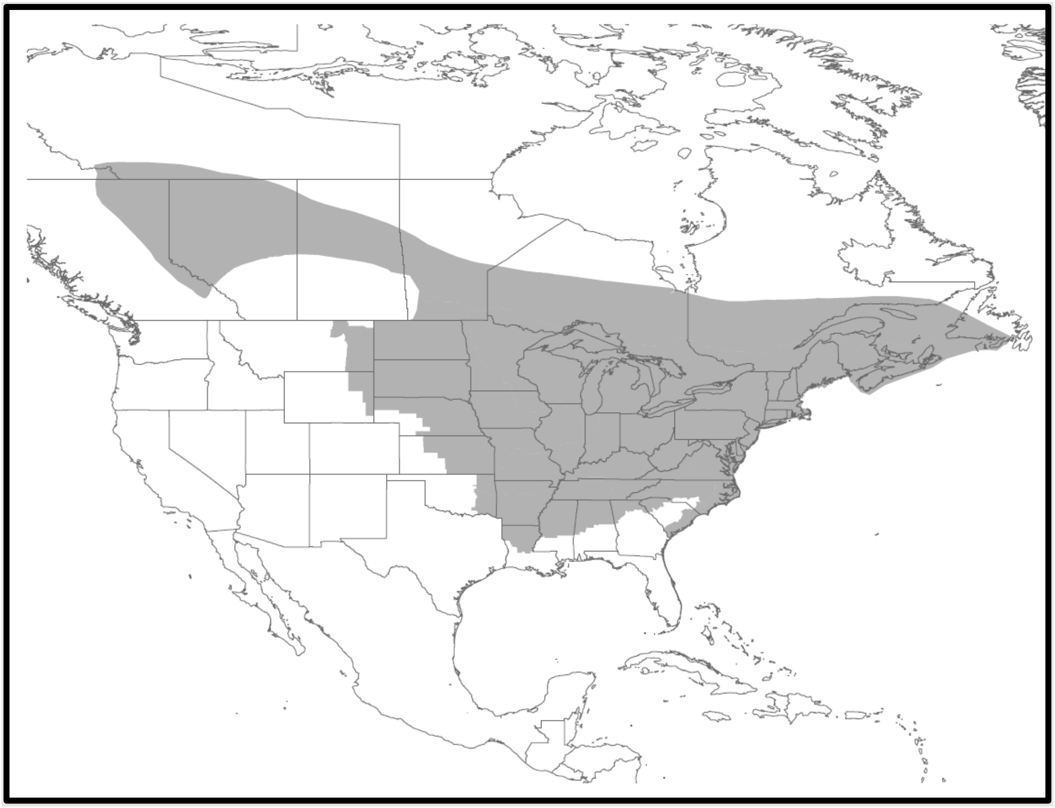


Figure 2.2. Range map for NLEB

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[2] The range map was developed using the USFWS's NLEB range map for the U.S. in combination with IUCN's map for Canada (https://www.iucnredlist.org/species/14201/22064312). USFWS maintains a range map using known locations of NLEB. The range boundary is updated as new information is received and can be found here: https://www.fws.gov/Midwest/Endangered/mammals/nleb/nlebRangeMap.html.

Individual Needs and Ecology

Below we describe the life history and ecological needs for NLEB individuals to survive and reproduce; ecological needs are summarized in Table 2.1. The generalized annual life history is summarized for NLEB in Figure 2.3.

Swarming (Fall)

The swarming season occurs between the summer and winter seasons (Lowe 2012, p. 50) and the purpose of swarming behavior may include: introduction of juveniles to potential hibernacula, copulation, and stopping over sites on migratory pathways between summer and winter regions (Kurta et al. 1997, p. 479; Parsons et al. 2003, p. 64; Lowe 2012, p. 51; Randall and Broders 2014, pp. 109–110). During this period, heightened activity and congregation of transient bats around caves and mines is observed, followed later by increased sexual activity and bouts of torpor prior to winter hibernation (Davis and Hitchcock 1965, pp. 304–306; Fenton 1969, p. 601; Parsons et al. 2003, pp. 63–64). For the NLEB, the swarming period may occur between July and early October, depending on latitude within the species' range (Hall and Brenner 1968, p. 780; Fenton 1969, p. 598; Caire et al. 1979, p. 405; Kurta et al. 1997, p. 479; Lowe 2012, p. 86;). The NLEB may investigate several cave or mine openings during the transient portion of the swarming period, and some individuals may use these areas as temporary daytime roosts or may roost in forest habitat adjacent to these sites (Kurta et al. 1997, pp. 479, 483; Lowe 2012, p. 51). Many of the caves and mines associated with swarming are also used as hibernacula for several species of bats, including the NLEB (Fenton 1969, p. 599; Whitaker and Rissler 1992, p. 132; Kurta et al. 1997, p. 484; Glover and Altringham 2008, p. 1498; Randall and Broders 2014, p. 109).

Winter Hibernation

NLEBs are thought to predominantly overwinter in hibernacula that include caves and abandoned mines. These hibernacula have relatively constant, cooler temperatures (0 to 9 degrees Celsius [°C] or 32 to 48 degrees Fahrenheit [°F]) (Raesly and Gates 1987, p. 18; Caceres and Pybus 1997, p. 2; Brack 2007, p. 744), with high humidity and no strong currents (Fitch and Shump 1979, p. 2; van Zyll de Jong 1985, p. 94; Raesly and Gates 1987, p. 118; Caceres and Pybus 1997, p. 2). NLEBs are typically found roosting singly or in small numbers in cave or mine walls or ceilings, often in small crevices or cracks, sometimes with only the nose and ears visible and thus are easily overlooked during surveys (Griffin 1940a, pp. 181–182; Barbour and Davis 1969, p. 77; Caire et al. 1979, p. 405; van Zyll de Jong 1985, p. 9; Caceres and Pybus 1997, p. 2; Whitaker and Mumford 2009, pp. 209–210).

NLEBs have also been observed overwintering in other types of habitat that have similar conditions (e.g., temperature, humidity levels, air flow) to cave or mine hibernacula. The species may use these alternate hibernacula in areas where caves or mines are not present (Griffin 1945, p. 22). NLEBs have been found using the following alternative hibernacula: abandoned railroad tunnels (USFWS 2015, p. 17977), the entrance of a storm sewer in central Minnesota (Goehring 1954, p. 435), a hydroelectric dam facility in Michigan (Kurta et al. 1997, p. 478), an aqueduct in

Massachusetts (Massachusetts Department of Fish and Game 2012, unpublished data), a dry well in Massachusetts (Griffin 1945, p. 22). More recently, NLEBs were found in a crawl space within a dwelling in Massachusetts (Dowling and O'Dell 2018, p. 376) and a rock crevice in Nebraska (White et al. 2020, p. 114). Further, Girder et al. (2016, p. 11) found NLEB to be present and active year round on the coastal plain of North Carolina, where there is no known non-cavernicolous (cave-like) hibernacula; therefore, it is possible this population was not (traditionally) hibernating. Also, in coastal North Carolina, NLEB were observed to be active the majority of the winter, and although torpor was observed, time spent in torpor was very short with the longest torpor bout (i.e., hibernation period) for each bat averaging 6.8 days (Jordan 2020, p. 672).

Summer Roosting

Roosting habitat–NLEBs typically roost singly or in maternity colonies underneath bark or more often in cavities or crevices of both live trees and snags (Sasse and Pekins 1996, p. 95; Foster and Kurta 1999, p. 662; Owen et al. 2002, p. 2; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 222; Timpone et al. 2010, p. 119). Males' and non-reproductive females' summer roost sites may also include cooler locations, including caves and mines (Barbour and Davis 1969, p. 77; Amelon and Burhans 2006, p. 72). Studies have documented the NLEB's selection of both live trees and snags (Sasse and Pekins 1996, p. 95; Foster and Kurta 1999, p. 668; Lacki and Schwierjohann 2001, p. 484; Menzel et al. 2002, p. 107; Carter and Feldhamer 2005, p. 262; Perry and Thill 2007, p. 224; Timpone et al. 2010, p. 118). NLEBs are flexible in tree species selection and while they may select for certain tree species regionally, likely are not dependent on certain species of trees for roosts throughout their range; rather, many tree species that form suitable cavities or retain bark will be used by the bats opportunistically (Foster and Kurta 1999, p. 668; Silvis et al. 2016, p. 12; Hyzy 2020, p. 62). Carter and Feldhamer (2005, p. 265) hypothesized that structural complexity of habitat or available roosting resources are more important factors than the actual tree species. Further, Silvis et al. (2012, p. 7) found forest successional patterns, stand and tree structure to be more crucial than tree species in creating and maintaining suitable long-term roosting opportunities. To a lesser extent, NLEBs have also been observed roosting in colonies in human-made structures, such as in buildings, in barns, on utility poles, behind window shutters, in bridges, and in bat houses (Mumford and Cope 1964, p. 72; Barbour and Davis 1969, p. 77; Cope and Humphrey 1972, p. 9; Burke 1999, pp. 77–78; Sparks et al. 2004, p. 94; Amelon and Burhans 2006, p. 72; Whitaker and Mumford 2009, p. 209; Timpone et al. 2010, p. 119; Bohrman and Fecske 2013, pp. 37, 74; ; Feldhamer et al. 2003, p. 109; Sasse et al. 2014, p. 172; USFWS 2015, p. 17984; Dowling and O'Dell 2018, p. 376). It has been hypothesized that use of human-made structures may occur in areas with fewer suitable roost trees (Henderson and Broders 2008, p. 960; Dowling and O'Dell 2018, p. 376). In north-central West Virginia, NLEBs were found to more readily use artificial roosts as distance from large forests (greater than 200 hectares [494 acres]) increased, suggesting that artificial roosts are less likely to be selected when there is greater availability of suitable roost trees (De La Cruz et al. 2018, p. 496).

Roosting behavior–Maternity colonies, consisting of females and young, are generally small, numbering from about 30 (Whitaker and Mumford 2009, p. 212) to 60 individuals (Caceres and Barclay 2000, p. 3); however, larger colonies of up to 100 adult females have been observed (Whitaker and Mumford 2009, p. 212). Most studies have found that the number of individuals roosting together in a given roost typically decreases from pregnancy to post-lactation (Foster and Kurta 1999, p. 667; Lacki and Schwierjohann 2001, p. 485; Garroway and Broders 2007, p. 962; Perry and Thill 2007, p. 224; Johnson et al. 2012, p. 227). NLEBs exhibit fission-fusion behavior (Garroway and Broders 2007, p. 961), where members frequently coalesce to form a group (fusion), but composition of the group is in flux, with individuals frequently departing to be solitary or to form smaller groups (fission) before returning to the main spatially discrete unit or network (Barclay and Kurta 2007, p. 44). As part of this behavior, NLEBs switch tree roosts often (Sasse and Pekins 1996, p. 95), typically every 2 to 3 days (Foster and Kurta 1999, p. 665; Owen et al. 2002, p. 2; Carter and Feldhamer 2005, p. 261; Timpone et al. 2010, p. 119). Patriquin et al. (2016, p. 55) found that NLEB roost switching and use varies regionally in response to differences in ambient conditions (e.g., precipitation, temperature). Adult females give birth to a single pup (Barbour and Davis 1969, p. 104). Birthing within the colony tends to be synchronous, with the majority of births occurring around the same time (Krochmal and Sparks 2007, p. 654). Parturition (birth) may occur as early as late May or early June (Easterla 1968, p. 770; Caire et al. 1979, p. 406; Whitaker and Mumford 2009, p. 213) and may occur as late as mid-July (Whitaker and Mumford 2009, p. 213). Juvenile volancy (flight) often occurs by 21 days after birth (Kunz 1971, p. 480; Krochmal and Sparks 2007, p. 651) and has been documented as early as 18 days after birth (Krochmal and Sparks 2007, p. 651).

Foraging (Spring, Summer, Fall)

Diet–NLEBs are nocturnal foragers and use hawking (catching insects in flight) and gleaning (picking insects from surfaces) behaviors in conjunction with passive acoustic cues (Nagorsen and Brigham 1993, p. 88; Ratcliffe and Dawson 2003, p. 851). The NLEB has a diverse diet including moths, flies, leafhoppers, caddisflies, and beetles (Griffith and Gates 1985, p. 452; Nagorsen and Brigham 1993, p. 88; Brack and Whitaker 2001, p. 207), with diet composition differing geographically and seasonally (Brack and Whitaker 2001, p. 208). The most common insects found in the diets of NLEBs are lepidopterans (moths) and coleopterans (beetles) (Brack and Whitaker 2001, p. 207; Lee and McCracken 2004, pp. 595–596; Feldhamer et al. 2009, p. 45; Dodd et al. 2012, p. 1122), with arachnids also being a common prey item (Feldhamer et al. 2009, p. 45).

Foraging behavior–Most foraging occurs above the understory, 1 to 3 m (3 to 10 ft) above the ground, but under the canopy (Nagorsen and Brigham 1993, p. 88) on forested hillsides and ridges, rather than along riparian areas (LaVal et al. 1977, p. 594; Brack and Whitaker 2001, p. 207). This coincides with data indicating that mature forests are an important habitat type for foraging NLEBs (Caceres and Pybus 1997, p. 2; White et al. 2017, p. 8). Foraging also takes place over small forest clearings and water, and along roads (van Zyll de Jong 1985, p. 94). NLEB seem to prefer intact mixed-type forests with small gaps (i.e., forest trails, small roads, or forest-covered creeks) in forest with sparse or medium vegetation for forage and travel rather than fragmented habitat or areas that have been clear cut (USFWS 2015, p. 17992). Foraging patterns indicate a peak activity period within 5 hours after sunset followed by a secondary peak within 8 hours after sunset (Kunz 1973, pp. 18–19). Brack and Whitaker (2001, p. 207) did not find significant differences in the overall diet of NLEBs between morning (3 a.m. to dawn) and evening (dusk to midnight) feedings; however there were some differences in the consumption of particular prey orders between morning and evening feedings. Additionally, no significant differences existed in dietary diversity values between age classes or sex groups (Brack and Whitaker 2001, p. 208).

Staging (Spring)

Spring staging for the NLEB is the time period between winter hibernation and spring migration to summer habitat (Whitaker and Hamilton 1998, p. 80). During this time, bats begin to gradually emerge from hibernation, exit the hibernacula to feed, but re-enter the same or alternative hibernacula to resume daily bouts of torpor (state of mental or physical inactivity) (Whitaker and Hamilton 1998, p. 80). The staging period for the NLEB is likely short in duration (Caire et al. 1979, p. 405; Whitaker and Hamilton 1998, p. 80). In Missouri, Caire et al. (1979, p. 405) found that NLEBs moved into the staging period in mid-March through early May. Sasse et al. (2014, p. 172) found pregnant NLEB using a mine in late April and May in Arkansas. In Michigan, Kurta et al. (1997, p. 478) determined that by early May, two-thirds of the *Myotis* species, including the NLEB, had dispersed to summer habitat. Variation in timing (onset and duration) of staging for Indiana bats (*Myotis sodalis*) was based on latitude and weather (USFWS 2007, pp. 39–40, 42); similarly, timing of staging for NLEBs is likely based on these same factors.

Migration (Spring and Fall)

While information is lacking, short regional migratory movements between seasonal habitats (summer roosts and winter hibernacula) of 56 kilometer (km) (35 mi) to 89 km (55 mi) have been documented (Griffin 1940b, pp. 235, 236; Caire et al. 1979, p. 404; Nagorsen and Brigham 1993 p. 88). The spring migration period typically runs from mid-March to mid-May (Easterla 1968, p. 770; Caire et al. 1979, p. 404; Whitaker and Mumford 2009, p. 207); fall migration typically occurs between mid-August and mid-October.

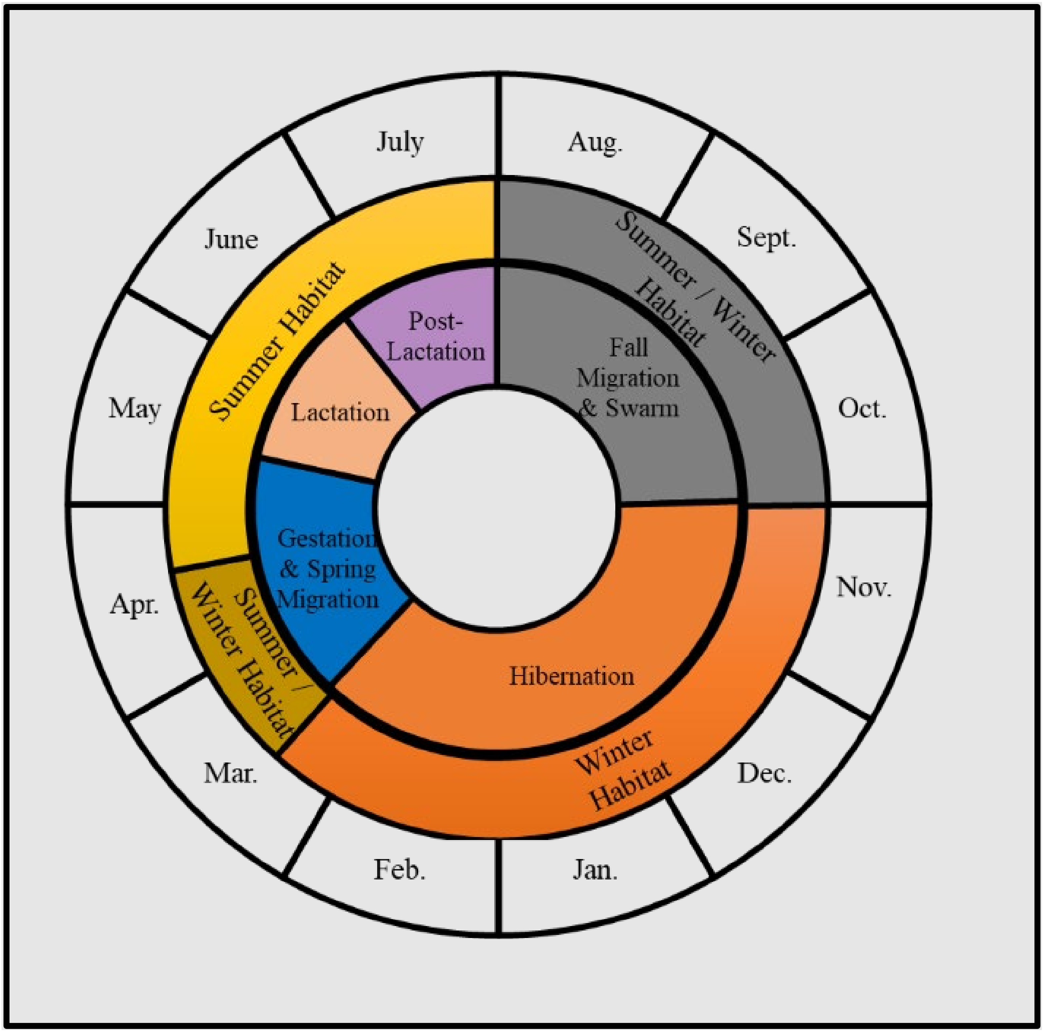

Figure 2.3. Generalized annual life history diagram for NLEB (adapted from Silvis et al. 2016, p. 1).

Table 2.1. The ecological requisites for survival and reproductive success of individuals.

LIFE STAGE	**SEASON**			
	Spring	**Summer**	**Fall**	**Winter**
Pups		Roosting habitat with suitable conditions for lactating females, and for pups to stay warm and protected from predators while adults are foraging.		
Juveniles		Other maternity colony members (colony dynamics, thermoregulation); Suitable roosting and foraging habitat near abundant food and water resources.	Suitable roosting and foraging habitat near abundant food and water resources.	Habitat with suitable conditions for prolonged bouts of torpor and shortened periods of arousal.
All Adults	Suitable roosting and foraging habitat near abundant food and water resources. Habitat connectivity and open air space for safe migration between winter and summer habitats.	Summer roosts and foraging habitat near abundant food and water resources.	Suitable roosting and foraging habitat near abundant food and water resources; Cave and/or mine entrances (or other similar locations, e.g., culvert, tunnel) for conspecifics to swarm and mate; Habitat connectivity and open air space for safe migration between winter and summer habitats.	Habitat with suitable conditions for prolonged bouts of torpor and shortened periods of arousal.
Reproductive Females		Other maternity colony members (colony dynamics); Network of suitable roosts (i.e.,		

LIFE STAGE	SEASON			
		multiple summer roosts in close proximity) near conspecifics and foraging habitat near abundant food and water resources.		

Population-level Needs

To be self-sustaining, a population must be demographically, genetically, and physically healthy (see Redford et al. 2011, entire). Demographically healthy means having robust survival, reproductive, and growth rates. Genetically healthy populations have large effective population sizes (*Ne*), high heterozygosity, and gene flow between populations. Physically healthy means individuals have good body condition. The population-level ecological requirements of a healthy NLEB population are discussed further below and summarized in (Figure 2.4 and Table 2.2).

Similar to other temperate bat species, NLEB hibernation conditions, prey availability, summer roosting habitat, and connectivity between habitats influence population growth rates and reproduction rates (Figure 2.4). For NLEB populations to be demographically healthy, their growth rate (lambda, or λ) must be sufficient to withstand natural environmental fluctuations. For a population to remain stable (or increasing) over time, λ must be greater than or equal to one. Although variations to summer and winter habitat conditions may result in lower demographic health of a population, NLEB does not generally experience extreme variation in demographics year-to-year due to their selection of summer and winter habitat with narrow microclimate conditions (see *Individual-Level Ecology and Requirements*). During favorable hibernation and summer habitat conditions, NLEB survival and therefore reproductive rates are greater (increasing λ); conversely, when environmental conditions are unfavorable, survival and reproductive rates are lower (decreasing λ).

To support a strong growth rate, NLEB populations benefit from large population sizes and sufficient quality and quantity of habitat to accommodate all life stages. Large effective population size is crucial in maintaining genetic health along with and withstanding environmental variability. Habitat requirements for NLEB are described under *Individual-level Ecology and Needs*. The necessary quantity of habitat is likely to vary among populations, but will likely hinge on the availability of roosting habitat in the summer and suitable hibernacula in the winter. Research has found the minimum summer roost area (i.e., area encompassing all known roost locations) for individual female NLEB ranges between 5.4 hectares and 26 hectares (13 acres and 65 acres), but most studies found the summer roost area to be leaning toward the smaller end of the range (Owen et al. 2003, p. 353; Broders et al. 2006, p. 1177; Badin 2014, p. 75).

To support all life stages, NLEB populations require a matrix of interconnected habitats that support spring migration, summer maternity colony formation and foraging, fall swarming, and winter hibernation. For these populations, movement among habitats is needed to maintain genetic diversity and to allow recolonization in the event of local extirpation. NLEB may migrate short distances between seasonal habitats (summer roosts and winter hibernacula) between 56 km (35 mi) and 89 km (55 mi), as previously mentioned (Griffin 1940b, pp. 235, 236; Caire et al. 1979, p. 404; Nagorsen and Brigham 1993 p. 88). There is evidence that NLEBs have an affinity for less fragmented habitat (interior forest) (Broders et al,. 2006, p. 1181; Henderson et al. 2008, p. 1825). Therefore, increased fragmentation may negatively impact connectivity between summer and winter habitat and between roosting and foraging habitat.

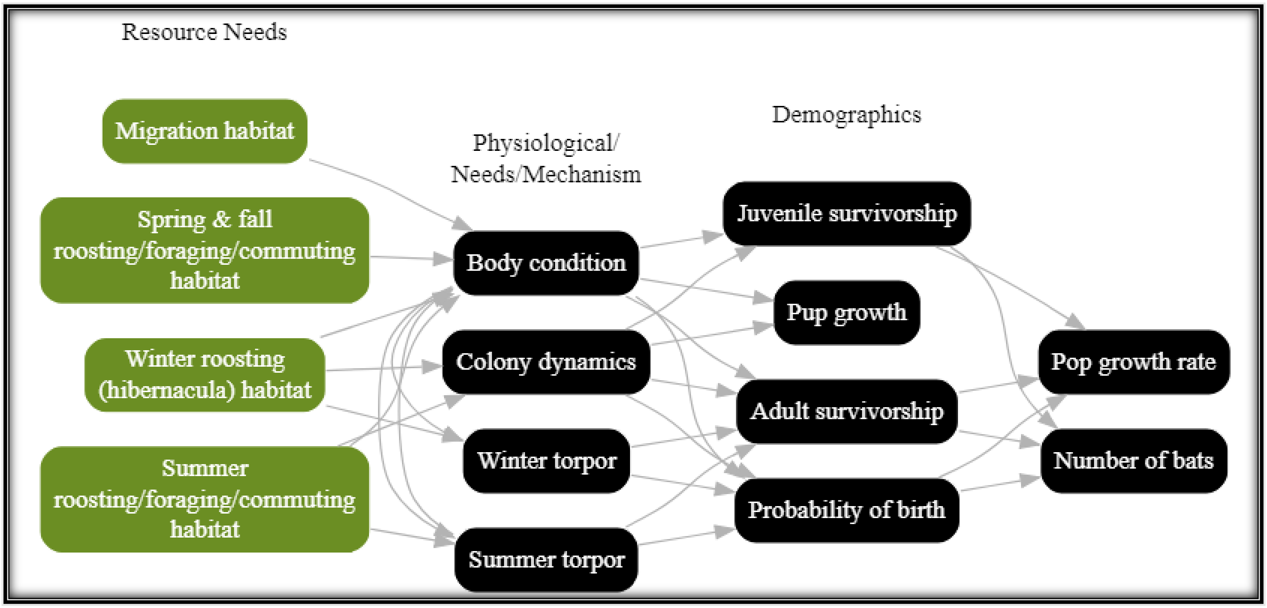

Figure 2.4. Conceptual model showing the connections between resource needs and the physiological needs and demographic rates of a NLEB population (population-level resiliency).

Table 2.2. Population level requirements for a healthy population.

Parameter	Requirements
Population growth rate, λ	At a minimum, λ must be ≥ 1 for a population to remain stable over time.
Population size, N	Sufficiently large N to allow for essential colony dynamics and to be resilient to environmental fluctuations.
Winter roosting habitat	Safe and stable winter roosting sites with suitable microclimates.
Migration habitat	Safe space to migrate between spring/fall habitat and winter roost sites.
Spring and fall roosting, foraging, and commuting habitat	A matrix of habitat of sufficient quality and quantity to support bats as they exit hibernation (lowest body condition) or as they enter into hibernation (need to put on body fat).
Summer roosting, foraging, and commuting habitat	A matrix of habitat of sufficient quality and quantity to support maternity colonies.

Species-level Needs

The ecological requisites at the species level include having a sufficient number and distribution of healthy populations to ensure NLEB can withstand annual variation in its environment (resiliency), catastrophes (redundancy), and novel or extraordinary changes in its environment (representation). We describe NLEB's requirements for resiliency, redundancy, and representation below, and summarize the key aspects in Table 2.3.

Resiliency

NLEB's ability to withstand stochastic events requires maintaining healthy populations across spatially heterogeneous conditions. Healthy populations-- demographically, genetically, or physically robust--are more likely to withstand and recover from environmental and demographic variability and stochastic perturbations. The greater the number of healthy populations, the more likely NLEB will withstand perturbations and natural variation, and hence, have greater resiliency. Additionally, occupying a diversity of environmental conditions and being widely distributed helps guard against populations fluctuating in synchrony (i.e., being exposed to adverse conditions concurrently). Asynchronous dynamics among populations minimizes the chances of concurrent losses, and thus, provides species' resiliency. Lastly, maintaining the natural patterns and levels of connectivity between populations also contributes to NLEB resiliency by facilitating population-level heterozygosity via gene flow and demographic rescue following population decline or extinction due to stochastic events.

Redundancy

NLEB's ability to withstand catastrophic events requires having multiple, widely distributed populations relative to the spatial occurrence of catastrophic events. In addition to guarding against population extirpation, redundancy is important to protect against losses in NLEB's adaptive capacity. Multiple, widely distributed populations within areas of unique diversity will guard against losses of adaptive capacity due to catastrophic events, such as extreme winter events, epizootics, and hurricanes.

Representation

NLEB's ability to withstand ongoing and future novel changes is influenced by its capacity to adapt (referred to as adaptive capacity). NLEB may adapt to novel changes by either moving to new, suitable environments or by altering (via plasticity or genetic change) its physical or behavioral traits to match the new environmental conditions. There are multiple intrinsic factors that limit the species ability to adapt to a rapidly changing environment (see Appendix 2-B). Below we describe NLEB's ability to colonize new areas and to alter its physical traits.

NLEB's capacity to colonize new areas (or track suitable conditions) is a function of its physical capability and behavioral tendencies to disperse. NLEB exhibits capabilities (e.g., flight) and behavior (e.g., fission-fusion) that allows for colonization of new areas. NLEB switch summer roosts for a variety of reasons, including temperature, precipitation, predation, parasitism, sociality, and ephemeral roost sites (Carter and Feldhamer 2005, p. 264; Patriquin et al. 2016, p. 55). In addition, although to a lesser extent, NLEB has been found using human-made structures for summer roosts (see *Individual-level Ecology and Needs*). It has been suggested that use of human-made structures may occur in areas with fewer suitable roost trees or lower proximity to larger patches of habitat (Henderson and Broders 2008, p. 960; De La Cruz et al. 2018, p. 496; Dowling and O'Dell 2018, p. 376). Therefore, NLEB has the ability to inhabit new summer roosting habitat at the local level provided that suitable habitat (see *Individual-level Ecology and Needs)* is in close proximity. However, the species may lack the capacity for rapid, large shifts in response to broad-scale novel changes to summer habitat. Maintaining suitable habitat within local home-ranges and beyond is needed to allow for any capacity to shift their range to track suitable conditions. With regard to NLEB's ability to colonize new winter hibernacula, although the species is capable of arousing from torpor and moving between hibernacula during the winter (Griffin 1940a, p. 185; Whitaker and Rissler 1992, p. 131; Caceres and Barclay 2000, pp. 2–3), arousal and movement come at a high energetic cost (Thomas and Geiser 1997, p. 585). NLEB's high degree of site fidelity for a hibernaculum (Pearson 1962, p. 30) also limit their capabilities to inhabit new hibernacula at a broad-scale.

NLEB's capacity to alter its physical or behavioral traits (phenotypes) to match the new environmental conditions is driven by the breadth of adaptive genetic variation. Thus, maintaining populations across the breadth of variation preserves NLEB's capacity to adapt to ongoing and future changes. In addition to preserving the breadth of variation, it is also necessary to maintain the key evolutionary processes through which adaptation occurs, namely, natural selection, gene flow, and genetic drift. Maintaining healthy NLEB populations across a diversity of environments and climatic conditions as well as keeping natural networks of genetic connections between populations allows for such adaptation, via natural selection or gene flow; and preserving large effective population abundances, ensures genetic drift does not act unduly upon the species (see Chapter 1 for further explanation).

For reasons explained in Chapter 1, we rely on proxies to identify species' adaptive genetic variation. We identified and delineated the genetic variation across NLEB's range into geographical representation units using the following proxies: variation in biological traits, neutral genetic diversity, peripheral populations, habitat niche diversity, and steep environmental gradients. These representation units (RPUs) are described below and displayed in Figure 2.5. Bailey's Eco-Divisions (Bailey 2016, entire) were overlayed on these proxies to identify approximate boundaries due to the associated climatic differences (i.e., precipitation levels, patterns and temperatures) that may be influential in driving the species' adaptive ability. By establishing these RPUs (a combination of proxies and Bailey's Eco-Divisions) the underlying adaptive variation of NLEB (at a broad scale) is preserved.

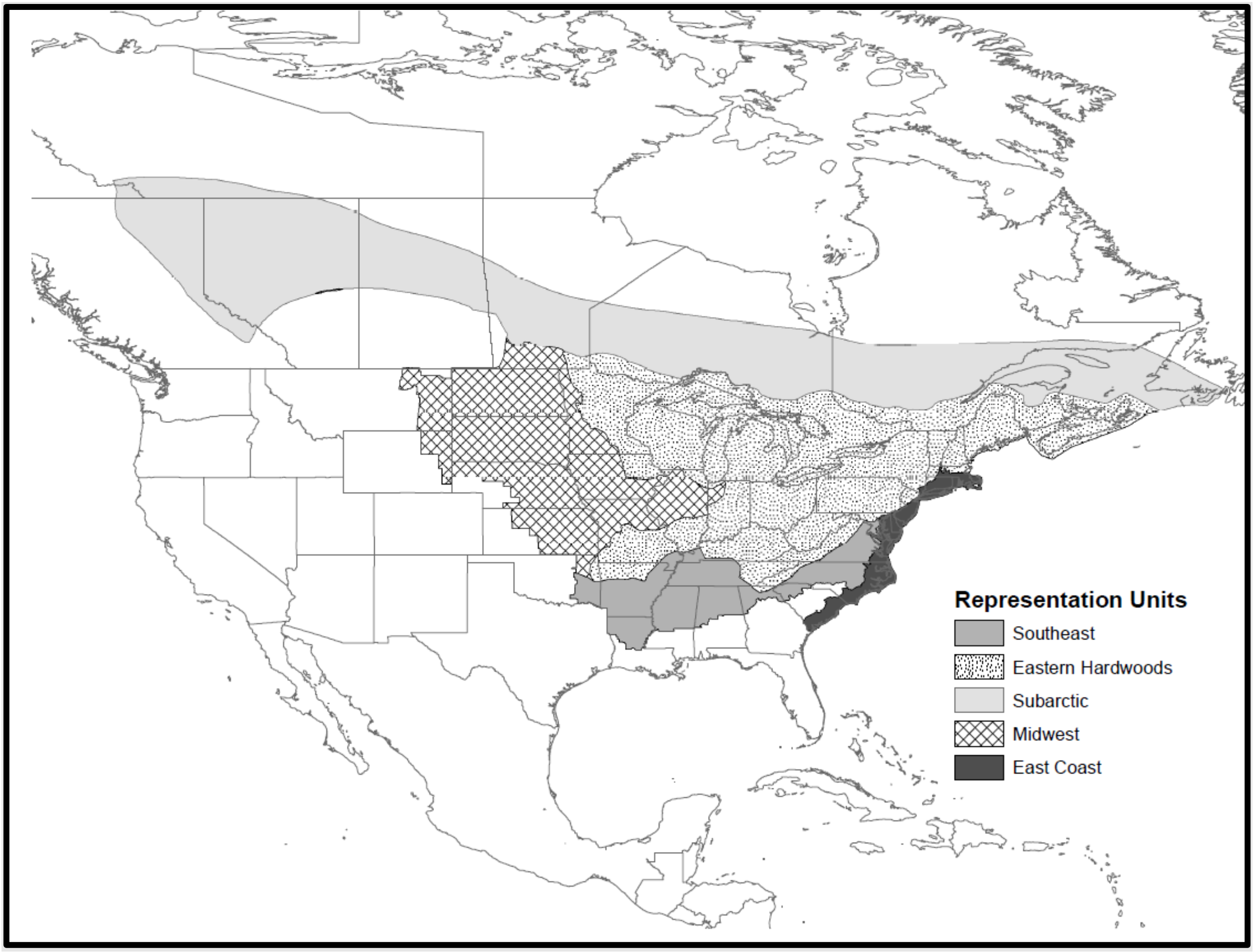

Figure 2.5. Range of NLEB organized into five Representation Units.

1. *Southeast RPU:* In general, NLEB have shorter hibernation periods in this unit (in comparison to the Eastern hardwoods and Subarctic units). Hibernation period correlates with average minimum temperatures and other climatic features, and thus, we used the minimum average temperature zones, specifically zones 6 and 7 in combination with Bailey's Ecoregions "Hot continental" and "Subtropical" divisions to circumscribe variation in hibernation periods.

2. *Eastern Hardwoods RPU:* The Eastern hardwoods Unit was established based on differences in hibernation duration and landcover. NLEB have longer hibernation periods in the Eastern hardwoods unit (in comparison with the Southeast unit). The northern border of this unit was separated from the Subarctic unit based on minimum average temperature zone lines, specifically zones 2 and 3 in combination with Bailey's Ecoregions "Warm continental" and "Subarctic" divisions.

3. *Subarctic RPU:* The Subarctic unit was established based on assumed longer hibernation periods relative to the Eastern hardwoods and Southeastern units. Unlike for the Eastern hardwoods and Southeast units, data on hibernation duration is lacking for the Subarctic unit. However, given hibernation is influenced by minimum winter temperatures, we

assume longer hibernation periods in northern portions of the species' range. The line that was established between the Eastern hardwoods and Subarctic units is described above under the Eastern hardwoods unit description.

4. *Midwest RPU:* The Midwest unit was established based primarily on markedly different landcover than other units, with limited or fragmented forested habitat prevailing throughout much of this unit. Unlike the other units, the Midwest Unit is largely non-forested landcover (e.g., grassland/pasture, cultivated crops, and pasture/hay; Appendix 4-D, NLCD 2016).
5. *East Coast RPU:* The Coastal unit was established based on observed NLEB atypical behavior (e.g., year-round activity, use of non-cavernicolous hibernacula). Southern coastal populations have been observed with similar activity levels year-round in areas with no known nearby traditional hibernacula (i.e., caves or mines; Girder et al. 2016, p. 11; Jordan 2020, p. 672). Further, northern coastal populations have been observed using alternate summer roosting habitat (e.g., human dwellings) and non-cavernicolous hibernacula (e.g., house crawl spaces, Dowling and O'Dell 2018, p. 376).

Table 2.3. Species-level ecology: Requisites for long-term viability (ability to maintain self-sustaining populations over a biologically meaningful timeframe).

3 Rs	**Requisites Long-Term Viability**	**Description**
Resiliency (populations able to withstand stochastic events)	Demographic, physically, and genetically healthy populations across a diversity of environmental conditions	Self-sustaining populations are demographically, genetically, and physiologically robust, have sufficient quantity of suitable habitat
Redundancy (number & distribution of populations to withstand catastrophic events)	Multiple and sufficient distribution of populations within areas of unique variation, i.e., Representation units	Sufficient number and distribution to guard against population losses and losses in species adaptive diversity, i.e., reduce covariance among populations; spread out geographically but also ecologically
Representation (genetic & ecological diversity to maintain adaptive potential)	Maintain adaptive diversity of the species	Populations maintained across breadth of behavioral, physiological, ecological, and environment diversity
	Maintain evolutionary processes	Maintain evolutionary drivers--gene flow, natural selection--to mimic historical patterns

CHAPTER 3 – HISTORICAL CONDITION

This chapter describes the number, health, and distribution of NLEB populations up to the present day. The historical condition provides the baseline condition from which we evaluated changes in NLEB viability over time (Figure 3.1).

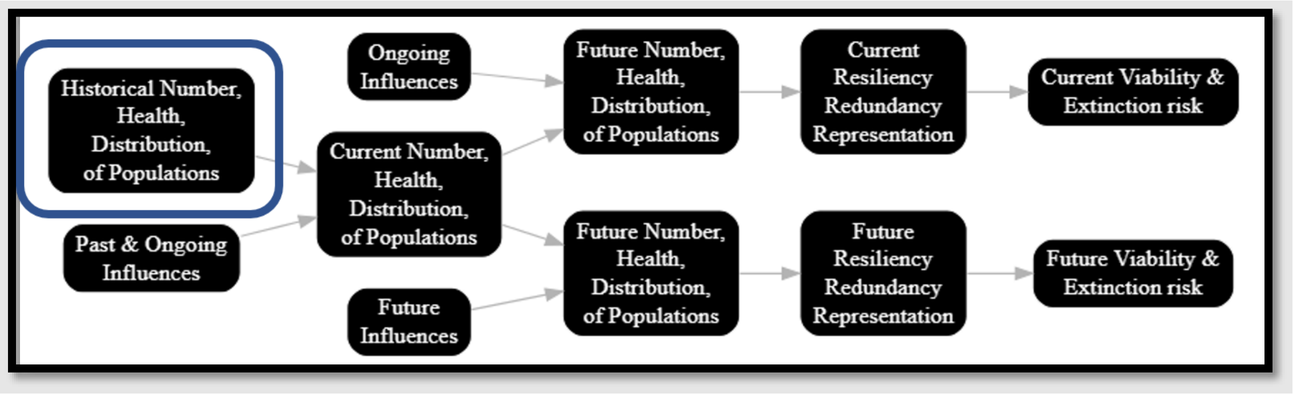

Figure 3.1. Highlighting (blue rectangle) the current step in our analytical framework.

Prior to 2006 (i.e., before WNS was first documented; see Chapter 4), NLEB was abundant and widespread throughout much of its range (despite having low winter detectability) with 737 occupied hibernacula, a maximum count of 38,181 individuals and its range being spread across >1.2 billion acres in 29 states and 3 Canadian provinces (Figure 3.2, Table A-3A1)[3]. NLEB numbers vary temporally and spatially, but abundance and occurrence on the landscape were stable (Cheng et al. 2022, p. 204; Wiens et al. 2022, p. 233). Winter colony sizes ranged from small (less than 100) to large (greater than 100), although the vast majority of individuals included in our dataset occupied a small subset of hibernacula; for example, in 2000, 16.6% (n = 66) of the known winter colonies contained 90% of total winter abundance.

Historically, the core of NLEB's range was centered in the Eastern Hardwoods RPU. This RPU encompasses approximately 90% of the total number of known hibernacula and 78% of the known winter abundance. The Southeast RPU contained 7% of the sites and 1% of total abundance, while the Subarctic RPU comprised 1% of the sites and 14% of the abundance. The Midwest and East Coast RPUs comprised 1% of the sites and 3% and 4% of the abundance, respectively (Table A-3A2).

The summer range for NLEB encompasses 37 states and 8 Canadian provinces (Figure 2.2). In this SSA, we have records of occurrences (i.e., NLEB acoustic calls, mist-net captures, and hibernacula records) from 37 states, the District of Columbia and 7 provinces (Figure 3.3).

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[3] Hibernacula count numbers, number of hibernacula, and spatial range only represent NLEB available (i.e., usable format, provided within certain timeframe) winter records submitted to NABat (NABat 2021) for use in this SSA; we acknowledge historical NLEB abundance, number of hibernacula, and spatial range were likely higher.

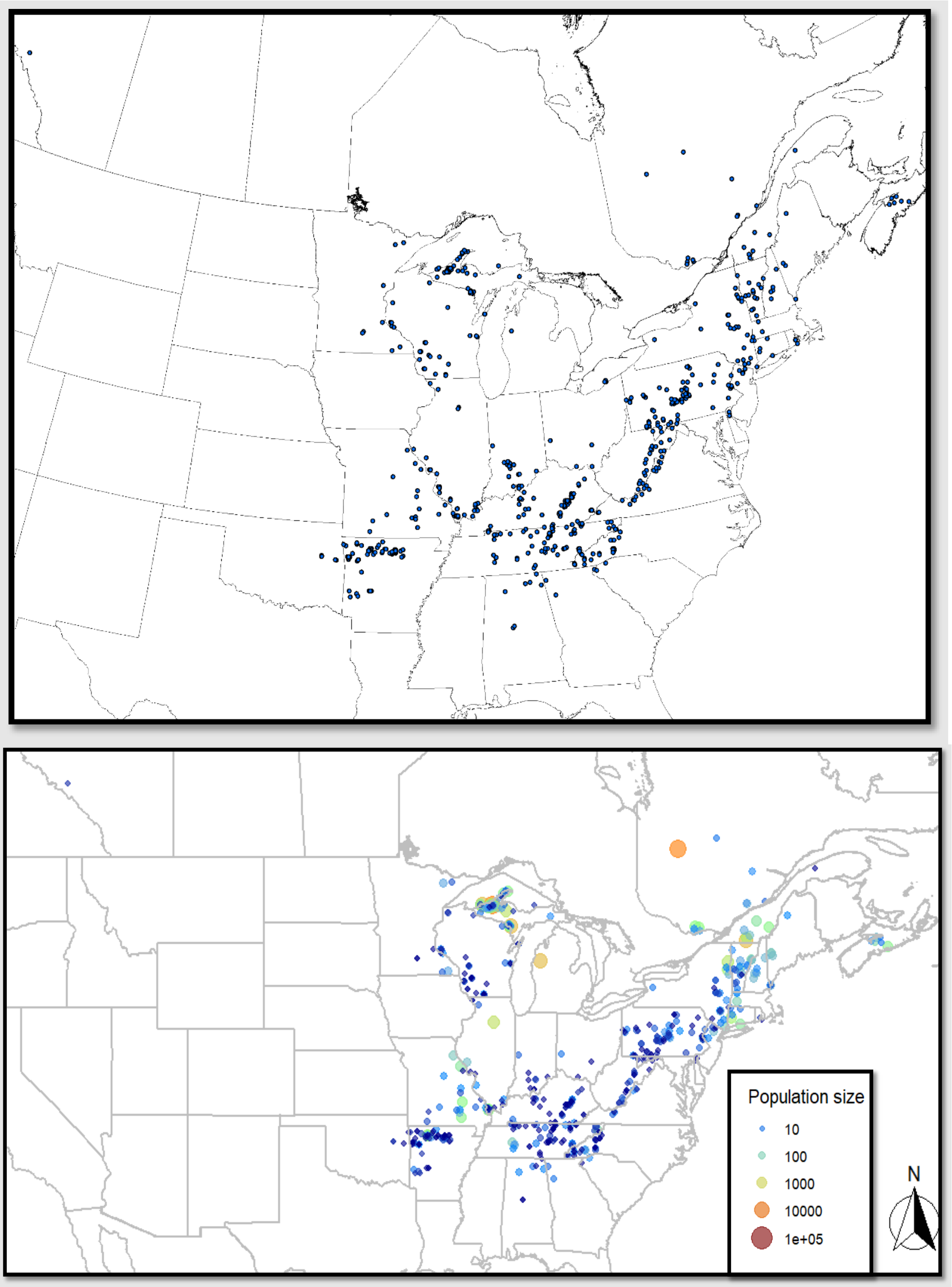


Figure 3.2. All known historical hibernacula (top figure) and winter abundances at hibernacula in 2000 (bottom figure). Point color and size corresponds to maximum colony count size at a hibernaculum.

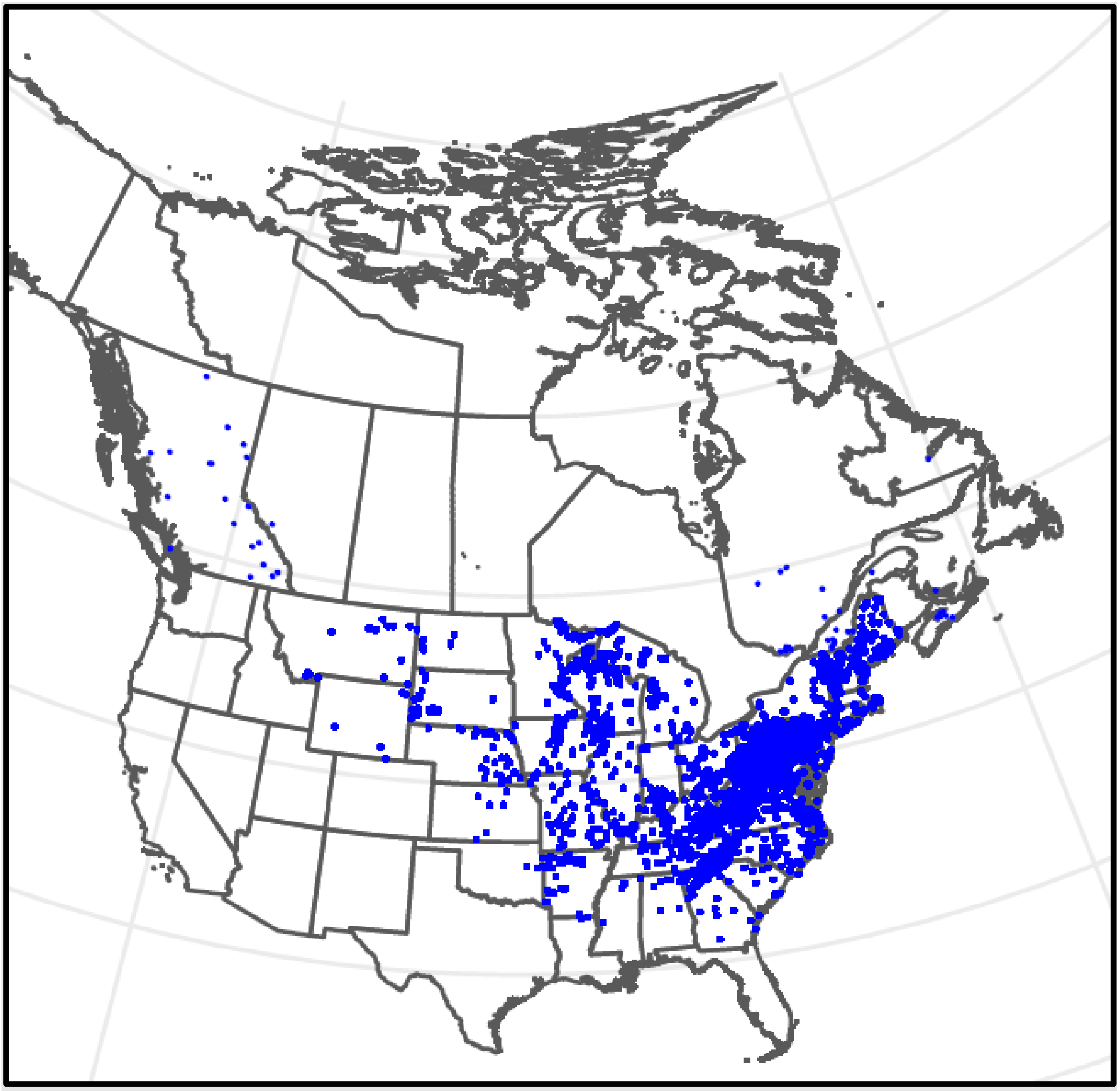


Figure 3.3. Documented range of NLEB as known from available acoustic calls, captures, and hibernacula records (records indicated by blue dots) in the U.S. and Canada. (Map credit: B. Udell, U.S. Geological Survey, Fort Collins Science Center. Disclaimer: Provisional information is subject to revision). This map shows data provided to the SSA and does not replace the accepted species range (Figure 2.2).

CHAPTER 4 – PRIMARY INFLUENCES ON VIABILITY

Recognizing there are myriad influences operating on NLEB, this chapter describes the primary threats that have most likely led to its current condition: WNS, wind related mortality, effects from climate change, and habitat loss (Figures 4.1 and 4.2). We similarly describe the primary past and ongoing conservation efforts that may be ameliorating these threats. Lastly, for WNS and wind related mortality we describe the plausible future condition for each threat. To capture the uncertainty in our future projections, we identified the lowest plausible and highest plausible state for each primary threat. These lower and upper impact states for each threat were then combined to create composite plausible “low impact” and “high impact” scenarios. For climate change and habitat loss, we provide qualitative assessments.

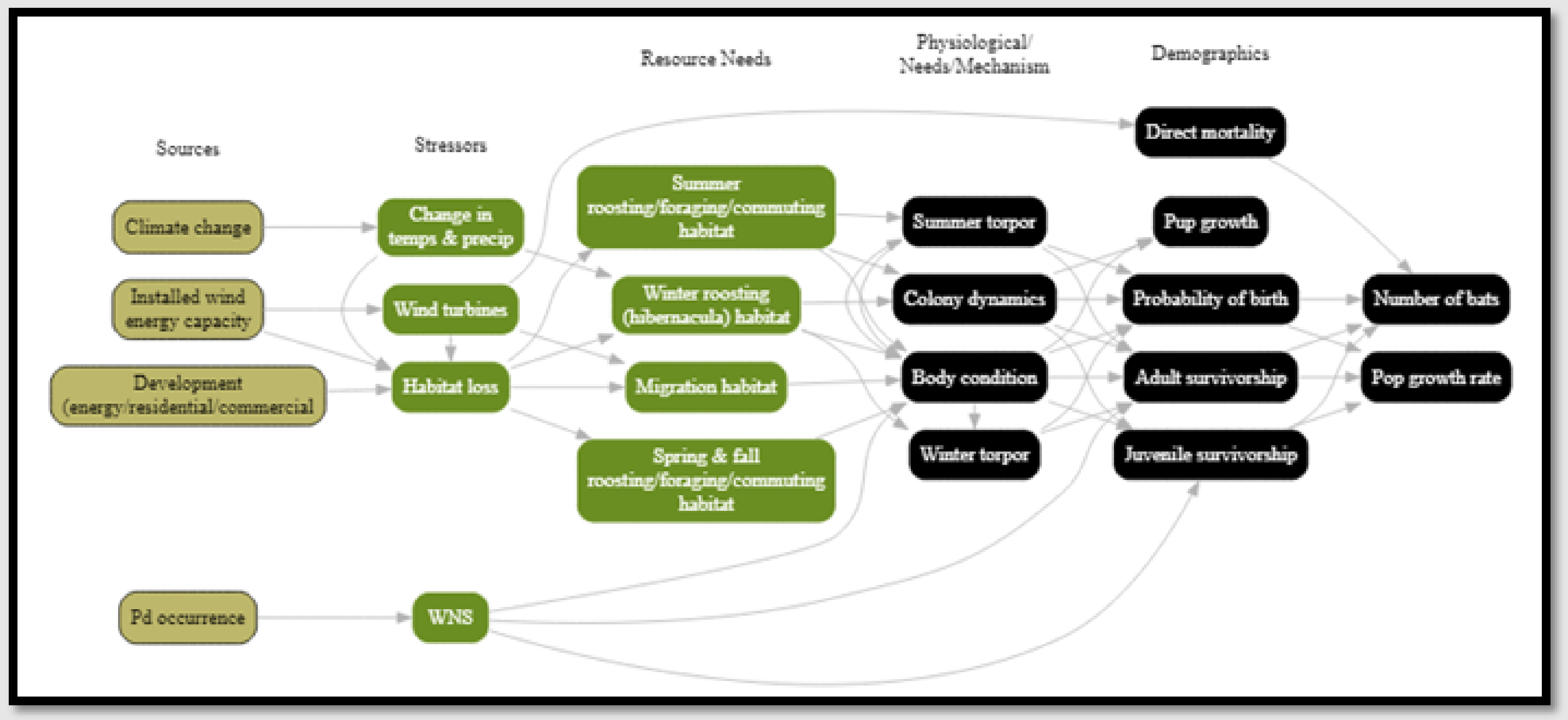


Figure 4.1. Visual diagram showing relationships between the primary threats and population needs.

Current Threat Conditions

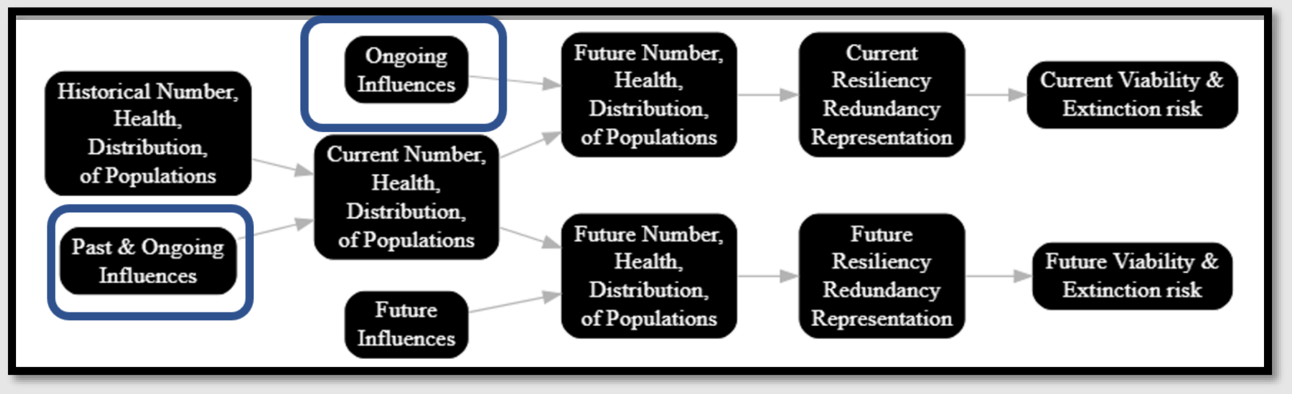
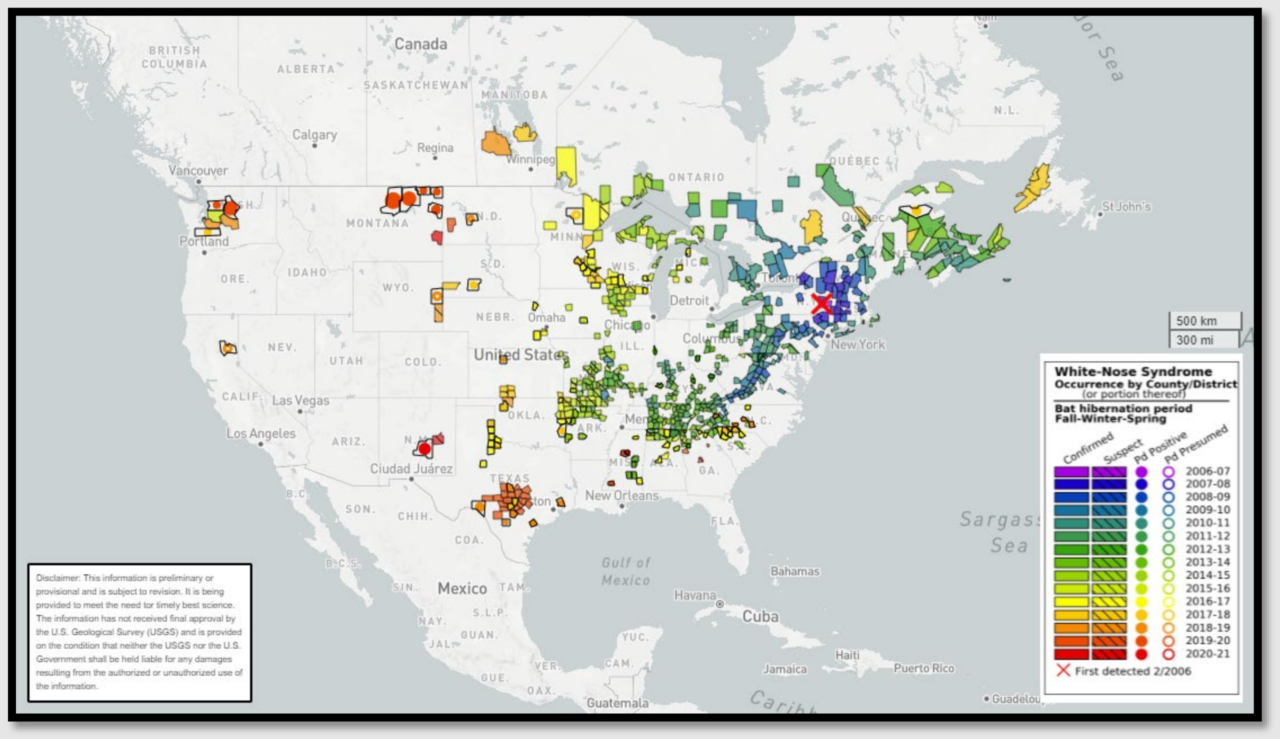


Figure 4.2. Highlighting (blue rectangle) the current step in our analytical framework.

White-nose Syndrome

For over a decade, WNS has been the foremost stressor on NLEB. WNS is a disease of bats that is caused by the fungal pathogen *Pd* (Blehert et al. 2009, entire; Turner and Reeder 2009, entire; Lorch et al. 2011, entire; Coleman and Reichard 2014, entire; Frick et al. 2016, entire; Puechmaille and Willis et al. 2017, entire; Bernard et al. 2020, entire; Hoyt et al. 2021, entire). The disease and pathogen were first discovered in eastern New York in 2007 (with photographs showing presence since 2006) (Meteyer et al. 2009, p. 411), and since then has spread to 39 states and 7 provinces in North America (Figure 4.3). *Pd* invades the skin of bats, initiating a cascade of physiological and behavioral processes that often lead to mortality (Warnecke et al. 2013, p. 3; Verant et al. 2014, pp. 3–6). Infection leads to increases in the frequency and duration of arousals during hibernation and raises energetic costs during torpor bouts, both of which cause premature depletion of critical fat reserves needed to survive winter (Reeder et al. 2012, p. 5; McGuire et al. 2017, p. 682; Cheng et al. 2019, p. 2). Bats that do not succumb to starvation in hibernacula often seek riskier roosting locations near entrances to roosts or emerge from roosts altogether, where they face exposure to winter conditions and scarce prey resources on the landscape (Langwig et al. 2012, p. 2). The weeks following emergence from hibernation also mark a critical period because prey availability is still limited, energetic costs of healing from WNS are high, and the potential for immune reconstitution inflammatory syndrome that can lead directly to mortality or impact reproductive success (Reichard and Kunz 2009, p. 461; Meteyer et al. 2012, p. 3; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8). As of May 2021, WNS has been confirmed in 12 species in North America, including NLEB, and numerous other species in Europe and Asia (www.whitenosesyndrome.org, accessed May 13, 2021; Hoyt et al. 2021, Suppl. material).



Disclaimer: This information is preliminary or provisional and is subject to revision. It is being provided to meet the need for timely best science. The information has not received final approval by the U.S. Geological Survey (USGS) and is provided on the condition that neither the USGS nor the U.S. Government shall be held liable for any damages resulting from the authorized or unauthorized use of the information.

Figure 4.3. Occurrence of Pd and WNS in North America based on surveillance efforts in the U.S. and Canada: disease confirmed (color-coded), suspected (stripes), Pd detected but not confirmed (solid circles), and Pd detected but inconclusive lab results (open circles). Pd and WNS occurrence records generally reflect locations of winter roosts and are not representative of the summer distribution of affected bats (www.whitenosesyndrome.org, accessed May 13, 2021).

The fungal pathogen is spread primarily via bat-bat and bat-environment-bat movement and interactions (Lindner et al. 2011, p. 246; Langwig et al. 2012, p. 1055). With the arrival of *Pd* (year 0) to a new location, WNS progresses through "stages" similarly to many emerging infectious diseases: pre-invasion, invasion, epidemic, and establishment (Langwig et al. 2015, p. 196; Cheng et al. 2021, entire). During *invasion* (years 0–1), the fungus arrives on a few bats and spreads through the colony as a result of swarming and roosting interactions until most individuals are exposed to the pathogen. Such interactions may occur in hibernacula or at nearby roosts where conspecifics engage in mating activity (Neubaum and Siemers, 2021, p. 2). As the amount of *Pd* on bats and in the environmental reservoir increases, the *epidemic* (years 2–4) proceeds with high occurrence of disease and mortality. By the fifth year after arrival of *Pd*, the pathogen is *established* (years 5–7), and 8 years after its arrival, *Pd* is determined to be *endemic* in a population (Langwig et al. 2015, p. 196; Cheng et al. 2021, entire).

The effect of WNS on NLEB has been extreme, such that most summer and winter colonies experienced severe declines following the arrival of WNS. Just 4 years after the discovery of WNS, for example, Turner et al. (2011, pp. 18–19) estimated that NLEB experienced a 98% decline in winter counts across 42 sites in Vermont, New York and Pennsylvania. Similarly, Frick et al. (2015, p. 5) estimated the arrival of WNS led to a 10–fold decrease in NLEB colony size. Most recently, Cheng et al. (2021, entire) used data from 27 states and 2 provinces to conclude WNS caused estimated population declines of 97–100% across 79% of NLEB's range. Although variation exists among sites, the arrival of *Pd* caused marked decrease in population abundance during invasion, epidemic, and established stages of the disease (Figure 4.4), with few exceptions (Figure 4.5). These analyses were extended to include additional data and years by Cheng et al. (2022, p. 212; Figure 4.4, Figure 4.5).

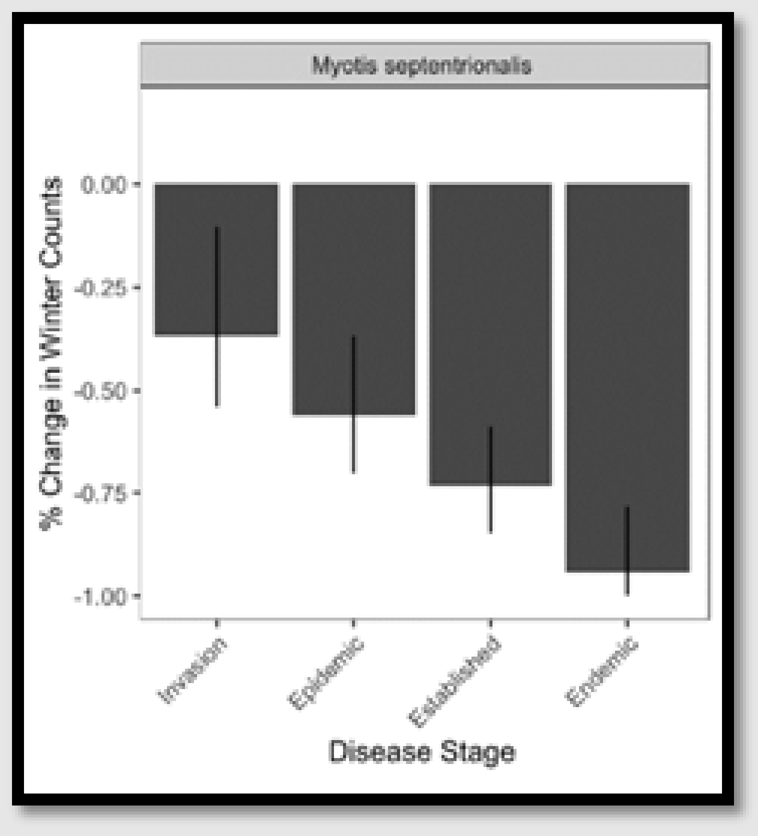


Figure 4.4. Percent change in winter colony counts by disease stage relative to predicted median count prior to arrival of Pd (with 95% credible interval) (Cheng et al. 2022, p. 212).

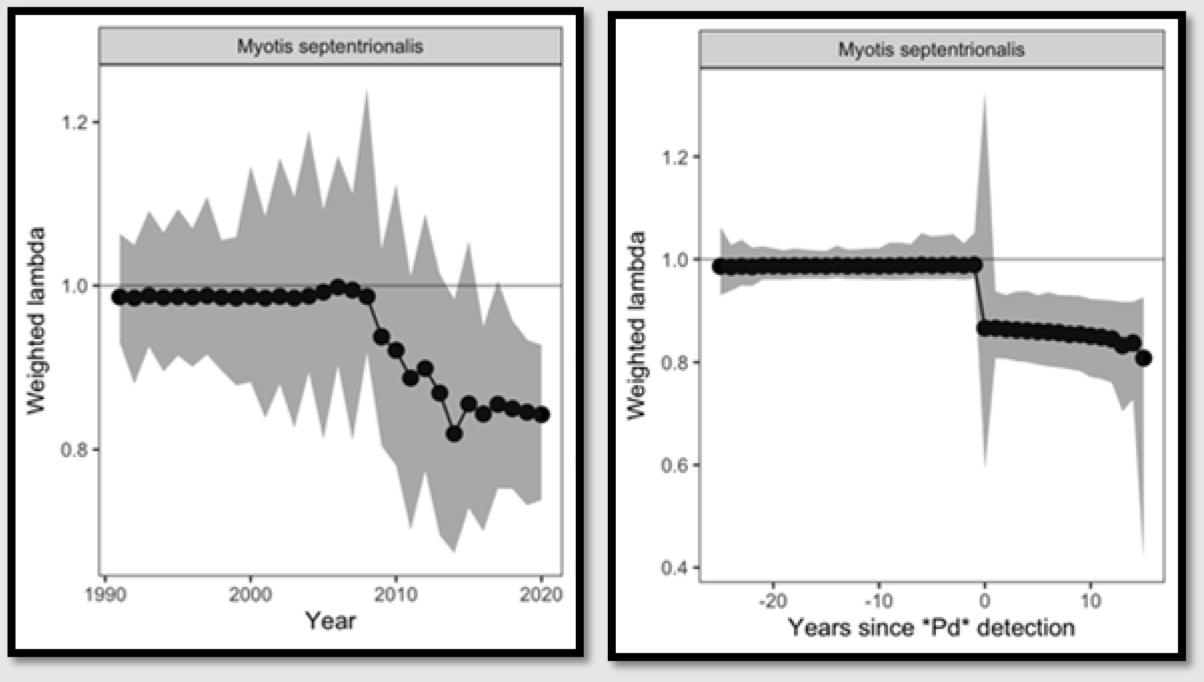


Figure 4.5. Estimated weighted lambda (function of growth rate and colony size) by year (left) and by year since arrival of Pd (right) (Cheng et al. 2022, p. 211).

Building off work of Cheng et al. (2022, entire), Wiens et al. (2022, entire) used available data from hibernacula surveys to estimate the annual impacts of WNS relative to the year of arrival of *Pd*, adding additional analysis of an endemic stage. Their analysis applied two models of *Pd* spread to interpolate WNS occurrence to all documented hibernacula. The analysis predicted *Pd* is present at 99–100% of documented hibernacula for NLEB (Appendix 2-A). Although variation exists among sites, an overwhelming majority of hibernating colonies of NLEB have developed WNS and experienced serious impacts within 2–3 years after the arrival of *Pd* (Cheng et al. 2021, entire; Wiens et al. 2022, pp. 231–247) (Figure 4.5).

A variety of factors may contribute to the differences observed amongst hibernacula. Year-round temperature profiles may affect the environmental reservoir of *Pd*, thus reducing the source of reinfection when bats return to the locations each fall, which would be more likely to delay than preclude infection (Hoyt et al. 2020, pp. 7257–7258). However, it is important to acknowledge that bats likely encounter multiple subterranean environments during swarming activity, during which they can encounter reservoirs of *Pd* (Neubaum and Siemers, 2021, pp. 3–4). Over winter temperature and climate may also affect the physiology of hibernating bats in these sites or offer foraging opportunities that make it possible for them to avoid more serious infections, but these mechanisms have not been tested. Regardless, the vast majority of NLEB colonies exposed to *Pd* have developed and will continue to develop WNS and experience impacts from the disease (Cheng et al. 2021, entire; Wiens et al. 2022, pp. 231–247) (Figure 4.6).

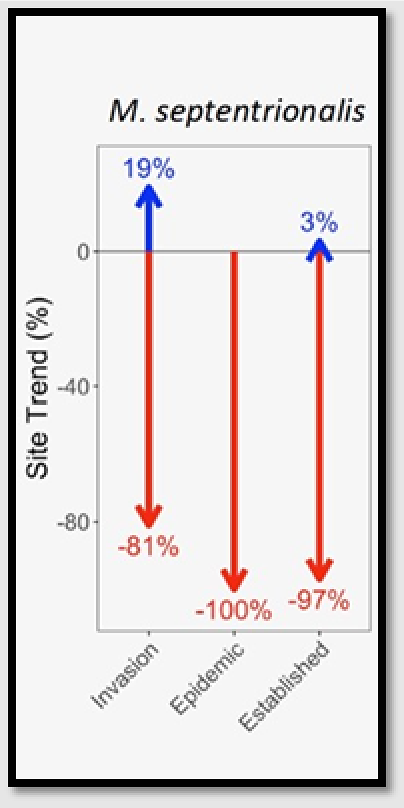


Figure 4.6. Percentage of accessible winter colonies with increasing (blue) and decreasing (red), colony trend relative to WNS pre-arrival stage for invasion, epidemic, and established stages (Cheng et al. 2021, entire; appendix S3).

There are multiple national and international efforts underway in attempt to reduce the impacts of WNS. To date, there are no proven measures to reduce the severity of impacts. See Appendix 4-A for more information regarding WNS impacts.

Wind Related Mortality

Wind related mortality, overshadowed by the disproportionate impacts to tree bats and by the enormity of WNS, is also proving to be a consequential stressor at local and RPU levels. Wind power is a rapidly growing portion of North America's energy portfolio in part due to changes in State energy goals (NCSL 2021, web) and recent technological advancements (Berkeley Lab 2020, web) and declining costs (Wiser et al. 2021, entire), allowing turbines to be placed in less windy areas. As of 2019, wind power was the largest source of renewable energy in the country, providing 7.2% of U.S. energy (American Wind Energy Association (AWEA) 2020, p. 1). Modern utility-scale wind power installations (wind facilities) often have tens or hundreds of turbines installed in a given area, generating hundreds of MW of energy each year. Installed wind capacity in the U.S. as of October 2020 was 104,628 MW (Hoen et al. 2018, entire; USFWS unpublished data).

The remarkable potential for bat mortality at wind facilities became known around 2003, when post-construction studies at the Buffalo Mountain, Tennessee, and Mountaineer, West Virginia, wind projects documented the highest bat mortalities reported at the time[4] (31.4 bats/MW and 31.7 bats/MW, respectively; Kerns and Kerlinger 2004, p. 15; Nicholson et al. 2005, p. 27). Bat mortalities continue to be documented at wind power installations across North America and Europe. We describe mechanisms leading to bat fatalities in Appendix 4-B.

Bat fatality varies across facilities, between seasons, and among species. Consistently, three species–hoary bats (*Lasiurus cinereus*), silver-haired bats (*Lasionycteris noctivagans*), and eastern red bats (*Lasiurus borealis)*–comprise the majority of all known bat fatalities (e.g., 74–90%). The disproportionate amount of fatalities involving these species has resulted in less attention and concern for other non-listed bat species. However, there is notable spatial overlap between NLEB occurrences and wind facilities (Figure 4.7) along with NLEB mortality documented. At the 2020 installed MW capacity, we estimated 122 NLEB are killed annually at wind facilities (Table 4.1). Analyses using data from Wiens et al. (2022, pp. 236–247) and analyses by Whitby et al. (2022, entire) suggest that the impact of wind related mortality is discernible in the ongoing decline of NLEB. Based on data from Wiens et al. (2022, pp. 236–247) comparing a no wind baseline scenario to current and future wind scenarios, the projected abundance decreases 24–33% by 2030 under the current wind scenario and up to 83% by 2060 under the future high impact wind scenario (Tables A-3D1 and A-3D2). Whitby et al. (2022, entire) found a decline in the predicted relative abundance of NLEB as wind energy risk index increased. To reduce bat fatalities, some facilities "feather" turbine blades (i.e., pitch turbine blades parallel with the prevailing wind direction to slow rotation speeds) at low wind speeds when bats are more at risk (Hein and Straw, p. 28). The wind speed at which the turbine blades begin to generate electricity is known as the "cut-in speed," and this can be set at the manufacturer's speed, or at a higher threshold, typically referred to as curtailment. The effectiveness of feathering below various cut-in speeds differs among sites and years (Arnett et al. 2013, entire; Berthinussen et al. 2021, pp. 94–106); nonetheless, most studies have shown all-bat fatality reductions of >50% associated with feathering below wind speeds of 4.0–6.5 meters per second (m/s) (Arnett et al. 2013, entire; USFWS unpublished data). The effectiveness of curtailment at reducing species-specific fatality rates for NLEB, however, has not been documented. Hereafter, we refer to feathering below the manufacturer's cut-in speed or higher wind speeds collectively as curtailment.

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[4] Higher wind fatality rates have since been reported (e.g., Schirmacher et al. 2018, p. 52; USFWS 2019, p. 32 and 69).

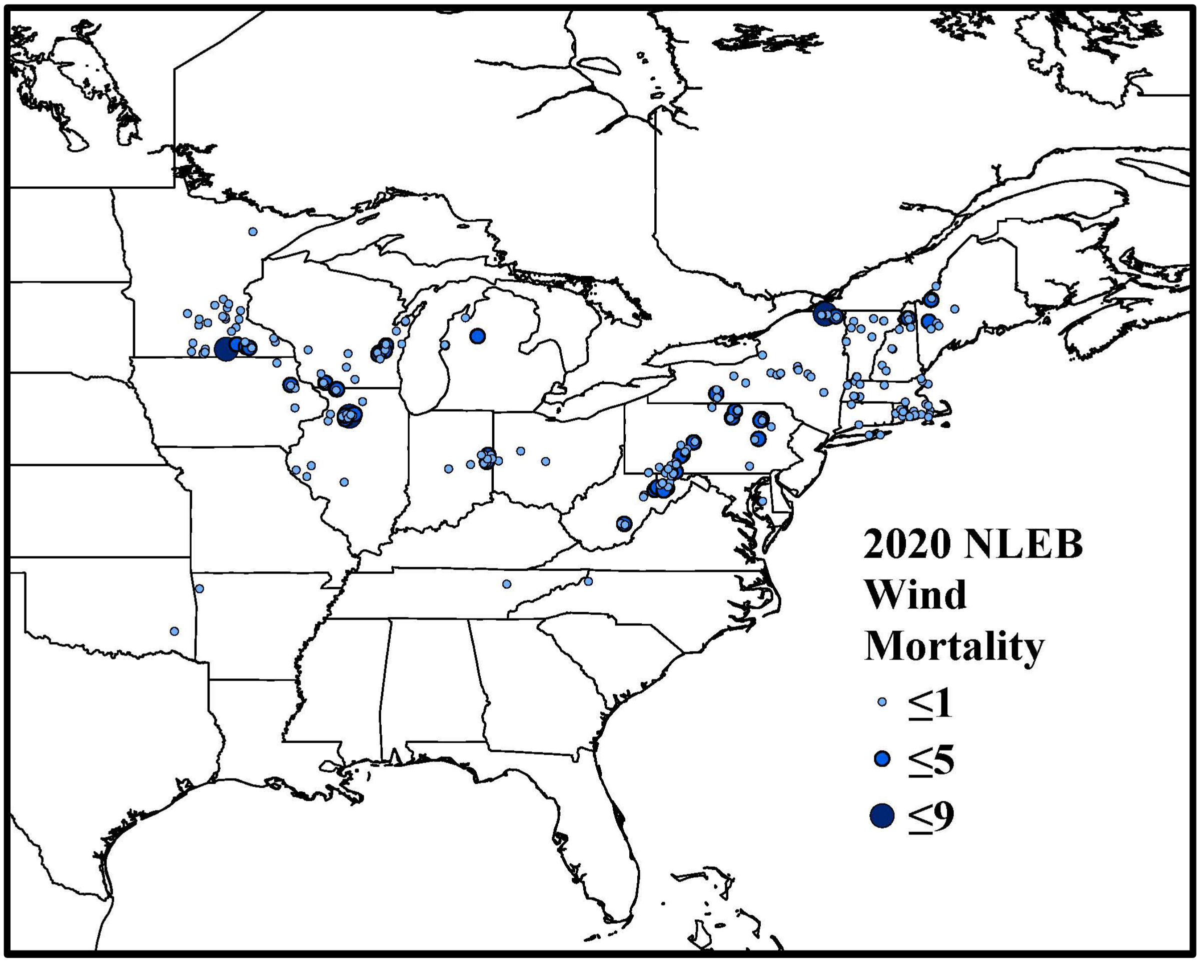

Figure 4.7. Estimated total annual NLEB mortality at wind facilities in 2020. Mortality is shown at U.S. wind turbines as summed by 11x11-km NREL grid cell within the migratory range of extant NLEB hibernacula. Note that because MW were summed by Province centroid in Canada (and none were within the migratory range of hibernacula), the only NLEB mortality that was allocated to Canadian hibernacula (Quebec) was that occurring at U.S. turbines within the migratory range. See Udell et al. 2022, pp. 265–266 and Appendix 2 for details on the wind mortality analysis.

Table 4.1. Estimated annual NLEB mortality from wind facilities allocated to hibernacula by USFWS Region (Figure A-2A6) and Canada, based on installed MW capacity in October 2020 (Udell et al. 2022, pp. 265–266).

Location	**Mean Annual Mortality (n)**	**Lower CI**	**Upper CI**
Region 2	0	0	0
Region 3	59	19	72
Region 4	1	0	1
Region 5	58	17	72
Quebec	4	1	5
Total	**122**	**38**	**150**

There are many ongoing efforts to improve our understanding of bat interactions with wind turbines and explore additional strategies for reducing bat mortality at wind facilities. To date, operational strategies (e.g., feathering turbine blades when bats are most likely to be active) are the only broadly proven and accepted measures to reduce the severity of impacts. See Appendix 4-B for more information.

Climate Change

There is growing concern about impacts to bat populations in response to climate change (Jones et al. 2009, entire; Jones and Rebelo 2013, entire, O'Shea et al. 2016, p. 9). Jones et al. (2009, p. 94) identified several climate change factors that may impact bats, including changes in hibernation, mortality from extreme drought, cold, or excessive rainfall, cyclones, loss of roosts from sea level rise, and impacts from human responses to climate change (e.g., wind turbines). Sherwin et al. (2013, entire) reviewed and discussed potential impacts of climate change, including effects to bat foraging, roosting, reproduction, and biogeography. Climate change is also likely to influence disease dynamics as temperature, humidity, phenology and other factors affect the interactions between *Pd* and hibernating bats (Hayman et al. 2016, p. 5; McClure et al. 2020, p. 2; Hoyt et al. 2021, p. 8). However, the impact of climate change is unknown for most species (Hammerson et al. 2017, p. 150). Climate change may impact these bats in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of various insect hatches not aligning with key life history periods of spring emergence, pregnancy, lactation, or fall swarming). In addition, there may be shifts in distribution of forest communities, invasive plants, invasive forest pest species, or insect prey. Long-term increases in global temperatures are correlated with shifts in butterfly ranges (Parmesan et al. 1999, entire; Wilson et al. 2007, p. 1880; Breed et al. 2013, p. 142) and similar responses are anticipated in moths and other insect prey. Milder winters may result in range expansions of insects or pathogens with a distribution currently limited by cold temperatures (e.g., hemlock woolly adelgid (*Adelges tsugae*), southern pine beetle (*Dendroctonus frontalis*)) (Haavik 2019).

While there are a number of changing climatic variables, our analysis focused solely on changes in temperature and precipitation. These variables influence NLEB resource needs, such as suitable roosting habitat (all seasons), foraging habitat, and prey availability (Figure 4.1). Global average temperature has increased by 1.7 degrees F (0.9 degrees C) between 1901 and 2016 (Hayhoe et al. 2018, p. 76). Over the contiguous U.S., annual average temperature has increased by 1.2 degrees F (0.7 degrees C) for the period of 1986 to 2016 relative to 1901 to 1960 (Hayhoe et al. 2018, p. 86). Temperatures increased during that time at a regional scale as well, with the largest changes (average increases of more than 1.5 degrees F (0.8 degrees C)) in Alaska, the Northwest, the Southwest, and the Northern Great Plains and the least change in the Southeast (increase of 0.46 degrees F (0.26 degrees C); Vose et al. 2017, pp. 186–187; Hayhoe et al. 2018, p. 86). Annual average precipitation has increased by 4% since 1901 across the entire U.S. with increases over the Northeast, Midwest and Great Plains and decreases over parts of the West, Southwest and Southeast (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events across the U.S. have increased more than the increases in average precipitation (Hayhoe et al. 2018, p. 88).

NLEB risk of exposure to changes in the climate is rangewide. However, the magnitude, direction, and seasonality of climate variable changes is not consistent rangewide. In addition, the resiliency of populations and inherent differences (e.g., genetics, summer roost microclimates) among populations may result in differing ability for NLEB to respond to the same types of changes across the range. Therefore, the overall impact of climate change for such a wide-ranging species is challenging to describe. Although there may be some benefit to NLEB from a changing climate, overall negative impacts are anticipated. Although we lack species-specific observations for NLEB, observed impacts to date for other insectivorous bats, such as the little brown bat, include reduced reproduction due to drought conditions leading to decreased availability of drinking water (Adams 2010, pp. 2440–2442) and reduced adult survival during dry years (drought) in the Northeast (Frick et al. 2010, pp. 131–133). While sufficient moisture is important, too much precipitation during the spring can also result in negative consequences to insectivorous bats. During the anticipated heavier precipitation events there may be decreased insect availability and reduced echolocation ability (Geipel et al. 2019, p. 4) resulting in decreased foraging success. Precipitation also wets bat fur, reducing its insulating value (Webb and King 1984, p. 190; Burles et al. 2009, p. 132) and increasing a bat's metabolic rate (Voigt et al. 2011, pp. 794–795). Bats are likely to reduce their foraging bouts during heavy rain events and reduced reproduction has been observed during cooler, wetter springs in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Responses will vary throughout NLEB range based on the extent of annual temperature rise in the future. For additional information on climate change see Appendix 4-C.

Habitat Loss

Roosting/Foraging/Commuting Habitat Loss

As discussed in Chapter 2, NLEB require suitable habitat for roosting and foraging, and commuting between those habitats during spring, summer, and fall. Forest is a primary component of roosting, foraging, and commuting habitat. Wetlands and water features are important foraging and drinking water sources. Loss of these habitats influences survival and reproduction of NLEB colonies.

We reviewed changes in various NLCD landcover classes within each RPU from 2006 to 2016 in the continental U.S. Overall, forest landcover was fairly stable in all RPUs with slight annual increases (27,000 to 50,000 acres/year) in all but Midwest RPU (loss of 23,000 acres/year). However, deciduous forest landcover decreased across all RPUs by 1.4 million acres for an average loss of 140,000 acres per year. Other cover types that provide foraging opportunities such as emergent wetland cover types decreased across all RPUs by 1.4 million acres. See Appendix 4-D for additional information.

These changes in landcover may be associated with losses of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality. While temporary or permanent habitat loss may occur throughout all states within the species' range, impacts to NLEB typically occur at a more local-scale (i.e., individuals and potentially colonies).

Impacts to the NLEB from loss of habitat vary depending on the timing, location, and extent of the removal.

Impacts from forest habitat removal may range from minor (e.g., removal of a small portion of foraging habitat in unfragmented forested area with a robust NLEB population) to significant (e.g., removal of roosting habitat in highly fragmented landscape with small, disconnected population). Adverse impacts are more likely in areas with little forest or highly fragmented forests (e.g., western U.S. and central Midwestern states), as there is a higher probability of removing roosts or causing loss of connectivity between roosting and foraging habitat. There are a variety of conservation measures that can either serve to reduce effects from habitat loss or help maintain or enhance habitat. See Appendix 4-D for examples.

Winter Roost Loss and Disturbance

As discussed in Chapter 2, NLEB require hibernation sites with specific microclimates and NLEB exhibit high interannual fidelity to their hibernacula. Therefore, the complete loss of or modification of winter roosts (such that the site is no longer suitable) can result in impacts to individuals or at the population level. In addition, disturbance within hibernacula can render a site unsuitable or can pose harm to individuals using the site.

Modifications to bat hibernacula (e.g., erecting physical barriers to control cave and mine access, intentional or accidental filling or sealing of entries, or creation of new openings) can alter the ability of bats to access the site (Spanjer and Fenton 2005, p. 1110) or affect the airflow and alter microclimate of the subterranean habitat, and thus the ability of the cave or mine to support hibernating bats, such as NLEB. These well-documented effects on cave-hibernating bat species were discussed in the USFWS's *Indiana Bat Draft Recovery Plan* (USFWS 2007, pp. 71–74). In addition to altering the thermal or humidity regime and ability of the site to support hibernating bats, bats present during any excavation or filling can be crushed or suffocated. Sources of these stressors include fill from adjacent activities, mining, and intentional closures of abandoned mines or cave openings to restrict access.

Human entry or other disturbance to hibernating bats results in additional arousals from hibernation which require an increase in total energy expenditure at a time when food and water resources are scarce or unavailable. This is even more important for sites where a species is impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Willis and Boyles 2012, p. 96).

There are many conservation efforts and protections (e.g., bat-friendly gates, closure of caves during hibernation) in place that attempt to reduce the risk of modifications to hibernacula and disturbance to overwintering bats. See Appendix 4-D for more information.

Conservation Efforts

Conservation efforts associated with reducing the effects of WNS, wind related mortality, and habitat loss are mentioned above and discussed further within associated appendices. In addition to those efforts, below we highlight the regulatory protections afforded to NLEB in parts of its range.

Federal, State, Provincial Protection

NLEB was listed as threatened under the Endangered Species Act on April 2, 2015 (USFWS 2015, entire). We also developed a final 4(d) rule, which published in the *Federal Register* on January 14, 2016 (USFWS 2016, entire). The 4(d) rule specifically defines the "take" prohibitions. NLEB was listed as endangered on Schedule 1 of Canada's Species at Risk Act in 2014. This provided the NLEB protection from being killed, harmed, harassed, captured, or taken in Canada. Environment and Climate Change Canada finalized a recovery strategy for NLEB in 2018 (Environment and Climate Change Canada 2018, entire).

In addition, NLEB receives varying degrees of protection through state laws as it is designated as Endangered in Arkansas, Connecticut, Delaware, Indiana, Maine, Massachusetts, Missouri, New Hampshire, Vermont; Threatened in Georgia, Illinois, Louisiana, Maryland, New York, Ohio, Pennsylvania, Tennessee, Virginia, and Wisconsin; and Special Concern in Alabama, Iowa, Michigan, Minnesota, Mississippi, Oklahoma, South Carolina, South Dakota, West Virginia, and Wyoming.

Synopsis of Current Threat Conditions

To provide a comparative assessment of the primary influences, we summarize the scope, severity, and impact of each of the four influences using criteria defined by Master et al. (2012, pp. 28–35; Table 4.2). Currently, WNS is the greatest threat to NLEB, with WNS related population declines occurring over 78% (pervasive in scope) of NLEB's range of an estimated 97–100% (extreme severity; Cheng et al. 2021, entire). Wind mortality, although large in scope (occurring over 49% of range) has a "medium" level impact to NLEB due to a moderate to serious severity based on differences in the two models (current population-level decline of 24–33% (Table A-3D1)). A "medium" impact level for wind mortality was decided on in part due to mortality rates being kept constant for projections in the model and as declines increase, presumably so will exposure to wind mortality, which reduces overall impact. While confidence in impact to NLEB from WNS and wind were "moderate to high" due to availability of quantitative data, our confidence analysis of the impact of habitat loss and climate change remain "moderate to low" due to minimal quantitative data. Both habitat loss and climate change are pervasive, occurring across the species' range, while severity of population level declines are predicted to be slight. Conservation efforts, such as protection of winter hibernacula from disturbance and habitat protections for NLEB and other listed species, may provide some benefit to NLEB populations. Lastly, habitat loss (e.g., disturbance to or loss of maternity colony, tree removal) and climate change (e.g., precipitation levels, rising temperatures) are anticipated to vary regionally, but have more localized impacts on the species viability.

Table 4.2 Assessment of current impact to NLEB from primary threats (adapted from Master et al. 2012). See Chapter 1 for definitions of the criteria (Figure 1.2).

Criteria	**WNS**	**Wind Mortality**	**Habitat Loss**	**Climate Change**
Scope	Pervasive	Large	Pervasive	Pervasive
Severity	Extreme	Moderate	Slight	Slight
Impact	**Very High**	**Medium**	**Low**	**Low**
Confidence level	High	Moderate	Moderate	Low

Future Threat Conditions

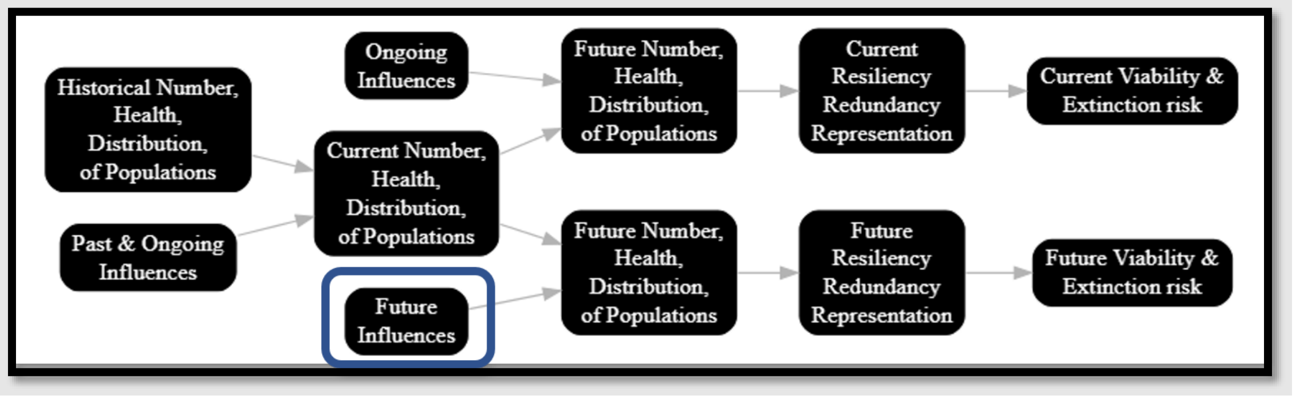

Figure 4.8. Highlighting (blue rectangle) the current step in our analytical framework.

To assess how NLEB will respond to foreseeable changes in *Pd* and wind energy capacity, we identified the plausible future state of these influences (Figure 4.8). We developed realistic lower and upper bounds for both and combined them to create composite plausible "high impact" and "low impact" scenarios. The composite future scenarios for WNS and wind mortality is summarized in Table 4.3. These scenarios and their underlying rationales are described below, along with the future projected conditions for habitat loss and climate change. We provide further rationale for our low and high impact scenarios in Appendix 5.

Table 4.3. NLEB composite plausible future scenarios.

Plausible Scenario	WNS Spread	WNS Duration	Wind Capacity	All-bat Fatality Rate	% Species Composition	*Pd* rate
Low impact	*Pd* occurrence model 1	15-yr species-specific survival rates	Lower build-out	Regional-specific	U.S. - combined, Canada - regional-specific	No
High impact	*Pd* occurrence model 2	40-yr species-specific survival rates	Higher build-out	Regional-specific	U.S. - combined, Canada - regional-specific	No

White-nose Syndrome

To project future impacts of WNS, we relied on 1) predicted current and future occurrence of *Pd* on the landscape using two different models (hereafter, "*Pd* occurrence models") and 2) the WNS impacts schedule. For the latter, we assumed winter colonies that are exposed to *Pd* in the future will respond similarly to those currently exposed (i.e., colonies exposed in the future will follow the same WNS impacts schedule) (see Chapter 1, *Step 3. Identify the Primary Drivers (Influences)* and Appendix 5 for more detail).

To project future spread of WNS, we relied upon two *Pd* occurrence models, *Pd* occurrence model 1 (derived by Wiens et al. 2022, pp. 226–229) and *Pd* occurrence model 2 (derived by Hefley et al. 2020, entire); both models are briefly described in Appendix 2. For a low impact scenario, we used *Pd* occurrence model 1 for predicted year of arrival (YOA) and assumed that the WNS impacts schedule continues for 15 years after arrival of *Pd*, after which the colonies return to pre-WNS survival rates for the remainder of the simulation (i.e., no WNS impacts applied after 15 years since *Pd* arrival). Return to pre-WNS growth rates at YOA 15 is the earliest year we can reasonably assume (given data show impacts continue occurring 14-years since the first detection in New York). For the high impact scenario, we used *Pd* occurrence model 2 for predicted YOA and assumed that WNS impacts continue through 2060 (i.e., after YOA 0 to 6, survival rates remain in the endemic phase).

Wind Related Mortality

To project future installed wind capacity, we relied upon National Renewable Energy Laboratory's (NREL; Cole et al. 2020) and Canadian Energy Regulator's (CER) (CER 2020) projections for the U.S. and Canada, respectively (Figure 4.9). Our low impact scenario (i.e., lower wind build-out) was based on NREL's *High Wind Cost* scenario and CER's *Reference Scenario* (Figure 4.10). Our high impact scenario (i.e., higher wind build-out) was based on NREL's *Low Wind Cost* scenario and CER's *Evolving Scenario* (Figure 4.11). For both scenarios, we calculated NLEB fatalities per MW using the species composition approach (see

Chapter 1 methods and Appendix 2-A for additional detail). The annual mortality associated with the future low and high impact scenarios by Year 2050 is provided in Table 4.4.

We selected NREL's scenarios per consultation with the U.S. Department of Energy's (USDOE) Wind Energy Technology Office (P. Gilman 2020, Program Manager, personal communication). The NREL scenarios model future deployment levels based on projected trends in electricity demand, technology cost trajectories, and existing Federal and state energy policies (Cole et al. 2020, p. iii; see Appendix 5 for details). NREL's 2020 (Cole et al. 2020) report presents 45 power sector scenarios that consider present day through 2050. We chose the *High Wind Cost* and *Low Wind Cost* scenarios as reasonable lower and upper bounds of future wind build-out, respectively. NREL agreed that use of the *High Wind Cost* and *Low Wind Cost* scenarios provides a reasonable range of future wind build-out (W. Cole 2020, personal communication).

CER's *Canada's Energy Future* report is published annually and provides up-to-date projections for wind build-out in Canada. CER uses economic and energy models to project future scenarios based on assumptions about *trends in* "*technology, energy and climate policies, energy markets, human behavior and the structure of the economy*" (CER 2019, p. 1). Annual wind build-out projections are produced at the province/territory level and data are continually refined based on current trends. We chose the *Reference Scenario* as our lower-impact scenario (i.e., lower wind build-out) and the *Evolving Scenario* as our higher-impact scenario (i.e., higher wind build-out; see Appendix 5 for details).

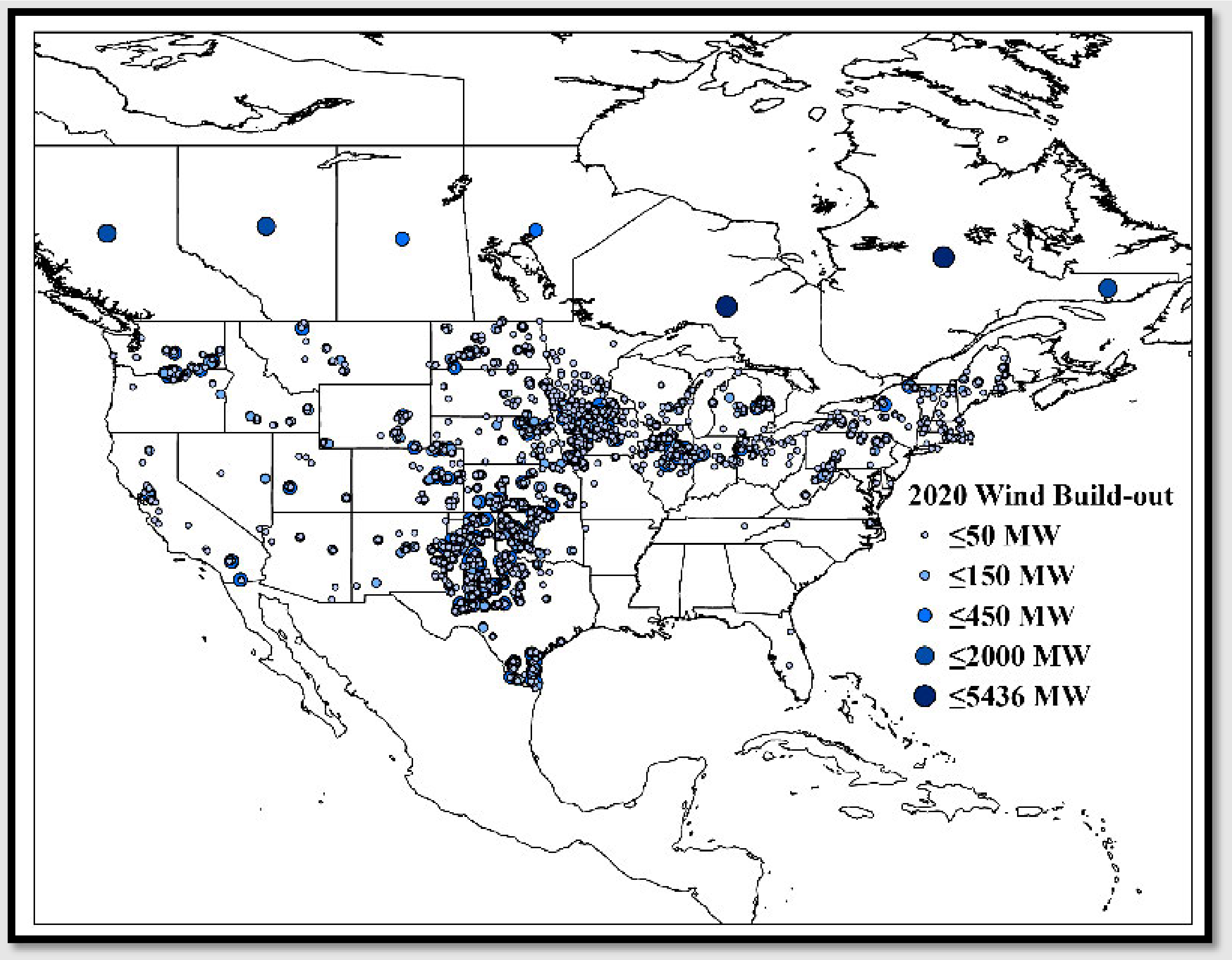

Figure 4.9. Wind build-out as of October 2020 for the U.S. and Canada (Udell et al. 2022, entire). U.S. capacity is summed by 11x11-km NREL grid cell and Canadian capacity by province.

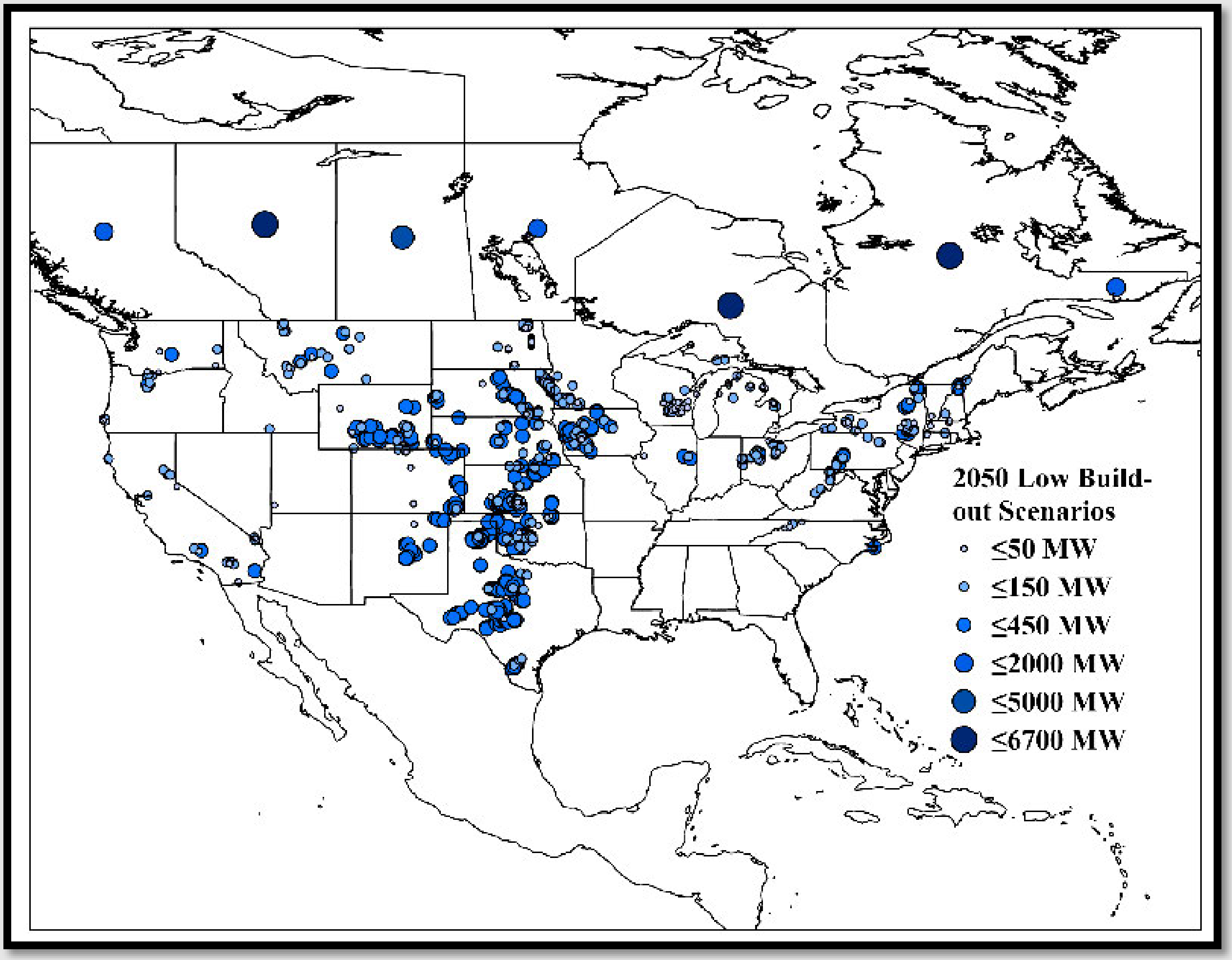

Figure 4.10. Projected wind build-out for the year 2050 per low build-out scenarios for the U.S. and Canada (NREL 2020; CER 2020; Udell et al. 2022, entire). U.S. future capacity is summed by 11x11-km NREL grid cell and Canadian future capacity by province.

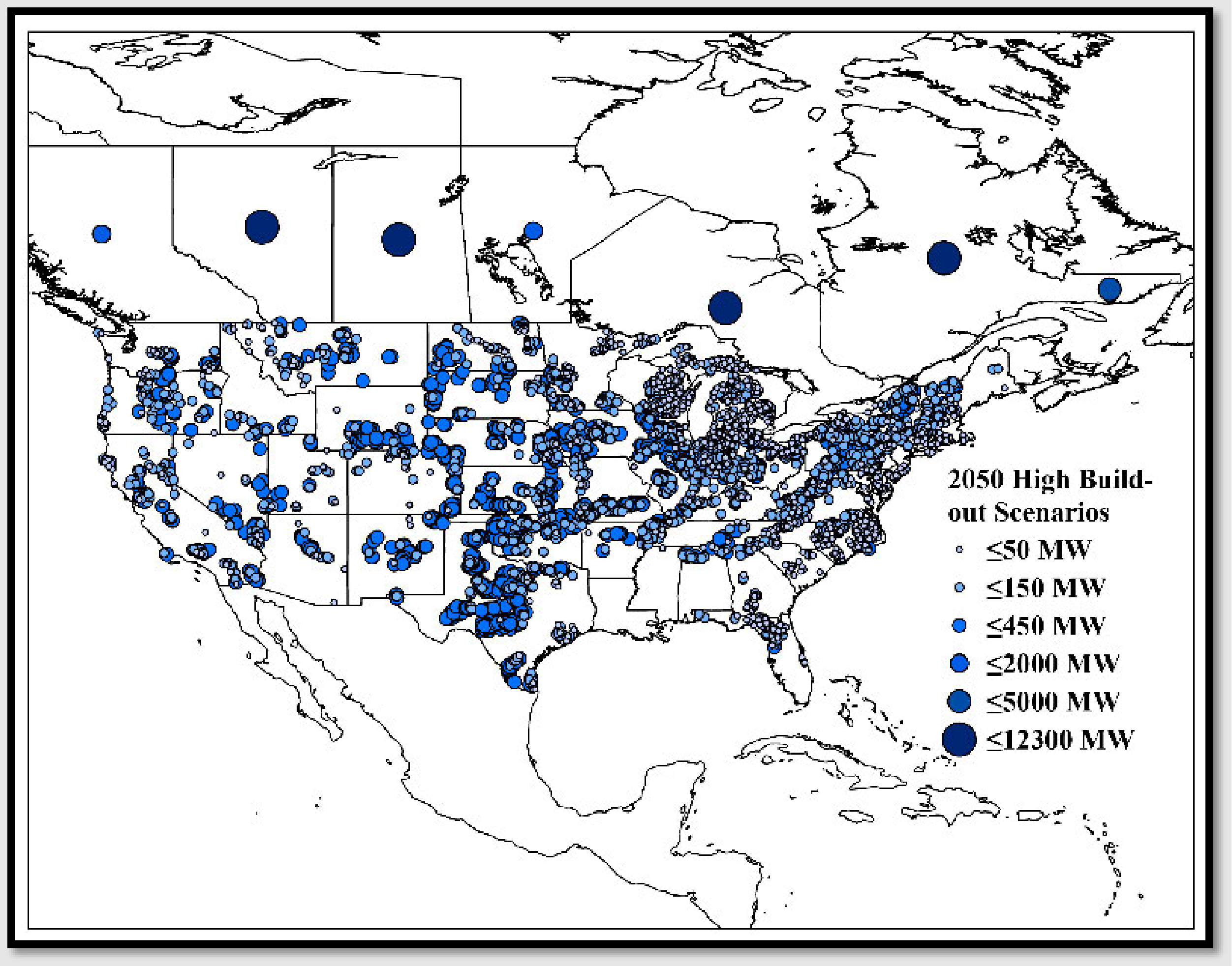

Figure 4.11. Projected wind build-out for the year 2050 per high build-out scenarios for the U.S. and Canada (NREL 2020; CER 2020; Udell et al. 2022, entire). U.S. future capacity is summed by 11x11-km power grid and Canadian future capacity by province.

Table 4.4. Predicted annual NLEB mortality[5] (25th–75th percentile) by USFWS Region and Canada, based on projected 2050 installed wind capacity under low and high build-out scenarios (Udell et al. 2022, entire).

Location	**Low build-out**	**High build-out**
Region 2	0 (0–0)	33 (11–33)
Region 3	57 (18–70)	1,395 (447–1,703)
Region 4	3 (1–4)	307 (93–380)
Region 5	138 (42–172)	1,157 (349–1,440)
Quebec	4 (1–5)	35 (11–43)
Total	**202 (62–250)**	**2,926 (911–3,600)**

Climate Change

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[5] It is likely that percent composition will decline as the species declines over time. To capture insights on the sensitivity of the results to wind energy mortality, we ran scenarios with zero and 50% reduction in wind energy mortality (see Appendix 1-B).

Over the next few decades, annual average temperature over the contiguous U.S. is projected to increase by about 2.2 degrees F (1.2 degrees C) relative to 1985 to 2015**,** regardless of any currently used representative concentration pathway (RCP 2.6 to RCP 8.5) (Hayhoe et al. 2018, p. 86). Larger increases are projected by late century of 2.3 to 6.7 degrees F (1.3 to 3.7 degrees C) under RCP4.5 and 5.4 to 11.0 degrees F (3.0 to 6.1 degrees C) under RCP8.5, relative to 1986 to 2015 (Hayhoe et al. 2018, p. 86).

For the period of 2070 to 2099 relative to 1986 to 2015, precipitation increases of up to 20% are projected in winter and spring for northcentral U.S., with decreases by 20% or more in the Southwest in spring (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events are expected to continue to increase across the U.S., with the largest increases in the Northeast and Midwest (Hayhoe et al. 2018, p. 88). Projections show large declines in snowpack in the western U.S. and shifts of snow to rain in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 91).

NLEB's responses to these changes are expected to be similar to what has already been observed in North American insectivorous bats, such as little brown bat (see above and Appendix 4-C). This includes reduced reproduction due to drought conditions leading to declines in available drinking water (Adams 2010, pp. 2440–2442), reduced adult survival during periods of drought (Frick et al. 2010, pp. 131–133), or reduced reproduction during cooler, wetter springs in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Magnitudes of responses will vary depending throughout the ranges of the species' and on how much the annual temperature actually rises in the future.

Habitat Loss

The 2010 Resources Planning Act (RPA) Assessment (USFS 2012, entire) and 2016 RPA Update (USFS 2016, entire) summarized findings related to the status, trends, and projected future of U.S. forests and rangeland resources (we have nothing comparable for Canada). This assessment was influenced by a set of future scenarios with varying assumptions regarding global and U.S. population, economic growth, climate change, wood energy consumption, and land use change from 2010 to 2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million hectares (16–34 million acres or 4–8% of 2007 forest area) across the conterminous U.S., and forest loss is expected to be concentrated in the southern U.S., with losses of 3.6–8.5 million hectares (9–21 million acres) (USFS 2012, p. 12). The 2010 Assessment projected limited climate effects to forest lands spread throughout the U.S. during the projection period, but effects were more noticeable in the western U.S. The projections were dominated by conversions of forested areas to urban and developed land cover (USFS 2012, p. 59). The 2016 Update incorporated several scenarios including increasing forest lands through 2022 and then leveling off or declines of forest lands (USFS 2016, p. 8–7). However, regenerating young forests temporarily lack large roosts that provide space and thermal needs for NLEB colonies. While past and projected forest loss and forest regeneration rates can provide a coarse assessment of long-term trends, they are not particularly meaningful for determining the magnitude of impact unless overlaid where the species actually occurs. Loss of essential population needs of roosts and foraging and commuting habitat within NLEB home range where they remain is the issue. Furthermore, loss of roosting and foraging habitat compounds the impacts from WNS (see Appendix 4-D).

Synopsis of Future Threat Conditions

Using the available data and information summarized above and in Chapters 5 and 6, we assigned the scope, severity, and impact given the projected future state conditions for each of the primary influences (Table 4.5). WNS continues to be the greatest threat to NLEB, due to the expected future declines in population abundances (98–100% in known hibernacula) over most to all of its range (Wiens et al. 2022, pp. 226–229). Confidence in impact to NLEB from WNS and wind were "high" due to availability of quantitative data. Wind mortality impact is expected to be pervasive in scope and increase in severity, with population impacts reaching 83% by 2060 (Table A-3D2). Although the increasing severity of wind energy related mortality suggests that a *High* to *Very high* ranking is appropriate, we believe that the fatality rates are likely to decline as the abundance declines. The data were too limited (therefore, our confidence level was "moderate") to discern whether fatality rates have declined as the species' abundance precipitously decreases, so our scenarios did not account for this likelihood. For this reason, we assumed the severity of wind energy mortality will stay constant ("moderate") over time along with the overall impact level (*Medium*).

Our confidence in analysis on the impact of habitat loss and climate change remain "low" to "moderate" due to minimal quantitative data. Both habitat loss and climate change are forecasted to remain pervasive across the species' range, while the severity of population level declines are predicted to range from slight to moderate due a reduction in the spatial distribution of the species across the range. Given NLEB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies), the severity of habitat loss at occupied sites will vary between slight (e.g., limited tree removal within summer habitat) to extreme (e.g., loss of a hibernaculum or maternity colony). Therefore, impacts from habitat loss in the future may vary between *Low Impact* and *Very High Impact*. Lastly, increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase in the future leading to increased negative effects to NLEB (e.g., increased mortality, reduced reproductive success); therefore, our impact analysis predicts *Medium Impact* from climate change under future state conditions.

Table 4.5 Assessment of future impact from primary threats (adapted from Master et al. 2012 and Cheng et al. 2021, p. 5). See criteria definitions in Chapter 1 (Figure 1.2).

Criteria	WNS	Wind Mortality	Habitat Loss	Climate Change
Scope	Pervasive	Pervasive	Pervasive	Pervasive
Severity	Extreme	Moderate	Slight-Extreme	Moderate
Impact	**Very High**	**Medium**	**Low** **Very High**	**Medium**
Confidence Level	High	Moderate	Moderate	Low

CHAPTER 5 – CURRENT CONDITION

In this chapter, we describe the current demographic conditions and the projected number, health, and distribution of NLEB populations given these current conditions (Figure 5.1). Current state conditions encompass the current abundance, growth rate, WNS occurrence, and installed wind energy capacity. We projected abundance under current state conditions to garner insight into viability, which we describe in Chapter 7.

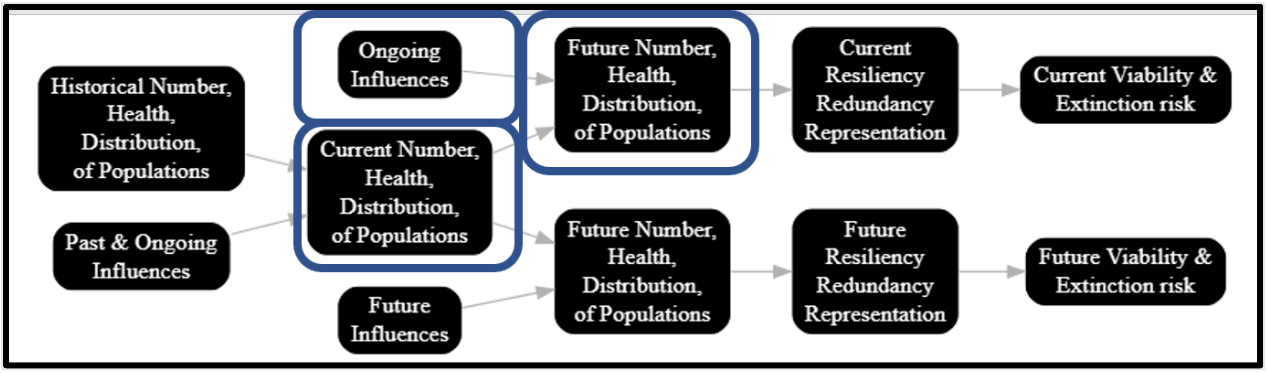

Figure 5.1. Highlighting (blue rectangles) the current step in our analytical framework.

Current demographic conditions– Available evidence indicates NLEB abundance has and will continue to decline substantially over the next 10 years under current conditions (Figure 5.2). Evidence of the past decline is demonstrated in available data in both winter and summer. For example, rangewide winter abundance has declined by 49% and the number of extant winter colonies (populations) by 81% (Figure 5.2, Table A–3A1). There has also been a noticeable shift towards smaller colony sizes, with a 96–100% decline in the number of large hibernacula (≥100 individuals) (Figure 5.3). Although the declines are widespread, the magnitudes of the winter declines vary spatially (Figure 5.4). In the Eastern Hardwoods, the core of NLEB range, abundance declined by 56% and the number of sites by 88%. Abundance and the number of sites declined in the remaining 4 RPUs (87% and 82% - East Coast RPU, 90% and 44% - Midwest RPU, 24% and 70% - Southeast RPU, and 0% and 40% - Subarctic RPU, respectively; Table A–3B3). Across all RPUs, the potential of population growth is low; the probability of RPU growth rates (λ) ≥1 ranges from 0 to 11% (Table A-3B2).

Declining trends in abundance and occurrence are also evident across much of NLEB summer range. Based on derived rangewide summaries from Stratton and Irvine (2022, p. 102), rangewide occupancy has declined by 80% from 2010–2019 (Table A-3B4, Figure 5.7). Although these declines attenuate westward, the probability of occupancy declined in all RPUs (Table A-3B4). Similarly, Whitby et al. (2022, p. 160), using data collected from mobile acoustic transects, found a 79% decline in rangewide relative abundance from 2009–2019. Measurable declines were also found in the Midwest RU (91%) followed by the Eastern Hardwoods (85%), East Coast (71%), and Southeast (57%) RPUs (Table A–3B4). Data were not analyzed in the Subarctic RPU due to a lack of observations. Finally, Deeley and Ford (2022, p. 18, 21–23) observed a significant decrease in mean capture rate post-WNS arrival. Estimates derived from their results indicted a 43–77% decline in summer mist net captures compared pre and post arrival of WNS (Table A–3B4).

Future projections based on current conditions - Collectively, these data indicate NLEB has declined and given the declining trajectories, will continue to decline. Future projections from the BatTool, assuming no further WNS spread nor increases in wind capacity (current stressor conditions), show sharp declines in rangewide abundance, number of hibernacula, and spatial extent into the future.

- By 2030 (~ 1 generation), rangewide abundance declines by 95% (CI 75–99%; Figure 5.2).
- The number of extant hibernacula declines by 99%, with 11 of the 737 historically occupied hibernacula extant by 2030 (Figure 5.5) and 1 extant hibernaculum by 2040.
- The winter colony sizes also become reduced, with the number of large hibernacula (≥100 bats) declining from 53 in 2000, 20 in 2020, to 1 hibernaculum (98% decline from 2020) by 2030 (Figure 5.3).
- Subsequent to declines in the number of hibernacula, NLEB's known winter range declines by 75% (Table A-3B1), with the vast majority (90%) of individuals becoming concentrated in a smaller number of hibernacula, going from 66 hibernacula in 2000, 29 in 2020 to 6 by 2030.

The projected declines are widespread across the RPUs.

- Median hibernacula abundances in the Southeast, East Coast, Midwest, and Subarctic RPUs decline to 2–16 (CI 0–4,118) or 99–100% decline, with corresponding low probabilities of persistence by 2030 (Tables A-3B3, Figure 5.6).
- In the Eastern Hardwoods RPU, median abundance declines 99%, with bats persisting in 10 hibernacula by 2030. Of the projected extant hibernacula, 1 is projected to be large (≥100 individuals; Figure 5.3).

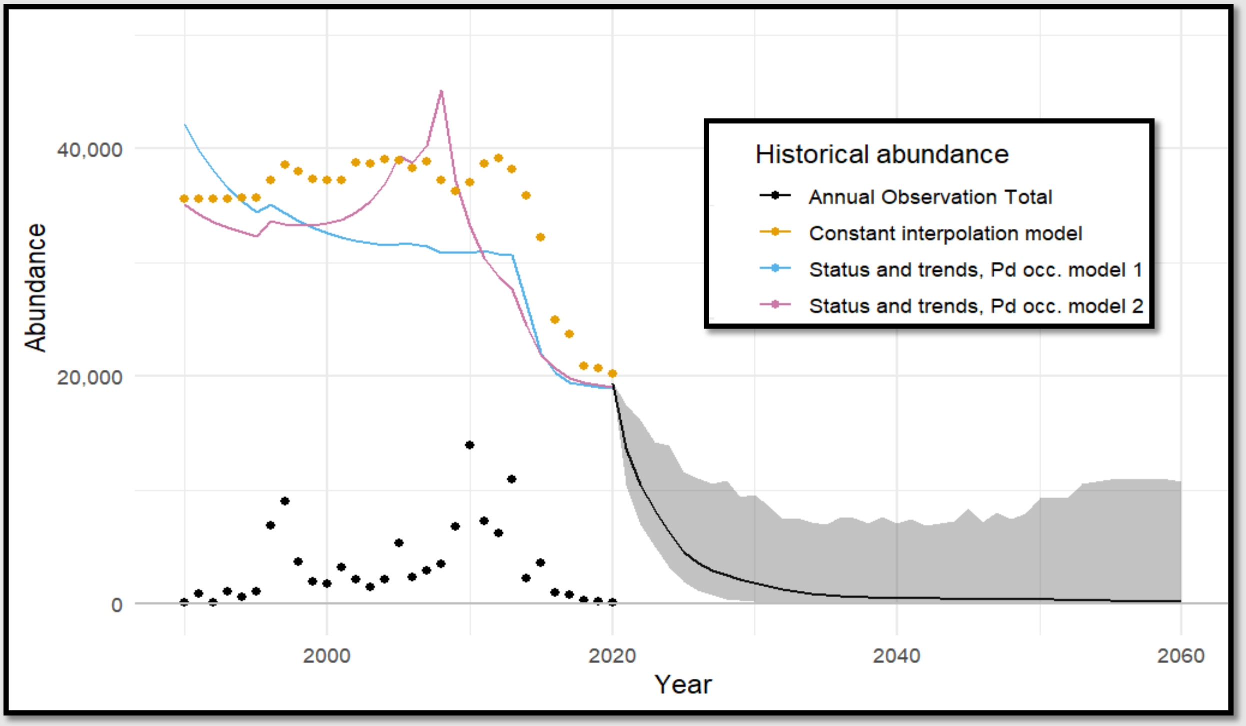

Figure 5.2. Median projected rangewide abundance (black line) and 90% CI (gray) given CURRENT state conditions (current abundance, growth rate, WNS occurrence, and installed wind energy capacity). Abundance from 1990 – 2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status & trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

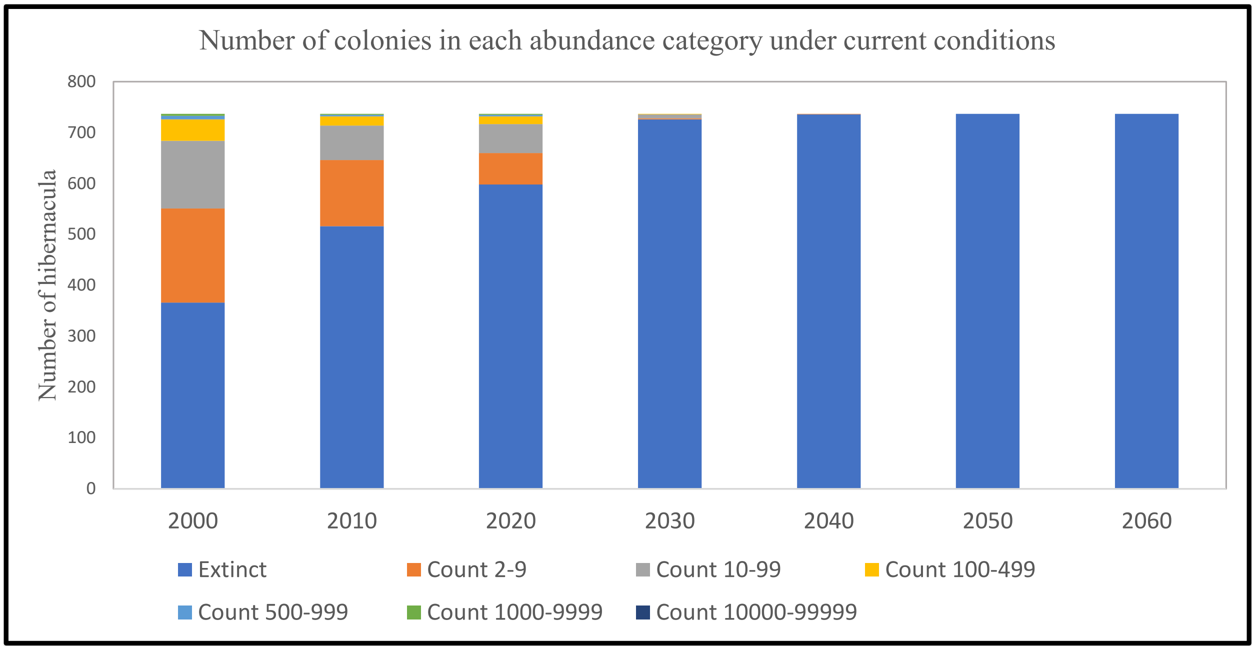

Figure 5.3. The number of hibernacula in each colony abundance category under current state conditions.

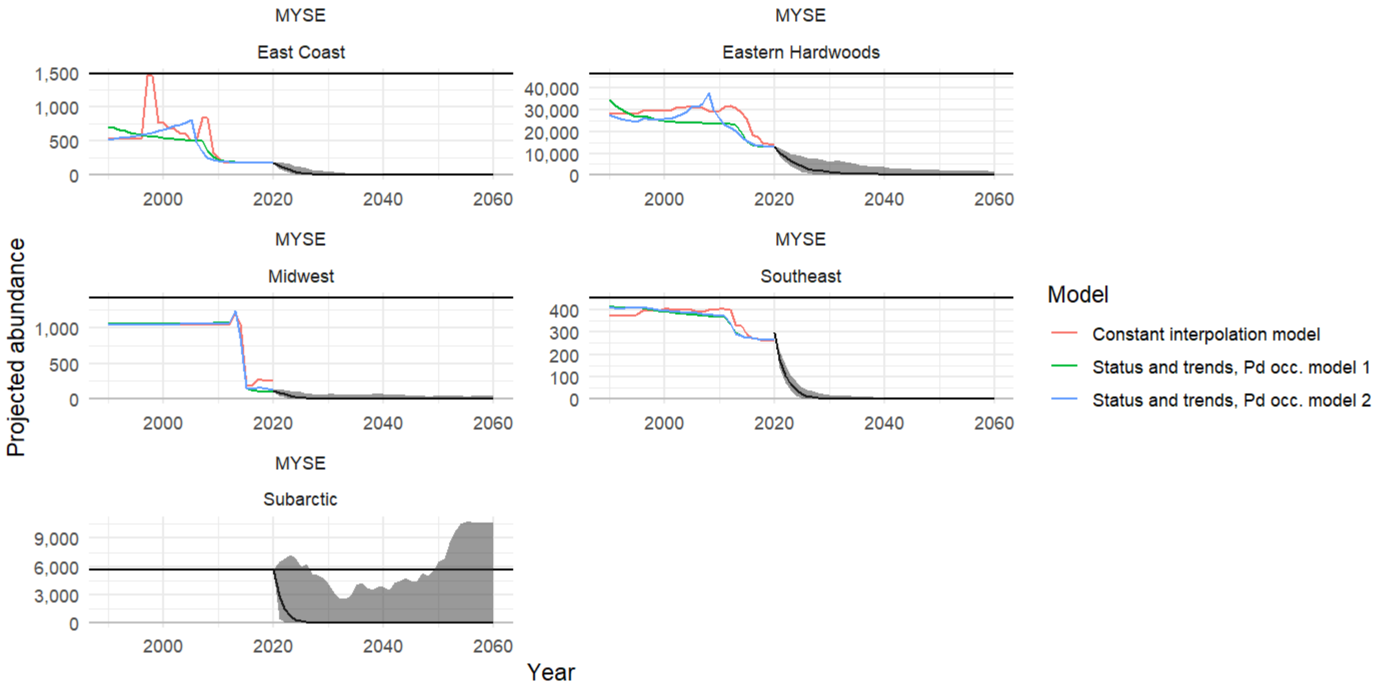

Figure 5.4. Median projected RPU abundance (black line) and 90% CI (gray) under CURRENT state conditions (current abundance, growth rate, WNS occurrence, and installed wind energy capacity for the 3 RPUs. Abundance from 1990–2020 derived from winter colony count data using a) constant interpolation (red line), b) status and trend model informed by Pd occurrence model 1 (green line) and c) status and trend model informed by Pd occurrence model 2 (blue line).

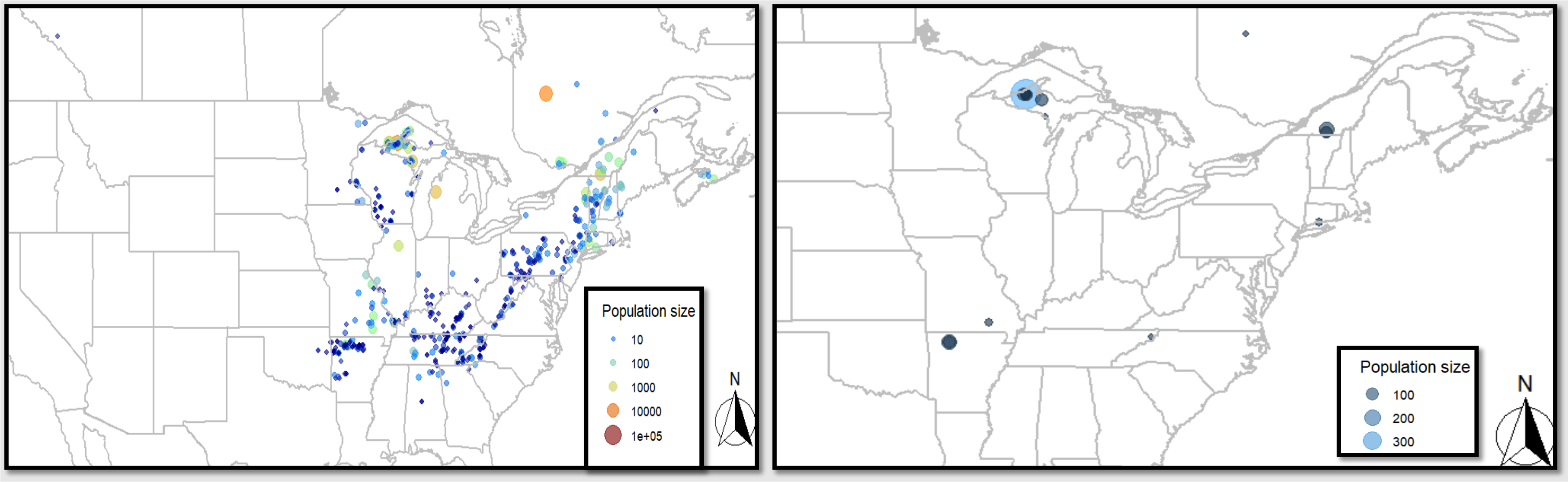

Figure 5.5. NLEB extant hibernacula at year 2000 (left) and projected at 2030 (right) given CURRENT state conditions. Color and size reflect median hibernacula abundance.

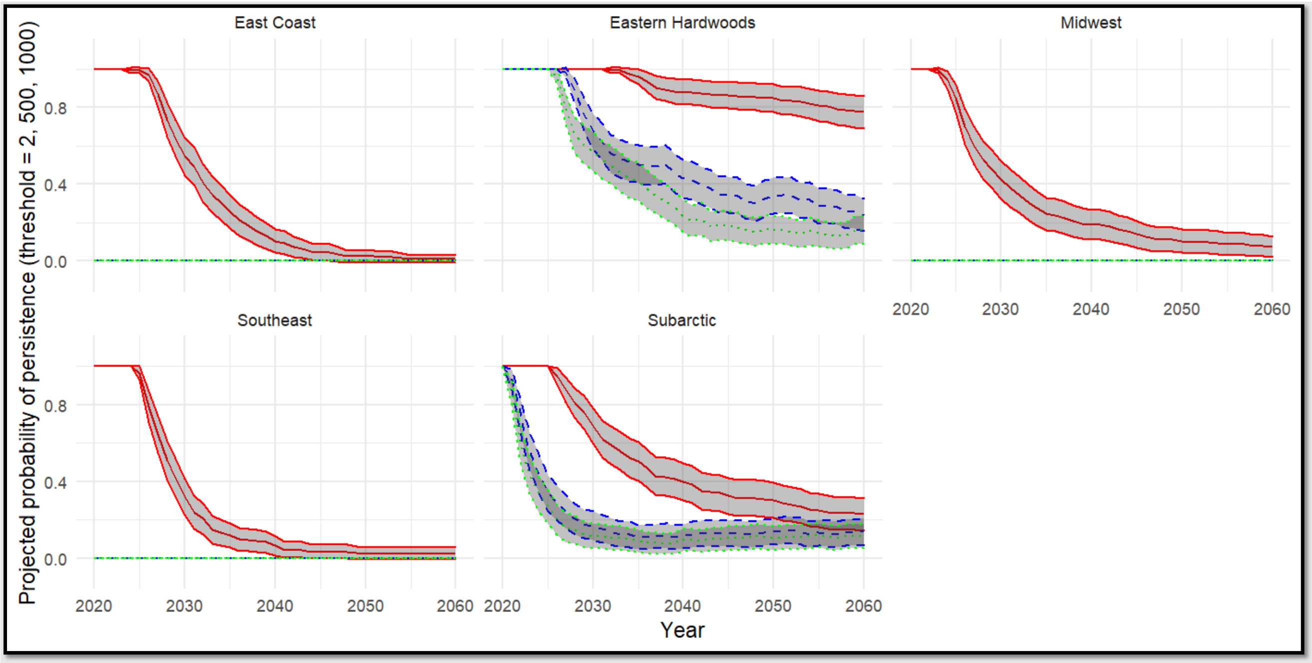

Figure 5.6. Probability of RU-abundance remaining above X individuals given CURRENT state conditions, x= 2 bats (red), x=500 bats (blue), and b) x=1000 bats (green).

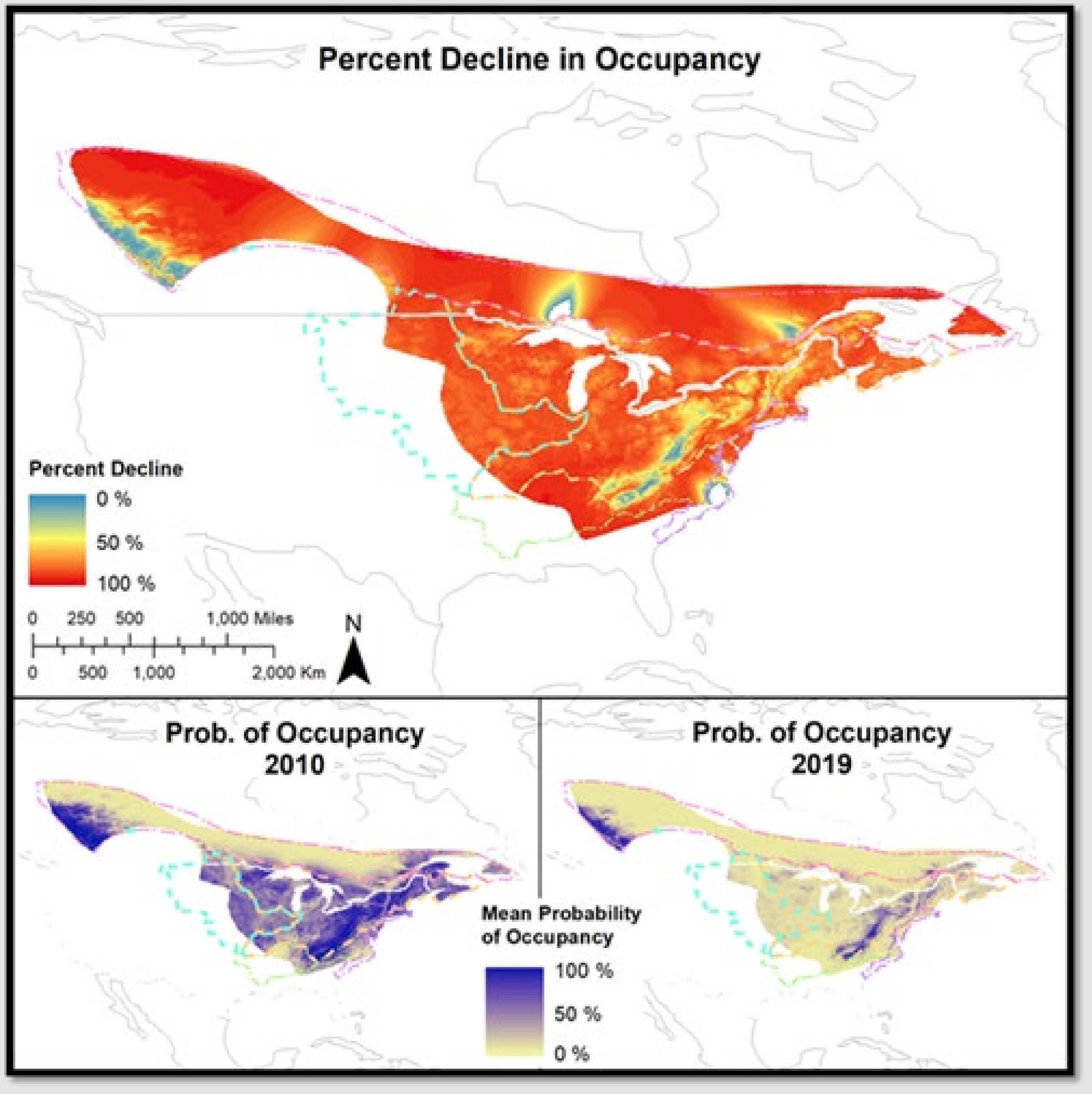

Figure 5.7. Predicted percent decline in probability of occupancy (top) and probability of NLEB summer occupancy in 2010 (bottom left) and 2019 (bottom right) based on data collected from stationary and mobile transect acoustic monitoring and capture records summarized at the 10km x 10km NABat grid cell (Stratton and Irvine 2022, entire). Dotted boundaries correspond to RPUs. Cooler colors represent lower percent declines (top panel) or higher probability of occupancy (bottom panels).

CHAPTER 6—FUTURE CONDITION

Future viability is the ability of NLEB to sustain healthy populations into the future given its current demographic condition and future condition of the influences (Figure 6.1). To assess NLEB future viability, we again used the BatTool to project hibernaculum abundance over time given projected *Pd* spread and wind energy build-out (see Chapter 4, *Future Scenarios* subsection, for further description). Projection of future number, distribution, and health of populations is needed to understand NLEB's future ability to withstand normal stochasticity, stressors, catastrophic events, and novel environmental changes (i.e., its viability under future influences). In this chapter, we describe the projected number, health, and distribution of NLEB given future state conditions (i.e., future *Pd* occurrence and future installed wind energy capacity) and describe the viability implications under future influences in Chapter 7.

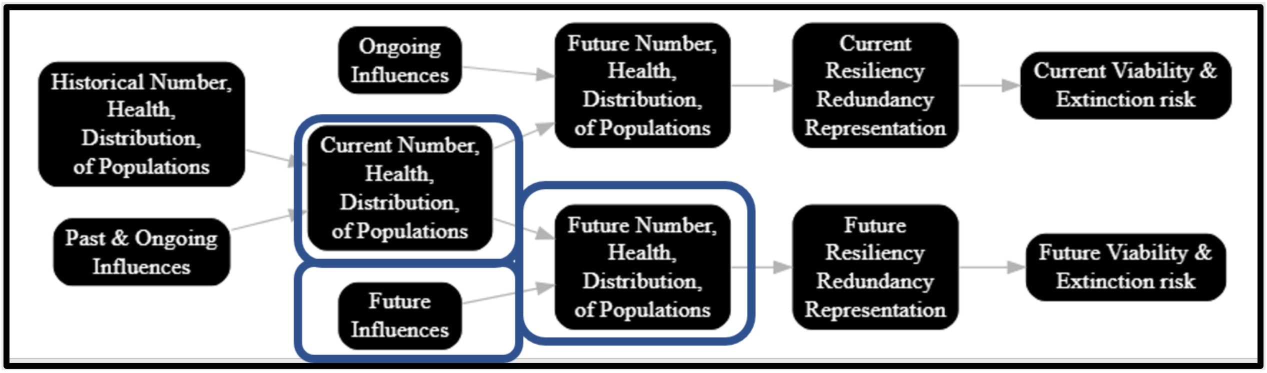

Figure 6.1. Highlighting (blue rectangles) current step in our analytical framework

Under both future scenarios, the declines worsen precipitously.

- Median rangewide abundance declines 95% by 2030 (74–100% CI) and reaches 99% by 2060 (67–100% CI). Under the future scenarios, the decline trajectory continues (despite no impacts due to WNS being applied 15 years after *Pd* arrival; Figure 6.2, Table A–3C1).
- The number of extant hibernacula decline to 9 by 2030 and 0 hibernacula by 2050 (Figure 6.3, Table A–3C1).
- Colony sizes continue to shift towards smaller sizes, with 89% of the projected extant colonies in 2030 having fewer than 100 bats (Figure 6.4).
- Spatially, NLEB's winter range declines by 75% by 2030 (100% by 2040) (Table A-3C1).

As projected under the future conditions, declines are widespread and there is limited chance for persistence.

- Median abundances in the Southeast, East Coast, Midwest, and Subarctic RPUs decline to 2–22 (CI 2–6,199) or 99–100% decline, with corresponding low probabilities of persistence by 2030 (Figure 6.6, Table A-3C3).

- In the Eastern Hardwoods RPU, median abundance declines 95%, with bats persisting in 8 hibernacula by 2030. By 2060, all populations at all hibernacula are projected to be extinct (Figure 6.5, Table A-3C2).

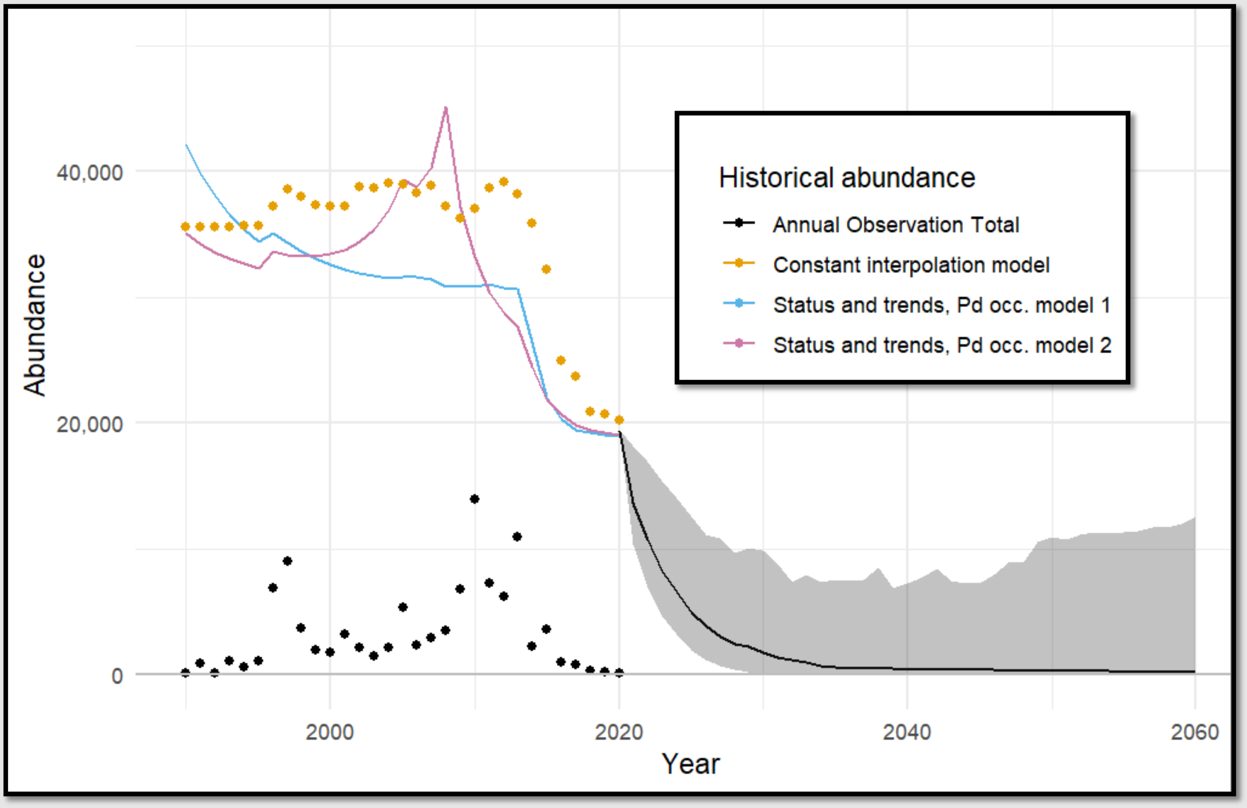

Figure 6.2. Projected median rangewide abundance (black line) and 90% CI (gray shading) under FUTURE state conditions. Abundance from 1990–2020 derived from raw data (black dots) using a) constant interpolation (yellow dots), b) status & trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

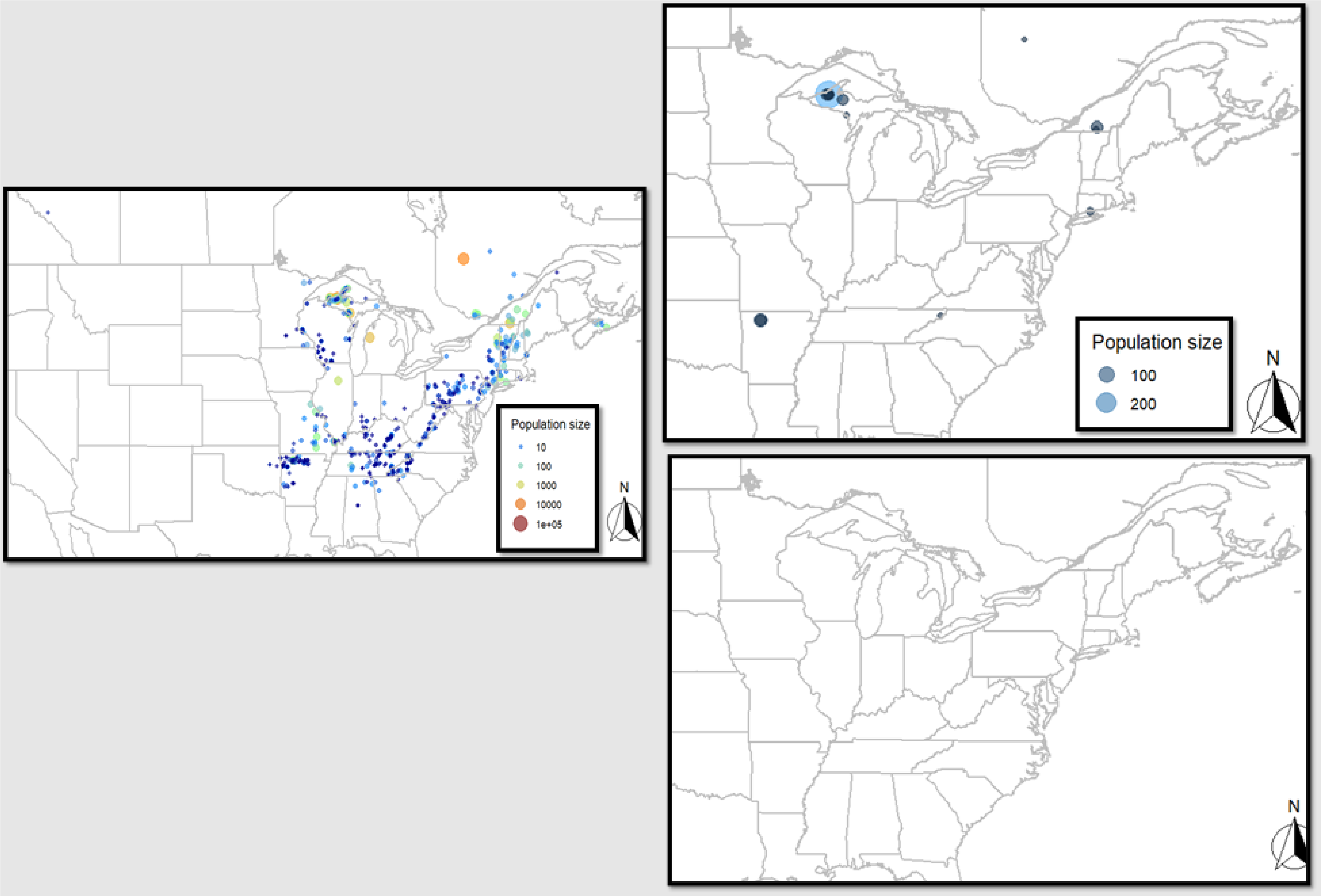

Figure 6.3. NLEB extant hibernacula in 2000 (left) and projected 2030 (upper right) and 2060 (bottom right) given FUTURE state conditions. Color and size reflect medium hibernacula abundance.

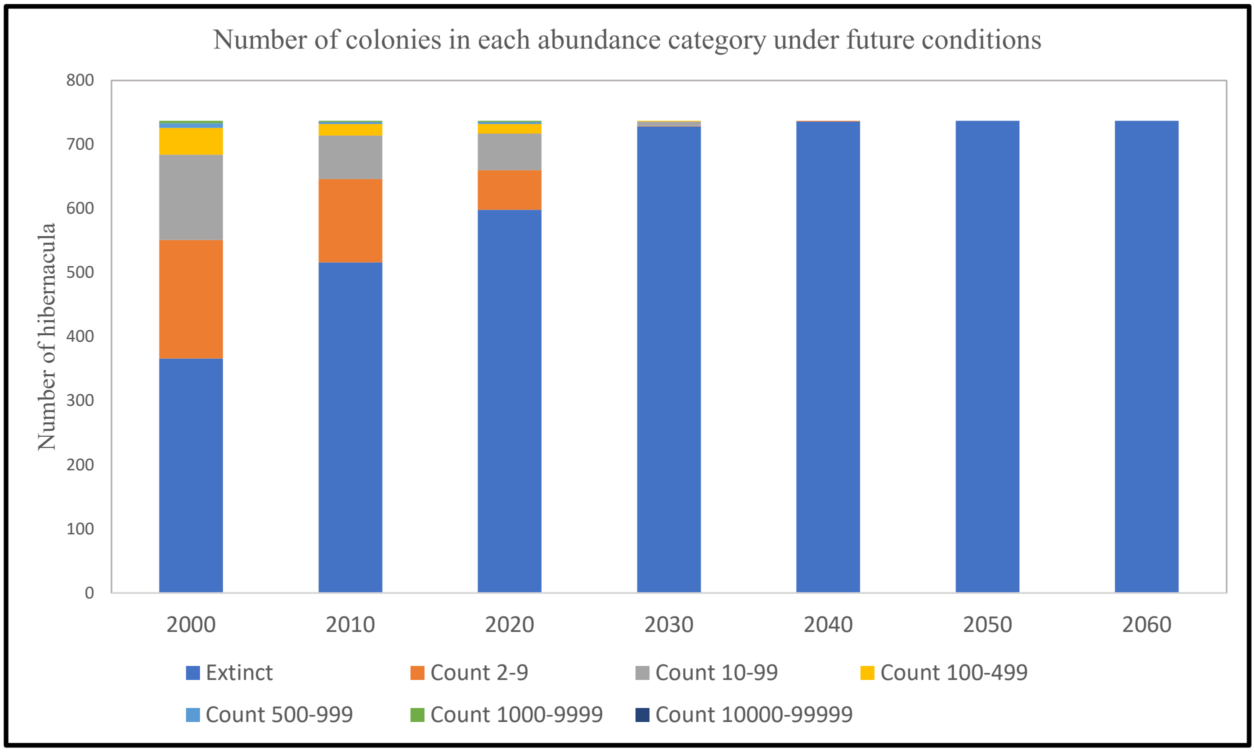

Figure 6.4. The projected number of hibernacula in each colony abundance category under FUTURE state conditions.

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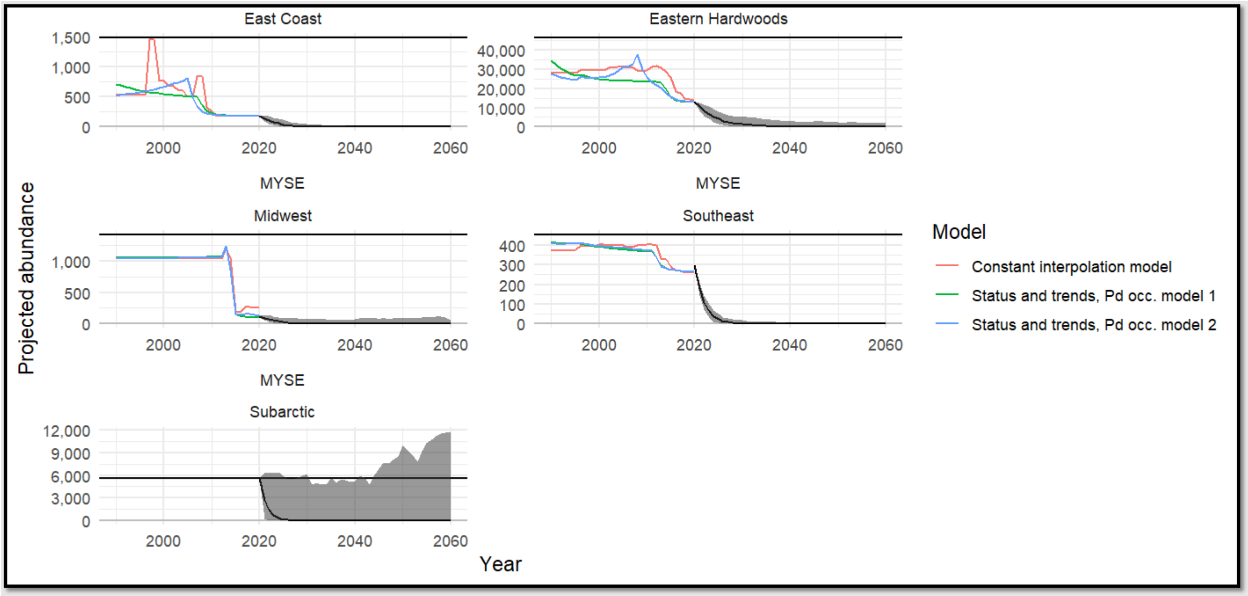

Figure 6.5. Projected median (black line) and 90% CI (gray shading) for RPU abundance under FUTURE state conditions for the 5 RPUs. Abundance from 1990 –2020 derived from raw data using a) constant interpolation (red line), b) status & trend model

informed by Pd occurrence model 1 (green line) and c) status & trend model informed by Pd occurrence model 2 (blue line).

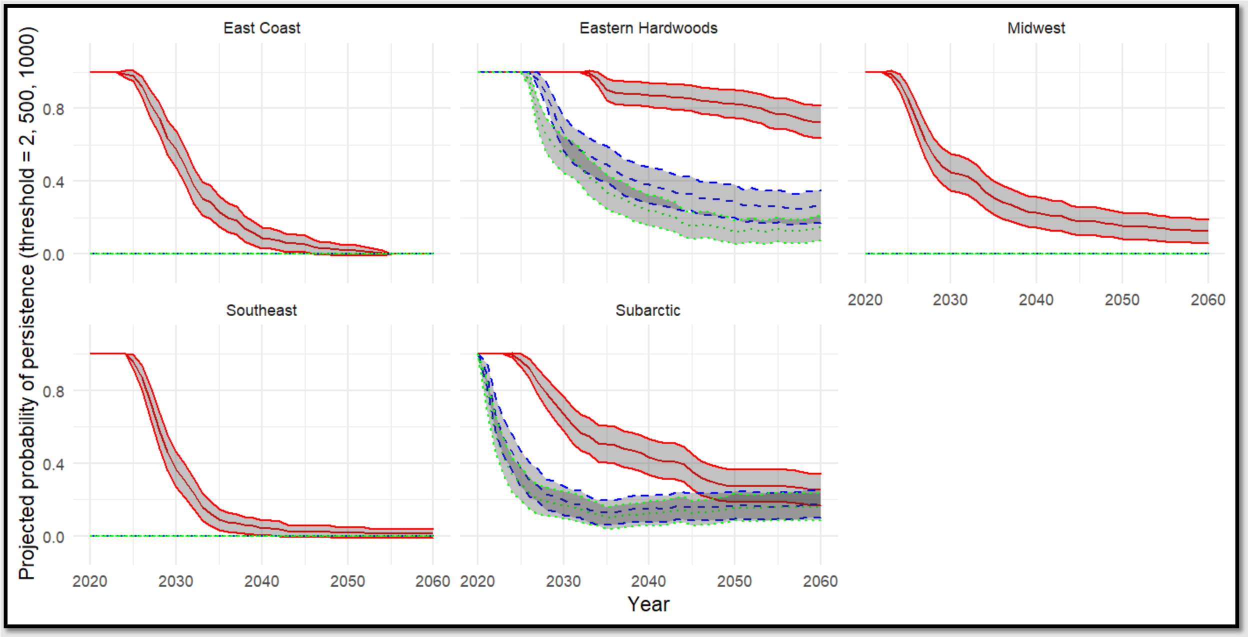

Figure 6.6. Probability of RPU-abundance remaining above X individuals given FUTURE state conditions, x= 2 bats (red), x=500 bats (blue), and x=1000 bats (green).

Habitat Loss and Climate Change

As discussed previously, we did not incorporate habitat loss and the effects of climate change into our quantitative modeling efforts (i.e., not included in the projections depicted in Figures 6.2– 6.6). Ongoing effects from habitat loss and climate change likely continue into the future and may even be exacerbated based on reduced abundance and distribution anticipated under our current and future scenarios. See Table 4.5 for a description of the scope, severity, and impact of future habitat loss and climate change impacts. Additionally, future impacts from habitat loss and climate change are discussed more thoroughly in Appendix 4.

CHAPTER 7—SPECIES VIABILITY

This chapter synthesizes the results from our historical, current, and future analyses and discusses the consequences for NLEB viability (Figure 7.1). NLEB viability is influenced by the number, health, and distribution of populations. Across the range and within all RPUs, NLEB abundance and distribution has decreased. Multiple data types and analyses indicate downward trends in NLEB population abundance and distribution over the last 14 years (2006–2020; Table 7.1), and we found no evidence to suggest that this downward trend will change in the future (Figure 7.2). As is the case for all species status assessments we do not have perfect information on NLEB's occurrence, but the best available data suggest that bats at unknown hibernacula will undergo similar declines observed at known winter colonies. We outline the key uncertainties in our analyses and our resolution of them in Appendix 1.

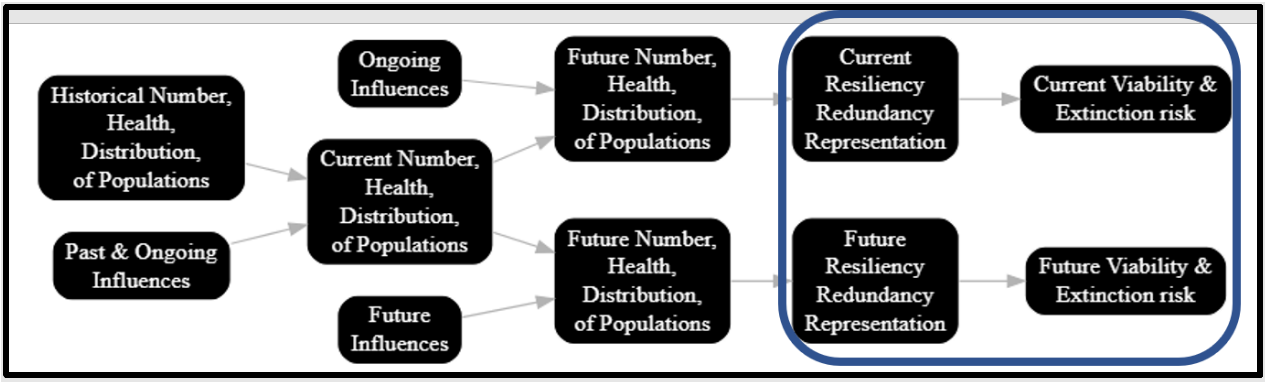

Figure 7.1. Highlighting (blue rectangle) the current step in our analytical framework.

Table 7.1. Summary of recent NLEB population trends from multiple data types and analyses. Winter Colony analysis – derived from Wiens et al. (2022, entire) data; Summer Occupancy analysis –Stratton and Irvine (2022, entire); Summer Capture analysis – Deeley and Ford (2022, entire); and Summer Mobile Acoustic analysis – Whitby et al. (2022, entire).[1] No data available.

Representation Unit	Winter Colony	Summer Occupancy	Summer Capture	Summer Mobile Acoustic
Southeast	-24%	-85%	-47%	-50%
Eastern Hardwoods	-56%	-78	-43%	-87%
Subarctic	-0%	-63%	-[1]	-[1]
Midwest	-90%	-87%	-77%	-99.9%
East Coast	-87%	-79%	-43%	-69%
Rangewide	-49%	-80%	-43–77%	-79%

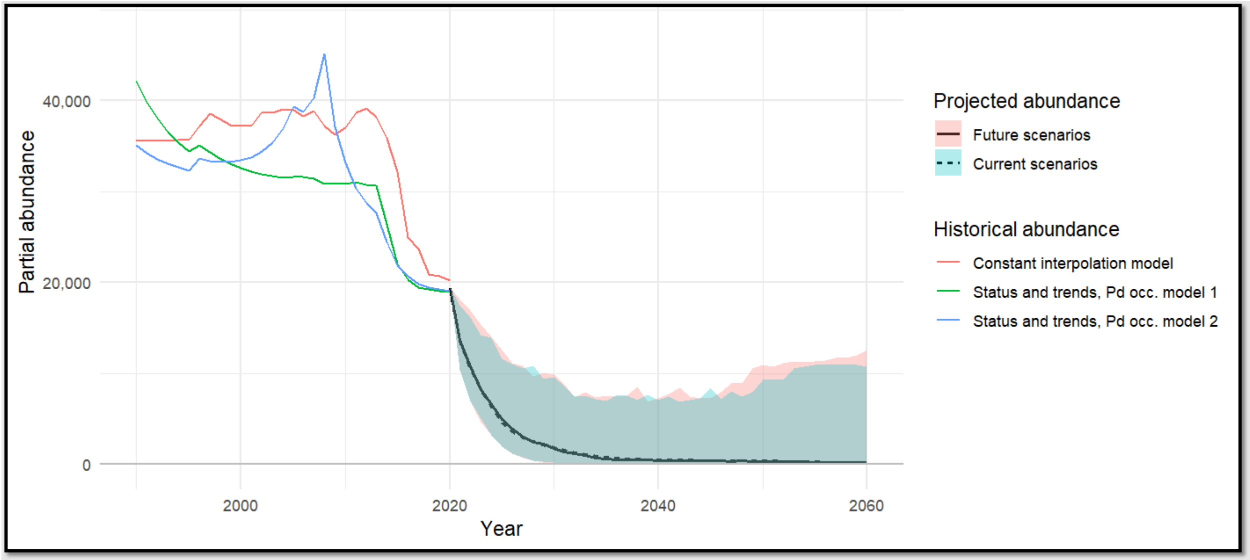

Figure 7.2. The projected NLEB abundance over time given current (blue) WNS spread and installed wind capacity and plausible future scenarios (pink) for WNS spread and increased installed wind energy capacity. The dotted and solid lines represent the median abundance under current and future scenarios, respectively. Historical abundance from 1990 –2020 derived from a) constant interpolation (red), b) status & trend model informed by Pd occurrence model 1 (green line) and c) status & trend model informed by Pd occurrence model 2 (blue line).

The viability of a species depends upon its ability to sustain populations in the face of normal environmental and demographic stochasticity, catastrophes, and novel changes in its environment. For example, demographically and physically healthy populations better withstand and recover from environmental variability and disturbances. Additionally, populations spread across heterogeneous conditions are unlikely to be exposed at the same time to poor environmental conditions, thereby guarding against synchronous population losses. Similarly, species with genetically healthy populations (large N_e, which begets genetic diversity) spread across the breadth of genetic and phenotypic diversity preserve a species' adaptive capacity, which is essential for adapting to their continuously changing environment (Nicotra et al. 2015, p. 1269). Without such variation, species are less responsive to change and more prone to extinction (Spielman et al. 2004, p. 15263). Lastly, having multiple healthy populations widely distributed guards against losses of adaptive diversity and RPU-level extirpation in the face of catastrophic events.

We quantitatively assessed NLEB's current viability by projecting the species' abundance and distribution given current WNS occurrence (no further spread) and current installed wind energy capacity, and future viability given future plausible scenarios of further WNS spread and increased wind energy capacity. We also qualitatively considered impacts from climate change, habitat loss, and conservation efforts. All existing data and our qualitative and quantitative analyses suggest that NLEB's viability has and will continue to steeply decline over time under the current and plausible future conditions.

Unquestionably, WNS is the primary driver (or influence) that has led to the species' current condition and is predicted to continue to be the primary influence into the future (Table 7.2). Currently, WNS occurs across 59% of NLEB's range (Cheng et al. 2021, p. 7) and is impacting 99–100% hibernacula (Wiens et al. 2022, pp. 226–229, 231–247). In addition, WNS is predicted to reach 100% of the species' range in the U.S. by 2025 (Wiens et al. 2022, pp. 226–229). Prior to WNS, NLEB was abundant and widespread, and abundance and occupancy were generally stable (Cheng et al. 2022, p. 204). WNS impacts have resulted in most winter colonies experiencing a 97–100% decline in abundance compared to historical conditions (Cheng et al. 2021, entire).

Wind energy related mortality, although not currently acting as a driver in NLEB's viability, is projected to be more impactful in the future as it will increase in pervasiveness and severity (Table 7.2). Based on 2020 wind build-out, an estimated 38 to 150 (mean = 122) NLEBs are killed annually at wind facilities and annual mortality is projected to increase to 202 to 2,926 individuals by 2050 under the future low and high build-out scenarios, respectively (Figures 4.10 and 4.11, Tables 4.1 and 4.4). Wind related mortality is discernible, particularly in future scenarios, even with ongoing declines from WNS (Figure A-1B2; see also Whitby et al. 2022, pp. 151–153). NLEB abundance is projected to decline 18 and 77% from 2030 to 2060 from wind related mortality alone under current conditions and from 28 to 80% under the future scenarios. Consequently, mortality from wind turbines likely has and will continue to cause detectable declines in NLEB abundance.

Although we consider habitat loss pervasive across NLEB range, impacts to NLEB and its habitat are often realized at the individual or colony level. Loss of hibernation sites (or modifications such that the site is no longer suitable) can result in impacts to winter colonies. Impacts from forest loss (e.g., roosting or foraging habitat) vary depending on the timing, location, and extent of the removal. Given how common and wide-ranging NLEB was throughout much of its range prior to the arrival of WNS, we assume the range-wide magnitude of impact from habitat loss was low. However, as NLEB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and summer colonies and remaining populations are anticipated to be less resilient), habitat loss at occupied sites will vary from slight (e.g., limited tree removal within summer habitat) to extreme (e.g., loss of a hibernaculum or maternity colony). Therefore, impacts from habitat loss in the future may vary between low to very high (Table 7.2).

Climate change impacts are challenging to describe for wide-ranging species, such as NLEB. The changing climate has and will likely continue to have a multitude of impacts on species throughout North America (Foden et al. 2018, p. 9). Despite being pervasive; however, we believe the rangewide magnitude of impact is currently low (Table 7.2). In addition, there are questions about whether some negative effects are currently offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). Although there may be some offsetting of effects under current climate conditions, increasing negative impacts are anticipated in the future (Table 7.2). Increasing incidence of climatic extremes (e.g., drought, excessive summer precipitation) will likely increase, leading to increased NLEB mortality and reduced reproductive success. As mentioned above, as NLEB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies) and populations anticipated to be less resilient, effects from climate change may be more impactful than if the populations were well distributed and robust.

Table 7.2. Threat (impact) level for the primary influences currently and projected future low and high impact scenarios.

	WNS	Wind Mortality	Habitat Loss		Climate Change
Current	Very High	Medium	Low		Low
Low Impact	Very High	Medium	Low	Very High	Medium
High Impact	Very High	Medium	Low	Very High	Medium

While we focused our analyses on ongoing and anticipated effects from WNS, wind, climate change and habitat loss, we also recognize that novel threats (e.g., new disease or invasive species) may emerge for NLEB. NLEB's mobility and roost-shifting behaviors provide mechanisms for individual bats to respond to changes in temperature, prey availability and roost suitability. However, as discussed in Chapter 2 and Appendix 2-B, temperate zone insectivorous bats including NLEB have several inherent traits that limit their ability to respond to changes in the environment, especially to rapid changes. These include their high site fidelity (winter and summer), specialized winter habitat requirements and summer roost microclimate needs, and low reproductive output. We have already observed the extremely limited ability for NLEB to respond to the novel threat WNS.

Viability under Current Conditions

Under current conditions, NLEB abundance, number of occupied hibernacula, spatial extent, probability of persistence, summer habitat occupancy (measured by bat captures and acoustic recordings) across the range and within all RPUs are decreasing (Chapter 5 and Table 7.1). Since the arrival of WNS, NLEB abundance steeply declined, with most (91%) winter colonies having fewer than 100 individuals. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, NLEB's ability to recover from these low abundances is limited given their low reproduction output (1 pup per year). Therefore, NLEB's resiliency is greatly compromised in its current condition. Additionally, NLEB's spatial extent is projected to decline, with 75% reduction by 2030. As NLEB's abundance and spatial extent decline, NLEB will also become more vulnerable to catastrophic events

In addition to reduced redundancy and resiliency, NLEB's representation has also been reduced. As explained above, NLEB's capacity to adapt is constrained by its life history and the level of its intraspecific diversity (e.g., genetic, phenotypic, behavioral, ecological variability). The steep and continued declines in abundance have likely led to reductions in genetic diversity, and thereby reduced NLEB adaptive capacity. Further, the projected widespread reduction in the distribution of hibernacula will lead to losses in the diversity of environments and climatic conditions occupied, which will impede natural selection and further limit NLEB's ability to adapt. Moreover, at its current low abundance, loss of genetic diversity via genetic drift will likely accelerate. Consequently, limiting natural selection process and decreasing genetic diversity will further lessen NLEB's ability to adapt to novel changes (currently ongoing as well as future changes) and exacerbate declines due to continued exposure to WNS, mortality from wind turbines, and impacts associated with habitat loss and climate change. Thus, even without further *Pd* spread and additional wind energy development, NLEB's viability is likely to rapidly decline over the next 10 years (Figures 7.2 and 7.3).

Viability under Future Scenarios

Under the projected range of plausible future scenarios, WNS spread reaches close to 100% of NLEB's entire range (Wiens et al. 2022, pp. 226–229) and wind energy related mortality increases by 66% to 2,298% (Udell et al. 2022, entire; see Table 4.4). By 2060, NLEB abundance declines by 99% (Figure 7.2) and the number of extant hibernacula declines by 100% (Figure 7.3). Under the future scenario, by 2040, only one hibernaculum is projected to remain in the Eastern Hardwoods RPU. By 2050, no hibernacula remain in any of the RPUs (Figure 7.3). Given the projected low abundance and the few number and restricted distribution of winter colonies, NLEB's currently impaired ability to withstand stochasticity, catastrophic events, and novel changes will worsen under the range of plausible future scenarios.

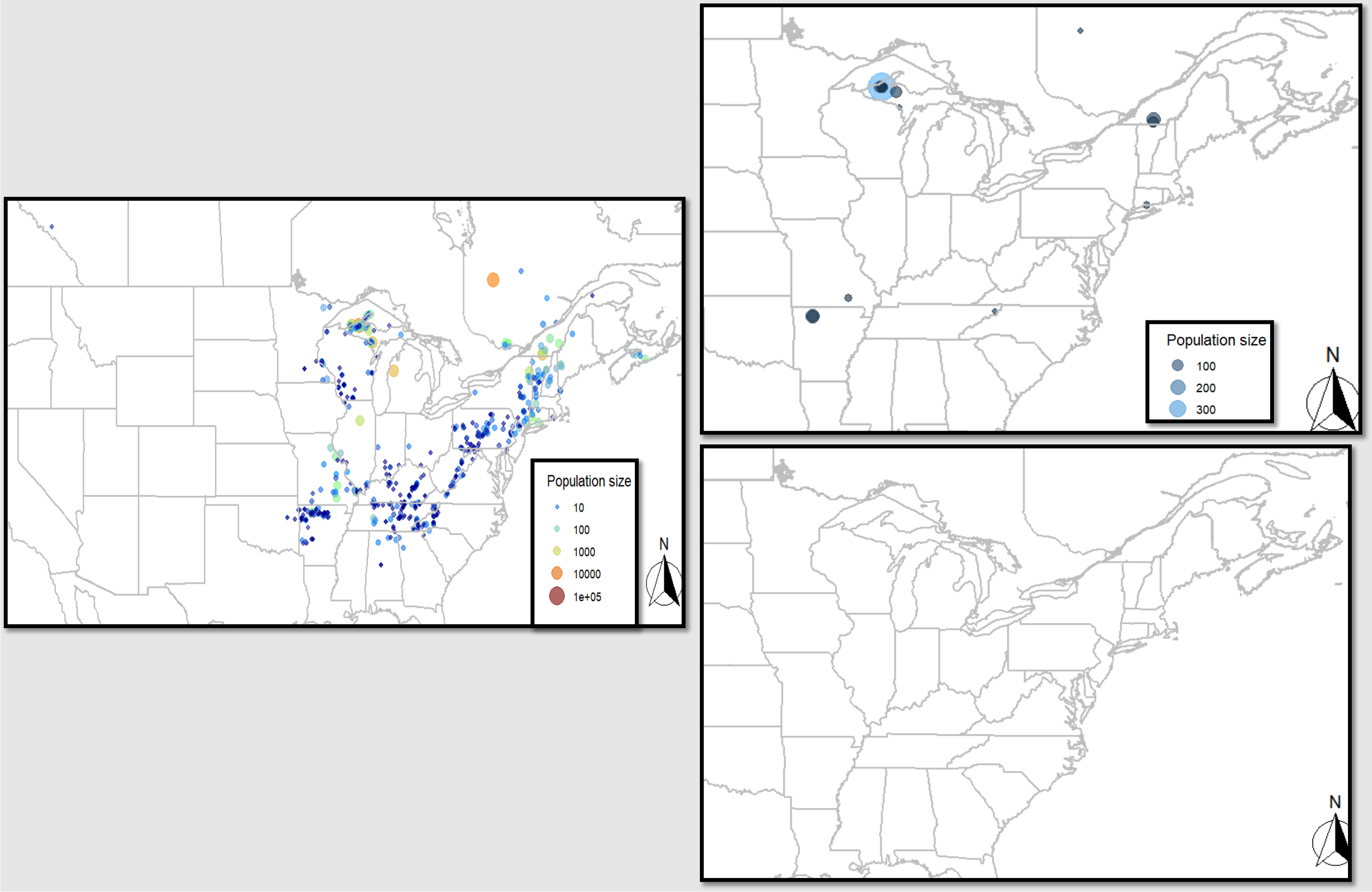

Figure 7.3. Projected change in NLEB winter distribution over time: 2000 (far left); 2030 under current conditions (top right), and 2060 under future conditions (bottom right).

REFERENCES CITED[6]

Adams, R.A. 2010. Bat reproduction declines when conditions mimic climate change projections for western North America. Ecology 91(8):2437–2445.

Adams, R.A., and M.A. Hayes. 2008. Water Availability and Successful Lactation by Bats as Related to Climate Change in Arid Regions of Western North America. Journal of Animal Ecology 77(6):1115–1121.

Alves, D.M.C.C, L.C. Terribile, and D. Brito. 2014. The Potential Impact of White-Nose Syndrome on the Conservation Status of North American Bats. PLoS ONE 9(9): e107395. doi:10.1371/journal.pone.0107395.

Amelon, S., and D. Burhans. 2006. Conservation assessment: *Myotis septentrionalis* (northern long-eared bat) in the eastern United States. Pages 69–82 *in* Conservation assessments for five forest bat species in the eastern United States, Thompson, F. R., III, editor. U.S. Department of Agriculture, Forest Service, North Central Research Station, General Technical Report NC-260. St. Paul, Minnesota. 82pp.

American Wind Energy Association (AWEA). 2020. Wind Facts at a Glance. Available at: https://www.awea.org/wind-101/basics-of-wind-energy/wind-facts-at-a-glance.

American Wind Wildlife Institute (AWWI). 2020. AWWI Technical Report: 2nd Edition: Summary of Bat Fatality Monitoring Data Contained in AWWIC. Washington, DC. Available at www.awwi.org. © 2020 American Wind Wildlife Institute.

Arnett, B., J. P. Hayes, and M. M. P. Huso. 2006. Patterns of preconstruction bat activity at a proposed wind facility in south-central Pennsylvania. An annual report submitted to the Bats and Wind Energy Cooperativer. Bat Conservation Internationa, Austin, TX. 46 pp. Available at: https://batsandwind.org/wp-content/uploads/precon_pa.pdf

Arnett, E B, W.K Brown, W.P. K Fiedler, B.L Hamilton, T.H. Henry, A. Jain, G.D. Johnson, J. Kerns, R. R. Koford, C. P. Nicholson, T. J. O'Connell, M. D. Piorkowski, and R. D. Tankersley, Jr. 2008. Patterns of bat fatalities at wind energy facilities in North America. Journal of Wildlife Management 72:61–78.

Arnett E.B., C.D. Hein, M.R. Schirmacher, M.M.P. Huso, and J.M. Szewczak. 2013. Evaluating the effectiveness of an ultrasonic acoustic deterrent for reducing bat fatalities at wind turbines. PLoS ONE 8(6):e65794.

Anthony, E.L.P., M.H. Stack, and T.H. Kunz. 1981. Night roosting and the nocturnal time budget of the little brown bat, *Myotis lucifugus*: Effects of reproductive status, prey density, and environmental conditions. Oecologia 51:151–156.

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[6] Includes references from body of document and appendices.

Arnold, B.D. 2007. Population Structure and Sex-biased Dispersal in the Forest Dwelling Vespertilionid Bat, *Myotis septentrionalis*. The American Midland Naturalist, 157(2):374–384.

Badin, H.A. 2014. Habitat selection and roosting ranges of northern long-eared bats (*Myotis septentrionalis*) in an experimental hardwood forest system. Muncie, IN: Ball State University. M.S. thesis.

Baerwald, E.F., G. H. D'Amours, B.J. Klug, and R.M.R.Barclay. 2008. Barotrauma is a significant cause of bat fatalities at wind turbines. Current Biology, 18(16):695–696.

Baijnath-Rodino, J.A., C. Duguay, and E. LeDrew. 2018. Climatological trends of snowfall over the Laurentian Great Lakes Basin. International Journal of Climatology 38:3942–3962.

Bailey, R.G. 2016. Ecoregions of the United States. [online] https://www.fs.usda.gov/rmrs/ecoregions-north-america [Accessed 1 March, 2021].

Barbour, R.W., and W.H. Davis. 1969. Bats of America. The University of Kentucky Press, Lexington, Kentucky. 311pp.

Barclay, R.M.R., J. Ulmer, C.J.A. MacKenzie, M.S. Thompson, L. Olson, J. McCool, E. Cropley, and G. Poll. 2004. Variation in the reproductive rate of bats. Canadian Journal of Zoology 82:688–693.

Barclay, R.M.R., and A. Kurta. 2007. Ecology and Behavior of Bats Roosting in Tree Cavities and Under Bark. Chapter 2: pp. 17–60 *in* Bats in Forests: Conservation and Management, M.J. Lacki, J.P. Hayes, and A. Kurta, editors. The Johns Hopkins University Press, Baltimore, Maryland, 352pp.

Beever, E.A., J. O'Leary, C. Mengelt, J.M West, S. Julius, N. Green, D. Magness , L. Petes, B. Stein , A.B. Nicotra, and J.J. Hellmann. 2016. Improving conservation outcomes with a new paradigm for understanding species' fundamental and realized adaptive capacity. Conservation Letters, 9:131–137.

Belwood, J.J. 2002. Endangered bats in suburbia: observations and concerns for the future. In The Indiana bat: biology and management of an endangered species (A. Kurta and J. Kennedy, editors). Bat Conservation International, Austin, Texas, 193–198.

Bennett, V.J., and A.M. Hale. 2018. Texturizing wind turbine towers to reduce bat mortality DE-EE007033. Texas Christian University. Final Technical Report. Available at: https://www.osti.gov/servlets/purl/1502997.

Berkeley Lab. 2020. Electricity Markets and Policy. Wind Technologies Market Report. https://emp.lbl.gov/wind-technologies-market-report. Accessed 21 May 2021.

Bernard, R.F. and G.F. McCracken. 2017. Winter behavior of bats and the progression of white-nose syndrome in the southeastern United States. Ecology and Evolution 7:1487–1496. https://doi.org/10.1002/ece3.2772

Bernard, R.F., J.D. Reichard, J.T.H. Coleman, J.C. Blackwood, M.L. Verant, J.L. Segers, J.M. Lorch, J. Paul White, M.S. Moore, A.L. Russell, R.A. Katz, D.L. Lindner, R.S. Toomey, G.G. Turner, W.F. Frick, M.J. Vonhof, C.K.R. Willis, and E.H.C. Grant. 2020. Identifying research needs to inform white-nose syndrome management decisions. Conservation Science and Practice 2(8):e220.

Berthinussen, A., O.C. Richardson, and J.D. Altringham. 2021. Bat Conservation: Global Evidence for the Effects of Interventions. Conservation Evidence Series Synopses. University of Cambridge, Cambridge, UK. Available at: www.conservationevidence.com.

Bird Studies Canada, Canadian Wind Energy Association, Environment and Climate Change Canada and Ontario Ministry of Natural Resources and Forestry. 2018. Wind Energy Bird and Bat Monitoring Database Summary of the Findings from Post-construction Monitoring Reports. 56 pp. Available at: https://www.bsceoc.org/resources/wind/2018_Database_Summary_Report.pdf.

Blehert, D.S., A.C. Hicks, M.J. Behr, C.U. Meteyer, B.M. Berlowski-Zier, E.L. Buckles, J. Coleman T.H., S.R. Darling, A. Gargas, R. Niver, J.C. Okoniewski, R.J. Rudd, and W.B. Stone. 2009. Bat White-Nose Syndrome: an emerging fungal pathogen? Science, 323:227.

Bohrman, J., and D. Fecske. 2013. White-Nose Syndrome Surveillance and Summer Monitoring of Bats at Great Swamp National Wildlife Refuge, Morris County, New Jersey. A final report prepared for United States Fish and Wildlife Service, 115pp.

Boyles, J.G. 2016. Microclimate of Indiana bat hibernacula in southern Indiana. Unpublished technical report. Prepared by Eko Consulting LLC, Cobden, Illinois for U.S. Fish and Wildlife Service, Indiana Ecological Services Field Office, Bloomington, Indiana. 41 pp.

Boyles, J.G., and A.E. McKechnie. 2010. Energy Conservation in Hibernating Endotherms: Why "Suboptimal" Temperatures Are Optimal. Ecological Modelling, 221:1644–1647.

Brack V. Jr. 2007. Temperatures and Locations Used by Hibernating Bats, Including *Myotis sodalis* (Indiana Bat), in a Limestone Mine: Implications for Conservation and Management. Journal of Environmental Management, 40:739–746.

Brack V. Jr., and J.O. Whitaker. 2001. Foods of the northern myotis, *Myotis septentrionalis*, from Missouri and Indiana, with notes on foraging. Acta Chiropterologica, 3(2):203–210.

Brack, V. Jr., C.W. Stihler, R.J. Reynolds, C.M. Butchkoski, and C.S. Hobson. 2002. Effect of climate and elevation on distribution and abundance in the Midwestern United States. Pp. 21–28 in The Indiana bat: biology and management of an endangered species (A. Kurta and J. Kennedy, eds.). Bat Conservation International, Austin, Texas.

Breed, G., S. Stichter, and E. Crone. 2013. Climate-driven changes in northeastern US butterfly communities. Nature Climate Change, 3:142–145.

Briggler, J., and J. Prather. 2003. Seasonal Use and Selection of Caves by the Eastern Pipistrelle Bat (*Pipistrellus subflavus*). The American Midland Naturalist, 149(2):406–412.

Broders, H.G., G.J. Forbes, S. Woodley, and I.D. Thompson. 2006. Range Extent and Stand Selection for Roosting and Foraging in Forest-Dwelling Northern Long-Eared Bats and Little Brown Bats in the Greater Fundy Ecosystem, New Brunswick. Journal of Wildlife Management 70(5):1174–1184. 2006.

Broders, H.G., L.E. Burns, and S.C. McCarthy. 2013. First Records of the Northern Myotis (*Myotis septentrionalis*) from Labrador and Summer Distribution Records and Biology of Little Brown Bats (*Myotis lucifugus*) in Southern Labrador. The Canadian Field Naturalist, 127:266–269.

Brown, L., and C. Duguay. 2010. The response and role of ice cover in lake-climate interactions. Progress in Physical Geography, 34:671–704.

Burke, H.S. Jr. 1999. Maternity colony formation in *Myotis septentrionalis* using artificial roosts: the rocket box, a habitat enhancement for woodland bats? Bat Research News, 40:77–78.

Burles, D.W., R.M Brigham, R.A., Ring, R.A., and T.E. Reimchen. 2009. Influence of weather on two insectivorous bats in a temperate Pacific Northwest rainforest. Canadian Journal of Zoology 87:132–138.

Caceres, M.C., and M.J. Pybus. 1997. Status of the northern long-eared bat (*Myotis septentrionalis*) in Alberta. Alberta Environmental Protection, Wildlife Management Division, Wildlife Status Report No. 3, Edmonton, AB, 19pp.

Caceres, M.C., and R.M.R. Barclay. 2000. *Myotis septentrionalis*. Mammalian Species, 634:1–4.

Caire, W., R.K. LaVal, M.L. LaVal, and R. Clawson. 1979. Notes on the ecology of *Myotis keenii* (Chiroptera, Vespertilionidae) in Eastern Missouri. American Midland Naturalist, 102(2): 404–407.

Canada Energy Regulator (CER). 2019. Canada's Energy Future 2019: Energy Supply and Demand Projections to 2040. 52 pp. Available at: https://www.cer-rec.gc.ca/en/data-analysis/canada-energy-future/2019/index.html

Canada Energy Regulator (CER). 2020. Canada's Energy Future 2020: Energy Supply and Demand Projections to 2050. 104 pp. Available at: https://www.cer-rec.gc.ca/en/data-analysis/canada-energy-future/2020/canada-energy-futures-2020.pdf

Carter, T.C., and G. Feldhamer. 2005. Roost tree use by maternity colonies of Indiana bats and northern long-eared bats in southern Illinois. Forest Ecology and Management, 219:259– 268.

Cheng, T.L., A. Gerson, M.S. Moore, J.D. Reichard., J. DeSimone, C.K.R. Willis, and A.M. Kilpatrick. 2019. Higher fat stores contribute to persistence of little brown bat populations with white-nose syndrome. Journal of Animal Ecology, 88:591–600.

Cheng, T.L., J.D. Reichard, J.T.H. Coleman, T.J. Weller, W.E. Thogmartin, B.E. Reichert, A.B. Bennett, H.G. Broders, J. Campbell, K. Etchison, D.J. Feller, R. Geboy, T. Hemberger, C. Herzog, A.C. Hicks, S. Houghton, J. Humber, J.A. Kath, R.A. King, S.C. Loeb, A. Massé, K.M. Morris, H. Niederriter, G. Nordquist, R.W. Perry, R.J. Reynolds, D.B. Sasse, M.R. Scafini, R.C. Stark, C.W. Stihler, S.C. Thomas, S.C. Thomas, G.G. Turner, S. Webb, B. Westrich, and W.F. Frick. 2021. The scope and severity of white-nose syndrome on hibernating bats in North America. Conservation Biology 2021:1–12.

Cheng, T., B. E. Reichert, W. E. Thogmartin, B. J. Udell, A. M. Wiens, M. Whitby, W. Frick, J.D Reichard, and J. Szymanski. 2022. Winter Colony Count Analysis for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter D in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Chevin L.M., R. Lande, and G.M. Mace. 2010. Adaptation, plasticity, and extinction in a changing environment: Towards a predictive theory. PLOS Biology 8(4):e1000357.

Cole, W. National Renewable Energy Laboratory (personal communication, August 3, 2020).

Cole, W.S.C., N. Gates, T. Mai, and P. Das. 2020. 2020 Standard Scenarios Report: A U.S. Electricity Sector Outlook. Golden, CO: National Renewable Energy Laboratory. Available at NREL/TP-6A20-77442.

Coleman, J.T.H., and J.D. Reichard. 2014. Bat White-Nose Syndrome in 2014: A brief assessment seven years after discovery of a virulent fungal pathogen in North America. Outlooks on Pest Management, 25(6):374–377.

Cope, J.B., and S.R. Humphrey. 1972. Reproduction of the bats *Myotis Keenii* and *Pipistrellus subflavus* in Indiana. Bat Research News, 13:9–10.

Crandall K.A., O.R. Bininda-Emonds, G.M. Mace, and R.K. Wayne. 2000. Considering evolutionary processes in conservation biology. Trends in Ecology and Evolution, 15:290–295.

Cryan, P.M., and R.M.R. Barclay. 2009. Causes of bat fatalities at wind turbines: Hypotheses and predictions. Journal of Mammalogy, 90:1330–1340. https://doi.org/10.1644/09-MAMM-S-076R1.1.

Cryan, P.M., C. Meteyer, J.G. Boyles, and D.S. Blehert. 2010. Wing pathology of white-nose syndrome in bats suggests life-threatening disruption of physiology. BioMed Central Biology 8(1):135–143.

Cryan, P.M., P.M. Goressen, C.D. Hein, M.R. Schirmacher, R.H. Diehl, M.M. Huso, D.T.S. Hayman, P.D. Fricker, F.J. Bonaccorso, D.H. Johnston, K. Heist, and D. C. Dalton. 2014. Behavior of bats at wind turbines. Proceedings of the National Academy of Sciences 111(42):15126–15,131. https://doi.org/10.1073/pnas.1406672111.

Cryan, P.M., P. Marcos Gorresen, D. Dalton, S. Wolf , and F. Bonaccorso. 2016. Ultraviolet illumination as a means of reducing bat activity at wind turbines. Presented at the Wind Wildlife Research Meeting XI in Broomfield, Colorado. 2 December 2016. Available at https://www.nationalwind.org/wp-content/uploads/2017/04/83_Cryan.pdf

Davis, W.H., and H.B. Hitchcock. 1965. Biology and Migration of the Bat, *Myotis lucifugus*, in New England. Journal of Mammalogy, 46(2):296–313.

Davy, C.M., G.F. Mastromonaco, J.L. Riley, J.H. Baxter-Gilbert, H. Mayberry, and C.K.R. Willis. 2017. Conservation implications of physiological carry-over effects in bats recovering from white-nose syndrome. Conservation Biology 31(3):615–624. DOI: 10.1111/cobi.12841. Epub 2016 Dec 5. PMID: 27641049.

Deeley, S.M., and W.M. Ford. 2022. Summary Assessment of USFWS Capture Data for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter A in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

De la Cruz, J.L., R.L. Ward, and E.S. Schroder. 2018. Landscape characteristics related to use of artificial roosts by northern long-eared bats in north-central West Virginia. Northeastern Naturalist, 25(3):487–501.

Dobony, C.A., A.C. Hicks, K.E. Langwig, R.I. von Linden, J.C. Okoniewski, and R.E. Rainbolt. 2011. Little brown myotis persist despite exposure to white-nose syndrome. Journal of Fish and Wildlife Management, 2:190–195.

Dodd, L.E., E.G. Chapman, J.D. Harwood, M.J. Lacki, and L.K. Rieske. 2012. Identification of prey of *Myotis septentrionalis* using DNA-based techniques. Journal of Mammalogy, 93(4):1119–1128.

Dowling, Z.R., and D.I. O’Dell. 2018. Bat use of an island off the coast of Massachusetts. Northeastern Naturalist, 25(3):362–382.

Dwyer, P.D. 1971. Temperature regulation and cave-dwelling in bats: An evolutionary perspective. Mammalia, 35(3):424–455.

Easterla, D.A. 1968. Parturition of Keen's Myotis in Southwestern Missouri. Journal of Mammalogy, 49(4):770.

Easterling, D.R., K.E. Kunkel, J.R. Arnold, T. Knutson, A.N. LeGrande, L.R. Leung, R.S. Vose, D.E. Waliser, and M.F. Wehner. 2017. Precipitation change in the United States. In: Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., D.W.

Environment and Climate Change Canada. 2018. Recovery Strategy for the Little Brown Myotis (*Myotis lucifugus*), the Northern Myotis (*Myotis septentrionalis*), and the Tri-colored Bat (*Perimyotis subflavus*) in Canada. Species at Risk Act Recovery Strategy Series. Environment and Climate Change Canada, Ottawa. ix + 172 pp.

Environment Yukon. 2011. Yukon Bats. Government of Yukon, Environment Yukon, Whitehorse, Yukon. 22pp.

Erickson, J.L., and S.D. West. 2002. The influence of regional climate and nightly weather conditions on activity patterns of insectivorous bats. Acta Chiropterologica, 4:17–24

Erickson, W., G. Johnson, D. Young, D. Strickland, R. Good, M. Bourassa, K. Bay, and K. Sernka. 2002. Synthesis and comparison of baseline avian and bat use, raptor nesting and mortality information from proposed and existing wind developments. WEST, Inc. 2003 Central Avenue, Cheyenne, Wyoming 82001.

Erickson, R.A., W.E. Thogmartin, and J.A. Szymanski. 2014. BatTool: an R package with GUI for assessing the effect of white-nose syndrome and other take events on Myotis spp. of bats. Source Code Biol Med 9, 9. https://doi.org/10.1186/1751-0473-9-9.

Erkert, H. G. 1982. Ecological aspects of bat activity rhythms. Pages 201–242 in Ecology of bats. T. H. Kunz (ed). Plenum Press, New York, New York, USA. 425pp.

Farrow L.J., and H.G. Broders. 2011. Loss of forest cover impacts the distribution of the forest-dwelling tri-colored bat (*Perimyotis subflavus*). Mammalian Biology, 72(2):172–179. doi: 10.1016/j.mambio.2010.04.004.

Feldhamer, G.A., T.C. Carter, A.T. Morzillo, and E.H. Nicholson. 2003. Use of bridges as day roosts by bats in southern Illinois. Transactions of the Illinois State Academy of Science, 96(2): 107–112

Feldhamer, G.A., T.C. Carter, and J.O. Whitaker Jr. 2009. Prey Consumed by Eight Species of Insectivorous Bats from Southern Illinois. The American Midland Naturalist, 162(1):43– 51.

Fenton, M.B. 1969. Summer activity of *Myotis lucifugus* (Chiroptera:Vespertilionidae) at hibernacula in Ontario and Quebec. Canadian Journal of Zoology, 47(4)597–602.

Field, K.A., J.S. Johnson, T.M. Lilley, S.M. Reeder, E.J. Rogers, M.J. Behr, and D.M. Reeder. 2015. The White-Nose Syndrome transcriptome: activation of anti-fungal host responses in wing tissue of hibernating little brown myotis. PLoS Pathog 11(10):e1005168.

Field, K.A., B.J. Sewall, J.M. Prokkola, G.G. Turner, M. Gagnon, T.M. Lilley, J P. White, J. S. Johnson, C.L. Hauer, and D.M. Reeder. 2018. Effect of torpor on host transcriptomic responses to a fungal pathogen in hibernating bats. Molecular Ecology, 27: 3727–3743.

Fitch, J.H., and K.A. Shump, Jr. 1979. *Myotis keenii.* Mammalian Species, 121:1–3.

Foden, W.B., B.E. Young, H.R. Akcakaya, R.A. Garcia, A.A. Hoffmann, B.A. Stein, C.D. Thomas, C.J. Wheatley, D. Bickfod, J.A. Carr, D.G. Hole, T. G. Martin, M. Pacifici, J.W. Pearce-Higgins, P.J. Platts, P. Cisconti, J.E.M. Watson, and B. Huntley. 2018. Climate change vulnerability assessment of species. WIREs Climate Change 10(e551):1–36. https://doi.org/10.1002/wcc.551.

Forsman A. 2014. Effects of genotypic and phenotypic variation on establishment are important for conservation, invasion, and infection biology. Proceedings of the National Academy of Sciences of the United States of America 111:302–307.

Foster, R.W., and A. Kurta. 1999. Roosting ecology of the Northern bat (*Myotis septentrionalis*) and comparisons with the endangered Indiana bat (*Myotis sodalis*). Journal of Mammalogy, 80(2):659–672.

Francl, K.E., W.M. Ford, D.W. Sparks, and V. Brack. 2012. Capture and reproductive trends in summer bat communities in West Virginia: assessing the impact of white-nose syndrome. Journal of Fish and Wildlife Management 3(1):33–42. DOI: https://doi.org/10.3996/062011-JFWM-039

Frick, W.F., D.S. Reynolds, and T.H. Kunz. 2010. Influence of climate and reproductive timing on demography of little brown myotis *Myotis lucifugus*. Journal of Animal Ecology, 79:128–136.

Frick, W.F., S.J. Puechmaille, J.R. Hoyt, B.A. Nickel, K.E. Langwig, J.T. Foster, K.E. Barlow, T. Bartonička, D. Feller, A. Haarsma, C. Herzog, I. Horaček, J. Kooij, B. Petrov, R. Reynolds, L. Rodrigues, C.W. Stihler, G.G. Turner, and A.M. Kilpatrick. 2015. Disease alters macroecological patterns of North American bats. Global Ecology and Biogeography, 24:741–749.

Frick, W.F., S.J. Puechmaille, and C.K.R. Willis. 2016. White-nose syndrome in bats. Pages 245–262 *In* C.C.Voigt, and T. Kingston, eds. Bats in the Anthropocene: conservation of bats in a changing world, Springer International Publishing, Cham, Heidelberg, New York, Dordrecht, London.

Frick, W.F., T.L. Cheng, K.E. Langwig, J.R. Hoyt, A.F. Janicki, K.L. Parise, J.T. Foster, and

A.M. Kilpatrick. 2017. Pathogen dynamics during invasion and establishment of white-nose syndrome explain mechanisms of host persistence. Ecology 98: 624–631.

Fuller, N.W., J.D. Reichard, M.L. Nabhan, S.L. Fellows, L.C. Pepin, and T.H. Kunz. 2012. Free-ranging little brown myotis (*Myotis lucifugus*) heal from wing damage associated with white-nose syndrome. EcoHealth, 8:154–162.

Fuller, N.W., L.P. McGuire, E.L. Pannkuk, T. Blute, C.G. Haase, H.W. Mayberry, T. S. Risch, and C K.R. Willis. 2020. Disease recovery in bats affected by white-nose syndrome. Journal of Experimental Biology 223: jeb211912 doi: 10.1242/jeb.211912.

Funk W.C., B.R. Forester, S.J. Converse, C. Darst, and S. Morey. 2019. Improving conservation policy with genomics: A guide to integrating adaptive potential into U.S. Endangered Species Act decisions for conservation practitioners and geneticists. Conservation Genetics 20:115–134.

Gargas A., M.T. Trest, M. Christensen, T.J. Volk, and D.S. Blehert. 2009. *Geomyces destructans* sp. nov. associated with bat white-nose syndrome. Mycotaxon, 108:147–154.

Garroway, C.J., and H.G. Broders. 2007. Nonrandom association patterns at northern long-eared bat maternity roosts. Canadian Journal of Zoology, 85:956–964.

Geipel, I., M. J. Smeekes, W. Halfwerk, and R.A. Page. 2019. Noise as an informational cue for decision-making: the sound of rain delays bat emergence Journal of Experimental Biology:222, jeb192005. doi:10.1242/jeb.192005

Ghalambor C.K., J.K. McKay, S.P. Carroll, and D.N. Reznick. 2007. Adaptive versus non-adaptive phenotypic plasticity and the potential for contemporary adaptation in environments. Functional Ecology, 21:394–407.

Gilman, P. U.S. Department of Energy (personal communication, July 14, 2020).

Girder, J.F., A.L. Larson, J.A. Homyack, and M.C. Kalcounis-Rueppell. 2016. Winter activity of coastal plain populations of bat species affected by White-Nose Syndrome and wind energy facilities. PLoS ONE 11(11): e0166512. doi:10.1371/journal.pone.0166512.

Glover, A.M., and J.D. Altringham. 2008. Cave selection and use by swarming bat species. Biological Conservation, 141:1493–1504.

Goehring, H.H. 1954. *Pipistrellus subflavus obscurus, Myotis keenii*, and *Eptesicus fuscus fuscus* hibernating in a storm sewer in central Minnesota. Journal of Mammalogy, 35(3):434–436.

Government of Canada. 2020. Canadian Wind Turbine Database. Available at: https://open.canada.ca/data/en/dataset/79fdad93-9025-49ad-ba16-c26d718cc070

Griffin, D.R. 1940a. Reviewed notes on the life histories of New England cave bats. Journal of Mammalogy, 21(2):181–187.

Griffin, D.R. 1940b. Migrations of New England bats. Bulletin of the Museum of Comparative Zoology, 86(6):215–246.

Griffin, D.R. 1945. Travels of banded cave bats. Journal of Mammalogy, 26(1):15–23.

Griffith, L.A., and J.E. Gates. 1985. Food habits of cave-dwelling bats in the central Appalachians. Journal of Mammalogy, 66(3):451–460.

Grindal, S., Collard, T., Brigham, R., & Robert M. R. Barclay. 1992. The Influence of Precipitation on Reproduction by Myotis Bats in British Columbia. The American Midland Naturalist, 128(2):339–344. doi:10.2307/2426468

Grodsky, S.M., M.J. Behr, A. Gendler, D. Drake, B.D. Dieterle, R.J. Rudd, and N.L. Walrath. 2011. Investigating the causes of death for wind turbine-associated bat fatalities. Journal of Mammalogy 92:917–925.

Groves, C.R., D. B. Jensen, L.L. Valutis, K.H. Redford, M.L. Shaffer, J.M. Scott, J.V. Baumgartner, J.V. Higgins, M.W. Beck, M.G. Anderson. 2002. Planning for biodiversity conservation: putting conservation science into practice. BioScience 526:499–512.

Haavik, L. 2019. Northeastern US Forest Pests. (February, 15, 2019). U.S. Department of Agriculture, Forest Service, Climate Change Resource Center. http://www.fs.fed.us/ccrc/topics/northeastern-us-forest-pests [accessed 3/21/21].

Hall, J.S., and F.J. Brenner. 1968. Summer netting of bats at a cave in Pennsylvania. Journal of Mammalogy, 49(4):779–781.

Hallam, T.G., and P. Federico. 2012. The Panzootic White-nose Syndrome: An Environmentally Constrained Disease? Transboundary & Emerging Diseases 59:269–278.

Hammerson G.A., M. Kling, M. Harkness, M. Ormes, and B.E. Young. 2017. Strong geographic and temporal patterns in conservation status of North American bats. Biological Conservation 212:144–152.

Hanski I., and M. Gilpin, eds. 1997. Metapopulation Biology: Ecology, Genetics, and Evolution. London, UK: Academic Press. 512 pp.

Hayes, M.A., L.A. Hooton, K.L. Gilland, C. Grandgent, R.L. Smith, S.R. Lindsay, J.D. Collins, Susan M. Schumacher, P.A. Rabie, J. C. Gruver, and J. Goodrich-Mahoney. 2019. A smart curtailment approach for reducing bat fatalities and curtailment time at wind energy facilities. Ecological Applications: e01881.

Hayhoe, K., D.J. Wuebbles, D.R. Easterling, D.W. Fahey, S. Doherty, J. Kossin, W. Sweet, R. Vose, and M. Wehner, 2018: Our changing climate. *In* Impacts, Risks, and Adaptation in the United States: Fourth National Climate Assessment, Volume II [Reidmiller, D.R., C.W. Avery, D.R. Easterling, K.E. Kunkel, K.L.M. Lewis, T.K. Maycock, and B.C. Stewart (eds.)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 72–144. doi: 10.7930/NCA4.2018.CH2.

Hayman, D. T. S., J. R. C. Pulliam, J. C. Marshall, P. M. Cryan, C. T. Webb. 2016. Environment, host, and fungal traits predict continental-scale white-nose syndrome in bats. Sci. Adv. 2, e1500831.

Hefley, T.J., R.E. Russel, A.E. Ballmann, and H. Zhang. 2020. When and where: estimating the date and location of introduction for exotic pests and pathogens. Available at: https://arxiv.org/abs/2006.16982

Hein, C.D., S.B. Castleberry, and K.V. Miller. 2009. Site-occupancy of bats in relation to forested corridors. Forest Ecology and Management 257(4):1200–1207.

Hein, C., and B. Straw. 2021. Proceedings from the State of the Science and Technology for Minimizing Impacts to Bats from Wind Energy. 80 pp. Available at: https://tethys.pnnl.gov/publications/proceedings-state-science-technology-minimizing-impacts-bats-wind-energy

Hendry A.P., Kinnison M.T., Heino M., Day T, Smith T.B., Fitt G., Bergstrom CT, Oakeshott J., Jørgensen P.S., Zalucki M.P., Gilchrist G. 2011. Evolutionary principles and their practical application. Evolutioxnary Applications 4:159–183.

Henderson, L.E., and H.G. Broders. 2008. Movements and resource selection of the northern long-eared myotis (*Myotis septentrionalis*) in a forest-agriculture landscape. Journal of Mammalogy, 89(4):952–963.

Henderson, L.E., L.J. Farrow, and H.G. Broders. 2008. Intra-specific effects of forest loss on the distribution of the forest-dependent northern long-eared bat (*Myotis septentrionalis*). Biological Conservation, 141:1819–1828.

Hoffmann A.A., Sgro C.M. 2011. Climate change and evolutionary adaptation. Nature 470:479–485.

Hoen, B.D., J.E. Diffendorfer, J.T. Rand, L.A. Kramer, C.P. Garrity, and H.E. Hunt. 2018. United States Wind Turbine Database (V3.2, October 14, 2020): U.S. Geological Survey, American Clean Power Association, and Lawrence Berkeley National Laboratory data release, https://doi.org/10.5066/F7TX3DN0. Data downloaded 3 November 2020.

Holderegger R., U. Kamm, and F. Gugerli. 2006. Adaptive vs neutral genetic diversity: Implications for landscape genetics. Landscape Ecology 21:797–807.

Horn, J. W., E. B. Arnett, M. Jensen, and T. H. Kunz. 2008a. Testing the effectiveness of an experimental acoustic bat deterrent at the Maple Ridge wind farm. The Bats and Wind Energy Cooperative and Bat Conservation International, Austin, TX. 30 pp.

Horn, J.W., E.B. Arnett, and T.H. Kunz. 2008b. Behavioral responses of bats to operating wind turbines. Journal of Wildlife Management 72:123–132.

Hoyt, J.R., A.M. Kilpatrick, and K.E. Langwig. 2021. Ecology and impacts of white-nose syndrome on bats. Nature Reviews Microbiology 19:196–210.

Hoyt, J.R., K.E. Langwig, K. Sun, K.L. Parise, A. Li, Y. Wang, X. Huang, L. Worledge, H. Miller, J.P. White, H.M. Kaarakka, J.A. Redell, T. Görföl, S.A. Boldogh, D. Fukui, M. Sakuyama, S. Yachimori, A. Sato, M. Dalannast, A. Jargalsaikhan, N. Batbayar, Y. Yovel, E. Amichai, I. Natradze, W.F. Frick, J.T. Foster, J. Feng, and A.M. Kilpatrick. 2020. Environmental reservoir dynamics predict global infection patterns and population impacts for the fungal disease white-nose syndrome. Proceedings of the National Academy of Sciences 117:7255–7262.

Humphrey, S.R. 1975. Nursery roosts and community diversity of nearctic bats. Journal of Mammalogy 56(2):321–346.

Humphrey, S., A. Richter, and J. Cope. 1977. Summer Habitat and Ecology of the Endangered Indiana Bat, *Myotis sodalis*. Journal of Mammalogy, 58(3):334–346. doi:10.2307/137933.

Humphries, M., Thomas, D. & Speakman, J. 2002. Climate-mediated energetic constraints on the distribution of hibernating mammals. Nature 418, 313–316 (2002). https://doi.org/10.1038/nature00828.

Humphries, M.M., J. Umbanhowar, and K.S. McCann. 2004. Bioenergetic Prediction of Climate Change Impacts on Northern Mammals, Integrative and Comparative Biology, 44(2):152–162, https://doi.org/10.1093/icb/44.2.152

Huso, M., T. Conkling, D. Dalthorp, M. Davis, H. Smith, A. Fesnock, and T. Katzner. Relative energy production determines effect of repowering on wildlife mortality at wind energy facilities. Journal of Applied Ecology DOI: 10.1111/1365-2664.13853

Hyzy, B.A., R.E. Russel, A. Silvis, W.M. Ford, J. Riddle, and K. Russel. 2020. Investigating maternity roost selection by northern long-eared bats at three sites in Wisconsin. Endangered Species Research, 41:55–65, 2020.

Ineson, M.I., T.J. O'Shea, C.W. Kilpatrick, K.L. Parise, and J.T. Foster. 2020. Ambiguities in using telomere length for age determination in two North American bat species. Journal of Mammalogy. 101(4):958–969, 2020 DOI:10.1093/jmammal/gyaa064

Johnson, J.B., W.M. Ford, and J.W. Edwards. 2012. Roost networks of northern myotis (*Myotis septentrionalis*) in a managed landscape. Forest Ecology and Management, 266:223-231.

Johnson, J.B., J.H. Roberts, T.L. King, J.W. Edwards, W.M. Ford, and D.A. Ray. 2014. Genetic structuring of northern myotis (*Myotis septentrionalis*) at multiple spatial scales. Acta Theriologica, 59(2):223–231.

Johnson, J.S., M.R. Scafini, B.J. Sewall, and G.G. Turner. 2016. Hibernating bat species in Pennsylvania use colder winter habitats following the arrival of white-nose syndrome. Pp. 181–199 in (C.M. Butchkoski, D.M. Reeder, G.G. Turner, and H.P. Whidden, eds.) Conservation and Ecology of Pennsylvania's Bats. The Pennsylvania Academy of Sciences.

Johnson, L.N.L., B.A. McLeod, L.E. Burns, K. Arseneault, T.R. Frasier, and H.G Broders. 2015. Population genetic Structure within and among seasonal site types in the little brown bat (*Myotis lucifugus*) and the northern long-eared bat (*M. septentrionalis*). PLoS ONE 10(5): e0126309. doi:10.1371/journal.pone.0126309.

Jones, G., D.S. Jacobs, T.H. Kunz, M.R. Willig, and P.A. Racey. 2009. Carpe noctem: the importance of bats as bioindicators. Endangered Species Research 8:93–115.

Jones, G. and H. Rebelo. 2013. Responses of bats to climate change: learning from the past and predicting the future. In: Adams, R., S. Pedersen. (eds) Bat Evolution, Ecology, and Conservation. Springer, New York, NY. https://doi.org/10.1007/978-1-4614-7397-8_22.

Jordan, G. W. 2020. Status of an Anomalous Population of Northern Long-Eared Bats in Coastal North Carolina. Journal of Fish and Wildlife Management 11(2):665–678.

Keller, S., J.M. Lorch, B. Berlowski-Zier, A. Ballman, and D.S. Blehert. 2021. Analysis of Archival Specimens Confirms White-nose Syndrome in Little Brown Bats (*Myotis lucifugus*) from New York, USA, in Spring 2007. Journal of Wildlife Diseases 57:1–4.

Kerns, J., and P. Kerlinger. 2004. A study of bird and bat collision fatalities at the Mountaineer Wind Energy Center, Tucker County, West Virginia Annual Report for 2003. Unpublished report. 39 pp.

Kilpatrick, A.M., J.R. Hoyt, R.A. King, H.M. Kaarakka, J.A. Redell, J.P. White, K.E. Langwig. 2020. Impact of censusing and research on wildlife populations. Conservation Science and Practice, 2: e264. https://doi.org/10.1111/csp2.264

Kindvall O. 1996. Habitat heterogeneity and survival in a bush cricket metapopulation. Ecology 77(1):207–214.

Krochmal, A.R., and D.W. Sparks. 2007. Timing of Birth and Estimation of Age of Juvenile *Myotis septentrionalis* and *Myotis lucifugus* in West–Central Indiana. Journal of Mammalogy, 88(3):649–656.

Kunz, T.H. 1971. Reproduction of Some Vespertilionid Bats in Central Iowa. American Midland Naturalist, 86(2):477–486.

Kunz, T.H. 1973. Temporal and Spatial Components of Bat Activity in Central Iowa. Journal of Mammalogy, 54(1):14–32.

Kunz, T.H., J.A. Wrazen, and C.D. Burnett. 1998. Changes in body mass and fat reserves in pre-hibernating little brown bats (*Myotis lucifugus*), Écoscience, 5(1):8–17.

Kunz, T.H., E.B. Arnett, W.P. Erickson, A.R. Hoar, G.D. Johnson, R.P. Larkin, M.D. Strickland, R.W. Thresher, and M. D Tuttle. 2007. Ecological impacts of wind energy development on bats: questions, research needs, and hypotheses. Frontiers in Ecology and Environment.5(6):315–324.

Kurta, A., J. Caryl, and T. Tipps. 1997. Bats and Tippy Dam: species composition, seasonal use, and environmental parameters. Michigan Academician, 29(4):473–490.

Kurta, A. 2005. Roosting ecology and behavior of Indiana bats (*Myotis sodalis*) in summer. Pp. 29–42 in K.C. Vories and A. Harrington (eds.), Proceedings of the Indiana bat and coal mining: a technical interactive forum. Office of Surface Mining, U.S. Department of the Interior, Alton, IL.

Kurta, A. 2013. Peer review of the 12-month Find on the Eastern Small-footed Bat and Northern Long-eared Bat. (dated 11/12/2013).

Kurta, A., and S.M. Smith. 2020. Changes in Population Size and Clustering Behavior of Hibernating Bats in the Upper Peninsula of Michigan After Arrival of White-Nose Syndrome. Northeastern Naturalist 27:763–772.

Kurta, A., R.W. Foster, B.A. Daly, A.K. Wilson, R.M. Slider, C.D. Rockey, J.M. Rockey, B.L. Long, G.G. Auteri, J.D. Collins, J.P. White, H.M. Kaarakka, J.A. Redell, and D.M. Reeder. 2020. Exceptional Longevity in Little Brown Bats Still Occurs, despite Presence of White-Nose Syndrome. Journal of Fish and Wildlife Management 11:583–587.

Lacki, M.J. 2013. Peer review of the Proposed Listing of the Northern Long-Eared Bat (*Myotis septentrionalis*) as Endangered. (dated 10/29/2013).

Lacki, M.J. and J.H. Schwierjohann. 2001. Day-roost characteristics of northern bats in mixed mesophytic forest. The Journal of Wildlife Management, 65(3):482–488.

Lacki, M.J., S.K. Amelon, and M.D. Baker. 2007. Foraging ecology of bats in forests. Pp. 83–127 In Lacki, M.J., J.P. Hayes, and A. Kurta eds.. 2007. Bats in forests: conservation and management. John Hopkins University Press, Baltimore, Maryland, USA.

Langwig, K.E., W.F. Frick, J.T. Bried, A.C. Hicks, T.H. Kunz, and A. Marm Kilpatrick. 2012. Sociality, density-dependence and microclimates determine the persistence of populations suffering from a novel fungal disease, white-nose syndrome. Ecology Letters 15(9):1050–1057.

Langwig, K.E., W.E. Frick, R. Reynolds, K.L. Parise, K.P. Drees, J.R. Hoyt, T.L. Cheng, T.H. Kunz, J.T. Foster, and A.M. Kilpatrick. 2015a. Host and pathogen ecology drive the seasonal dynamics of a fungal disease, white-nose syndrome. Proceedings of the Royal Society B 282(1799):1–7.

Langwig, K.E., J. Voyles, M.Q. Wilber, W.F. Frick, K.A. Murray, B.M. Bolker, J.P. Collins, T.L. Cheng, M.C. Fisher, J.R. Hoyt, D.L. Lindner, H.I. McCallum, R. Puschendorf, E.B. Rosenblum, M. Toothman, C.K.R. Willis, C.J. Briggs, and A.M. Kilpatrick. 2015b. Context-dependent conservation responses to emerging wildlife diseases. Frontiers in Ecology and the Environment 13(4):195–202.

Lankau R., P.S. Jørgensen, D.J. Harris, and A. Sih. 2011. Incorporating evolutionary principles into environmental management and policy. Evolutionary Applications 4:315–325.

LaVal, R.K., R.L. Clawson, M.L. LaVal and W. Caire. 1977. Foraging Behavior and Nocturnal Activity Patterns of Missouri Bats, with Emphasis on the endangered species *Myotis grisescens* and *Myotis sodalis.* Journal of Mammalogy, 58(4):592–599.

Lawson M., D. Jenne, R. Thresher, D. Houck, J. Wimsatt, and B. Straw. 2020. An investigation into the potential for wind turbines to cause barotrauma in bats. PLoS ONE 15(12): e0242485. https://doi.org/10.1371/journal.pone.0242485

Lee, Y.F. and G.F. McCraken. 2004. Flight activity and food habits of three species of Myotis bats (Chiroptera: Vespertilionidae) in sympatry. Zool. Studies. 43:589–597.

Lindner, D.L., A. Gargas, J.M. Lorch, M.T. Banik, J. Glaeser, T.H. Kunz, and D.S. Blehert. 2011. DNA-based detection of the fungal pathogen *Geomyces destructans* in soils from bat hibernacula. Mycologia 103(2):241–246.

Linton, D.M., and D.W. MacDonald. 2018. Spring weather conditions influence breeding phenology and reproductive success in sympatric bat populations. Journal of Animal Ecology 87:1080–1090.

Loeb, S. C., and E. A. Winters. 2012. Indiana bat summer maternity distribution: effects of current and future climates. Ecology and Evolution, 3:103–114.

Lorch, J.M., C.U. Meteyer, M.J. Behr, J.G. Boyles, P.M. Cryan, A.C. Hicks, A.E. Ballmann, J.T.H. Coleman, D.N. Redell, D.M. Reeder, and D.S. Blehert. 2011. Experimental infection of bats with *Geomyces destructans* causes white-nose syndrome. Nature 480:376–379.

Lorch, J.M., L.K. Muller, R.E. Russell, M. O'Connor, D.L. Lindner, and D.S. Blehert. 2013. Distribution and Environmental Persistence of the Causative Agent of White-Nose Syndrome, *Geomyces destructans*, in Bat Hibernacula of the Eastern United States. Applied and Environmental Microbiology 79(4): 1293–1301; DOI: 10.1128/AEM.02939-12

Lorch, J.M., J.M. Palmer, D.L. Linder, A.E. Ballman, K.G. George, K. Griffin, S. Knowles, J.R. Huckabee, K.H. Haman, C.D. Anderson, P.A. Becker, J.B. Buchanan, J.T. Foster, and D.S. Blehert. 2016. First detection of white-nose syndrome in western North America. mSphere DOI**:** 10.1128/mSphere.00148-16

Lowe, A.J. 2012. Swarming Behaviour and Fall Roost-Use of Little Brown (*Myotis lucifugus*), and Northern Long-Eared Bats (*Myotis septentrionalis*) in Nova Scotia, Canada. Master's Thesis. St. Mary's University, Halifax, Nova Scotia, Canada. 88pp.

Lucan, R.K., M. Weiser, and V. Hanak. 2013. Contrasting effects of climate change on the timing of reproduction and reproductive success of a temperate insectivorous bat. Journal of Zoology 290:151–159.

Lundy, M., I. Montgomery, and J. Russ. 2010. Climate change-linked range expansion of Nathusius' pipistrelle bat, *Pipistrellus nathusii* (Keyserling & Blasius, 1839). Journal of Biogeography 37:2232–2242.

Maher, S.P., A.M. Kramer, J.T. Pulliam, M.A. Zokan, S.E. Bowden, H.D. Barton, K. Magori, and J.M. Drake. 2012. Spread of white-nose syndrome on a network regulated by geography and climate. Nature Communications 3:1306.

Mangal, M., and Tier C. 1993. A simple direct method for finding persistence times of populations and application to conservation problems. Proceedings of the National Academy of Sciences of the United States of America 90:1083–1086.

Martínková, N., J. Pikula, J. Zukal, V. Kovacova, H. Bandouchova, T. Bartonička, A.D. Botvinkin, J. Brichta, H. Dundarova, T. Kokurewicz, N.R. Irwin, P. Linhart, O.L. Orlov, V. Piacek, P. Škrabánek, M.P. Tiunov and A. Zahradníková Jr. 2018. Hibernation temperature-dependent *Pseudogymnoascus destructans* infection intensity in Palearctic bats. Virulence 9:1734–1750. DOI: 10.1080/21505594.2018.1548685.

Master, L.L., D. Faber-Langendoen, R. Bittman, G.A. Hammerson, B. Heidel, L. Ramsay, K. Snow, A. Teucher, and A. Tomaino. 2012. NatureServe Conservation Status Assessments: Factors for Evaluating Species and Ecosystem Risk. NatureServe, Arlington, VA. 64 pp.

McClure, M. L., D. Crowley, C. G. Haase, L. P. McGuire, N. W. Fuller, D. T. S. Hayman, C. L. Lausen, R. K. Plowright, B. G. Dickson, and S. H. Olson. 2020. Linking surface and subterranean climate: implications for the study of hibernating bats and other cave dwellers. Ecosphere 11:e03274. 10.1002/ecs2.3274.

McGuire, L.P., H.W. Mayberry, and C.K.R. Willis. 2017. White-nose syndromeincreases torpid metabolic rate and evaporative water loss in hibernating bats. American Journal of Physiology; Regulatory, Integrative and Comparative Physiology 313: R680–R686.

McNab, B. 1982. Evolutionary Alternatives in Physiological Ecology of Bats. Pp. 151–200 In (Kunz, T.H., ed.) Ecology of Bats. Plenum Press, New York.

Meierhofer, M. B., J. S. Johnson, K. A. Field, S. S. Lumadue, A. Kurta, J. A. Kath, and D. M. Reeder. 2018. Bats recovering from white-nose syndrome elevate metabolic rate during wing-healing in spring. Journal of Wildlife Diseases 54(3): 480-490.

Menzel, M.A., S.F. Owen, W.M. Ford, J.W. Edwards, P.B. Wood, B.R. Chapman, and K.V.

Miller. 2002. Roost tree selection by northern long-eared bat (*Myotis septentrionalis*) maternity colonies in an industrial forest of the central Appalachian mountains. Forest Ecology and Management, 155:107–114.

Meteyer, C.U., E.L. Buckles, D.S. Blehert, A.C. Hicks, D.E. Green, V. Shearn-Bochsler, N.J. Thomas, A. Gargas, and M.J. Behr. 2009. Histopathologic criteria to confirm white-nose syndrome in bats (cited as: Pathology criteria for confirming white-nose syndrome in bats). Journal of Veterinary Diagnostic Investigation 21:411–414.

Meteyer, C.U., D. Barber, and J.N. Mandl. 2012. Pathology in euthermic bats with white-nose syndrome suggests a natural manifestation of immune reconstitution inflammatory syndrome. Virulence 3:583–588.

Miller-Butterworth, C.M., M.J. Vonhof, J. Rosenstern, G.G. Turner, and A.L. Russell. 2014. Genetic Structure of Little Brown Bats (*Myotis lucifugus*) Corresponds With Spread of White-Nose Syndrome Among Hibernacula. Journal of Heredity 105:354–364.

Moore, M.S., J.D. Reichard, T.D. Murtha, B. Zahedi, R.M. Fallier, and T.H. Kunz. 2011. Specific Alterations in Complement Protein Activity of Little Brown Myotis (*Myotis lucifugus*) Hibernating in White-Nose Syndrome Affected Sites. PLoS ONE (11):e27430.

Muller, L.K., J.M. Lorch, D.L. Lindner, M. O'Connor, A. Gargas, and D.S. Blehert. 2013. Bat white-nose syndrome: a real-time TaqMan polymerase chain reaction test targeting the intergenic spacer region of Geomyces destructans, Mycologia 105:(2):253–259. DOI: 10.3852/12-242

Mumford R.E., and J.B. Cope. 1964. Distribution and status of the chiroptera of Indiana. American Midland Naturalist, 72(2):473–489.

Murray, S. W., and A. Kurta. 2004. Nocturnal activity of the endangered Indiana bat (*Myotis sodali*s). Journal of Zoology 262:197–206.

Muthersbaugh, M.S., W.M. Ford, A.Silvis, and K.E.Powers. 2019. Activity Patterns of Cave-Dwelling Bat Species during Pre-Hibernation Swarming and Post-Hibernation Emergence in the Central Appalachians. Diversity, 11(9):159.

Nagorsen, D.W. and R.M. Brigham. 1993. Bats of British Columbia. Royal British Columbia Museum, Victoria, and the University of British Columbia Press, Vancouver. 164 pp.

National Conference of State Legislatures (NCSL). 2021. State Renewable Portfolio Standards and Goals. <https://www.ncsl.org/research/energy/renewable-portfolio-standards.aspx >. Accessed 12 May 2021.

National Research Council 2007. Environmental Impacts of Wind-Energy Projects. Washington, DC: The National Academies Press. https://doi.org/10.17226/11935.

Neubaum, D. J., and Siemers, J. L. 2021. Bat swarming behavior among sites and its potential for spreading white-nose syndrome. Ecology 00(00):e03325. 10.1002/ecy.3325

Nicholson, C.P., R.D. Tankersley Jr., J.K. Fiedler, N.S. Nicholas. 2005. Assessment and Prediction of bird and bat mortality at wind energy facilities in the southeastern United States. Tennessee Valley Authority, Knoxville, TN. 65 pp.

Nicotra, A.B., E.A. Beever, A.L. Robertson, G.E. Hofmann, J. O'Leary. 2015. Assessing the components of adaptive capacity to improve conservation and management efforts under global change. Conservation Biology 29:1268–1278.

North American Bat Monitoring Program (NABat 2020) Database v5.3.11 (Provisional Release): U.S. Geological Survey. Accessed 2020-11-18. NABat Request Number 10. https://doi.org/10.5066/P9UXA6CF.

North American Bat Monitoring Program (NABat 2021) Database v6.0.12 (Provisional Release): U.S. Geological Survey. Accessed 2021-02-10. NABat Request Number 12. https://doi.org/10.5066/P9UXA6CF.

Ofori, B., Stow, A., Baumgartner, J., & Beaumont, L. 2017. Influence of adaptive capacity on the outcome of climate change vulnerability assessment. Scientific Reports 7:1–12. [12979]. DOI: 10.1038/s41598-017-13245-y.

Oliver T., D.B. Roy, J.K. Hill, T. Brereton, C.D. Thomas. 2010. Heterogeneous landscapes promote population stability. Ecology Letters 13:473–484.

Olivera-Hyde, M., A. Silvis, E.M. Hallerman, W.M. Ford, and E.R. Brittzke. 2019 Relatedness within and among northern long-eared bat (*Myotis septentrionalis*) colonies at a local scale. Canada Journal of Zoology. 97: 724–735.

Osborn, S. California Department of Fish and Game (personal communication April 30, 2021).

O'Shea, T.J., P.M. Cryan, D.T.S. Hayman, R.K. Plowright, and D.G. Streicker. 2016. Mammal Review. 46(3): 175–190.

Oswalt, S.N., W.B. Smith, P.D. Miles, and S.A. Pugh. 2019. Forest Resources of the United States, 2017: a technical document supporting the Forest Service 2020 RPA Assessment. Gen. Tech. Rep. WO-97. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office. 223 p. https://doi.org/10.2737/WO-GTR-97

Owen, S.F., M.A. Menzel, W.M. Ford, J.W. Edwards, B.R. Chapman, K.V. Miller, and P.B. Wood. 2002. Roost tree selection by maternal colonies of Northern long-eared Myotis in an intensively managed forest. USDA Forest Service. Newtown Square, Pennsylvania. 10 pp.

Owen, S. F., M.A. Menzel, W. M. Ford, B. R. Chapman, K. V. Miller, J. W. Edwards, P. Bohall Wood. 2003. Home-Range Size and Habitat Used by the Northern Myotis (*Myotis septentrionalis*). American Midland Naturalist 150(2): 352–359.

Parmesan, C., N. Ryrholm, C. Stefanescu, J.K. Hillk, C.D. Thomas, H. Descimon, B. Huntley, L. Kaila, J. Kullberg, T. Tammaru, W.J. Tennent, J.A. Thomas, and M. Warren. 1999. Poleward shifts in geographical ranges of butterfly species associated with regional warming. Nature 399:579–583.

Parsons, S., K.L. Lewis, and J.M. Psyllakis. 2003. Relationships between roosting habitat of bats and decay of aspen in the sub-boreal forests of British Columbia. Forest Ecology and Management 177:559–570.

Patriquin, K.J., M.L. Leonard, H.G. Broders, and C.J. Garroway. 2010. Do social networks of female northern long-eared bats vary with reproductive period and age? Behavioral Ecology and Sociobiology 84:899–913.

Patriquin, K.J., M.L. Leonard, H.G. Broders, W.M. Ford, E.R. Britzke, and A. Silvis. 2016. Weather as a proximate explanation for fission–fusion dynamics in female northern long-eared bats. Animal Behaviour 122:47–57.

Pauli, B. P., P. A. Zollner, G. S. Haulton, G. Shao, and G. Shao. 2015. The simulated effects of timber harvest on suitable habitat for Indiana and northern long-eared bats. Ecosphere 6(4):58.

Pearson, E.W. 1962. Bats hibernating in silica mines in southern Illinois. Journal of Mammalogy, 43(1):27–33.

Perry, R.W., and R.E. Thill. 2007. Roost selection by male and female northern long-eared bats in a pine-dominated landscape. Forest Ecology and Management 247:220–226.

Perry. R.W. 2013. A review of factors affecting cave climates for hibernating bats in temperate North America. Environmental Reviews 21:28–39.

Peste, F., A. Paula, L. P. da Silva, J. Bernardino, P. Pereira, M. Mascarenhas, H. Costa, J. Vieira, C. Bastos, C. Fonseca, and M. J. R. Pereira. 2015. How to mitigate impacts of wind farms on bats? A review of potential conservation measures in the European context. Environmental Impact Assessment Review 51:10–22.

Platt, R.N., B.C. Faircloth, K. A. M. Sullivan, T. J. Kiernan, T. C. Glenn, M. W. Vandewege, T. E. Lee, R. J. Baker, R. D. Stevens, and D. A. Ray. 2018. ConflictingEvolutionary Histories of the Mitochondrial and Nuclear Genomes in New World Myotis Bats. Systematic Biology 67(2):236–249.

Plumb, G. and P. Budde 2011. Unpublished data from U.S. Fish and Wildlife Service Region 3 data request regarding status of 7 cave bats species (sent 03/30/2011).

Puechmaille, S.J., G. Wibbelt, V. Korn, H. Fuller, F. Forget, K. Mühldorfer, A. Kurth, W. Bogdanowicz, C. Borel, T. Bosch, T. Cherezy, M. Drebet, T. Görföl, A. Haarsma, F. Herhaus, G. Hallart, M. Hammer, C. Jungmann, Y. Le Bris, L. Lutsar, M. Masing, B. Mulkens, K. Passior, M. Starrach, A. Wojtaszewski, U. Zöphel, and E.C. Teeling. 2011. Pan-European Distribution of White-Nose Syndrome Fungus (Geomyces destructans) Not Associated with Mass Mortality. PLoS ONE 6(4):e19167.

Rabie, P.A., D. Riser-Espinoza, J. Studyvin, D. Dalthorp, and M. Huso. 2021. Performance of the GenEst mortality estimator compared to the Huso and Shoenfeld Estimators. American Wind Wildlife Institute Technical Report. 10 March 2021. 28 pp. Available at https://awwi.org/wp-content/uploads/2021/04/Performance-of-the-GenEst-Statistical-Mortality-Estimator.pdf

Racey, P. 1982. Ecology of Bat Reproduction. pp 57–104 in Ecology of Bats (ed Kunz, T.). Boston University, Boston, MA.

Racey, P.A., and S.M. Swift. 1981. Variations in gestation length in a colony of pipistrelle bats (Pipistrellus pipstrellus) from year to year. Journal of Reproductive Fertility 61:123–129.

Raesly, R.L., and J.E. Gates. 1987. Winter habitat selection by north temperate cave bats. American Midland Naturalist, 118(1):15–31.

Randall, J. and H.G. Broders. 2014. Identification and characterization of swarming sites used by bats in Nova Scotia, Canada. Acta Chiropterologica, 16:109–116.

Ratcliffe, J.M. and J.W. Dawson. 2003. Behavioral flexibility: the little brown bat, *Myotis lucifugus*, and the northern long-eared bat, *M. septentrionalis*, both glean and hawk prey. Animal Behaviour, 66:847–856.

Rebelo, H., P. Tarroso, and G. Jones. 2010.Predicted impact of climate change on European bats in relation to their biogeographic patterns.Global Change Biology 16:561–576.

Redford, K.H., G. Amato, J. Baillie, P. Beldomenico, E.L. Bennett, N. Clum, R. Cook, G. Fonseca, S. Hedges, F. Launay, S. Lieberman, G.M. Mace, A. Murayama, A. Putnam, J.G. Robinson, H. Rosenbaum, E.W. Sanderson, S.N. Stuart, P. Thomas, and J. Thorbjarnarson. 2011. What does it mean to conserve a (vertebrate) species? BioScience, 61:39–48.

Reeder, D. M., C. L. Frank, G. G. Turner, C. U. Meteyer, A. Kurta, E. R. Britzke, M. E. Vodzak, S. R. Darling, C. W. Stihler, A. C. Hicks, R. Jacob, L. E. Grieneisen, S. A. Brownlee, L. K. Muller, and D. S. Blehert. 2012. Frequent arousal from hibernation linked to severity of infection and mortality in bats with white-nose syndrome. Plos One 7:e38920. <http://dx.doi.org/10.1371%2Fjournal.pone.0038920>.

Reichard, J. D., and T. H. Kunz. 2009. White-nose syndrome inflicts lasting injuries to the wings of little brown myotis (*Myotis lucifugus*). Acta Chiropterologica 11:457–464. <https://doi.org/10.3161/150811009X485684>.

Reynolds, H.T., T. Ingersoll, and H.A. Barton. 2015. Modeling the environmental growth of

Pseudogymnoascus destructans and its impact on the white-nose syndrome epidemic. Journal of Wildlife Diseases 51: 318–331.

Rodenhouse, N.L., L.M. Christenson, D. Parry, and L.E. Green. 2009.Climate change effects on native fauna of Northeastern forests. Canadian Journal of Forest Research 39:249–263.

Rollins, K.E., D.K. Meyerholz, G.D. Johnson, A.P. Capparella and S.S. Loew. 2012. A Forensic Investigation into the Etiology of Bat Mortality at a Wind Farm: Barotrauma or Traumatic Injury? Veterinary Pathology 49: 362–371.

Romano, W.B., J.R. Skalski, R.L. Townsend, K.W. Kinzie, K.D. Coppinger, and M.F. Miller. 2019. Evaluation of an acoustic deterrent to reduce bat mortalities at an Illinois wind farm. Wildlife Society Bulletin 43(4):608–618.

Ruckelhaus M., P. McElhany, and. M.J. Ford. 2002. Recovering species of conservation concern: Are populations expendable? Pp. 305–329 in Kareiva P, Levin S., eds. The Importance of Species: Perspectives on Expendability and Triage. Princeton, NJ: Princeton University Press.

Sasse, D.B., and P.J. Pekins. 1996. Summer roosting ecology of northern long-eared bats *(Myotis septentrionalis)* in the white mountain national forest. Bats and Forests Symposium October 1995, Victoria, British Columbia, Canada, pp. 91–101.

Sasse, D.B., M. L. Caviness, M. J. Harvey, J. L. Jackson, and P. N. Jordan. 2014. Journal of the Arkansas Academy of Science 68(32):170–173.

Scheel, D, T.L.S. Vincent, and G.N. Cameron. 1996. Global Warming and the Species Richness of Bats in Texas. Conservation Biology 10(2)452–464.

Schirmacher, M.S., A. Prichard, T. Mabee, and C.D. Hein. 2018. Evaluating a Novel Approach to Optimize Operational Minimization to Reduce Bat Fatalities at the Pinnacle Wind Farm, Mineral County, West Virginia, 2015. An annual report submitted to NRG Energy and the Bats and Wind Energy Cooperative. Bat Conservation International. Austin, Texas, USA.

Schirmacher, M. R., 2020. Evaluating the Effectiveness of Ultrasonic Acoustic Deterrents in Reducing Bat Fatalities at Wind Energy Facilities." U.S. Department of Energy, DE-EE0007036.

Sgro C.M., Lowe A.J., Hoffmann A.A. 2011. Building evolutionary resilience for conserving biodiversity under climate change. Evolutionary Applications 4:326–337.

Shaffer ML, Stein BA. 2000. Safeguarding our precious heritage. Pp. 301–321 in Stein BA, Kutner LS, Adams JS, eds. Precious Heritage: The Status of Biodiversity in the United States. New York: Oxford University Press.

Sheffield, S.R., J.H. Shaw, G.A. Heidt, and L.R. McClenaghan, 1992. Guidelines for the protection of bat roosts. Journal of Mammalogy 73(3):707–710.

Sherwin, H.A., W.I. Montgomery, and M.G. Lundy. 2013. The impact and implications of climate change for bats. Mammal Review. 18 pp.

Silvis, A., W.M. Ford, E.R. Britzke, N.R. Beane, and J.B. Johnson. 2012. Forest succession and maternity day-roost selection by *Myotis septentrionalis* in a mesophytic hardwood forest. International Journal of Forestry Research 2012:1–8.

Silvis, A., W.M. Ford,,E.R. Britzke, and J.B. Johnson. 2014. Association, roost use and simulated disruption of *Myotis septentrionalis* maternity colonies. Behavioural Processes, 103:283–290.

Silvis, A., W.M. Ford, and E.R. Britzke. 2015. Effects of Hierarchical Roost Removal on Northern Long-Eared Bat (*Myotis septentrionalis*) Maternity Colonies. PLoS ONE, 10(1):1–17.

Silvis, A., R.W. Perry, and W.M. Ford. 2016. Relationships of three species of bats impacted by White-Nose Syndrome to forest condition and management. Gen. Tech. Rep. SRS–214. Asheville, NC: U.S. Department of Agriculture Forest Service, Southern Research Station. 48 p.

Simmons, N.B. 2005. Order Chiroptera: Subfamily Myotinae. p. 516 *in* Mammal species of the world: a taxonomic and geographic reference, D.E. Wilson and D.M. Reeder, editors. The John Hopkins University Press, Baltimore, Maryland. 2000 pp.

Solick, D., D. Pham, K. Nasman, and K. Bay. 2020. Bat activity rates do not predict bat fatality rates at wind energy facilities. Acta Chiroptera 22(1):135–146.

Spanjer, G.R., and M.B. Fenton. 2005. Behavioral responses of bats to gates at caves and mines. Wildlife Society Bulletin, 33:1101–1112.

Sparks, J.K., B.J. Foster, and D.W. Sparks. 2004. Utility pole used as a roost by a northern myotis, *Myotis septentrionalis*. Bat Research News, 45:94.

Spielman, D., B.W. Brook, and R. Frankham. 2004. Most species are not driven to extinction before genetic factors impact them. PNAS 101(42): 15261–15264.

Stratton, C., and K.M. Irvine. 2022. Summertime Analysis Statistical Report for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter B in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Straw, B. U.S. Geological Survey (personal communication, April 6, 2021).

Suriano, Z.J., D.A. Robinson, and D.J. Leathers. 2019. Changing snow depth in the Great Lakes basin (USA): Implications and trends. Anthropocene 26:1–11.

Taucher, J., T. Librandi Mumma, and W. Capouillez. 2012. Pennsylvania Game Commission Wind energy Voluntary Cooperation Agreement Third Summary Report. 79 pp.

Teplitsky C., Robinson M.R., Merilä J. 2014. Evolutionary potential and constraints in wild populations. Pp. 190–208 in Charmantier A, Garant D, and Kruuk LEB, eds. Quantitative Genetics in the Wild. Oxford, UK: Oxford University Press.

Thapa V, G.G. Turner, and M.J. Roossinck. 2021. Phylogeographic analysis of Pseudogymnoascus destructans partitivirus-pa explains the spread dynamics of white-nose syndrome in North America. PLoS Pathog 17(3): e1009236.

Thomas, D.W., and F. Geiser. 1997. Periodic arousals in hibernating mammals: is evaporative water loss involved? Functional Ecology 11:585–591.

Thomas, C., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. Erasmus, M. Ferreira de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega-Huerta, A. Townsend Peterson13, O.L. Phillips, and S.E. Williams. 2004. Extinction risk from climate change. Nature 427:145–148.

Thurman, L, B. Stein, E. Beever, W. Foden, S. Geange, N. Green, J. Gross, D. Lawrence, O. LeDee, J. O'Leary, J. Olden, L. Thompson, and B. Young. Persist in place or shift in space? Evaluating the adaptive capacity of species to climate change. Frontiers in Ecology and the Environment 18(9):520–528.

Timpone, J.C., J.G. Boyles, K.L. Murray, D.P. Aubrey, and L.W. Robbins. 2010. Overlap in roosting habits of Indiana Bats (*Myotis sodalis*) and northern bats (*Myotis septentrionalis*). American Midland Naturalist 163:115–123.

Turner, G.G., and D.M. Reeder. 2009. Update of white-nose syndrome in bats, September 2009. Bat Research News 50(3):47–53.

Turner, G.G., D.M. Reeder, and J. Coleman T.H. 2011. A Five-year Assessment of Mortality and Geographic Spread of White-nose Syndrome in North American Bats and a Look to the Future. Bat Research News 52(2):13–27.

Tuttle, M.D. and D.E. Stevenson. 1978. Variation in the cave environment and its biological implications. Pp. 108–121, in National Cave Management Symposium Proceedings. Albuquerque: Adobe Press, 1977.

Udell, B.J., B.R. Straw, J. Szymanski, M. Seymour, J. Wong, A.M. Wiens, B.E. Reichert, J.E. Diffendorfer, L. Kramer, and Z. Ancona. 2022. Spatially explicit method of predicting bat wind take at wind facilities and allocating it among winter hibernacula populations. Appendix E-1 in Straw, B.R., J.A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator

Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU.

U.S. Department of Agriculture, Forest Service (USFS). 2012. Future of America's forest and rangelands: Forest Service 2010 resources planning act assessment. Gen. Tech. Rep. WO-87. Washington, DC. 198 pp.

U.S. Department of Agriculture, Forest Service (USFS). 2013. Adaptive Management Strategy for White-nose Syndrome, Available at: http://www.fs.usda.gov/detail/r2/home/?cid=stelprdb5319926 Accessed 10/21/2020

U.S. Department of Agriculture, Forest Service (USFS). 2014. U.S. Forest Resource Facts and Historical Trends. Forest Service, FS-1035, 64 pp.

U.S. Department of Agriculture, Forest Service (USFS). 2016. Future of America's Forests and Rangelands: Update to the 2010 Resources Planning Act Assessment. Gen. Tech. Report WO-GTR-94. Washington, DC. 250 p.

U.S. Department of Energy (USDOE). 2015. Wind Vision: A New Era for Wind Power in the United States. 289 pp. Available at: https://www.energy.gov/sites/prod/files/WindVision_Report_final.pdf

U.S. Energy Information Administration (USEIA). 2020. Annual Energy Outlook 2020 with projections to 2050. 161 pp. Available at: https://www.eia.gov/outlooks/aeo/pdf/AEO2020%20Full%20Report.pdf

U.S. Fish and Wildlife Service (USFWS). 2007. Indiana Bat (*Myotis sodalis*) Draft Recovery Plan: First Revision. U.S. Fish and Wildlife Service, Fort Snelling, Minnesota, 258 pp.

U.S. Fish and Wildlife Service (USFWS). 2011. A national plan for assisting states, federal agencies, and tribes in managing white-nose syndrome in bats. U.S. Fish and Wildlife Service. Hadley, MA. 21 pp.

U.S. Fish and Wildlife Service (USFWS). 2016. Draft Midwest Wind Energy Multi-species Habitat Conservation Plan in collaboration with the States of Iowa, Illinois, Indiana, Michigan, Minnesota, Missouri, and Wisconsin and the American Wind Energy Association. Public Review Draft. April 2016. Available at: https://www.fws.gov/midwest/endangered/permits/hcp/r3wind/DraftHCPandEIS.html

U.S. Fish and Wildlife Service (USFWS). 2015. Endangered and Threatened Wildlife and Plants; Threatened Species Status for the Northern Long-Eared Bat With 4(d) Rule; Final Rule and Interim Rule. Federal Register 80(63): 17974–18033.

van Zyll de Jong, C.G. 1979. Distribution and systematic relationships of long-eared Myotis in western Canada. Canadian Journal of Zoology, 57:987–994.

van Zyll de Jong, C.G. 1985. Handbook of Canadian mammals. National Museums of Canada, Ottawa, Canada. 212pp.

Verant, M.L., J.G. Boyles, W. Waldrep Jr., G. Wibbelt, and D.S. Blehert. 2012. Temperature-Dependent Growth of *Geomyces destructans*, the Fungus That Causes Bat White-Nose Syndrome. PLoS ONE 7(9):e46280.

Verant, M., C. Meteyer, J. Speakman, P. Cryan, J. Lorch, and D.S. Blehert. 2014. White-nose syndrome initiates a cascade of physiologic disturbances in the hibernating bat host. BMC Physiology 14(1):10.

Voigt, C.C., K. Schneeberger, S. L. Voigt-Heucke, and D. Lewanzik. 2011. Rain increases the energy cost of bat flight.Biological Letters 7:793–795

Vose, R.S., D.R. Easterling, K.E. Kunkel, A.N. LeGrande, and M.F. Wehner. 2017. Temperature changes in the United States. In: Climate Science Special Report: Fourth National Climate Assessment, Volume I [Wuebbles, D.J., D.W. Fahey, K.A. Hibbard, D.J. Dokken, B.C. Stewart, and T.K. Maycock (eds.)]. U.S. Global Change Research Program, Washington, DC, USA, pp. 185–206.

Warnecke, L., J.M. Turner, T.K. Bollinger, J.M. Lorch, V. Misra, P.M. Cryan, G. Wibbelt, D.S. Blehert, and C.K.R. Willis. 2012. Inoculation of bats with European *Geomyces destructans* supports the novel pathogen hypothesis for the origin of white-nose syndrome. Proceedings of the National Academy of Sciences 109:6999–7003.

Warnecke, L., J.M. Turner, T.K. Bollinger, V. Misra, P.M. Cryan, D.S. Blehert, G. Wibbelt, and C.K.R. Willis. 2013. Pathophysiology of white-nose syndrome in bats: a mechanistic model linking wing damage to mortality. Biology Letters 9:20130177.

Weaver, S.P., C.D. Hein, T.R. Simpson, J.W. Evans, and I. Castro-Arellano. 2020. Ultrasonic deterrents significantly reduce bat fatality at wind turbines. Global Ecology and Conservation 24:e01099.

Webb, D.R., and J.R. King. 1984. Effects of wetting on insulation of bird and mammal coats. Journal of Thermal Biology 9(3)189–191.

Webb, P. I., J. R. Speakman, P. A. Racey. 1996. A review of the temperatures at which bats hibernate. Candadian Journal of Zoology https://doi.org/10.1139/z96-087

Whitaker, J.O., and L.J. Rissler. 1992. Seasonal activity of bats at copperhead cave. Proceedings of the Indiana Academy of Science, 101:127–134.

Whitaker, J.O., and W.J. Hamilton. 1998. Order Chiroptera: Bats. Chapter 3: pp.89–102 *in* Mammals of the eastern United States, Third Edition, Comstock Publishing Associates, a Division of Cornell University Press, Ithaca, New York, 608pp.

Whitaker, J.O., and R.E. Mumford. 2009. Northern Myotis. pp. 207–214. In Mammals of Indiana. Indiana University Press, Bloomington, Indiana. 688pp.

Whitby, M., B.J. Udell, A.M. Wiens, T. Cheng, W. Frick, B.E. Reichert, and J.D. Reichard. 2022. Summer Mobile Acoustic Transect Analysis for Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter C in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

White, J.W., P. Freeman, C. A. Lemen. 2017. Habitat selection by the Northern Long-eared Myotis (*Myotis septentrionalis*) in the Midwestern United States: Life in a shredded farmscape. Transactions of the Nebraska Academy of Sciences and Affiliated Societies 37: 1–10.

White, J.A., P.W. Freeman, H.W. Otto, and C.A. Lemen. 2020. Winter use of a rock crevice by northern long-eared myotis (*Myotis septentrionalis*) in Nebraska. Western North American Naturalist: Vol. 80: No. 1, Article 14.

Wiens, A.M., J. Szymanski, B.J. Udell, and W. E. Thogmartin. 2022. Winter Colony Count Data Assessment and Future Scenarios for the Little Brown, Northern Long-eared, and Tricolored Bat Species Status Assessment. Chapter E in Straw, B.R, J. A. Martin, J.D. Reichard, and B.E. Reichert, editors. Analytical Assessments in Support of the U.S. Fish and Wildlife Service 3-Bat Species Status Assessment. Cooperator Report prepared in cooperation with the U.S. Geological Survey, United States Fish and Wildlife Service and Bat Conservation International. https://doi.org/10.7944/P9B4RWEU

Wilcox, A., L. Warnecke, J.M. Turner, L.P. McGuire, J.W. Jameson, V. Misra, T.C. Bollinger, and C.K.R. Willis. 2014. Behaviour of hibernating little brown bats experimentally inoculated with the pathogen that causes white-nose syndrome. Animal Behaviour 88:157–164.

Wilder, A.P., W.F. Frick, K.L. Langwig, and T.H. Kunz. 2011. Risk factors associated with mortality from white-nose syndrome among hibernating bat colonies. Biology Letters 7: 950–953. http://doi.org/10.1098/rsbl.2011.0355

Wilson, R.J., D. Gutierrez, J. Guitierrez, and V.J. Monserrat. 2007. An elevational shift in butterfly species richness and composition accompanying recent climate change. Global Change Biology 13:1873–1887.

Wiser, R., J. Rand, J. Seel, P. Beiter, E. Baker, E. Lantz, and P. Gilman. 2021. Expert elicitation survey predicts 37% to 49% declines in wind energy costs by 2050. Nature Energy. Available at: https://doi.org/10.1038/s41560-021-00810-z

Zackay A. 2007. Random genetic drift and gene fixation. [online] https://www.metabolic-economics.de/pages/seminar_theoretische_biologie_2007/ausarbeitungen/zackay.pdf [Accessed 21 June 2018].

APPENDICES

1. **Key Uncertainties, Wind Energy Mortality Sensitivity Analyses, and State-of-the-Knowledge**
    - A. Key Uncertainties
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2. **Supplemental Methodology**
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    - B. Current Condition
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5. **Supplemental Future Scenarios Descriptions**

Appendix 1: Key Uncertainties, Wind Mortality Sensitivity Analyses, and State-of-the-Knowledge

Note: Appendices were created for a three bat (*Myotis lucifugus* [little brown bat, LBB], *Myotis septentrionalis* [northern long-eared bat, NLEB] and *Perimyotis subflavus* [tricolored bat, TCB] SSA). When reference is made to “these bats” or “these species,” we are referring to LBB, NLEB or TCB, or all three species.

B. Wind Energy Mortality Sensitivity Analysis

To discern the sensitivity of our results to uncertainty regarding wind energy related mortality, we ran various mortality scenarios. We compared four scenarios: 1) no wind energy related mortality, 2) current predicted mortality, 3) 50% of mortality corresponding to the future high impact scenario, and 4) full projected level of mortality corresponding the high impact scenario. Clearly, WNS is the driving force in the future trajectory of the species (see Figure A-1A1, comparing no WNS impacts to WNS impact scenarios), thus it is not unexpected that the general trend in abundance is unaffected by wind energy mortality (Figure A-1B1). The additive effect of wind energy mortality is, however, discernible as seen when comparing no wind energy related mortality to wind energy mortality scenarios (Figure A-1B2, see bar 1 vs 2 under current conditions and bar 3 vs 4 and 5 under future conditions). The results are markedly sensitive to the range of uncertainty in future mortality levels among scenarios (Figure A-1B2, see bar 4 vs 5).

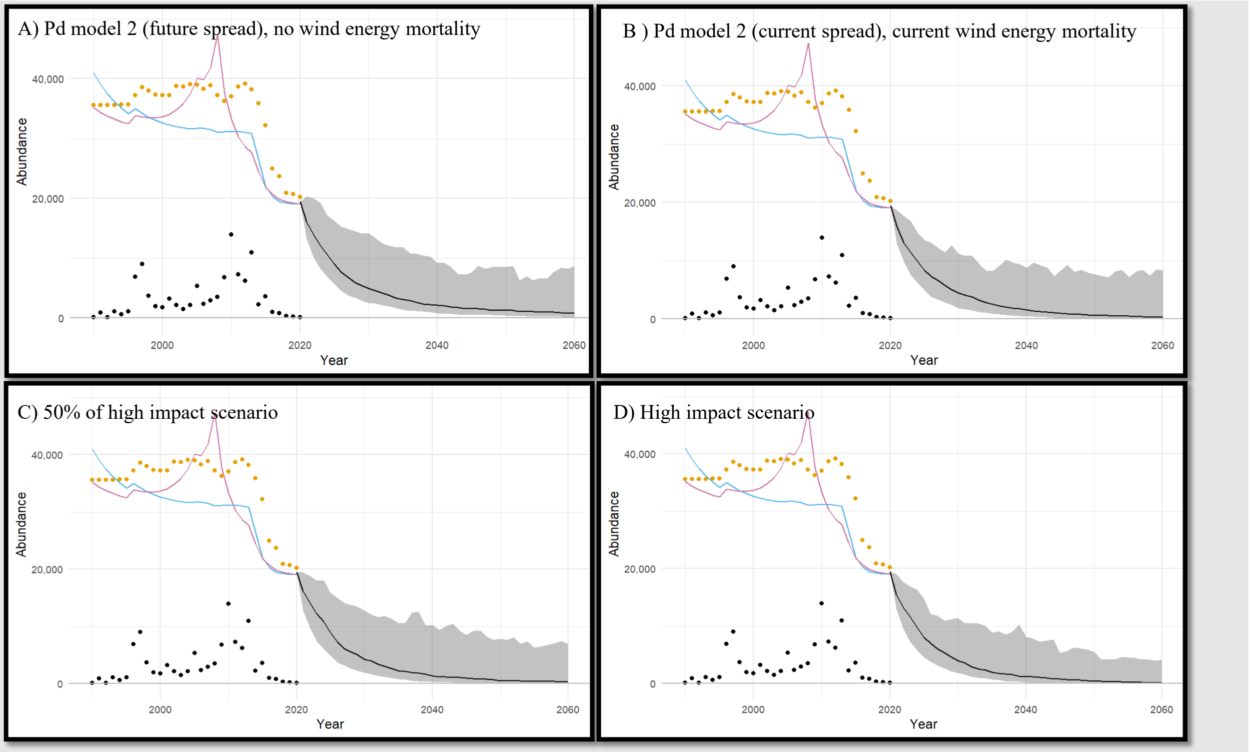


Figure A-1B1. NLEB projected abundance under various wind mortality levels: (A) Pd model 2 (future spread), no future wind energy mortality, (B) Pd model 2 (current spread), current wind energy mortality, (C) 50% of the future wind energy mortality under the high impact scenario, and (D) high impact scenario mortality. Abundance from 1990–2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

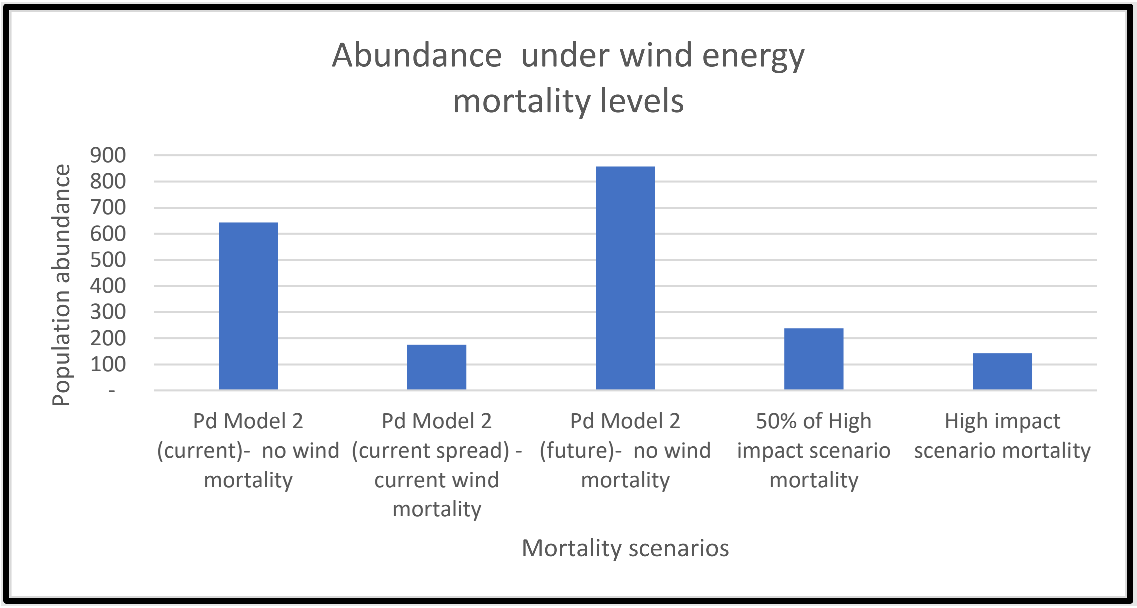


Figure A-1B2. NLEB projected 2060 median abundance under five wind energy related mortality levels: (A) Pd model 2 (future spread), no future wind energy mortality, (B) Pd model 2 (current spread), current wind energy mortality, (C) future mortality under low impact scenario, (D) 50% of the future mortality under the high impact scenario, and (E) future mortality under the high impact scenario.

A. Key Uncertainties

Our analysis includes both aleatory (i.e., inherent, irreducible) and epistemic (i.e., ignorance, reducible) uncertainty that we address by developing a range of future scenarios, adding environmental stochasticity to our model, and making reasonable assumptions. The key uncertainties are listed in Table A-1.1 and described below.

Table A-1.1. A list of key uncertainties addressed in the analysis.

Current Abundance and Trend	White-nose Syndrome Impacts	Wind Energy Related Mortality	Climate Change and Habitat Loss
Imperfect abundance data over time and space	*Pd* rate of spread*	Future wind energy capacity*	Response to climate change
	WNS impact schedule	Fatality rates	Response to habitat loss
	Duration of WNS impact*	Fatality risk over time and space	
	Bat response where WNS not yet arrived		
	Unknown hibernacula		

*Uncertainties are addressed directly in our high and low impact future scenarios (see Appendix 5).

Abundance and population trend

We do not have **perfect knowledge of current colony abundance and population trend** because hibernacula are not surveyed every year nor concurrently, and there are likely many undocumented hibernacula. Furthermore, bats can be hidden in crevices or inaccessible locations within roosts that are surveyed, and some species are difficult to identify accurately. We address this uncertainty by using predictive models developed by Cheng et al. (2021, entire) and Wiens et al. (2022, pp. 231–247) to predict current abundance and population growth rate (trend) for each known hibernaculum. Cheng et al. (2022, entire) explain that using a statistical model rather than inferring from data summaries is preferred because it can account for site-to-site variation, year-to-year variation, and survey effort, thereby allowing evaluation of the main effects of count over time and the impacts of WNS on counts. Further, statistical methods allow for objectively quantifying the relationships between variables while also quantifying the amount of uncertainty around those results. We summarized the state-of-the-knowledge (raw data summaries) that inform these statistical methods in Appendix 1-C.

The statistical models are constructed from the raw data available (in this case, 3,493 NLEB winter observations). Although these available data are biased towards the eastern portion of the U.S., these data represent the core of the species' known historical and current abundances, and thus are representative of the species' overall condition. Further, while the imminent threats (i.e., WNS, wind, habitat loss, and climate change) may vary temporally, the spatial distribution and overall severity of these threats are not likely to differ markedly (see WNS impacts assumptions below). Coupling this assumption with information concerning the narrow range of optimal conditions for hibernation, we believe these data provide the best available and reliable dataset to assess the current and future viability of the species.

Estimating bat population abundance and trends is challenging due to bats' cryptic nature, wide ranging habits, and variable detectability. A variety of methods have been developed and continue to be improved to fulfil this important information need, including winter and summer colony counts, mist-netting, acoustic monitoring, and mark-recapture studies. However, these efforts are often limited in scope or have been inconsistently applied across species' ranges. For several federally protected hibernating bats (e.g., Indiana bat, Virginia big-eared bat (*Plecotus townsendii virginianus*), and gray bat (*Myotis grisescens*)), successful population monitoring has been achieved through coordinated survey efforts at winter and summer roosts in caves. Fortunately, non-listed species have benefitted from these coordinated survey efforts and monitoring expertise where they overlap with either state or federally listed species. For this reason, estimates of overwintering colony abundance of NLEB are available through a substantial portion of the range over recent decades. Winter survey efforts for these and other hibernating species also increased when concerns about WNS were first raised in North America over 10 years ago. Other sources of data, to date, are more sporadic spatially and temporally but are still useful to inform population status.

We also do not have perfect knowledge of every hibernacula throughout the range of the species (**unknown hibernacula**). NLEBs are commonly found and counted during surveys in cavernicolous (cave-like places) hibernacula in eastern North America. Despite the expectation that many hibernating bats remain unobserved during winter, abundance estimates based on winter counts represent a sound estimate of the site-specific abundances, relative abundances, or at least trends of these species. Importantly, although these surveys do not produce a true census of the populations, they provide an estimate (or index) of abundance during winter when both sexes of these species are roosting together. Summer roost counts are possible, but much less feasible for NLEB due to their roost preferences and frequent roost switching. Mist-netting efforts to estimate capture per unit effort is another method for assessing trends, but these efforts are labor intensive and not commonly available rangewide (as efforts are often concentrated in certain selected areas). Finally, acoustic monitoring can be used to estimate occupancy or indices of abundance that are useful to estimate relative changes in populations but are very difficult to interpret as estimates of abundance. For these reasons, winter colony counts produce the most direct, representative, and feasible method for estimating abundance of NLEB, even if these data only represent minimum estimates of abundance.

Furthermore, WNS is typically detected and causes mortality either during winter or in spring after sick bats emerge from hibernation. Thus, estimating the impacts of this disease is best achieved by evaluating changes in winter colonies, where possible, in response to the arrival of the fungal pathogen. This approach allows for analyses that specify the year of arrival of the fungal pathogen and subsequent changes in population sizes. While winter counts provide the most direct method for estimating the impacts of WNS, additional data streams are used to verify the patterns observed in winter. Analyses of mobile acoustic monitoring and capture efforts provide estimates of changes in relative abundance, while stationary acoustic monitoring produces indices of bat activity. All of these together are also used in occupancy modelling to determine changes in occurrence on the landscape over time. While none of these methods provides a perfect estimate of bat population abundance, together they improve our understanding of the status of the species.

White nose syndrome impacts

To capture the uncertainty in the **rate of spread** of *Pd* by using two different *Pd* occurrence models, a faster spread rate (*Pd* occurrence model 1, Wiens et al. 2022, pp. 226–229) based on spread rates observed and annual changes in the occurrence of *Pd* and a slower spread rate (*Pd* occurrence model 2, Hefley et al. 2020, entire) that incorporates historic occurrence and multiple habitat covariates (Appendices 2A and 5). Both models rely on the same WNS surveillance dataset but each model performs differently in different geographic regions of the country based on the models' parameters. Thus, these two predictions provide a plausible range of the timing of *Pd* spread into the future.

Although we have empirical information on population-level impacts associated with WNS disease progression (on average, 98% decline by the endemic stage, Cheng et al. 2021, entire), there is variability among sites. We identified sites that trended differently (i.e., bats fared better) than most and assumed they do not experience further WNS impacts. For all remaining sites, we assumed they would follow the empirically derived yearly impacts schedule. Wiens et al. (2022, pp. 231–235) used random draws from the impact distribution for each year (Appendix 2-A).

Another source of uncertainty is the **duration of WNS impacts**. We captured the full breadth of uncertainty in our future scenarios. For all scenarios, WNS impacts ameliorate 6 years after the arrival of *Pd*, forming an endemic stage (see Appendix 2-A). Under the low impact scenario, we assumed a 9-year endemic stage and thus yielding a 15-year WNS impacts duration in total. This is the shortest conceivable timeframe based on our analysis of the data available. Under the high impact scenario, we assumed a 34-year endemic stage, thus yielding a 40-year WNS impacts duration in total (Appendix 5). Figure A-1A1 shows results assuming no further WNS impacts beginning in 2020, a 25-year impacts duration, and a 40-year impacts duration.

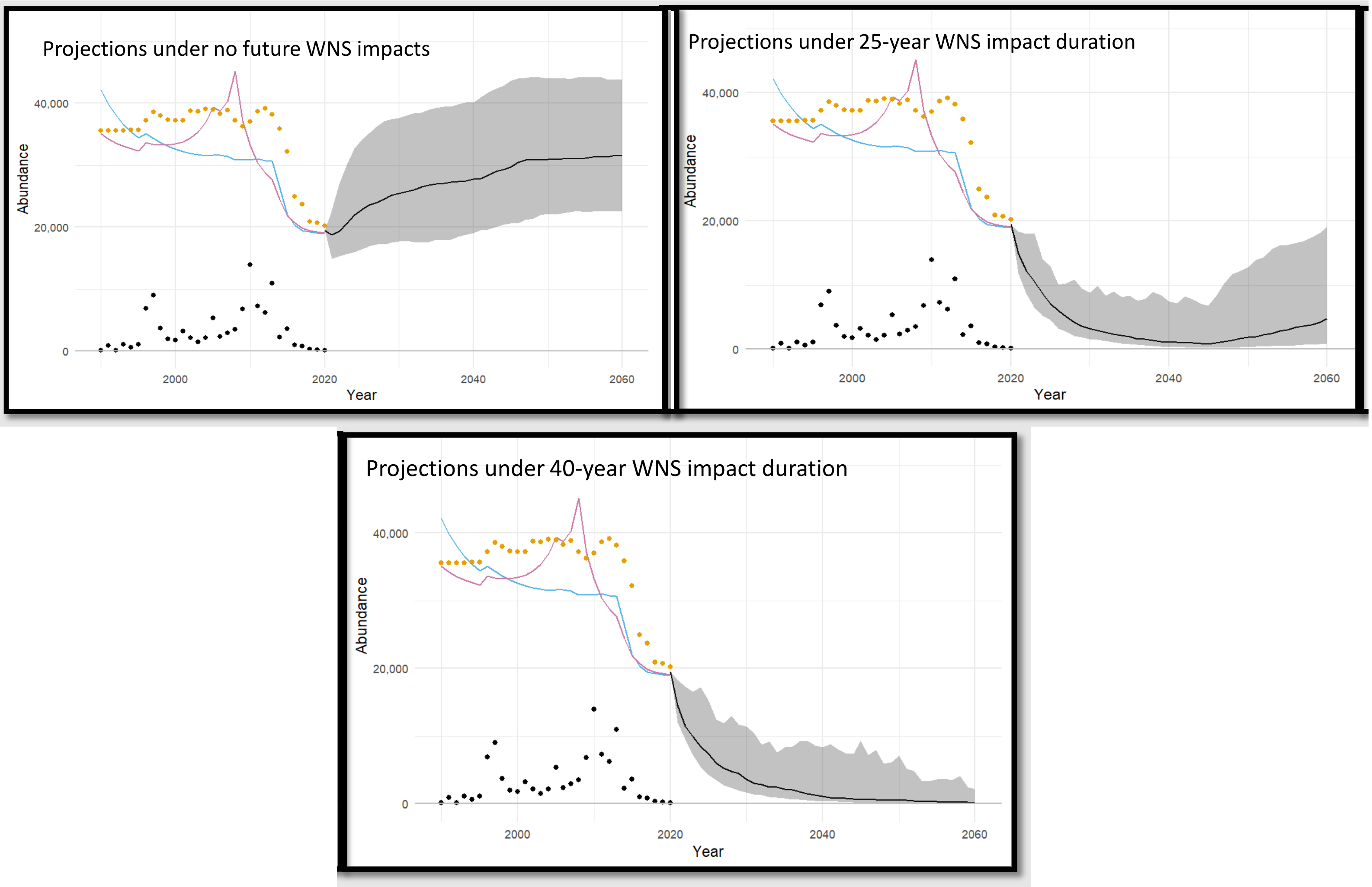


Figure A-1A1. Projected median rangewide abundance (median [black line], 90% CI [gray shading]) over time under no future WNS impacts (top left), a 25-year impacts duration (top right), and a 40-year impacts duration (bottom). Abundance from 1990–2020 derived from winter colony count data (black dots) using a) constant interpolation (yellow dots), b) status and trend model informed by Pd occurrence model 1 (blue line) and c) status & trend model informed by Pd occurrence model 2 (pink line).

Where disease dynamics of WNS have been observed (primarily, but not solely in the eastern half of North America and in cave-like hibernacula), very few colonies of NLEB have avoided severe impacts of the disease. A variety of site characteristics including colony size, temperature, and humidity may explain some of the variability that is observed in the degree of impact caused by WNS. Wilder et al. (2011) predicted that larger colonies will experience impacts of WNS sooner than smaller colonies. Further, Langwig et al. (2012, p. 6) determined that smaller colonies of NLEB may experience less severe impacts than larger colonies during the initial stages of the disease. Frick et al. (2015, p. 6) found that NLEB had a consistently high local extinction risk regardless of pre-WNS colony size. Environmental conditions may also influence impacts of disease. While it has been determined that colder roosts may reduce WNS infections, mortality from WNS has been documented at a wide range of temperatures, including sites with winter temperature approaching 0°C (Langwig et al. 2012, p. 6). Low humidity conditions may also lessen the severity of infection, at least for some species. For example, Indiana bat in drier hibernacula have shown to have less severe impacts from WNS, but this pattern was not observed in NLEB (Langwig et al. 2012, p. 6).

Physiological demands of hibernation limit the ranges of temperature and humidity in which bats can hibernate successfully, although these limits or preferences differ among species. Hibernacula temperatures that are too low present a risk of freezing or raise the energetic cost of torpor. Similarly, hibernacula that are too dry lead to dehydration or frequent arousal from torpor that will consume limited fat reserves. Thus, although these factors may delay or reduce the impacts of WNS, none of them would prevent the arrival of *Pd* or avoid impacts of WNS altogether. Because their winter roosts must be cold and humid to allow for successful hibernation and these conditions are also conducive to growth of *Pd*, it is valid to presume WNS impacts will be similar throughout the portions of the species' ranges where bats hibernate for extended periods, regardless of whether these hibernacula are unknown or human inaccessible.

Wind Mortality

We don't know the **future build-out of wind energy capacity** in the U.S. and Canada. We relied on the National Renewable Energy Laboratory's (NREL) (Cole et al. 2020, entire) and Canadian Energy Regulator's (CER) (CER 2020, entire) projections for the U.S. and Canada, respectively. To capture the uncertainty associated with these projections, we incorporated lower and upper bound capacity projections into our future scenarios. Our low impact scenario (i.e., lower wind build-out) was based on NREL's *High Wind Cost* scenario and CER's *Reference Scenario* (Figure 4.10). Our high impact scenario (i.e., higher wind build-out) was based on NREL's *Low Wind Cost* scenario and CER's *Evolving Scenario* (Chapter 4 and Appendix 5). These build-out scenarios provide reasonable bounds for future expectation of wind capacity in both the U.S. and Canada.

Fatality Rates vary across species, range, and seasons. We used regional specific data garnered from post construction monitoring efforts. We obtained nearly 300 reports spanning 20 states and 4 USFWS Regions. We calculated the mean fatality rate for the species within each USFWS Region using currently accepted methods to account for spatial variability (see Appendix 2).

We also are uncertain about how **fatality risk varies over time and space**. Although it is logical to assume fatality risk declines with decreasing abundance, the functional relationship is unknown. We evaluated fatality rates pre- and post-WNS arrival to discern a relationship between abundance and fatality risk. Where applicable, we applied pre- and post *Pd* fatality rates to account for the uncertainty in fatality risk as abundance changes over time (see Appendix 2). Additionally, we are uncertain of where bats killed at wind facilities originate. To address this uncertainty, we relied on the analysis completed by Udell et al. (2022, entire). Briefly, Udell et al. (2022, entire) created a distance decay function to allocate total wind mortality per 11x11-km NREL grid cell among hibernacula within the known average maximum migration distance, relative to the size of the hibernating populations as well as the distance from the grid cell centroids (i.e., hibernacula with larger colony counts and those closer to grid cell centroids were assigned higher proportions of the overall mortality). However, the analysis did not account for the possibility that some bats may originate from additional unknown hibernacula within the maximum recorded migration distance, or that bats may be migrating farther than previously documented. To look at how this latter uncertainty may affect the results, we ran a scenario in which wind mortality is 50% of what is projected under the high capacity scenario. The additive effect of wind energy mortality is discernible as seen when comparing a no wind to a wind scenario (Figure A-1B2); although from a viability perspective, the results do not appear sensitive to the range of uncertainty in future mortality levels (i.e., no marked changes in the overall trend in abundance).

Climate change

As we detail further in Chapter 4 and Appendix 4, both habitat loss and climate change are pervasive across the species' range and severity of population level declines are currently assumed to be slight (recognizing varying impacts by population). Thus, we believe overall climate change impacts are currently low. While there is uncertainty about the magnitude of future temperature increases and any associated changes in precipitation (e.g., regional changes, rate and intensity of extreme weather events), we have high confidence in the precipitation and temperature changes observed to date and that minimal projected temperature increases (2.2 degrees F (1.2 degrees C), relative to baseline) will occur. Similarly, we have high certainty in observed species responses to changes in temperature and precipitation (which vary geographically). However, we have less certainty about species responses that have not been observed, such as: death of individuals or alteration of hibernacula use due to increased risk of flooding from sea level rise or extreme weather events; reduced reproduction or survival due to increased habitat loss in wildfire prone areas; changes in phenology of bats and their prey; and changes in bat distribution. Lastly, we have uncertainty about possible beneficial impacts from climate change in portions of species' range. While possible, beneficial impacts (e.g., warmer temperatures may lead to shorter hibernation periods, which in turn may decrease the *Pd* exposure duration and thus reduce impacts) are more speculative, at least relative to the observed negative impacts reported in the literature. For this reason, our assessment of effects from climate change likely underestimates risk to these species.

Habitat Loss

We have high confidence that changes in vegetation cover types occur throughout the range of NLEB. We also have high confidence that these changes in landcover may be associated with losses of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality (during active season tree removal). Despite this knowledge, we have uncertainty about how much forest removal must occur within a home range before impacts associated with winter tree removal are realized. We also have imperfect knowledge of where roosts (summer and winter) for NLEB occur. Therefore, we have uncertainty about which colonies (summer and winter) are at greatest risk of impacts and ultimately the magnitude of risk associated with habitat loss. Also, we have high confidence of prior impacts to winter hibernacula and hibernating bats.

C. State-of-the-Knowledge

For reasons articulated in subsection A above, we relied upon statistical methods rather than raw data alone to assess the species' current status. We summarize the data underlying these methods here.

- We have 3,492 NLEB records from 737 hibernacula (90% of the sites are from the Eastern Hardwoods RPU).
- Based on these raw data:
  - Number of hibernacula with "Last observed = 0": 373 (1990-2020), 5 (2006-2009), 103 (2010–2015), 263 (2016–2020); the ratio (proportion) of extirpated to extant sites increased since WNS discovered in 2006 (Figure A-1C1)
  - Of the 364 potentially extant sites, 84 to 92% have uncertain status (304 and 335 sites do not have ≥ 1 record from 2017–2020 and 2019–2020, respectively)

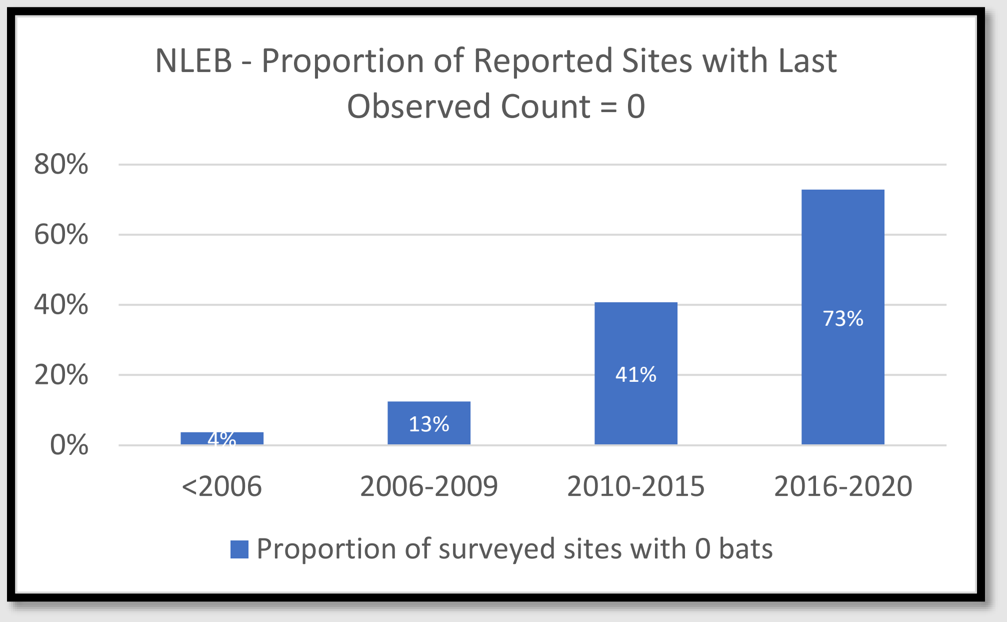
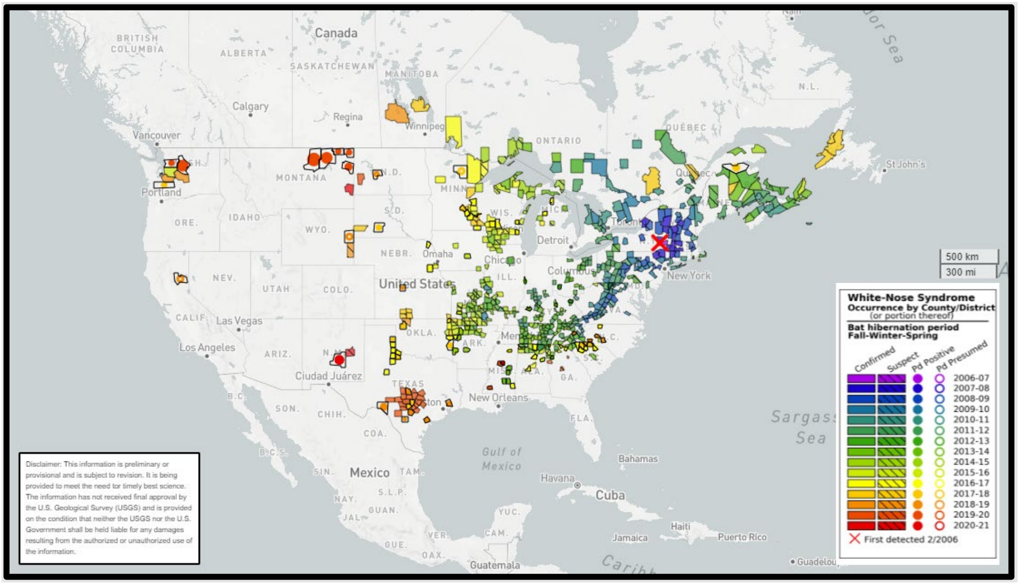


Figure A-1C1. The proportion of sites reported to NABat with 0 as the "last observed count." The proportion is the number of hibernacula with 0 counts divided by the total number of hibernacula surveyed.

- As of 2021, 580 counties across 40 states and 7 provinces have presumed or confirmed *Pd*/WNS (485 are confirmed WNS/*Pd*)(www.whitenosesyndrome.org, accessed May 13, 2021). WNS/*Pd* suspected/confirmed from Nova Scotia southward to South Carolina, westward to Texas, New Mexico, Wyoming, Montana, and Washington (Figure A-1C2).



Disclaimer: This information is preliminary or provisional and is subject to revision. It is being provided to meet the need for timely best science. The information has not received final approval by the U.S. Geological Survey (USGS) and is provided on the condition that neither the USGS nor the U.S. Government shall be held liable for any damages resulting from the authorized or unauthorized use of the information.

Figure A-1C2. WNS occurrence as of 5/12/2021 (www.whitenosesyndrome.org, accessed May 13, 2021)

- The number of NLEB hibernacula with suspected or confirmed WNS is not available; WNS has been confirmed in every RPU. However, *Pd* has not been detected in the northwestern arm of NLEB's range.
    - As of May 2021, there are 112 NLEB events. Events are winter or summer sites with suspected/confirmed WNS/*Pd* reported on the species of interest (i.e., a species event is recorded only when the species has *Pd*/WNS, even if the WNS/*Pd* confirmed/suspected on other species or the site, *www.whitenosesyndrome.org, accessed May 13, 2021*)
- Where WNS is present, severe declines have occurred, except in a few (3%) hibernacula. On average, NLEB colonies declined by mean 100% (95% CI 97 –100) by the endemic stage of WNS progression (Cheng et al. 2021, p. 7).

- Declines are discernible in summer data as well. Data availability vary among the data type (mobile transect acoustic, stationary acoustic, and mist-net capture data), however we incorporated all available data into the analyses.

    - Using mobile acoustic data from 2009 to 2019, Whitby et al. (2022, entire) found relative abundance declined 50% (Southeast RPU) to 99% (Midwest RPU) from 2009 to 2019. Insufficient data were available for the Subarctic RPU.

    - Using mist-net capture data from 1999 to 2019, Deeley and Ford (2022, entire) found a significant decrease in mean capture rates post-WNS arrival. Estimates derived from their data indicated a 43% (Eastern Hardwoods RPU) to 77%

    (Midwest RPU) decline in mean capture rates post-WNS arrival. Insufficient data were available for the Subarctic RPU.

  - Using all 3 data types (mobile transect acoustic, stationary acoustic, and mist-net capture data) from 2010 to 2019, Stratton and Irvine (2022, entire) looked at changes in probability of occupancy across the species' range. Although the declines attenuated westward, there was a decline in predicted occupancy across all RPUs (Stratton and Irvine (2022, p. 102). Estimates derived from their results showed declines in the probability of occupancy across all 5 RPUs, ranging from 63% (Subarctic RPU) to 87% (Midwest RPU) from 2010 to 2019.

Appendix 2: Supplementary Methodology

A: Analytical Framework

Below we describe our methods for assessing a species viability over time. Our approach entailed: 1) describing the historical condition (abundance, health, and distribution of populations prior to 2020), 2) describing the current condition (abundance, health, and distribution of populations in 2020), 3) identifying the primary influences leading to the species' current condition and projecting the future states (scope and magnitude) of these influences, 4) projecting the number, health, and distribution of populations given the current and future states of the influences, and 5) assessing the implications of the projected changes in the number, health, and distribution of populations for the species' viability (Figure A-2A1). Because of the difficulty of delineating individual populations for NLEB, we used winter colonies (hibernacula) to track the change in number, health, and distribution of populations over time. The terms populations, winter colonies, and hibernacula are used interchangeably.

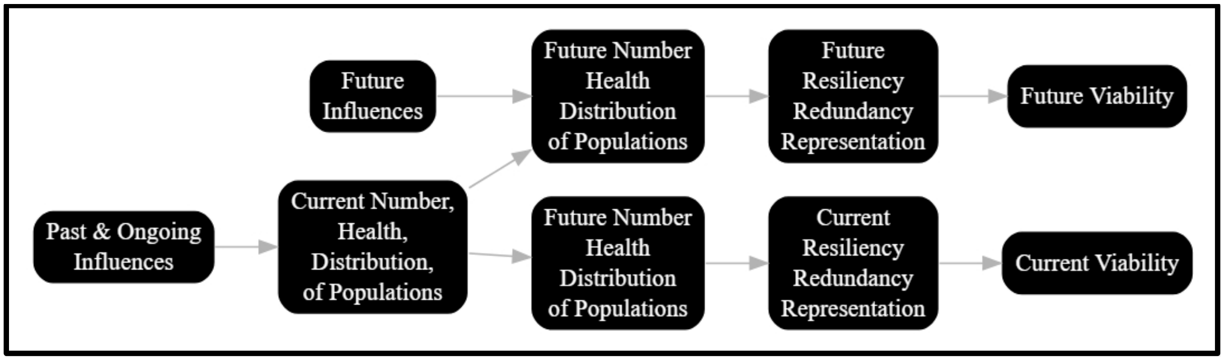


Figure A-2A1. Simplified conceptual diagram depicting the analytical framework for assessing bat viability over time.

Step 1. Historical Abundance, Health, and Distribution

We reached out to partners (Tribal, Federal, State and other) across the range to garner summer (capture data and stationary and mobile acoustic) and winter occurrence (hibernacula counts) data. Most of these data are maintained in the North American Bat Monitoring Program (NABat) database, unless otherwise requested by the data contributor or the data was not provided in a format that could be accepted by the database. These efforts yielded thousands of records across the range (Figure A-2A2) and one of the largest bat data repositories we aware of. Hibernacula counts were available for much of the range of NLEB, although occurrence information is limited for the species in parts of the western portion of the U.S. and Canada range. Consistent with the species' biology, we assumed that NLEB employs hibernation in cold, humid roosts even when these roosting locations are not observed by data collectors. Using this information, we compiled a list of all known hibernacula and associated yearly winter counts (winter hibernacula surveys; NABat 2021).

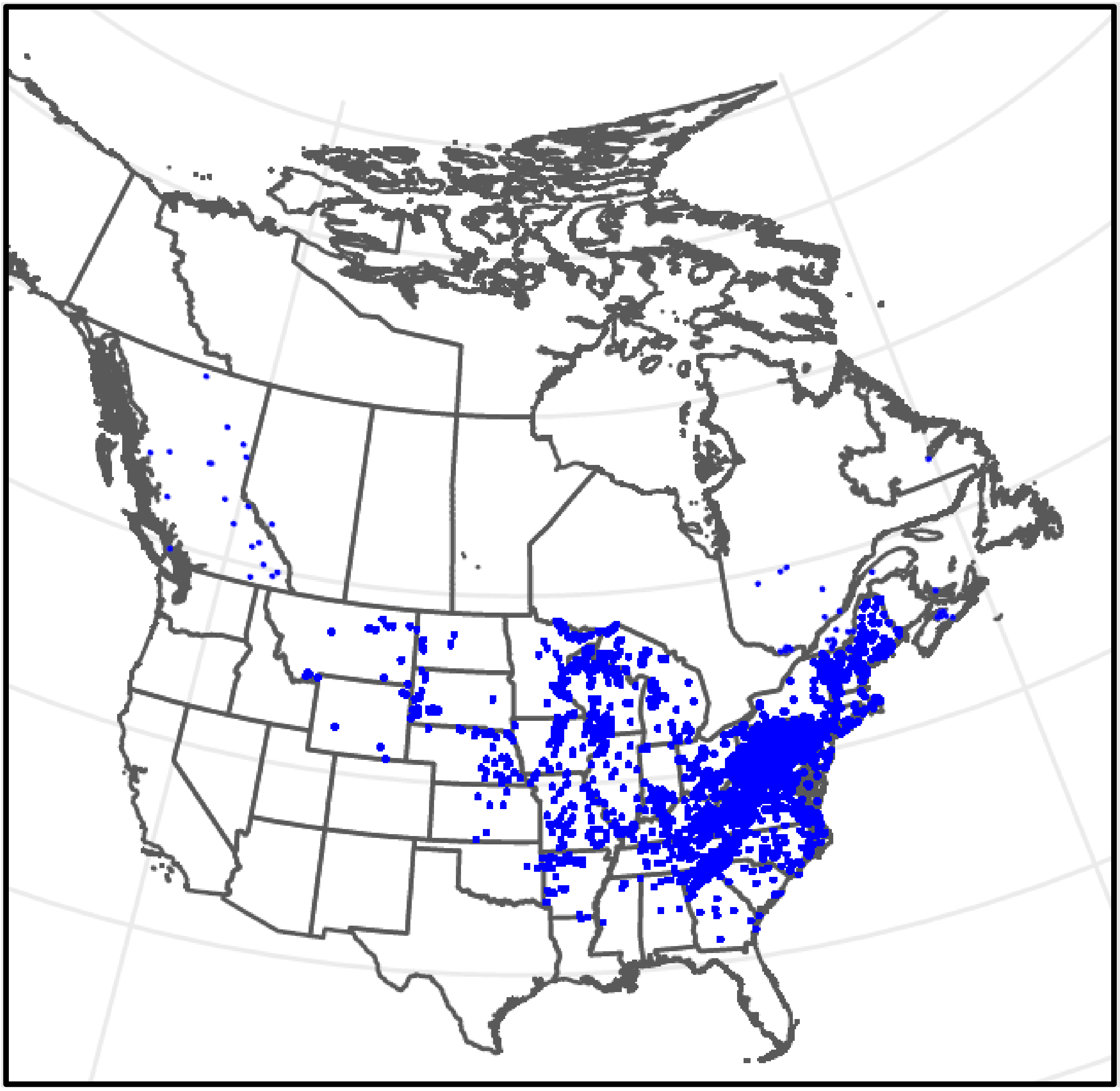


Figure A-2A2. Data available in and contributed to NABat for use in NLEB population analyses. These data show sampling effort and there may be some locations where NLEB was not detected at the survey site. Map credit: B. Udell, U.S. Geological Survey, Fort Collins Science Center. Disclaimer: Provisional information is subject to revision.

One way we measure population health was hibernacula abundance (*N*) and population trend (λ). Despite the thousands of winter counts, data are not available for all years and not necessarily both pre and post WNS arrival. Thus, to estimate historical *N* and λ, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233). For sites with more than 5 data-points (n = 297), they fit the data using a statistical linear mixed effects model (henceforth referred to as Status/Trends model) to estimate the yearly abundance for each hibernaculum from 1990 through 2020. For sites with fewer than 5 colony counts (n = 440, they used last observed count and used the λ from closest hibernaculum or complex of hibernacula. The Status/Trends model relies upon

WNS year of arrival, thus, *N* and *λ* estimates vary with the occurrence of *Pd*. Wiens et al. (2022, pp. 231–233) used two projections of *Pd* occurrence (referred to as *Pd* occurrence Model 1 and 2) to identify year of arrival for hibernacula lacking data (see *Current and Future Primary Drivers* subsection below) to capture uncertainty in the presence and spread rate of *Pd* at unknown and uncontaminated sites. Both models use available disease surveillance data documenting past detection of *Pd* but use different parameters to estimate occurrence of *Pd* beyond those detections. Hence, we have two estimates for yearly historical colony *N* and *λ*. See Appendix 5 for further details on the Status/Trends model.

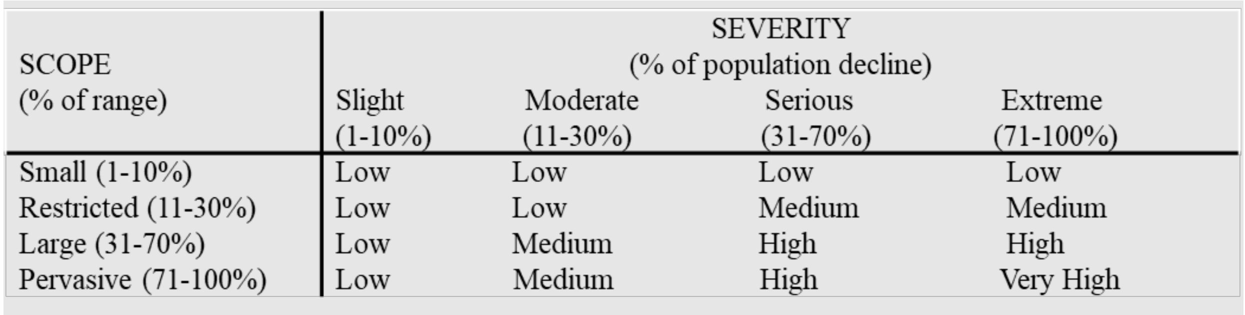
Step 2. Describe Current Abundance, Health, and Distribution

To estimate current conditions, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233) as described above. Additionally, because colony estimates are not available for all hibernacula and because bats occupying a given hibernaculum disperse to many different locations on the summer landscape, we also relied upon the results from USGS-led summer capture records and acoustic records analyses to garner insights on population trends at regional scales (see *Summer Data Analyses* subsection below).

Step 3. Identify the Primary Drivers (Influences)

We reviewed the available literature and sought out expert input to identify both the negative (threats) and positive (conservation efforts) influences of population numbers. We identified WNS, wind related mortality, habitat loss, and climate change as the primary negative influences on the species' abundance. We also identified several other potential influences but based on available information were either too local in scale or lacking data to assess species response.

Qualitative/Comparative Threat Analysis - We assessed the impact of the four influences using an approach adapted from Master et al. (2012, entire) to allow a comparison between influences. For each influence, we assigned a scope, severity, and impact level for both current and future states. Briefly, scope is the proportion of the populations that can be reasonably expected to be affected by the threat within 10 years (current). Severity is the level of damage to the species from the threat. Impact is the degree to which the species is directly or indirectly threatened based on the interaction between the scope and severity values. The criteria used to assign levels are shown in Figure A-2B3.



SCOPE (% of range)	SEVERITY (% of population decline) Slight (1-10%)	Moderate (11-30%)	Serious (31-70%)	Extreme (71-100%)
Small (1-10%)	Low	Low	Low	Low
Restricted (11-30%)	Low	Low	Medium	Medium
Large (31-70%)	Low	Medium	High	High
Pervasive (71-100%)	Low	Medium	High	Very High

Figure A-2B3. Comparative threat assessment criteria and definitions (adapted from Master et al. 2012, entire). Impact level (Low to Very High) is based upon the scope and severity assigned.

<u>Quantitative Threat Analysis</u> – We sought to model the impact of the four primary drivers, however, we did not have the time to rigorously determine the species response to changes in climate change and habitat loss. Although we have information on ongoing effects to North American insectivorous bats associated with climate change in specific geographic areas, given the differences in types and magnitude of climate change, the large range of these species, and the fact that we had finite time and resources, we were unable to reliably quantify each species' response in a manner that could be included in the population model (e.g., what specific changes to which specific demographic parameters should we include in response to projected changes in temperature or precipitation). Similarly, habitat loss or alteration can lead to locally consequential effects, especially with the compounding effects of WNS. We considered information on loss or alteration of hibernacula as well as information on changes in landcover types across each species' range; however, given our finite time and resources we were unable to project rangewide future landcover changes or the species associated response in a manner that could be included in the BatTool (e.g., what specific landcover changes would result in what specific changes to which demographic parameters). Instead, we provided a narrative on the spatial extent and magnitude of impact from these two stressors.

To assess the current and plausible future state conditions (magnitude and severity) for WNS and wind related mortality, we used published data, expert knowledge, and professional judgment. To capture the uncertainty in our future state projections, we identified plausible upper and lower bound changes for each influence. The lower and upper bounds for each influence were then combined to create composite plausible "low" and "high" impact scenarios. These scenarios were used as inputs to a population-specific demographic model (BatTool, Erickson et al. 2014, entire; explained Step 4 below) to project abundance given specified WNS and wind mortality scenarios.

WNS – To assess the current and future severity of WNS, we calculated disease-induced fatality rates from data gathered from winter colonies following *Pd* arrival (referred to as "WNS impacts schedule", see below). We assumed that the WNS impacts schedule (severity) will not change into the future, and hence, the only difference between the current and future WNS scenarios is the rate of spread (scope) of WNS. To estimate the current and future occurrence of WNS, we relied on two models (several others are available with similar predictions), Wiens et al. (2022, pp. 226–229) and Hefley et al. (2020, entire). We refer to these projections as "*Pd* occurrence model 1 and 2." Both models rely on the same WNS surveillance dataset but allowed us to capture uncertainty in spread rates. Additionally, each model performs differently in different geographic regions of the country, making one model better than the other in a certain area of the country and vice-versa.

Since 2007, collection and management of surveillance data for WNS and *Pd* on bats or in the environment has been coordinated by the National Response to WNS, led by USFWS. State agencies or other appropriate land-management entities conduct most sample collection for disease surveillance and are responsible for reporting county level-determinations of *Pd* status. WNS is confirmed by histopathological observation of lesions characteristic of the disease (Meteyer et al. 2009, entire), molecular detection of the fungus (Muller et al. 2013, entire), or characteristic field signs associated with WNS Case Definitions determined by USGS, National Wildlife Health Center. Year of arrival of WNS or *Pd* at a location is documented at a county-level resolution (available at www.whitenosesyndrome.org).

Wiens et al. (2022, pp. 226–229) used a Gaussian interpolation and projection using linear movement estimates based on observed rates of spread of *Pd* (see Appendix 5 for further information). Hefley et al. (2020, entire) used a diffusion and growth model, which estimates the prevalence (similar to abundance) of *Pd* at a location. In their model, prevalence is influenced by proximity to known occurrences and environmental covariates of percent canopy cover, terrain ruggedness index, waterways, locations of mines, and karst geology. Year of arrival of *Pd* at a location is assigned to the year in which prevalence exceeds 0.25 (this level was chosen by the SSA Core Team based on the prevalence value observed at a subset of sites where *Pd* has already been detected). Separate parameters were calculated to estimate current and future distribution of *Pd* in the Pacific Northwest, where the fungus is expected to have initiated a second epicenter after "jumping" from the nearest known previous occurrence (Lorch et al. 2016, p. 4). Using their estimates of spread rates, future distribution of *Pd* was projected on an annual scale for every 10 km x 10 km grid cell until *Pd* was predicted to be present throughout the entirety of the species' range (Wiens et al. 2022, pp. 226–229) or until statistical confidence interval in the model projection was too great for the value to be reliable (Hefley et al. 2020, entire). The projected *Pd* spread under the two models is shown in Figure A-2B4.

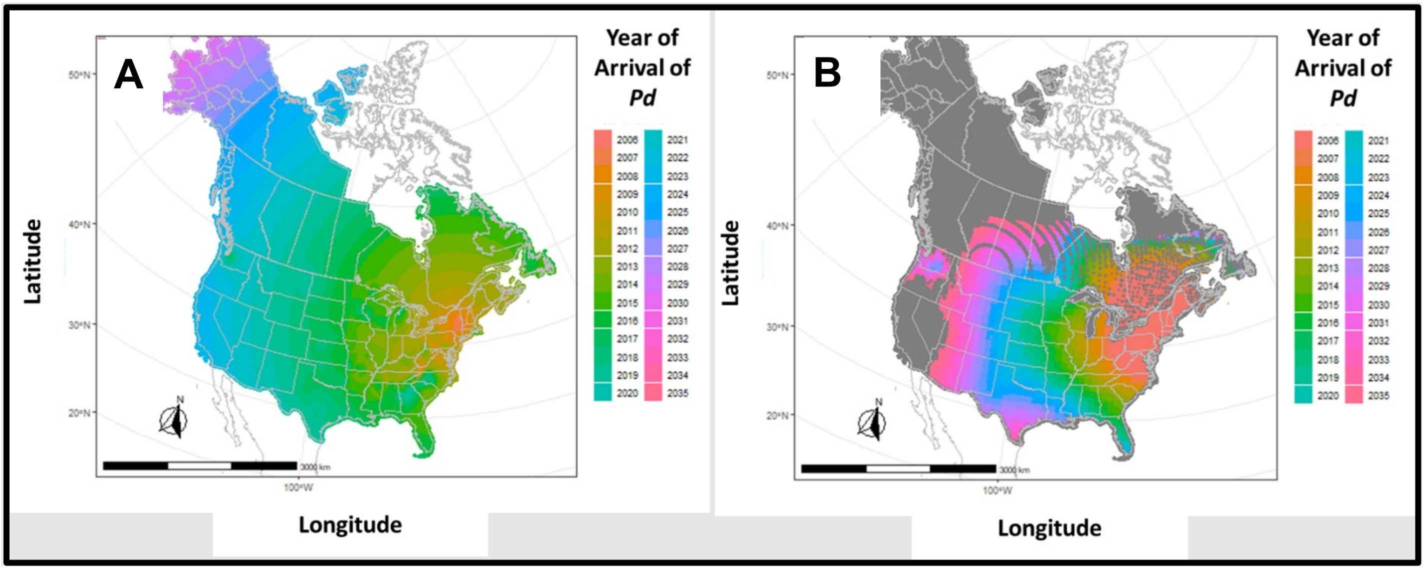


Figure A-2B4. Two models of Pd occurrence in North America since 2007 and into the future. A) A Gaussian interpolation map using spatial relationships and direct observations of Pd occurrence (Wiens et al. 2022, pp. 226–229). B) A diffusion and growth model using observed Pd prevalence in diagnostic samples to predict environmental prevalence of Pd based on spatial and environmental covariates (Hefley et al. 2020, entire).

To estimate current and future WNS impact (fatality rates), we relied on Wiens et al. (2022, pp. 233–235) derived "WNS impacts schedule"; a distribution of annual-specific changes to survival rates. They used data collected during winter hibernacula surveys from 1990 to 2020 and calculated the proportional change in size of the colony between calendar years and between years since arrival of *Pd*. Assuming that change in the estimated colony size was the result of WNS-induced mortality, these estimates of percent change in colony size were translated into changes in adult over-winter survival rate (a parameter in the BatTool). Lastly, they collated these site-specific over-winter survival rates to create annual distributions, i.e., WNS impacts schedule (Figure A-2A5.). This WNS impacts schedule was used in the BatTool to apply WNS impacts to hibernacula over time. For a few sites, the severity of WNS impact has deviated from the norm; for these exceptions, a colony-specific WNS impacts schedule was derived (Wiens et al. 2022, pp. 231–247). See Appendix 5 for additional information and further description of future scenarios.

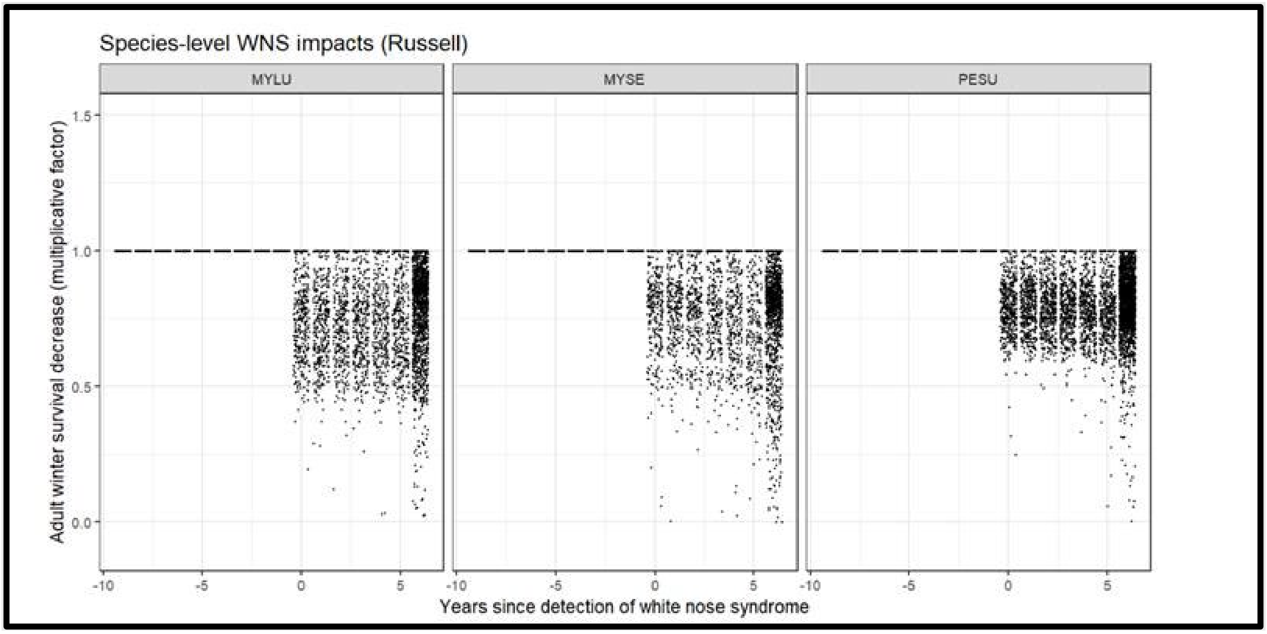


Figure A-2A5. Adult winter survival decreases annually after Pd detection for little brown bat (MYLU), NLEB (MYSE), and tricolored bat (PESU). These data were used to create the WNS impacts schedule. The data depicted for 6 years since detection of Pd include all years since detection ≥ 6.

Wind - To assess the current and future magnitude and severity of current and future wind energy development, we 1) estimated species-specific wind fatality rates (bats per megawatt (MW) per year), 2) applied current and projected future wind capacity within the species' range, and 3) applied species-specific fatality rates to current and future wind capacity to estimate wind related mortality for known hibernating populations. We assumed the only difference between the current and future wind scenarios is the amount of installed wind capacity. NLEB data were too limited to discern differences in percent species composition after WNS arrival, so we assumed no change in fatality rates over time.

To estimate wind fatality rates (severity), we reached out to the public, states, USFWS Ecological Services field offices, and other partners to request data from wind post-construction bat fatality monitoring at wind projects within the ranges of NLEB, little brown bat, and tricolored bat. We obtained 287 reports for wind projects in 20 states within USFWS Legacy Regions (Regions) 3, 4, 5, and 6 (Figure A-2A6).

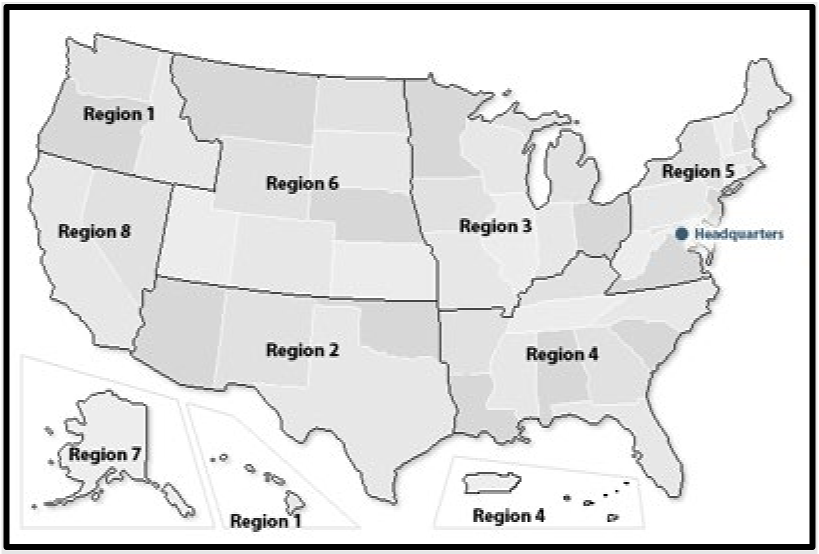


Figure A-2A6. U.S. Fish and Wildlife Service Regions.

For a subset (n = 155) of these reports (those that met our inclusion criteria, described below) we calculated [species]-specific per MW fatality rate using the following equation:

NLEB *per MW fat rate = Bfat * %Sp*

Where *Bfat* is the all-bat fatality rate per MW and *%Sp* is the species-specific percent composition of fatalities reported. *Bfat* was calculated for each Region by deriving annual all-bat per MW fatality rates for each study in our subset, applying corrections for unsearched areas and portions of the year as needed, and then averaging the corrected all-bat fatality rates across the studies in each Region. *%Sp* was calculated by dividing the total number of each species' carcasses reported in our subset of studies by the total number of bat carcasses.

To maximize consistency and comparability across studies in our database, we applied the following inclusion criteria:

1. Study must report a bats/megawatts (MW) or bats/turbine fatality rate, corrected for searcher efficiency (SE) and carcass persistence (CP). If bats/turbines is the only reported fatality rate, the report must also include the number of turbines and MW at the site in order to calculate bats/MW.
2. Turbines were operated without curtailment (i.e., no feathering below manufacturer's or other cut-in speeds) during the study period. In a few instances where studies tested certain cut-in speeds in a subset of turbines and reported separate fatality rates for curtailed versus control (uncurtailed) turbines, the control turbine fatality rate was used.
3. The study search interval was 7 days or less.
4. The study provided the range of dates when carcass searches were performed.
5. The study provided the search area (i.e., plot) dimensions.

Because we only obtained two reports from Region 4, and AWWIC (2020) did not report any NLEB-specific fatality rates, we combined our Region 4 and 5 studies to calculate *Bfat* and *%Sp* in these two Regions. NLEB data from Regions 1, 2, 6, 7, and 8, despite the considerable number of wind projects in these areas, were too limited to generate reliable estimates, and the data for Regions 3, 4 and 5 were too limited to support parsing out by Region. Therefore, we combined all available U.S. studies to derive a single *%Sp* for NLEB. For Canada, we used species composition rates (*%Sp*) reported in Bird Studies Canada et al. (2018). We detected no difference in NLEB *%Sp* by WNS stage; thus, we used rates pre- and post-WNS.

It should be noted that reported fatality rates in our USFWS database were derived using a variety of estimators with differing, imperfect assumptions and biases toward underestimating or overestimating mortality (i.e., see Rabie et al. 2021, entire). Additionally, a recent study by Huso et al. (2021, entire) found that bird and bat fatality rates were relatively constant per unit energy produced by turbines under similar environmental conditions regardless of their size, suggesting that the relative amount of energy produced, rather than simply the size, spacing, or nameplate capacity of turbines, determines the relative all-bat fatality rate. However, bat fatalities per turbine generally increased with turbine size or MW capacity (Huso et al. 2021, p. 4). Lacking information about the capacity factor (total energy produced relative to the theoretical maximum, or nameplate capacity), for all the turbines in our database, we relied on reported bats/MW fatality rates. As such, our averaged fatality rates may overestimate mortality for facilities with high capacity but low energy production (low capacity factor) or vice versa, but are more robust than bats/turbine fatality rates. Moreover, because they are averages across many facilities and states, they should capture the general capacity factor trends across regions, at least for built facilities as of October 2020.

To determine current and future wind capacity (magnitude), we obtained current wind capacity data from the U.S. Wind Turbine Database (USWTDB version 3.2) (Hoen et al. 2018, entire) and corrected/incorporated curtailment information based on facility-specific, unpublished USFWS data. For future projections, we used—at the counsel of experts at USDOE and NREL—the 2020 NREL High and Low Onshore Wind Cost Scenarios data (Cole et al. 2020, p. 26) as reasonable lower and upper bounds of future U.S. wind capacity by state. For Canada, we used Canada Energy Regulator's (CER) Evolving and Reference (baseline) scenarios as our upper and lower bounds, respectively (see Appendix 5 for further description of future scenarios).

Lastly, to calculate hibernacula-specific mortality, we relied upon the analysis by Udell et al. (2022, entire). Briefly, Udell et al. (2022, entire) summed wind capacity under the lower and upper bound scenarios for each 11x11-km NREL grid cell centroid and calculated a grid cell-specific mortality estimate. They then created a distance decay function to allocate the total mortality per 11x11-km grid cell among hibernacula, relative to the size of the hibernating populations and distance of hibernacula (within the known average maximum migration distance) from the grid cell centroid (i.e., hibernacula with larger colony counts and those closer to grid cell centroids were assigned higher proportions of the overall mortality). To account for mortality reductions associated with feathering below the manufacturer's cut-in speed or higher, we applied a 50% mortality reduction to turbines implementing any level of curtailment during the fall or summer and fall seasons, per our 2020 data (USFWS unpublished data). We then multiplied this 50% mortality reduction by t https://www.gray-robinson.com/attorneys-professionals/william-billy-dove he relative proportion of all-bat mortality reported by season in our post-construction mortality database (USFWS unpublished data; Table A–2A1). Based on these proportions, we applied an overall mortality reduction of 50% to turbines curtailing in both summer and fall and a 34% reduction to turbines curtailing in fall only (Table A–2A2).

Table A–2A1. Proportion of all-bat mortality by season (USFWS, unpublished data).

Season	Date Range	Proportion of All-bat Mortality
Spring	March – May 31	0.065
Summer	June 1 – July 30	0.252
Fall	August 1 – November 30	0.68
Summer + Fall	June 1 – November 30	1.0[7]

Table A–2A2. Curtailment categories by season and associated fatality reductions applied to turbine MW.

Category	Curtailment Season	Total Mortality Reduction Applied*
No Curtailment	None	N/A
Fall Only	Fall, Fall + Spring	0.34
Summer + Fall	Summer + Fall, Summer + Fall + Spring	0.50

**Reflects 50% mortality reduction for curtailment multiplied by seasonal proportion of all-bat fatality (Table A-2A1).*

Step 4. Project Future Number, Health, Distribution of Populations Under Current and Future Influences

To project future abundance and trend given current and future state conditions for WNS and wind, we used an existing bat population tool (BatTool, Erickson et al. 2014, entire). The BatTool is a demographic model that projects hibernaculum abundance over time given starting abundance (*N*), trend (λ), environmental stochasticity, WNS stage, annual WNS impacts schedule, and annual wind related mortality as specified by the wind capacity scenarios. Starting abundance (*N*) and trend (λ) were derived from the Status/Trends model described in Step 1 above. For each hibernaculum, the model was run for 100 simulations projecting 40 years into the future.

Using these projected abundance estimates, we calculated various hibernaculum-level and RPU-level (described in Chapter 2) metrics to describe the species' historic, current, and future number, health, and distribution of populations given current and future influences. Figure A-2A7 provides the conceptual framework for the BatTool, which includes the origins of model inputs.

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[7] Sum after rounding summer and fall curtailment to nearest tenth.

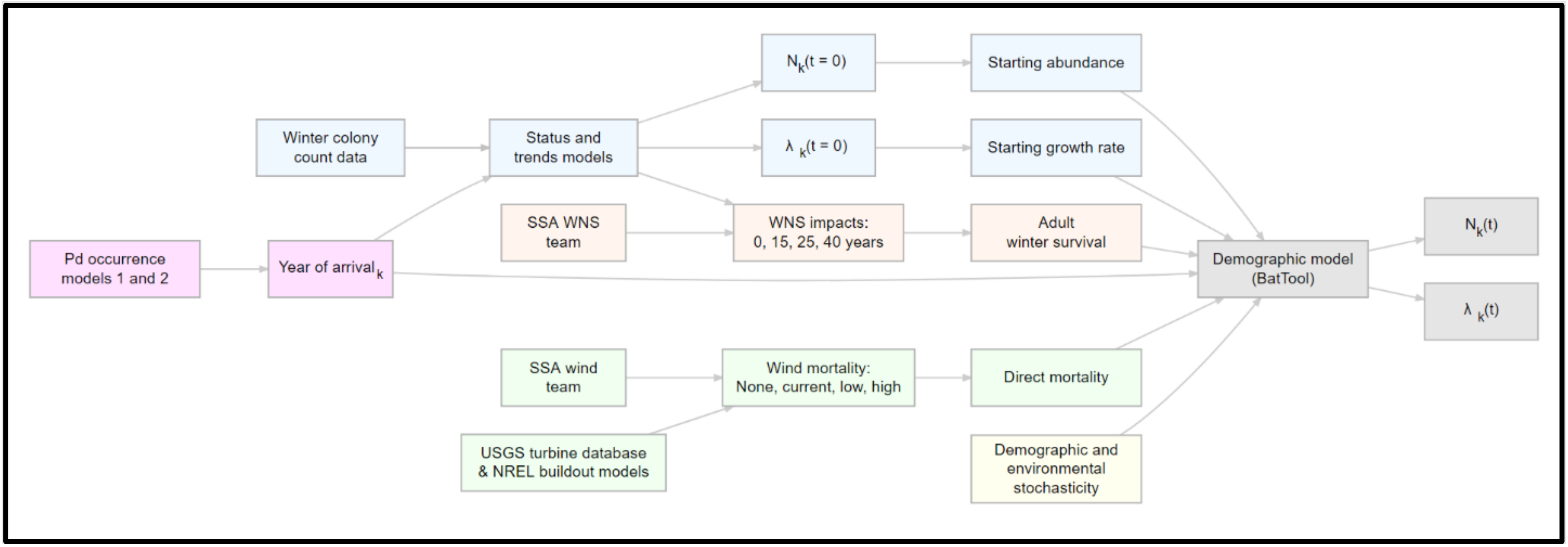

Figure A-2A7. A schematic of the BatTool, including origins of model inputs.

Summer Data Analyses

Because the population of bats monitored at a given hibernaculum disperse to many different locations on the summer landscape and because colony estimates are not available for all hibernacula, we also relied upon the results from USGS-led summer capture records and acoustic records analyses. These studies assessed the changes in occupancy (λ) and capture rates over time. We briefly describe their methodologies here; refer to Appendix 5 for further details.

Deeley and Ford (2022, entire) assessed the change in capture rates during summer surveys to garner insights on change in capture rates over time and to assess reproductive conditions of female bats, age structure, and body condition indices of male bats. Between 1999 and 2019, they analyzed NLEB in 9,885 sampling events in which 1,527 (3.6%) records had sufficient information. Rates of capture per unit effort or per sampling event were calculated for each species on an annual timescale by year and by year since arrival of *Pd* based on Wiens et al. (2022, pp. 226–229) *Pd*-occurrence estimates. Stratton and Irvine (2022, entire) assessed recent change in predicted summer occupancy using stationary and mobile acoustic detector records and capture records across NLEB's range. They developed a false-positive occupancy model to estimate probability of occurrence, annual rate of change in summertime occupancy (λ_{avg}), and total change in occupancy (λ_{tot}) from 2010 to 2019. Predicted occupancy was calculated for each 10km by 10km grid cell in NLEB's range and then aggregated to RPU and rangewide scales. The occupancy prediction used covariates of mean elevation, terrain ruggedness index, annual mean precipitation, annual mean temperature, distance to nearest wind farm, percent forest cover, and percent water cover to provide estimates in locations that were not sampled directly. Metrics of change were based on aggregating predicted occupancy between 2010 to 2019 at the RPU and rangewide scale. Whitby et al. (2022, entire) analyzed relative abundance of NLEB annually using acoustical data collected during mobile transect surveys. They analyzed the number of calls detected along driving routes and estimated changes in abundance over the past decade relative to the arrival of WNS and changes in installed wind energy facilities. These analyses were used to estimate rate of change in population at state and RPU scales.

B: Adaptive Capacity Analysis

To garner additional insights into the intrinsic (and historical) ability of NLEB to withstand stressors and adapt to novel changes in the environment, we used the framework put-forth by Thurman et al. (2020, entire). Specifically, Thurman et al. (2020, entire) developed an attribute-based framework for evaluating the adaptive capacity of a given species. Although the basis for the framework is climate change based, the attributes apply to other stressors and changes a species may be exposed to. They identified 12 "core" attributes out of their 36 potential attributes (Figure A-2B1), which collectively provide a comprehensive means of assessing adaptive capacity and are generally available for many species. For each attribute, a species is evaluated on a 5-level "low–moderate–high" scale, with criteria specified for each adaptive capacity level. They do not advise a composite level as many of the attributes interact and some may be "so important that they may overwhelm other considerations (i.e., "deal makers" or "deal breakers"). Using the criteria defined in Thurman et al. (2020, supporting information), we categorized NLEB's level of adaptive capacity for each of the 12 core attributes (Table A-2B1)

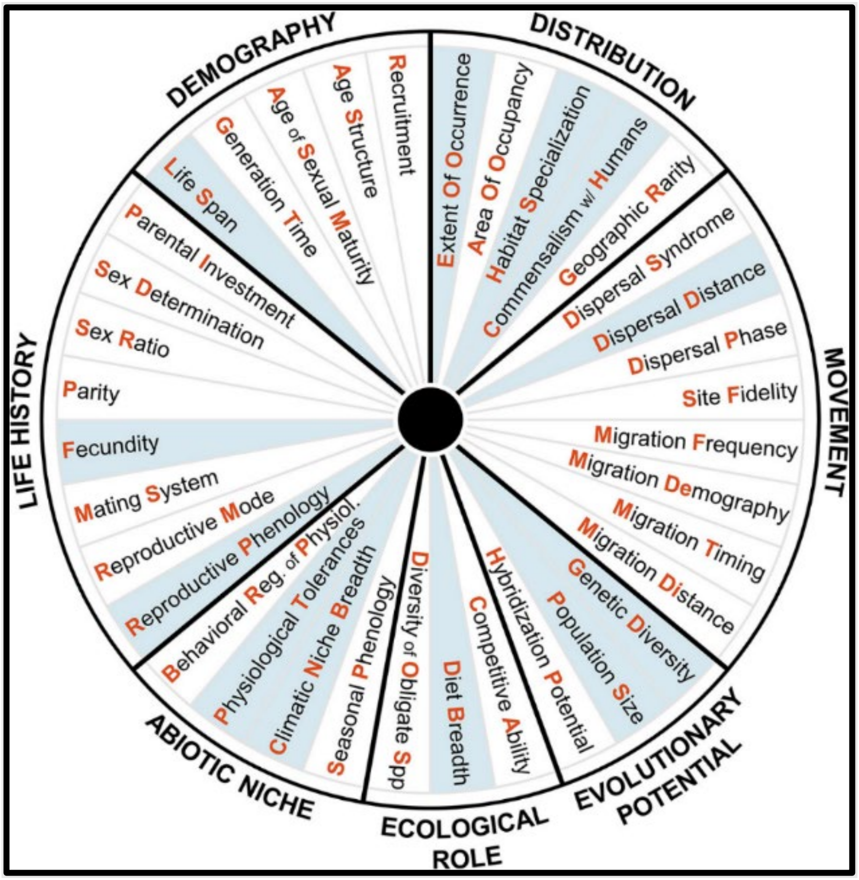

Figure A-2B1. The adaptive capacity "wheel", depicting 36 individual attributes organized by ecological complexes (or themes). Twelve core attributes, representing attributes of particular importance and for which data are widely available, are highlighted in light blue (from Thurman et al. 2020, Figure 1).

Table A-2B1. Assessment of 12 core attributes of NLEB adaptive capacity (from Thurman et al. 2020, Supporting Information).

Core Attribute	Relative Level	Evidence and Relevance
Extent of Occurrence	High	Broadly distributed; typically, a broader distribution is expected to confer higher adaptive capacity.
Habitat Specialization	Low/Moderate	Summer habitat: <u>generalist</u>; suitable roosting habitat includes trees and structures (to lesser degree). However, specific microclimates needed for successful pregnancy and recruitment. Breeding/Winter habitat: <u>specialist</u>; requires suitable hibernacula. High site fidelity in both summer and winter.
Commensalism with Humans	Moderate	Individuals or colonies infrequently utilize man-made structures as summer colony sites (e.g., barns, on utility

Core Attribute	Relative Level	Evidence and Relevance
		poles, behind window shutters, under bridges, and in bat houses) and more frequently use man-made infrastructure as hibernation sites (e.g., mines, tunnels, storm sewer, hydroelectric dam, aqueduct, dry well, crawl space). Use of human-made structures for summer roosts may occur in areas with fewer suitable roost trees (Henderson and Broders 2008, p. 960; Dowling and O'Dell 2018, p. 376).
Genetic Diversity	Low	Although there have been few wide-ranging genetic studies on this species, information collected to date indicates the species to be panmictic (random mating within a population). Johnson et al. (2014, entire) assessed nuclear genetic diversity at one site in New York and several sites in West Virginia, and found little evidence of population structure in NLEB at watershed or regional scales. In addition, studies conducted in Ohio, Nova Scotia and Quebec, Canada, and Kentucky showed variation in NLEB haplotypes at local levels; however, these studies also indicated relatively low levels of overall genetic differentiation between groups and high levels of diversity overall (Arnold 2007, p. 157; Johnson et al. 2015, p. 12; Olivera-Hyde et al. 2020, p.729).
Population Size	Low	Once common, populations have decreased significantly; adaptive capacity may decrease with smaller populations.
Dispersal Distance	Moderate/High	May migrate short distances- up to 89 km (55 mi) between summer and winter habitat (Griffin 1940b, pp. 235, 236; Caire et al. 1979, p. 404; Nagorsen and Brigham 1993 p. 88)
Climatic Niche Breadth	High	Broad climatic niche breadth across range; may indicate a broader tolerance to climate change because they currently encompass a broader array of climate conditions.
Physiological Tolerances	Moderate	NLEB can employ torpor during food shortages, if conditions allow (even in summer). Clustering and roost selection behavior help to limit the physiological stress from cold or heat waves.
Diet Breadth	High	Use hawking and gleaning foraging behavior. Diverse diet including moths, flies, leafhoppers, caddisflies, and beetles (Griffith and Gates 1985, p. 452; Nagorsen and Brigham

Core Attribute	Relative Level	Evidence and Relevance
		1993, p. 88; Brack and Whitaker 2001, p. 207), with diet composition differing geographically and seasonally (Brack and Whitaker 2001, p. 208). Lepidopterans and coleopterans (beetles) are most commonly found insects in NLEB diet (Brack and Whitaker 2001, p. 207; Lee and McCracken 2004, pp. 595–596; Feldhamer et al. 2009, p. 45; Dodd et al. 2012, p. 1122), with arachnids also being a common prey item (Feldhamer et al. 2009, p. 45)
Reproductive Phenology	Low	Copulation occurs in fall and winter. Females ovulate in the spring upon emergence from hibernacula and fertilization occurs soon after; duration of hibernation and timing of spring emergence is variable across the range. Copulation occasionally occurs again in the spring (Racey 1982, p. 73), and can occur during the winter as well (Kurta 2013, in litt.).
Life Span	Moderate/Low	Maximum NLEB lifespan is estimated to be up to 18.5 years (Hall et al. 1957, p. 407).
Fecundity	Low	A reproductive female can produce up to one offspring annually.

Appendix 3: Supplementary Results

A: Historical Condition

Table A-3A1. The historical number of states/provinces, spatial extent (Extent of Occurrence: EOO), winter abundance, and documented hibernacula rangewide.

# of States/Provinces	EOO (acres)	# of hibernacula	Abundance (max)
29/3	1.2 billion	737	38,131

Table A-3A2. The historical number of hibernacula and winter abundance by RPU.

RPU	# of Hibernacula	Abundance (max)
East Coast	8	1,460
Eastern Hardwoods	665	29,775
Midwest	9	1,218
Southeast	50	393
Subarctic	5	5,628

B: Current Condition

Table A-3B1. Projected yearly rangewide number of states, spatial extent (EOO in acres), number of hibernacula, and median abundance under ***current*** *conditions.*

Year	# of States	EOO (ac)	# of hibernacula	Abundance (median)
2020	18	644 million	139	19,356
2030	7	294 million	11	1,889
2040	2	0	1	540
2050	1	0	0	409
2060	0	0	0	230

Table A-3B2. Projected RPU-level number of hibernaculaand probability of population growth (λ)>1 (pPg) under ***current*** *conditions.*

RPU	Year	# of Hibs	pPg
Southeast	2020	1	0
	2030	1	0.24
	2040	0	0.04
	2050	0	0.03
	2060	0	0.02
Subarctic	2020	5	0.11
	2030	0	0.30

RPU	Year	# of Hibs	pPg
	2040	0	0.30
	2050	0	0.20
	2060	0	0.21
Eastern Hardwoods	2020	115	0
	2030	10	0.19
	2040	1	0.47
	2050	0	0.47
	2060	0	0.64
East Coast	2020	1	0.07
	2030	1	0.36
	2040	0	0.09
	2050	0	0.03
	2060	0	0.01
Midwest	2020	5	0.08
	2030	0	0.24
	2040	0	0.14
	2050	0	0.06
	2060	0	0.07

Table A-3B3. Projected RPU median abundance (90% CI) under current conditions.

RPU	2020	2030	2040	2050	2060
Southeast	298 (CI 298 – 298)	2 (CI 0 – 20)	0 (CI 0 – 6)	0 (CI 0 – 0)	0 (CI 0 – 0)
Subarctic	5,630 (CI 5,630 – 5,630)	16 (CI 0 – 4,118)	0 (CI 0 – 3,328)	0 (CI 0 – 6,459)	0 (CI 0 – 10,601)
Eastern Hardwoods	13,119 (CI 13,076 – 13,162)	1,576 (CI 149 – 6,151)	390 (CI 0 – 3,673)	252 (CI 0 – 2,482)	130 (CI 0 – 1,554)
East Coast	187 (CI 186 – 188)	4 (CI 0 – 50)	0 (CI 0 – 14)	0 (CI 0 – 0)	0 (CI 0 – 0)
Midwest	122 (CI 108 – 136)	4 (CI 0 – 72)	0 (CI 0 – 70)	0 (CI 0 – 44)	0 (CI 0 – 42)

Table A-3B4. Summary of recent NLEB population trends from multiple data types and analyses. Winter Colony analysis –(Chapter 5); Summer Occupancy analysis –Stratton and Irvine (2022, entire); Summer Capture analysis – Deeley and Ford (2022, entire); and Summer Mobile Acoustic analysis – Whitby et al. (2022, entire). [1] *No data available.*

Representation Unit	Winter colony	Summer occupancy	Summer capture	Summer mobile acoustic
Southeast	-24%	-85%	-47%	-50%
Eastern Hardwoods	-56%	-78	-43%	-87%
Subarctic	-0%	-63%	-[1]	-[1]
Midwest	-90%	-87%	-77%	-99.9%
East Coast	-87%	-79%	-43%	-69%
Rangewide	-49%	-80%	-43% – 77%	-79%

C: **Future Condition**

*Table A-3C1. Projected rangewide number of states and known hibernacula with 1 or more bats persisting, spatial extent (EOO), number of hibernacula, and population abundance under **future** scenarios.*

Year	**# of States**	**EOO (ac)**	**# of hibernacula**	**Abundance (median)**
2030	6	294 million	9	1,801
2040	4	0	1	460
2050	0	0	0	324
2060	0	0	0	201

*Table A-3C2. Projected RPU-level number of hibernacula and probability of population growth (λ)>1 (pPg) over time under **future** scenarios.*

RPU	**Year**	**# of Hibs**	**pPg**
Southeast			
	2030	0	0.19
	2040	0	0.06
	2050	0	0.03
	2060	0	0.02
Subarctic			
	2030	0	0.28
	2040	0	0.33
	2050	0	0.23

RPU	Year	# of Hibs	pPg
	2060	0	0.22
Eastern Hardwoods			
	2030	8	0.20
	2040	1	0.50
	2050	0	0.52
	2060	0	0.63
East Coast			
	2030	1	0.28
	2040	0	0.10
	2050	0	0.01
	2060	0	0.00
Midwest			
	2030	0	0.27
	2040	0	0.19
	2050	0	0.10
	2060	0	0.12

Table A-3C3. Projected RPU median abundance (90% CI) under future scenarios.

RPU	2020	2030	2040	2050	2060
Southeast	298 (CI 298 – 298)	2 (CI 0 – 18)	0 (CI 0 – 2)	0 (CI 0 – 0)	0 (CI 0 – 0)
Subarctic	5,630 (CI 5,630 – 56,30)	22 (CI 0 – 6,199)	0 (CI 0 – 5,199)	0 (CI 0 – 9,974)	0 (CI 0 – 11,830)
Eastern Hardwoods	13,119 (CI 13,076 – 13,162)	1,358 (CI 107 – 5,040)	294 (CI 0 – 3,385)	174 (CI 0 – 2,486)	66 (CI 0 – 2,297)
East Coast	187 (CI 186 – 187)	6 (CI 0 – 46)	0 (CI 0 – 12)	0 (CI 0 – 0)	0 (CI 0 – 0)
Midwest	122 (CI 108 – 136)	2 (CI 0 – 74)	0 (CI 0 – 73)	0 (CI 0 – 83)	0 (CI 0 – 65)

D: Qualitative/Comparative Threat Analysis

To estimate the proportion of NLEB's range with wind mortality risk in 2020, we took the following approach:

1. Buffer extant (known) hibernacula by avg. migration distance (89 km)
2. Buffer summer points by avg. migration distance (89 km)

3. Merge & dissolve buffered hibernacula and summer shapefiles into a “NLEB occupied” area, clip NLEB range by contiguous U.S. border for “NLEB U.S. range”, and clip NLEB occupied area by NLEB U.S. range.
4. Buffer & dissolve current turbines (Hoen et al. 2018) by avg. migration distance for “wind threat” area (89 km)
5. Clip wind threat area by NLEBs occupied area for “NLEB wind risk” area
6. Compare NLEB wind risk area with range area in U.S.: NLEB = 3,378,317 km^2 and 2020 wind risk area (U.S.): NLEB = 1,650,889 km^2 (49% of U.S. range) (Figure A-3D1)

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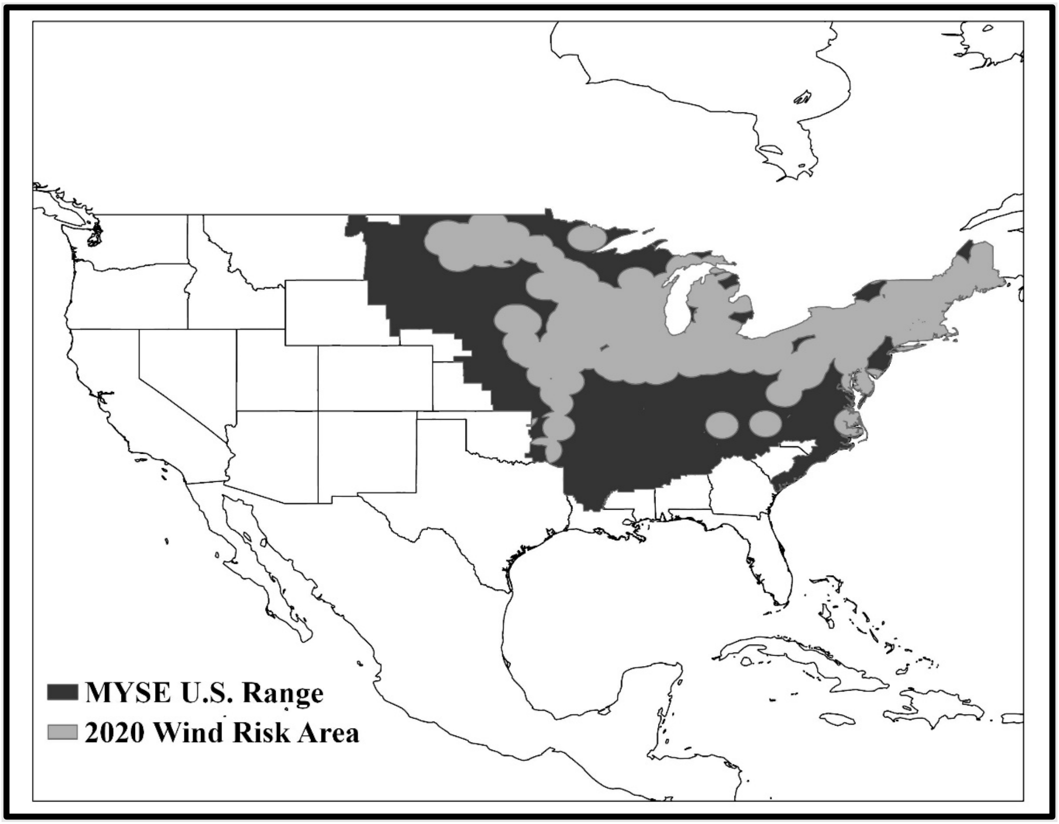

Figure A-3D1. Estimated extent of NLEB's U.S. range with wind mortality risk.

To estimate the proportion of NLEB's range with wind mortality risk in 2050 (per low and high build-out scenarios), we took the following approach:

1. 2050 Low Build-out Scenario:
    a. Buffer & dissolve 2050 High Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance for "future wind threat: area. *Note: Future MW summed by 11x11-km NREL grid cell so does not capture actual distribution of turbines on landscape* (89 km)
    b. Clip wind threat area by NLEB occupied areas for "NLEB 2050 low wind risk" area (U.S.)
    c. Compare NLEB 2050 low wind risk areas with range area in U.S.
        i. Range area (U.S.): 3,378,317 km^2
        ii. 2050 low wind risk areas = 937,019 km^2 (28% of U.S. range) (Figure A-3D2*)*
2. 2050 High Build-out Scenario:
    a. Buffer & dissolve 2050 Low Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance for "future wind threat" area. *Note: Future MW summed by 11x11-km NREL grid cell so does not capture actual distribution of turbines on landscape*: 89 km

b. Clip wind threat area by NLEB occupied areas for “NLEB 2050 high wind risk” area (U.S.)

c. Compare NLEB 2050 high wind risk areas with range area in U.S.
    i. Range area (U.S.): 3,378,317 km^2
    ii. 2050 high wind risk areas (U.S.): 2,374,707 km^2 (70% of U.S. range) (Figure A-3D3*)*

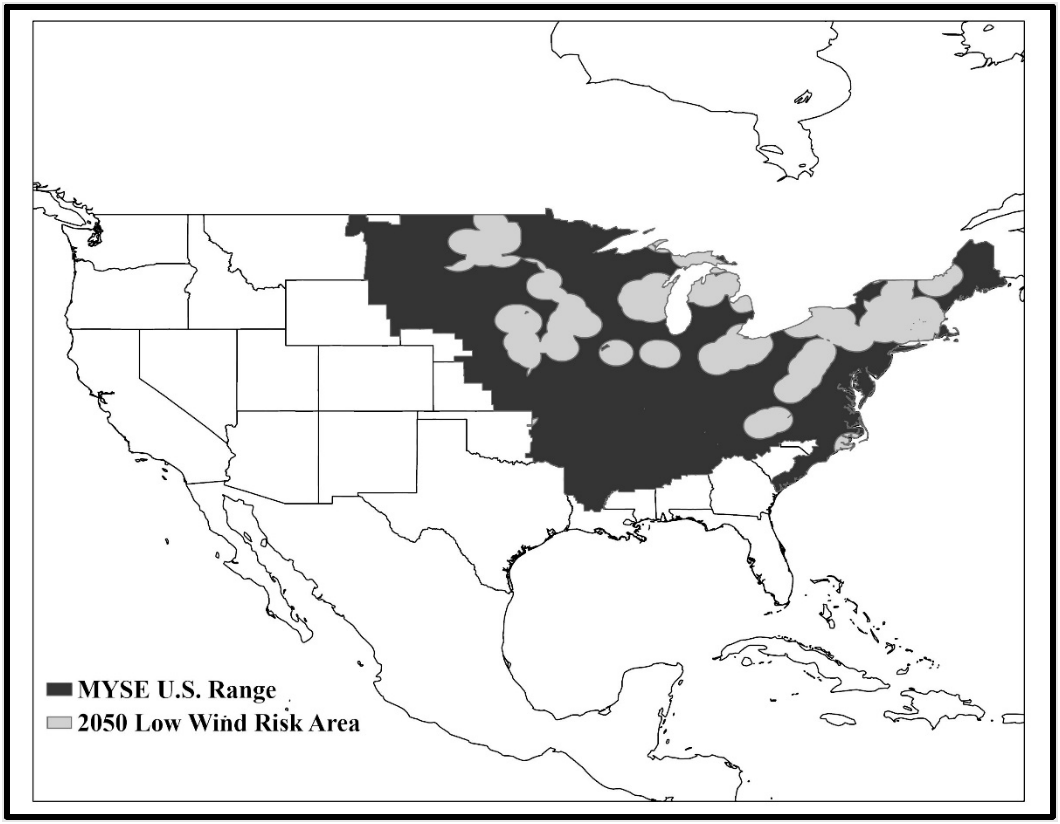

Figure A-3D2. Estimated extent of NLEB’s U.S. range with wind mortality risk in 2050 low build-out scenario.

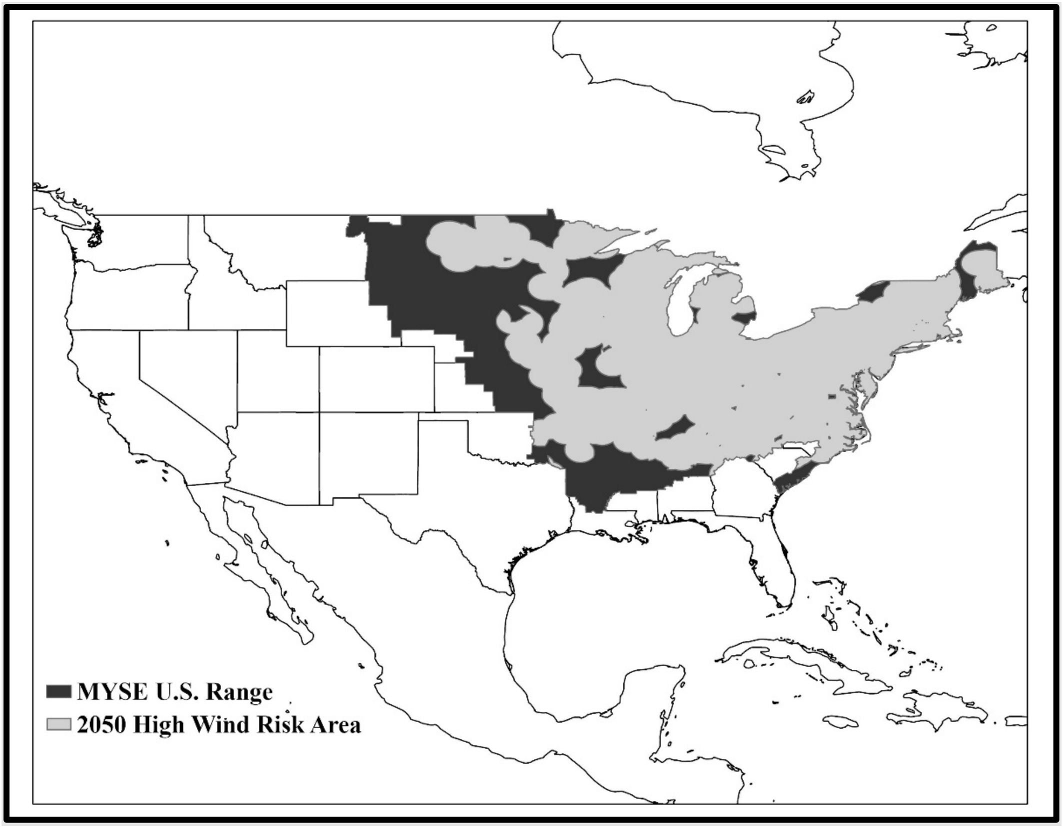

Figure A-3D3. Estimated extent of NLEB's U.S. range with wind mortality risk in 2050 high build-out scenario.

To estimate the severity of impact from wind energy related mortality, we compared scenarios to baseline scenarios without wind energy mortality. The results are presented in Tables A-3D1 and A-3D2.

Table A-3D1. Projected median rangewide abundance given wind energy mortality under 4 current conditions scenarios: 1) Pd model 1 and current wind energy related mortality, 2) Pd model 1 and no wind energy related mortality, 3) Pd model 2 and current wind energy related mortality, 4) Pd model 2 and no wind energy related mortality.

Scenario	2030	2040	2050	2060
Pd Model 1 – Current mortality	804	409	1,071	2,241
Pd Model 1 – No mortality	538	152	446	858
% change	-33%	-63%	-58%	-62%
Pd Model 2 – Current mortality	4,785	2,064	1,108	643
Pd Model 2 – No mortality	3,615	1,055	526	176
% change	-24%	-49%	-53%	-73%

Table A-3D2. Projected median rangewide abundance given wind energy mortality under 4 future conditions scenarios: 1) Pd model 1 and future wind energy related mortality, 2)

Pd model 1 and no wind energy related mortality, 3) Pd model 2 and future wind energy related mortality, 2) Pd model 2 and no wind energy related mortality.

Scenario	2030	2040	2050	2060
Pd Model 1 – low impact mortality	546	108	201	340
Pd Model 1 – future no mortality	719	383	1,026	1,831
% change	-24%	-72%	-80%	-81%
Pd Model 2 – high impact mortality	3960	1197	397	142
Pd Model 2 – future no mortality	4,938	2,210	1,278	857
% change	-20%	-46%	-69%	-83%

Appendix 4: Supplemental Threat and Future Scenario Information

A: WNS

Background

White-nose syndrome (WNS) is a disease of bats that is caused by the fungal pathogen *Pseudogymnoascus destructans (Pd)* (Blehert et al. 2009, entire; Turner et al. 2011, entire; Lorch et al. 2011, entire; Coleman and Reichard 2014, entire; Frick et al. 2016, entire; Bernard et al. 2020, entire; Hoyt et al. 2021, entire). The disease and pathogen were first observed in eastern New York in 2007 (with photographs showing presence since 2006; Meteyer et al. 2009, p. 411), although it is likely the pathogen existed in North America for a short time prior to its discovery (Keller et al. 2021, p. 3; Thapa et al. 2021, p. 17). Since then, *Pd* and WNS have spread to 39 states and 7 provinces, with lesions indicative of disease confirmed in 12 species of North America bats, including NLEB (Figure A-4A1, www.whitenosesyndrome.org; accessed May 13, 2021; Hoyt et al. 2021, Suppl. Material). *Pd* invades the skin of bats, leading to significant morbidity and mortality that causes drastic declines in multiple species of hibernating bats.

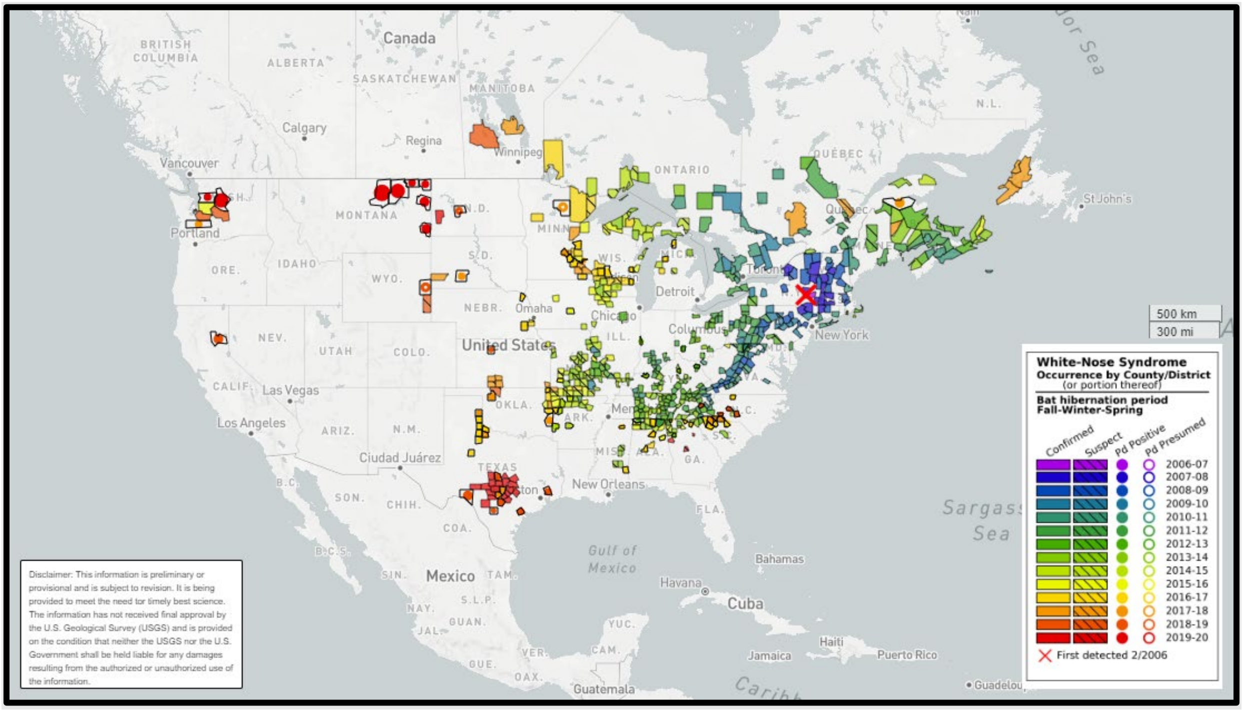

Figure A-4A1. Occurrence of Pd and WNS in North America based on surveillance efforts in the U.S. and Canada: disease confirmed (color-coded), suspected (stripes), Pd detected but not confirmed (solid circles), and Pd detected but inconclusive lab results (open circles) (www.whitenosesyndrome.org, accessed online: May 13, 2020).

White-nose Syndrome
As with any disease, there are three critical elements necessary for WNS to manifest: the pathogen, *Pd*; the host, hibernating bats; and a favorable environment for them to interact, the mainly subterranean hibernacula of bats (Turner et al. 2011, pp. 20–21).

- The *pathogen* that causes WNS, *Pseudogymnoascus destructans* (Gargas et al. 2009, pp. 151–152, Lorch et al. 2011, entire, Minnis and Lindner, 2013, p. 644) grows at cold temperatures ranging from 0–21 degrees C, with optimal growth temperature of 12–16 degrees C (Verant et al. 2012, p. 3), thus it is adapted to grow in conditions characteristic of bat hibernacula. It grows by invading the epidermis and underlying tissues of the face, ears and wings of bats (Meteyer et al. 2009, entire).
- The *hosts*, hibernating bats, are susceptible to infection by *Pd* in part because the physiological, physical and behavioral attributes associated with prolonged use of torpor present the opportunity for this cold-loving fungus to invade their tissues (Lorch et al. 2011, p. 2; Langwig et al. 2012, p. 4; Reeder et al. 2012, p. 4). In particular, hibernating bats overwinter in alternating states of torpor and euthermia (i.e., arousal) to survive prolonged periods without eating (McNab, 1982, p. 171). To use limited fat stores efficiently, metabolic rates are greatly reduced, along with immune functioning and other physiological processes (Moore et al. 2011, p. 8).
- The *environment* where *Pd* and bats interact to cause disease is typically a winter roost location where bats engage in fall swarming and hibernation. The conditions of these locations overlap with the suitable growth requirements for *Pd* (Verant et al. 2012, p. 4). Hibernacula are often assumed to be caves and mines that provide overwinter shelter for large aggregations of hibernating bats, but these essential habitats take many forms and are used by individual bats to large, multi-species colonies. In North America, bats have been documented overwintering in caves, mines, rock crevices, talus, tunnels, bunkers, basements, bridges, aqueducts, trees, earthen burrows, leaf litter, and a variety of other roosts. For bats to hibernate successfully, the most important conditions are relatively stable- low temperatures, but generally above freezing, and high humidity (Perry, 2013, p. 28). Notably, many North American hibernating bats select winter roosts that range between –4 and 16 degrees C (0.6 degrees C to 13.0 degrees C for NLEB; summarized in Webb et al. 1996, p. 763). The overlap of these roost conditions and suitable growth conditions for *Pd* (reported above), combined with the behavioral and physiological characteristics of their torpid state, are the primary factors making hibernating bats so susceptible to infection by *Pd*.

WNS is diagnosed histologically with the identification of "cup-like erosions" as *Pd* invades the skin tissue causing dehydration (Meteyer et al. 2009, p. 412). This fungal invasion destroys the protective skin tissue and disrupts water and electrolyte balance that is important to sustaining homeostasis through hibernation (Cryan et al. 2010, pp. 3–4; Warnecke et al. 2013, pp.3–4). Likely in response to the homeostatic imbalance and irritation of the skin, *Pd* infection leads to increases in the frequency and duration of arousals during hibernation and raises energetic costs during torpor bouts, both of which cause premature depletion of critical fat reserves (Reeder et al. 2012, p. 5; McGuire et al. 2017, p. 682; Cheng et al. 2019, p. 2). As a result, WNS leads to starvation as sick bats run out of fat needed to support critical biological functions.

Bats suffering from WNS may exhibit a variety of behavioral changes that can alter the course of morbidity from the disease. In addition to altered arousal patterns, bats have been observed relocating to different areas of hibernacula where conditions may be advantageous for hibernation or disadvantageous for *Pd* growth (Turner et al. 2011, p. 22; Langwig et al. 2012, p. 2; Johnson et al. 2016, p. 189). Observed changes in clustering behavior such that a greater proportion of bats in a colony are seen hibernating solitarily after WNS is present rather than huddled with roost mates may point to a behavioral factor that affects severity of WNS (Langwig et al. 2012, p. 2; Kurta and Smith 2020, p. 769), but may also be a maladaptive response to experiencing symptoms of WNS (Wilcox et al. 2014, p. 162). In many situations, infected bats have been documented exiting hibernacula earlier than usual and prior to when surface conditions are suitable for spring emergence. Early emergence has also been observed during daylight hours when diurnal predators such as hawks and ravens can take advantage of bats weakened by disease. It is possible that bats may find water to drink and insects to prey upon at this time, especially in more moderate climates, thus supplementing depleted energy reserves (Bernard and McCracken, 2017, p. 1492–1493), but in much of NLEB's range, exposure to winter conditions and predation pose a significant threat to animals evacuating from hibernacula. Whether within the roost or on the landscape, WNS causes high rates of mortality during the hibernation season for multiple species (Turner et al. 2011, entire; Cheng et al. 2021, entire).

The weeks following emergence from hibernation also mark a critical period when bats incur energetic costs of clearing infection and recovering from over-winter sickness (Reichard and Kunz 2009, p. 461; Meteyer et al. 2012, p. 3; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8). Meteyer et al. (2012, p. 3) proposed that bats with WNS can also suffer from immune reconstitution inflammatory syndrome, or IRIS. In this potentially fatal condition, deep or systemic infections that developed during hibernation while immune function was down-regulated trigger an excessive inflammatory response as immune function is upregulated in the spring (Meteyer et al. 2012, p. 5). Additionally, heavily compromised wing conditions resulting from overwinter infections and healing processes are likely to further limit foraging efficiency as the integrity of flight membranes is altered (Reichard and Kunz 2009, p. 462; Fuller et al. 2012, p. 6). These post-emergence complications can lead directly to mortality in addition to impacting reproductive success as a result of energetic constraints and trade-offs (Reichard and Kunz 2009, p. 462; Frick et al. 2010, p. 131; Field et al. 2015, p. 20; Fuller et al. 2020, pp. 7–8).

Transmission of Pd among bats

The fungus is spread via bat-bat and bat-environment-bat movement interactions (Lindner et al. 2011, p. 246; Langwig et al. 2012, p. 1055). Transmission occurs primarily in the fall and winter months when bats aggregate in hibernacula (Langwig et al. 2015a, p. 4). In spring, bats that survive a winter exposed to *Pd* can rid themselves of the fungus such that individuals are largely free of *Pd* at summer roosts (Dobony et al. 2011, p. 193; Langwig et al. 2015a, p. 4). However, it is not uncommon for some bats to be found carrying viable *Pd* later into summer (Dobony et al. 2011, p. 193; Ineson, 2020, p. 104) and *Pd* is capable of remaining viable in hibernacula without bats for extended periods (Lorch et al. 2013, p. 1298). The cool, humid conditions of hibernacula likely serve as environmental reservoirs for the fungal pathogen where it can survive and even proliferate until bats return in the fall (Reynolds et al. 2015, p. 320; Hoyt et al. 2020, p. 7259). Generally, bats return to winter roosts in the fall and engage in social interactions that lead to rapid spread of *Pd* from the environmental reservoir to the population (Hoyt et al. 2020, p. 7256). However, because hibernacula may be used throughout the year by males and non-reproductive females who hibernate there, as well as by other species that are more transient, including long distance migrants, some transmission is likely to occur year round and by other mechanisms.

Expansion of Pd in North America

Since it was first detected in New York, the range of *Pd* in North America has increased steadily via bat to bat transmission, although activities of humans, including scientific research, recreational activity, and shipping are also likely to contribute to some short and long distance movements (Bernard et al. 2020, p. 5–6). Simply, *Pd* has spread from just a small number of sites in New York in 2007 to hundreds of locations across the continent in just 14 years. Several predictive models have identified biological, geological, climatic, ecological and behavioral variables correlated with the patterns and timing of its expansion (Hallam and Federico, 2012, p. 2; Maher et al. 2012, p. 3; Alves et al. 2014, p. 2; Hefly et al. 2020, pp. 10–11). Putative barriers to *Pd* expansion have been hypothesized, but these generally have provided very short-term delays in *Pd's* steady progression into uncontaminated areas (Miller-Butterworth et al. 2014, p. 9; Hoyt et al. 2021 p. 3). While these obstacles to natural disease spread may delay arrival of *Pd*, when the fungus does pass them either via dispersing bats or via inadvertent transport by humans, it has led to disease and continued spread of the fungus on the other side (Miller-Butterworth et al. 2014, p. 9; Lorch et al. 2016, p. 4). Because the above published models have fallen behind reality in their predictions, we used two models to describe past occurrence of *Pd* and to predict its future expansion in North America (see *Figure A-2A4, methods described above).*

Establishment of Pd

With the arrival of *Pd* at a new location, progression of the disease proceeds similarly to many emerging infectious diseases through stages of invasion, epidemic, and establishment (Langwig et al. 2015b, p. 196; Cheng et al. 2021, entire). During *invasion* (years 0–1), the fungus arrives on a few bats and spreads through the colony until most individuals are exposed to and carry it. As the amount of *Pd* on bats and in the environmental reservoir increases, the *epidemic* (years 2–4) proceeds with high occurrence of disease and mortality. By the fifth year after arrival of *Pd*, the pathogen is *established* (years 5–7) in the population. Then 8 years after its arrival, *Pd* is determined to be *endemic* (Langwig et al. 2015b, p. 196; Cheng et al. 2022, p. 205). Although methods for detecting *Pd* have changed over time, it is apparent with few exceptions that morbidity and mortality associated with WNS occurs within a year or two after *Pd* has been observed in a population (Frick et al. 2017, pp. 627–629; Hoyt et al. 2020 p. 7259). With the publication by Muller et al. (2013, entire), the use of polymerase chain reaction (PCR) to confirm the presence of *Pd* became the gold standard for diagnosing WNS. This technique provided greater confidence in *Pd* detection and improved our understanding of the disease progression.

Langwig et al. (2015a, pp. 3–4) and Hoyt et al. (2020, p. 7257) quantified the proportion of bats on which *Pd* is detected (prevalence) and the amount of *Pd* on bats (load) in the years after *Pd* invades and establishes itself in a site. In general, when *Pd* is first detectable (by PCR), a relatively small number of bats carry the fungus in low loads. These values increase throughout the first winter at varying rates among species. By the end of the first winter, *Pd* is detectable both on bats and on surfaces of the roost. In the second year after detection, *Pd* loads and prevalence pick up near where they were the previous year; prevalence and load are at significantly higher levels in the fall and early winter, and prevalence approaches 1 (i.e., all bats are infected) by mid-winter for NLEB (Frick et al. 2017, p. 627).

There are a few exceptions in which evidence of *Pd* has been detected in a site and then not detected at that site in subsequent years. These occurrences may represent failed invasions by *Pd*. In Iowa, for example, molecular tests revealed evidence suggestive of *Pd* being present, but WNS was not confirmed at that location for several more years. In California, *Pd* has not been detected in two subsequent years after initial evidence was detected (S. Osborne 2021, California Department of Fish and Game, personal communication). There are also examples that do not fit the expected disease progression described above. At Tippy Dam in Michigan, *Pd* has been present for over 5 years without indication of WNS in little brown bats, although NLEB are no longer observed at this location (Kurta et al. 2020, p. 584). The factors contributing to this atypical scenario are under investigation. It has also been posited that WNS may have a southern limit where disease is less likely to impact populations (Hallam and Federico 2012 p. 9; Hoyt et al. 2021, pp. 6–7). Nevertheless, the overwhelming pattern has been that WNS develops in a population soon after the arrival of *Pd*. Still, because environmental reservoirs of the pathogen play an important role in its transmission, hibernacula that become unsuitable for *Pd* during summer (e.g., too warm or dry) may reduce the amount of fungus in the environment between hibernation seasons, leading to lesser or delayed development of WNS (Hoyt et al. 2020, pp. 7257–7258). To date, these exceptions where colonies experience less severe impacts from WNS compared to the majority of colonies are not reliably predictable based on geographic or biological features, although see *"Persistence of impacted populations"* below.

Impacts of WNS

The impacts of white-nose syndrome are severe among species that were the first observed with the disease. This pattern has remained true over a large area as *Pd* has continued to expand its range affecting previously unexposed colonies of hibernating bats. Four years after the discovery of WNS, Turner et al. (2011) estimated total declines of 98% for NLEB at 42 sites with WNS in Vermont, New York, and Pennsylvania. Later, with data from six states (Vermont, New York, Pennsylvania, Maryland, Virginia, West Virginia), Frick et al. (2015) estimated that median colony size decreased by 90% and NLEB was extirpated from 69% of historical hibernacula (Frick et al. 2015. P. 5). Hoyt et al. (2021, p. 7) summarized overall declines from WNS to be "drastic" for NLEB in both the Northeast and Midwest regions. Using data from 27 states and 2 provinces, the most complete dataset available at the time, Cheng et al. (2021, entire) reported similar patterns. They estimated that WNS has caused 97–100% decline in NLEB across 79% of their range (Figure 4.4; Cheng et al. 2021, entire). Although there are ecological and environmental differences across the currently affected regions of North America, WNS has consistently caused significant declines in populations of NLEB (Figure 4.4), with very few examples of colonies that are avoiding the impacts (Figure 4.6).

Conservation Measures Associated with WNS

There are multiple national and international efforts underway in an attempt to reduce the impacts of WNS. To date, there are no proven measures to reduce the severity of impacts.

Efforts associated with the national response to WNS were initially aimed at determining the cause of the disease and reducing or slowing its spread. The response broadened and was formalized by the *National Plan for Assisting States, Federal Agencies, and Tribes in Managing White-nose Syndrome in Bats* which provides the strategic framework for implementation of a collaborative, national response to WNS by State, Federal, Tribal and non-governmental partners (USFWS 2011, entire). The U.S. plan integrates closely with a sister plan for Canada, assuring a coordinated response across much of North America. Implementation of the WNS National Plan is overseen by executive and steering committees comprising representation from the Department of Interior, Department of Agriculture, Department of Defense, and state wildlife agencies under the authority of a multi-species recovery team under the ESA, with the USFWS serving the lead coordinating role. In 2021, the WNS National Plan is being revised to reflect current state of knowledge and identify key elements to continue to effectively respond to this disease. Goals and actions address the greatest needs and knowledge gaps to be pursued, including: coordinated disease surveillance and diagnostic efforts; inter-programmatic data management; development and implementation of disease management, conservation and recovery strategies; and communication and outreach among partners and with the public. These efforts are also supported by the North American Bat Monitoring Program (NABat), which is co-led by USGS and USFWS, to integrate data across jurisdictional borders in support of population level information that supports management decisions at different scales. Actions under the National Plan are intended to be supported through multiple funding programs in different agencies. For several years, many state, Federal, Tribal, and private partners have annually provided funding and physical efforts or both toward WNS research. For its part, the USFWS supports management activities of many partners, research to address key information needs, and development and application of management solutions. The USFWS maintains a website (www.whitenosesyndrome.org) and social media accounts to address many of the communication needs for both internal and external audiences.

Over 100 state and Federal agencies, Tribes, organizations and institutions are engaged in this collaborative work to combat WNS and conserve affected bats. Partners from all 37 states in NLEB’s range, Canada, and Mexico are engaged in collaborations to conduct disease surveillance, population monitoring, and management actions in preparation for or response to WNS.

B: Wind

Background

Wind power is a rapidly growing portion of North America’s clean energy sector due to its small footprint, lack of carbon emissions, changes in state’s renewable energy goals and recent technological advancements in the field allowing turbines to be placed in less windy areas. As of 2019, wind power was the largest source of renewable energy in the country, providing 7.2% of U.S. energy (American Wind Energy Association 2020, p. 1). Modern utility-scale wind power installations (wind facilities) often have tens or hundreds of turbines installed in a given area, generating hundreds of MW of energy each year. Installed wind capacity in the U.S. as of 2020 was 104,628 MW (Hoen et al. 2018, entire; USFWS unpublished data).

Wind related NLEB mortality, while often overshadowed by the disproportionate impacts to tree bats and by the enormity of WNS, is also proving to be a consequential stressor at local and regional levels. The remarkable potential for bat mortality at wind facilities became known around 2003, when post-construction studies at the Buffalo Mountain, Tennessee, and Mountaineer, West Virginia, wind projects documented the highest bat mortalities reported at the time[8] (31.4 bats/MW and 31.7 bats/MW, respectively; Kerns and Kerlinger 2004, p. 15; Nicholson et al. 2005, p. 27). Bat mortalities continue to be documented at wind power installations across North America.

Mechanism behind bat mortality

Most bat mortality at wind energy projects is caused by direct collisions with moving turbine blades (Grodsky et al. 2011, p. 920; Rollins et al. 2012, p. 365). Barotrauma--a rapid air pressure change causing tissue damage to air-containing structures such as the lungs—may also contribute to bat mortality (Baerwald et al. 2008, pp. 695–696; Cryan and Barclay 2009, p. 1331; Rollins et al 2012, p. 368–369; Peste et al. 2015, p. 11), although impact trauma is likely the cause of most wind-related bat mortality (Lawson et al. 2020; entire).11). Grodsky et al. (2011, 924) further hypothesize that direct collision with turbine blades may cause delayed lethal effects (i.e., injured bats may leave the search area before succumbing to injuries; turbines may damage bats' ears, negatively affecting their ability to echolocate, navigate, and forage), thus causing an underestimation of true bat mortality.

Bats may be attracted to turbines (Solick et al. 2020, entire; Richardson et al. 2021, entire), though support for this is limited. Some hypotheses for bat attraction to wind turbines include the sound of moving blades, blade motion, insect aggregations near these structures, turbines as potential roost structures, and turbines as mating locations (Kunz et al. 2007, pp. 317–319, 321; National Research Council 2007, p. 97; Cryan and Barclay 2009, pp. 1334–1335, Cryan et al. 2014 p. 15128). Horn et al. (2008a, p. 14; 2008b, p. 126) observed bats flying within the turbine blade's rotor swept zone at wind projects in New York and West Virginia and noted that bats were actively feeding and foraging around moving and non-moving blades (2008b, p. 130), while Cryan et al. (2014, p. 15127) observed bats altering course towards turbines using thermal imagery.

Bat mortality tends to exhibit a seasonal pattern, with mortality peaking generally in the late summer and early fall (Erickson et al. 2002, p. 39; Arnett et al. 2008, p. 65; Taucher et al. 2012, pp. 25–26; Bird Studies Canada et al. 2018, pp. 28, 32, 33, 46). Based on our analysis, 6.5, 25.5, and 68.0% of bat fatalities occur during the spring, summer, and fall periods, respectively (USFWS 2016, pp. 4-12, 4-15). Temperature and wind speed may also indirectly influence bats risk of collision risk with wind turbines. Bat activity is higher during nights of low wind speed and warmer temperatures (Arnett et al. 2006, p. 18), and is lower during periods of rain, low temperatures, and strong winds (Anthony et al. 1981, 154–155; Erkert 1982, pp. 201–242; Erickson and West 2002, p. 22; Lacki et al. 2007, p. 89).

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[8] Higher wind fatality rates have since been reported (e.g., Schirmacher et al. 2018, p. 52; USFWS 2019, p. 32 and 69).

Bat Mortality

Bat mortality varies across wind facilities, between seasons, and among species. Consistently, three species–hoary bats (*Lasiurus cinereus*), silver-haired bats (*Lasionycteris noctivagans*), and eastern red bats (*Lasiurus borealis)*–comprise the majority of all known bat fatalities (e.g., 74–90%). The disproportionate amount of fatalities involving these species has resulted in less attention and concern for other non-listed bat species. However, there is notable spatial overlap between NLEB occurrences and wind facilities along with NLEB mortality documented (Figure 4.7). Based on October 2020 installed MW capacity (Hoen et al. 2018, USFWS unpublished data), we estimated 122 NLEB are annually killed at wind facilities (Table 4.1; Udell et al. 2022, entire). Data from Whitby et al. (2022, entire) analyses suggest that the impact of wind related mortality is discernible in the ongoing decline of NLEB. We compared a no wind baseline scenario to current and future wind scenarios. The percent change in abundance relative to the baseline no wind scenario ranges from a 24% decrease by 2030 under the current wind scenario to a 83% decrease by 2060 under the future high impact wind scenarios (see Tables A-3D1–2). Whitby et al. (2022, entire) found a decline in the predicted relative abundance of NLEB as wind energy risk index increased.

Conservation Measures

To reduce bat fatalities, some facilities “feather” turbine blades (i.e., pitch turbine blades parallel with the prevailing wind direction to slow rotation speeds) at low wind speeds when bats are more at risk (Hein et al. 2021, p. 28). The wind speed at which the turbine blades begin to generate electricity is known as the "cut-in speed," and this can be set at the manufacturer's speed or at a higher threshold, typically referred to as curtailment. The effectiveness of feathering below various cut-in speeds differs among sites and years (Arnett et al. 2013, entire; Berthinussen et al. 2021, pp. 94–106); nonetheless, most studies have shown all-bat fatality reductions of >50% associated with raising cut-in speeds by 1.0–3.0 meters per second (m/s) above the manufacturer’s cut-in speed (Arnett et al. 2013, entire; USFWS unpublished data). The effectiveness of curtailment at reducing species-specific fatality rates for NLEB has not been documented.

Our wind threat analysis incorporated available curtailment data for existing facilities, and to a limited degree, accounted for future curtailment (see Appendix 2-A). Although effective, curtailment results in energy and revenue losses, which may limit the viability of widespread implementation (Hein and Straw 2021, p. 28). Based on available data (USFWS, unpublished data), most current curtailment is implemented as part of Habitat Conservation Plans developed to support Incidental Take Permits or Technical Assistance Letters detailing methods to avoid incidental take of Indiana bat, and these areas with risk to Indiana bat do not fully overlap with those where NLEB and other species may be susceptible to mortality.

However, there are many ongoing efforts to improve our understanding of bat interactions with wind turbines and explore additional strategies for reducing bat mortality at wind facilities. For example, the use of ultrasonic acoustic bat deterrents mounted on turbine towers, blades, and nacelles is an emerging research field showing some promise at reducing bat fatalities (Arnett et al. 2013, entire; Romano et al. 2019, entire; Schirmacher et al. 2020, entire; Weaver et al. 2020, entire; Berthinussen et al. 2021, pp. 88–91). Acoustic-activated “smart” curtailment aims to focus operational curtailment when bat activity is detected in real time (e.g., Hayes et al. 2019, entire; Berthinussen et al. 2021, pp. 105–106; Hein and Straw 2021, pp. 29–30). Additionally, USGS is testing whether illuminating turbines with dim ultraviolet light may deter bats from approaching them (Cryan et al. 2016, entire; Berthinussen et al. 2021, p. 91; Hein and Straw 2021, pp. 23–24). Further, researchers have tested applying a textured coating to the surface of the turbine to alter bats’ perception of the turbine (Bennett and Hale 2019, entire; Berthinussen et al. 2021, pp. 87–88; Hein and Straw 2021, p. 24). These and other methods of reducing bat mortality are still in the research phase, and to date, there are no broadly proven and accepted measures to reduce the severity of impacts beyond various operational strategies (e.g., feathering turbine blades when bats are most likely to be active).

C: Climate Change

Background

There is growing concern about impacts to bat populations in response to climate change (for example, Jones et al. 2009, entire; Jones and Rebelo 2013, entire, O’Shea et al. 2016, p. 9). Jones et al. (2009, p. 94) identified several climate change factors that may impact bats including changes in hibernation, mortality from extreme drought, cold, or rainfall, cyclones, loss of roosts from sea level rise, and impacts from human responses to climate change (e.g., wind turbines). Sherwin et al. (2013, entire) reviewed potential impacts of climate change on foraging, roosting, reproduction, and biogeography of bats and also discussed extreme weather events and indirect effects of climate change. However, the impact of climate change is unknown for most species (Hammerson et al. 2017, p. 150). In particular, there are questions about whether some negative effects will be offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). For example, Lucan et al. (2013, p. 157) suggested that while rising spring temperatures may have a positive effect on juvenile survival, increasing incidence of climatic extremes, such as excessive summer precipitation, may counter this effect by reducing reproductive success. While there may be a variety of ways that climate change directly or indirectly effects NLEB, here we summarize information on the effect of increasing temperatures and changes in precipitation.

Increased annual temperature

Global average temperature has increased by 1.7 degrees F (0.9 degrees C) between 1901 and 2016 (Hayhoe et al. 2018, p. 76). Over the contiguous U.S., annual average temperature has increased by 1.2 degrees F (0.7 degrees C) for the period of 1986 to 2016 relative to 1901 to 1960 (Hayhoe et al. 2018, p. 86). At a regional scale, each National Climate Assessment region also increased in temperature during that time with the largest changes in the west with average increases of more than 1.5 degrees F (0.8 degrees C) in Alaska, the Northwest, the Southwest and the Northern Great Plans and the least change in the Southeast (Hayhoe et al. 2018, p. 86).

Increased annual temperatures are likely to change bat activity and phenology. For example, increased winter temperatures may reduce hibernation period due to longer fall activity or earlier spring emergence (Jones et al. 2009, p. 99). Rodenhouse et al. (2009, p. 250) suggest that hibernation may be shortened by 4 to 6 weeks by the end of this century. Reduced hibernation periods may decrease the duration that an individual bat is exposed to *Pd* and effects of WNS (Langwig et al. 2015a, p. 5).

With increasing temperatures, earlier spring emergence has been documented for cave-roosting bats in Virginia (Muthersbaugh et al. 2019, p. 1). After earlier arrival to summer habitat, if spring weather remains favorable (warm, dry and calm nights providing suitable foraging conditions for bats), this could result in earlier parturition (Racey and Swift 1981, pp. 123–125; Jones et al. 2009, p. 99; Linton and MacDonald 2018, p. 1086) and increased reproductive success (Frick et al. 2010, p. 133; Linton and MacDonald 2018, p. 1086). However, earlier emergence increases the risk of exposure to lethal cold snaps (Jones et al. 2009, p. 99).

Increased temperatures may expand the suitable window for nightly foraging opportunities thereby increasing per night caloric intake. Low ambient temperatures reduce flying insect activity and bat foraging (Anthony et al. 1981, p. 155), while higher average temperatures may result in more frequent suitable foraging nights, particularly during the pre-hibernation fattening period.

Bats that hibernate in temperate regions require temperatures above freezing but cool enough to save energy through torpor (Perry 2013, p. 28). Increased ambient surface temperatures change hibernacula temperatures which then influences their ability to meet the needs of hibernating bats. However, increased ambient surface temperatures will not affect all hibernacula or all parts of a given hibernaculum equally. Hibernaculum microclimate is influenced by a variety of factors including the size, complexity, and location of the site (Tuttle and Stevenson 1978, pp. 109–113). In addition, temperatures of microsites near entrances are strongly correlated to external ambient temperatures compared to microsites deep within hibernacula (Dwyer 1971, p. 427; Boyles 2016, p. 21). Therefore, changes in ambient temperatures are anticipated to result in the greatest changes to portions of hibernacula nearest entrances.

In warmer regions, caves and mines that trap cold air produce beneficial conditions for hibernacula, while in colder regions sites that trap warm air will be more suitable (Perry 2013, p. 33; Kurta and Smith 2014, p. 595). Consequently, a northern site that is suitable today in part for its ability to trap warm air while surface temperatures are very low may become unsuitable as mean annual surface temperature increases.

Indiana bats have been documented to use a wide variety of microclimates within hibernacula and Boyles (2016, p. 34) suggests that the most valuable caves for protection might be the ones with the widest variety of microclimates available. Briggler and Prather (2003, p. 411) similarly found that more tricolored bats were found in caves with wide temperature gradients available. These more complex hibernacula will be less influenced by changes in surface ambient temperatures.

Variations in ambient temperature increase energy expenditure of hibernating bats (Boyles and McKechnie 2010, p. 1645); therefore, stable microsites may be advantageous. Increased ambient temperatures may reduce reliance on relatively stable temperatures associated with underground hibernation sites (Jones et al. 2009, p. 99). However, variation in ambient temperature (e.g., increases in spring) may decrease the energetic costs of arousing from hibernation and serve as a signal that surface conditions are suitable for emergence and foraging (Boyles 2016, p. 36).

Increased hibernacula temperatures may influence overwinter survival rates. If more frequent bat arousals occur, bats will burn through fat reserves more quickly. While insect abundance may also increase in winter, it is unknown whether they will become sufficiently abundant to offset the increased energetic costs associated with more frequent arousal by bats (Rodenhouse et al. 2009, p. 251; Jones and Rebelo 2013, p. 464). Changes to hibernacula temperatures could potentially alter the severity of WNS in these sites (Martínková et al. 2018, p. 1747). For example, a hibernaculum with temperature below the optimal growth rate for *Pd* could shift into the optimal temperature range, thus increasing infection at the site.

Lastly, increased temperatures may result in range shifts of the bats, forest communities, and invasive species. With increasing temperatures, a poleward range expansion of temperate-zone species is predicted (Humphries et al. 2004, p. 154). Kuhl's pipistrelle (*Pipistrellus kuhlii*) has already undergone a substantial northward range shift over the past 15 years (Jones et al. 2009, p. 100), and Lundy et al. (2010, entire) suggested that the migratory Nathusius' pipistrelle (*Pipistrellus nathusii*) has expanded its range in the United Kingdom in response to climate change and will continue to do so. The ranges of European bats are forecasted to show considerable shifts, with species in the Boreal Zone experiencing the greatest change and risk of extinction (Rebelo et al. 2010, p. 568). Many species have little or no overlap between their current and predicted range and face enhanced extinction risk (Rebelo et al. 2010, p. 572). Loeb and Winters (2012, pp. 5–8) found the suitability of an area for Indiana bat maternity colonies declines once the average summer maximum temperature reaches 27.4 degrees C (81.3 degrees F) and predicts a range contraction and northward shift based on climate projections.

Any northern range shifts, however, will be limited based on availability of suitable hibernacula and energetic requirements for hibernation and migration. Humphries et al. (2002, p. 315) predicted that minimum accumulated fat stores of little brown bats are currently inadequate for surviving hibernation throughout the northern portions of the Canadian provinces and the maximum possible fat stores are inadequate for most of Alaska and Canadian territories. When considering a predicted increase of 6 to 8 degrees C (10.8 to 14.4 degrees F), the region of suitable hibernation is expected to expand with a northward shift of approximately 6 km (3.7 mi) per year over the next 80 years (Humphries et al. 2002, pp. 315–316) (Figure A-4C1).

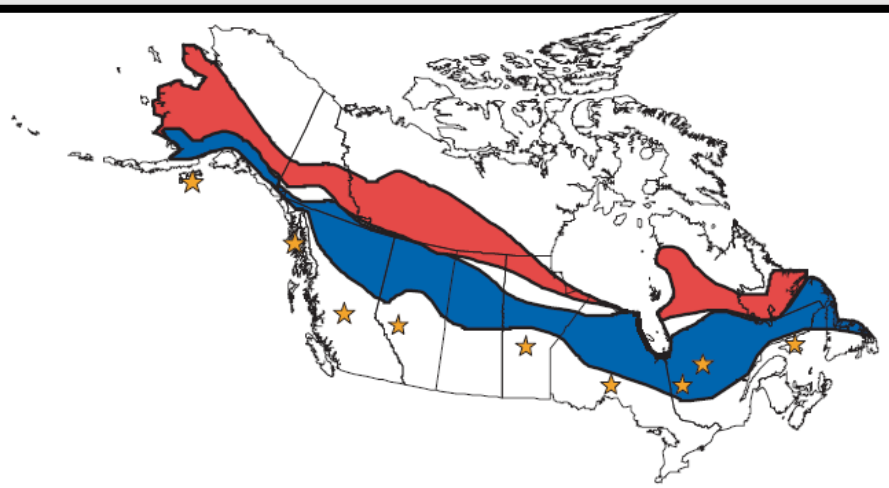

Figure 3 Observed and predicted *M. lucifugus* range distributions in northern North America. Confirmed northern locations of hibernacula (refs 10, 14, 19 and 29, and M. Gauthier, personal communication) are indicated with yellow stars, and the predicted distributional limit of hibernacula is indicated by the blue area; the width of the area represents variation in predictions resulting from cave temperatures exceeding average annual temperature by 0–2 °C, fat stores varying from 1.5 to 3.5 g, and all other parameters varying by up to ±5%. Thus, the lower margin of the blue zone indicates the limit beyond which to the south almost all bats could successfully hibernate, and the upper margin indicates the limit beyond which to the north almost no bats could successfully hibernate. Observed hibernacula approach and only marginally exceed the lower margin of this zone and there is a significant positive correlation between the latitude of observed hibernacula and the lower margin of this zone (predictions generated for each hibernaculum longitude, $n = 9$, $r = 0.93$, $P < 0.001$; intercept not significantly different from 0). The red area represents the predicted 2080 winter range limit of *M. lucifugus* due to global warming. Climate projections are based on the Hadley Centre coupled global climate model (HadCM2—GAX)[12]. The width of the red zone represents the same as does the blue zone.

Figure A-4C1. Observed and predicted little brown bat range distributions in northern North America (from Humphries et al. 2002, Figure 3).

While more northerly sites may become suitable for hibernation, there may be other constraints on successful recruitment at higher latitudes. The active season is shorter in higher latitudes or elevations which may be particularly important for juveniles. Juvenile little brown bats take longer than adults to gain sufficient fat stores for hibernation and shorter active seasons limit their capacity to grow and fatten before their first winter (Kunz et al. 1998, pp. 10–13; Humphries et al. 2002, p. 315). Higher elevations have similar climatic influences as higher latitudes and significantly fewer reproductive female little brown bats are captured at higher elevations in Pennsylvania, West Virginia and Virginia with a similar pattern for tricolored bats in West Virginia (Brack et al. 2002, pp. 24–26).

While bats may be more flexible than other mammals in shifting their ranges, given their ability to fly, the ability of individuals to reach new climatically suitable areas will be impacted by loss and fragmentation of habitat (Thomas et al. 2004, p. 147). The availability of ample suitable roosts may be one of the most limiting resources for bats (Scheel et al. 1996, p. 453). This may be of special concern for tree-dwelling bats since the rate of climate change may be too fast to allow the development of mature forests in the new climatically suitable areas in the north (Rebelo et al. 2010, p. 573).

Changes in Precipitation

Increased temperatures interact with changes in precipitation patterns and results may differ regionally. Annual average precipitation has increased by 4% since 1901 across the entire U.S. with increases over the Northeast, Midwest and Great Plains and decreases over parts of the Southwest and Southeast (Easterling et al. 2017, p. 208; Hayhoe et al. 2018, p. 88) (Figure A-4C2). The frequency and intensity of heavy precipitation events across the U.S. have increased more than increases in average precipitation (Hayhoe et al. 2018, p. 88).

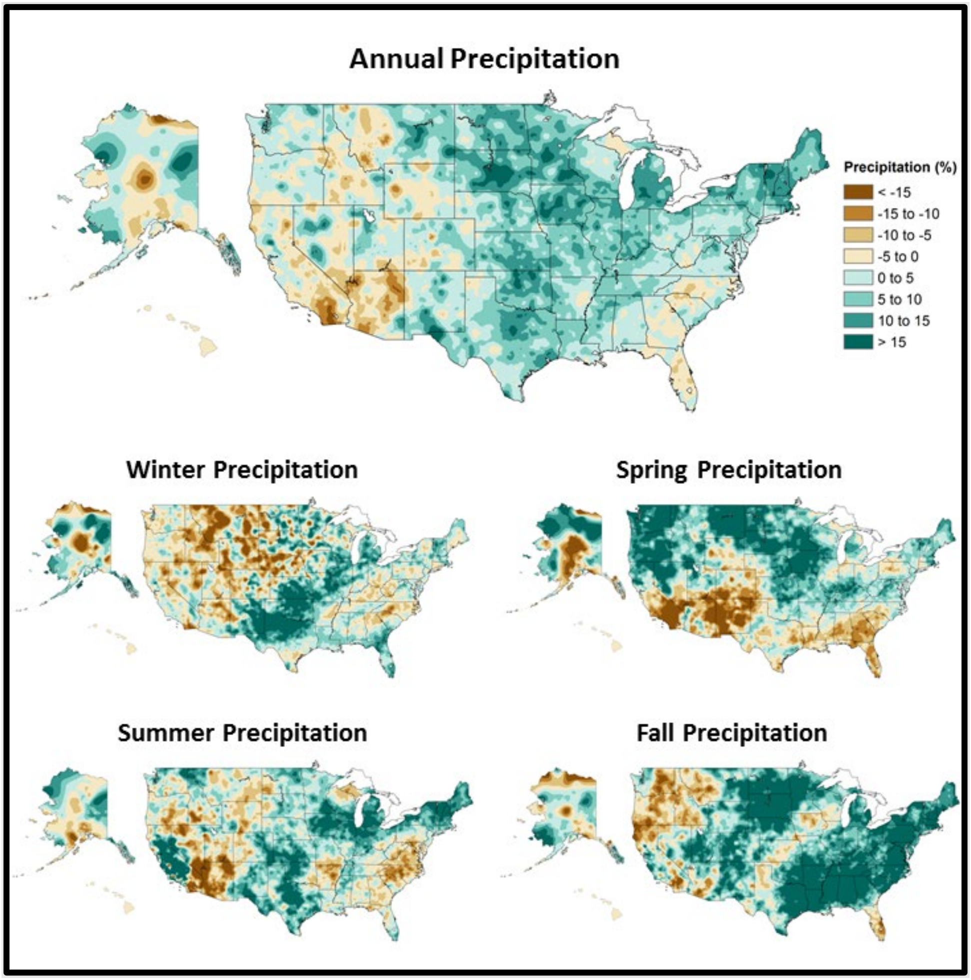

Figure A-4C2. Annual and seasonal changes in precipitation over the U.S. Changes are the average for present-day (1986–2015) minus the average for the first half of the last century (1901–1960 for the contiguous U.S., 1925–1960 for Alaska and Hawai'i) divided by the average for the first half of the century (Easterling et al. 2017, Figure 7.1).

In arid regions, any further reductions in water availability from human uses, reductions in snowpack, or droughts will amplify existing constraints. Spring snow cover extent and maximum snow depth has declined in North America and snow water equivalent and snowpack has declined in the western U.S. (Hayhoe et al. 2018, p. 90). Bats rely on access to free water for thermoregulation, foraging, and reproduction (Adams and Hayes 2008, pp. 1117–1119). In the Rocky Mountains, drought and reduced standing water appears associated with decreased reproduction in bats (Adams 2010, entire). Years that were hotter and drier had a higher incidence of non-reproductive females for all species and 64% of adult female little brown bats were non-reproductive in the drought years of 2007 and 2008 (Adams 2010, pp. 2440–2442) (Figure A-4C3). While cooler and wetter springs resulted in shifts in parturition dates (Grindal et al. 1992, p. 342; Linton and MacDonald 2018, p. 1086), drought years resulted in an overall reduction in the percentage of bats that were reproductive <u>at all</u> (Adams 2010, p. 2442). Readily available water sources appear to be particularly important during lactation (Adams and Hayes 2008, pp. 1117–1120).

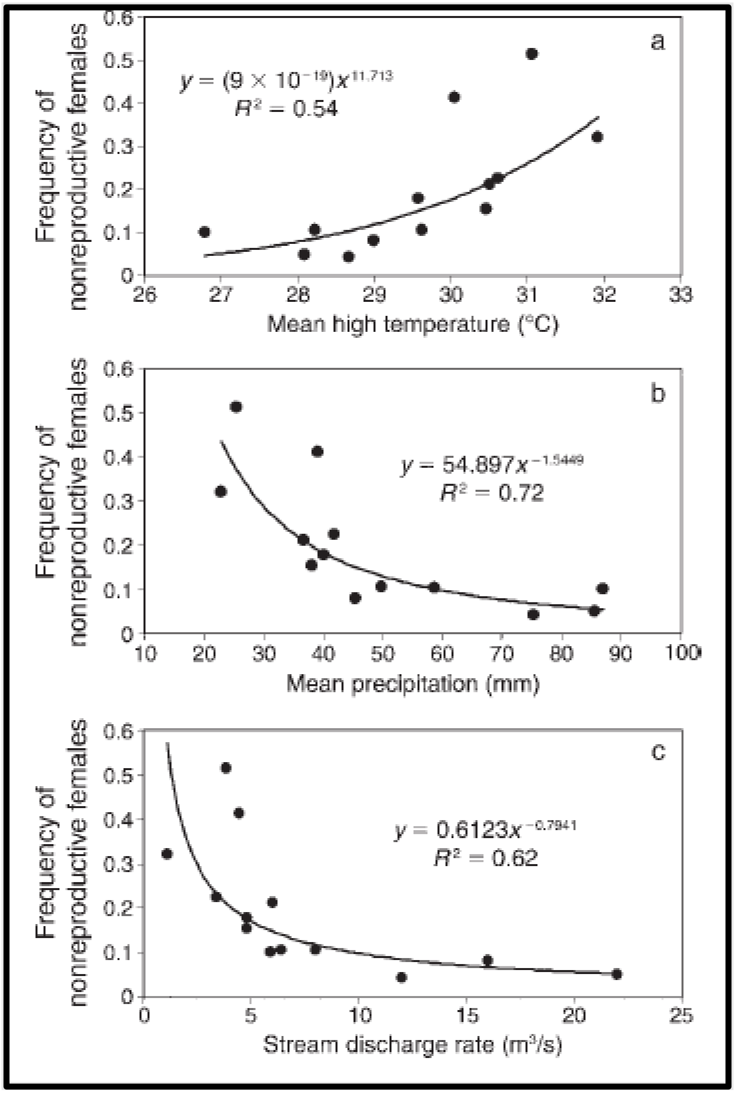

Figure A-4C3. Relationships between the frequency of non-reproductive females captured from 1996 through 2008 and (a) mean high temperature (R -0.74, P = 0.001), (b) mean precipitation (R -0.85, P = 0.0001), and (c) stream discharge rate (R -0.79, P = 0.001(Adams 2010, Figure 2).

In temperate regions, increased cumulative annual rainfall may lead to increases in the abundance of insects such as dipterans and lepidopterans and is correlated with higher little brown bat survival rates (Frick et al. 2010, pp. 131–133). They suggest that increased insect abundance associated with higher moisture availability was the likely driver and this relationship may vary based on the timing of precipitation (Frick et al. 2010, p. 133). Drying summer conditions may negatively impact aquatic insect prey and little brown bats in the northeastern U.S. (Rodenhouse et al. 2009, p. 250; Frick et al. 2010, p. 133). Small mammals with high energy demands like bats, may be particularly vulnerable to changes in food supply (Rodenhouse et al. 2009, p. 250).

More precipitation has been falling as rain rather than snow in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 90). For example, increased winter temperatures are associated with decreases in Great Lakes ice cover and increases in winter precipitation occurring as rain. The extent and duration of lake ice on the Great Lakes are two of the principal factors controlling the amount of lake-effect snow (provided the air temperatures are sufficiently cool). When large areas of the lakes are covered with ice, the moisture cycle that generates lake-effect snow systems is greatly diminished (Brown and Duguay 2010, p. 692). During the first half of the 20th century there was an increase in snowfall in the Great Lakes Basin; however, recent studies have shown a decline through the latter half of the 20th and early 21st century (Baijnath-Rodino et al. 2018, p. 3947). Similarly, Suriano et al. (2019, pp. 4) found a reduction in snow depth in the Great Lakes Basin of approximately 25% from 1960 to 2009. Trends in snowfall and snow depth during this timeframe are variable by subbasin (Suriano et al. 2019, pp. 5–6) and there was a significant increase of the number of ablation events (i.e., snow mass loss from melt, sublimation, or evaporation) in many areas (Suriano et al. 2019, pp. 6–7). These events are associated with rapid snow melt and often lead to localized flooding. Hibernacula that already faced periodic flooding would be expected to have an increased risk in these areas.

While sufficient moisture is important, too much precipitation during the spring can also result in negative consequences to insectivorous bats. During precipitation events there may be decreased insect availability and reduced echolocation ability (Geipel et al. 2019, p. 4) resulting in decreased foraging success. Precipitation also wets bat fur, reducing its insulating value (Webb and King 1984, p. 190; Burles et al. 2009, p. 132) and increasing a bat's metabolic rate (Voigt et al. 2011, pp. 794–795). Given these consequences, bats are likely to reduce their foraging bouts during these heavy rain events.

There is a balancing act that insectivorous bats perform, balancing the costs of flight, thermoregulation and reproduction versus energetic gains from foraging. When bats arrive at maternity areas in the spring, they are stressed after a lengthy hibernation period, a potentially long migration, and the demands of early pregnancy. During this period when their energetic and nutritional requirements are highest, food (flying insects) is relatively scarce, due to cool and wet weather (Kurta 2005, p. 20). Adverse weather, such as cold spells, increases energetic costs for thermoregulation and decreases availability of insect prey (the available energy supply). Bats may respond to a negative energy balance by using daily torpor which conserves consumed and stored energy, and probably minimizes mortality. This has significant implications for their survival or reproduction.

Also, as mentioned above, increased rainfall during pregnancy and lactation may delay parturition or reduce reproductive success (Racey and Swift 1981, pp. 123–125; Grindal et al. 1992, p. 128; Burles et al. 2009, pp. 135–136; Linton and MacDonald 2018, p. 1086). Some females may not bear a pup in years with adverse weather conditions (Barclay et al. 2004, p.

691). Young bats who are born and develop later in the season have less time to develop to successfully forage and to build the fat stores needed to meet the energy demands of migration and hibernation (Humphrey 1975, p. 339). Frick et al. (2010, pp. 131–132) found that little brown bats born even a few weeks later in the summer have significantly lower first-year survival rates and are significantly less likely to return to the maternity colony site to breed in their first year.

Early in the summer, females are under heavy energy requirements to supply their developing fetuses. After giving birth, the adult females experience increased energy needs due to the requirements of lactation and the need to return to the roost during night foraging times to feed their non-volant pups (Murray and Kurta 2004, p. 4). Later in the summer as the pups become volant, these inexperienced and relatively inefficient flyers must expend increased levels of energy as they are growing and learning to feed. Once weaned, young-of-the-year bats must consume enough on their own to migrate to hibernacula and store sufficient fat for the coming winter.

Interaction with WNS-affected Bats

Regardless of the source of increased stress (e.g., reduced foraging, reduced free standing water), because of WNS, there are additional energetic demands for bats. Because WNS causes premature fat depletion, affected bats have less fat reserves than non-WNS-affected bats when they emerge from hibernation (Warnecke et al. 2012, p. 2–3). In addition, WNS-affected bats have wing damage (Meteyer et al. 2009, entire; Reichard and Kunz 2009, entire) that makes flight (migration and foraging) more challenging and results in increased energetic demands associated with the healing process (Davy et al. 2017, pp. 619–612; Meierhofer et al. 2018, p. 487; Fuller et al. 2020, p. 8).

Females that migrate successfully to their summer habitat must partition energy resources between foraging, keeping warm, sustaining fetal development and recovering from the disease. Bats may use torpor to conserve energy during cold, wet weather when insect activity is reduced and increased energy is needed to thermoregulate. However, use of torpor reduces healing opportunities as immune responses are suppressed (Field et al. 2018, p. 3731).

Dobony et al. (2011, entire) observed a little brown bat colony prior to and after onset of WNS impacts and found evidence of lower reproductive rates in the years immediately after WNS was first documented to affect the colony. Francl et al. (2012, p. 36) observed a reduction in juveniles captured pre- and post-WNS in West Virginia, suggesting similarly reduced reproductive rates. Meierhofer et al. (2018, p. 486) found higher resting metabolic rates in WNS-infected (vs. uninfected) little brown bats, suggesting additional energy costs during spring in WNS survivors.

Future climate conditions

Over the next few decades, annual average temperature over the contiguous U.S. is projected to increase by about 2.2 degrees F (1.2 degrees C), relative to 1985 to 2015 regardless of future scenario (Hayhoe et al. 2018, p. 86; Figure A-4C4). Larger increases are projected by late century of 2.3 to 6.7 degrees F (1.3 to 3.7 degrees C) under RCP4.5 and 5.4 to 11.0 degrees F

(3.0 to 6.1 degrees C) and 5.4 to 11.0 degrees F (3.0 to 6.1 degrees C) under RCP8.5, relative to 1986 to 2015 (Hayhoe et al. 2018, p. 86). For the period of 2070 to 2099 relative to 1986 to 2015, precipitation increases of up to 20 and 30% are projected in winter and spring for north central U.S. and Alaska, respectively, with decreases by 20% or more in the Southwest in spring (Hayhoe et al. 2018, p. 88). The frequency and intensity of heavy precipitation events are expected to continue to increase across the U.S., with the largest increases in the Northeast and Midwest (Hayhoe et al. 2018, p. 88). Projections show large declines in snowpack in the western U.S. and shifts of snow to rain in many parts of the central and eastern U.S. (Hayhoe et al. 2018, p. 91).

NLEB's responses to these changes are expected to be similar to what has already been observed in North American insectivorous bats, such as little brown bat (see above). This includes reduced reproduction in the Rocky Mountains due to drought conditions leading to declines in available drinking water (Adams 2010, pp. 2440–2442) and reduced adult survival during dry years in the Northeast (Frick et al. 2010, pp. 131–133). However, the timing of rain events is also important as reduced reproduction has been observed during cooler, wetter springs in the Northwest (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Magnitudes of responses will likely vary throughout NLEB's rangedepending on how much the annual temperature actually rises in the future.

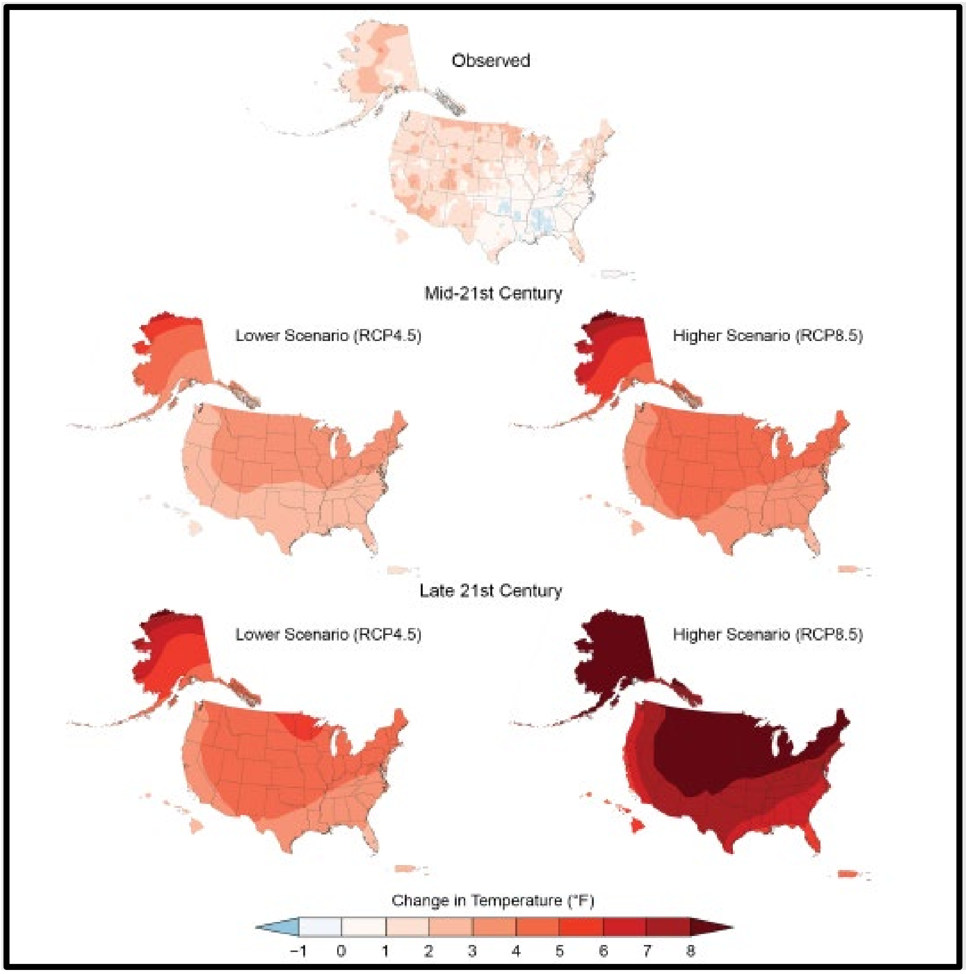

Figure A-4C4. Observed and Projected Changes in Annual Average Temperature (from Hayhoe et al. 2018, Figure 2.4, p. 87).

Climate change may additionally impact these bats in ways that are more difficult to measure. This may include phenological mismatch (e.g., timing of various insect hatches not aligning with key life history periods of spring emergence, pregnancy, lactation, or fall swarming). In addition, there may be shifts in distribution of forest communities, invasive plants, invasive forest pest species, or insect prey. Long-term increases in global temperatures are correlated with shifts in butterfly ranges (Parmesan et al. 1999, entire; Wilson et al. 2007, p. 1880; Breed et al. 2013, p. 142) and similar responses are anticipated in moths and other insect prey. Milder winters may result in range expansions of insects or pathogens with a distribution currently limited by cold temperatures (e.g., hemlock woolly adelgid, southern pine beetle) (Haavik 2019).

Climate change has also resulted in a rise of global sea level by about 7 to 8 inches (16 to 21 centimeters) since 1993 and relative to the year 2000, sea level is very likely to rise 1 to 4 feet (0.3 to 1.3 meters) (Hayhoe et al. 2018, p. 83). Relative sea level rise is projected to be greater than the global average along the coastlines of the U.S. Northeast and western Gulf of Mexico

(Hayhoe et al. 2018, p. 99), which may reduce access to cave roost along low-lying coastal areas (Jones et al. 2009, p. 101).

Additionally, there are questions about whether some negative effects will be offset by other positive effects, whether population losses in one part of a species' range will be offset by gains in other regions, and the degree to which bats can adapt by adjusting their ecological and phenological characteristics (Hammerson et al. 2017, p. 150). For example, Lucan et al. (2013, p. 157) suggested that while rising spring temperatures may have a positive effect on juvenile survival, increasing incidence of climatic extremes, such as excessive summer precipitation, may counter this effect by reducing reproductive success.

D: Habitat Loss

Background

As discussed in Chapter 2, NLEB require suitable habitat for roosting, foraging, and commuting between those habitats during spring, summer, and fall. Forest is a primary component of all of these habitat types, except for the far western portion of the range. Wetlands and water features are important foraging and drinking water sources.

There are a variety of reasons for roosting, foraging, and commuting habitat loss within the range of the NLEB. Hammerson et al. (2017, entire) assessed scope and severity of threat to bats with the highest projected threat impact including invasive species and diseases (particularly WNS); energy production and mining, especially wind energy; human intrusions and disturbance of primarily cave- or mine-dwelling species; and biological resource use, such as tree cutting and forestry practices. Tree cutting and wetland loss can occur from a variety of sources (e.g., development, energy production and transmission, transportation projects). As discussed in Chapter 4, these are increasing across much of the range of the NLEB (USFWS 2015, p. 17991; Oswalt et al. 2019, p. 17) and may result in impacts to the NLEB.

Past and Current

The USFS (2014, p. 7) summarized U.S. forest trends and found a decline from 1850 to the early 1900s, and a general leveling off since that time; therefore, conversion from forest to other land cover types has been fairly stable with conversion to forest (cropland reversion/plantings). In addition, the USFS reviewed U.S. forest trends through 2017 and found forest area trended upward from 1987 to 2012, but since 2012 appears to have reached a plateau (Oswalt et al. 2019, p. 4). About 9.6 million acres (1.4%) of U.S. forest land are affected by tree cutting and removal each year and on an average annual basis, twice as much forest land area (~19 million acres) is affected by natural disturbances that cause either mortality or damage to trees (Oswalt et al. 2019, p. 7). These forest disturbances are attributable to insects and disease (34 percent), fire (21%), weather (16%), and other causes (30%), with importance of disturbance agents varying greatly among geographic regions (Oswalt et al. 2019, p. 7).

In addition to reviewing these reports, we examined more recent (2006 to 2016) change in various NLCD landcover classes within each RPU in the continental U.S. Overall, forest landcover was fairly stable in all RPUs with slight annual increases (27,000 to 50,000 acres/year) in all but Midwest RPU (loss of 23,000 acres/year) (Table A-4D1). However, deciduous forest landcover decreased across all RPUs by 1.4 million acres for an average loss of 140,000 acres per year. Other cover types that provide foraging opportunities such as emergent wetland cover types decreased across all RPUs by 1.4 million acres.

Table A-4D1. Changes in land cover types in acres (NLCD 2006-2016) by NLEB RPU occurring within the continental U.S. (Subarctic RPU not included).

	NLEB Representative Units – Change (in acres)				
Land Cover Type	**Southeast RPU**	**Eastern Hardwoods RPU**	**Midwest RPU**	**East Coast RPU**	**All Units**
No Data	0	0	0	0	0
Open Water	-53228	-15513	645390	16451	593100
Developed, Open Space	86718	136193	58923	24639	306472
Developed, Low Intensity	133223	226024	90183	51348	500778
Developed, Medium Intensity	162539	300223	106341	64206	633309
Developed, High Intensity	64748	135120	43896	23717	267481
Barren Land	16701	-767	65283	-3608	77609
Deciduous Forest	-717517	-638191	-24698	-49555	-1429962
Evergreen Forest	920674	-36455	-215544	214328	883003
Mixed Forest	218377	245548	15009	21098	500032
Shrub/Scrub	253128	971856	46115	-114649	1156451
Grassland/Herbaceous	-532118	-520196	-2944844	-164519	-4161676
Pasture/Hay	-888122	-1676022	-2000851	-19983	-4584978
Cultivated Crops	325615	788149	4498629	-37950	5574443
Woody Wetlands	77534	876487	-7942	88299	1034379
Emergent Herbaceous Wetlands	-68273	-792457	-375890	-113822	-1350443
Forest change over 10 years	499068	447390	-233175	274170	987453
Annual average forest change	49906	44738	-23317	27416	98745

Forest ownership varies widely across the species' range in the U.S. As of 2017, private landowners owned approximately 60% of forests (Oswalt et al. 2019, p. 7). Private lands may carry with them a higher risk for conversion than do public forests (since they do not support the same level of regulatory certainty as public lands) a factor that must be considered when assessing risk of forest loss now and in the future (USFWS 2015, p. 17990). Private land ownership is approximately 81% in the East and 30% in the western U.S. (USFS 2014, p. 15). Of the timber harvested annually in the U.S., 89% comes from private lands (Oswalt et al. 2019, p. 9).

Future

The 2010 Resources Planning Act (RPA) Assessment (USFS 2012, entire) and 2016 RPA Update (USFS 2016, entire) summarized findings related to the status, trends, and projected future of U.S. forests and rangeland resources (we have nothing comparable for Canada). This assessment was influenced by a set of future scenarios with varying assumptions regarding global and U.S. population, economic growth, climate change, wood energy consumption, and land use change from 2010 to 2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million hectares (16–34 million acres or 4–8% of 2007 forest area) across the conterminous U.S., and forest loss is expected to be concentrated in the southern U.S., with losses of 3.6–8.5 million hectares (9–21 million acres) (USFS 2012, p. 12). The 2010 Assessment projected limited climate effects to forest lands spread throughout the U.S. during the projection period, but effects were more noticeable in the western U.S. The projections were dominated by conversions of forested areas to urban and developed land cover (USFS 2012, p. 59). The 2016 Update incorporated several scenarios including increasing forest lands through 2022 and then leveling off or declines of forest lands (USFS 2016, p. 8–7). However, regenerating young forests temporarily lack roosts until suitable tree sizes are reached to provide space and thermal needs for NLEB colonies. In addition, NLEB is not uniformly distributed across the landscape. Loss of essential population needs of roosts and foraging and commuting habitat within NLEB home range where they remain is the issue.

Impacts to bats

These changes in land cover may be associated with losses of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colony networks, and direct injury or mortality (during active season tree removal).

Bats may be directly affected by forest habitat loss by removal of occupied roost trees or loss of roosting and foraging habitat (Farrow and Broders 2011, p. 177). While roosting bats can sometimes flee during tree removal, removal of occupied roosts (during spring through fall) is likely to result in direct injury or mortality to some bats (Belwood 2002, p. 193; McAlpine et al. 2021, p. 2). This is particularly likely during cool spring months (when bats enter torpor) and if flightless pups or inexperienced flying juveniles are also present.

Removal of trees any time of year, including winter, can result in additional impacts depending upon the scope of the action (e.g., acreage of tree removal, locations, and landscape context of the projects) and current understanding or well-supported inferences regarding NLEB presence and use of the area.

Loss of roosts → colony fragmentation → smaller colonies → reduced thermoregulation, reduced information sharing → increased energy expenditure →

- reduced pregnancy success
- reduced pup survival

- reduced adult survival

Loss of roosts, foraging habitat, or travel corridors → displacement → increased flights → increased energy expenditure →

- reduced pregnancy success
- reduced pup survival
- reduced adult survival

Displacement from optimal roosts can also lead directly to increased energy expenditure.

For temperate bats, the requirements for roosting are more restricted and habitat suitable for roosting is rare relative to foraging habitat (Pauli et al. 2015, p. 16); therefore, removal of roosting habitat is more impactful than foraging habitat to these species.

For these species, although loss of a roost is a natural occurrence that temperate bat species must cope with regularly due to the ephemeral nature of tree roosts, the loss of many roosts or an entire home range may result in impacts at the colony level. Bats switch roosts for a variety of reasons, including temperature, precipitation, predation, parasitism, sociality, and ephemeral roost sites (Carter and Feldhamer 2005, p. 264; Barclay and Kurta 2007, p. 34). NLEB is known to switch roosts; therefore, NLEB can tolerate some loss of roosts, provided suitable alternative roosts are available (see Chapter 2). However, loss of central or important roosts can result in colony fragmentation. For example, Silvis et al. (2015, pp. 6–12) found a loss of approximately 17% of roosts may begin to cause colony fragmentation in NLEB. One of the most prominent advantages of colonial roosting is the thermoregulatory benefit (Humphrey et al. 1977, pp. 343–344). Therefore, smaller colonies are expected to provide fewer thermoregulatory benefits for adults in cool spring temperatures and for non-volant pups at any time.

If bats are required to search for new roosting or foraging habitat and to find the same habitats as the rest of their colony finds in the spring, it is reasonable to conclude that this effort places additional stress on pregnant females at a time when fat reserves are low or depleted and they are already stressed from the energy demands of migration and pregnancy. In addition, removal of roosting or foraging habitat may result in longer travel distances between sites used for roosting and foraging. The increased energetic cost of longer commuting distances may result in maternity colony disruption and may be particularly important for pregnant and lactating females and therefore, reproductive success (Lacki et al. 2007, p. 89). NLEB emerge from hibernation with their lowest annual fat reserves, and return to their summer home ranges. Loss or alteration of roosting or foraging habitat puts additional stress on species such as NLEB with strong summer site (i.e., roosting area) fidelity (Foster and Kurta 1999, p. 665; Patriquin et al. 2010, p. 908; Broders et al. 2013, p. 1180), when returning to summer roosting or foraging areas after hibernation. Reproduction is one of the most energetically demanding periods for temperate-zone bats (Broders et al. 2013, p. 1174). Female NLEB produce a maximum of one pup per year; therefore, loss of just one pup results in loss of that entire year's recruitment for females. Limited reproductive potential severely limits the ability of bat populations to respond quickly to perturbations.

Interaction with WNS-affected Bats

Similar to climate change, there are interacting effects of habitat loss with effects from WNS. Regardless of the source of increased stress on bats (roost or foraging habitat removal), because of WNS, there are additional energetic demands for bats associated with healing (Fuller et al. 2020, p. 7). Because WNS causes more frequent arousals (Reeder et al. 2012, pp. 6–9) and fat depletion, affected bats have less fat reserves than non-WNS-affected bats when they emerge from hibernation (Warnecke et al. 2012, p. 7001) and have wing damage (Meteyer et al. 2009, entire; Reichard and Kunz 2009, entire) that makes flight (migration and foraging) more challenging. Females that migrate successfully to their summer habitat must partition energy resources between foraging, keeping warm, sustaining fetal development and recovering from the disease. With increased flights to find suitable habitat or between roosting and foraging habitat comes a trade-off for sufficient energy for survival, recovering from WNS, successful pregnancy or successful rearing of pups.

Roosting/Foraging/Commuting Habitat Loss Conservation Measures

All states have active forestry programs with a variety of goals and objectives. Several states have established habitat protection buffers around known Indiana bat hibernacula that will also serve to benefit other bat species by maintaining sufficient quality and quantity of swarming habitat. Some states conduct some of their own forest management activities in the winter within known listed bat home ranges, as a measure that would protect maternity colonies and non-volant pups during summer months. The USFWS routinely works with project sponsors and Federal agencies to minimize the amount of forest loss associated with their projects and to provide mitigation for impacts associated with forest loss within the range of the federally listed Indiana bat. Examples of largescale efforts to address impacts associated with habitat loss include the rangewide transportation consultation for Indiana bats and NLEB, NiSource Habitat Conservation Plan, and rangewide in-lieu fee program for Indiana bats. Many of the beneficial actions associated with these and similar efforts may benefit other bats if they occur in overlapping ranges. Depending on the type and timing of activities, forest management can be beneficial to bat species (e.g., maintaining or increasing suitable roosting and foraging habitat).

Forest management that results in heterogeneous (including forest type, age, and structural characteristics) habitat may benefit tree roosting bat species (Silvis et al. 2016, p. 37). For example, creation of small canopy openings could increase solar exposure to roosts, leading to warmer conditions that result in more rapid development of NLEB young (Perry and Thill 2007, p. 224). In central Arkansas, female NLEB roosts were more often located in areas with partial harvesting than males, with more male roosts (42%) in unharvested stands than female roosts (24%) (Perry and Thill 2007, pp. 223–224). Silvicultural practices can meet both male and female NLEB roosting requirements by maintaining large-diameter snags in early stages of decay, while allowing for regeneration of forests (Lacki and Schwierjohann 2001, p. 487). Although loss of a roost is a natural phenomenon that bats must deal with regularly, the loss of multiple roosts due to a variety of reasons likely stresses individual bats, as well as the social structure of the colony. Therefore, maintaining roost networks is essential for maternity colony dynamics as colonies may fragment (split into multiple colonies) temporarily with the loss of a primary (central node) roost or multiple alternate roosts (Silvis et al. 2014, pp. 287, 289).

Summary

In summary, U.S. forest area trends have remained relatively stable with some geographic regions facing more loss than others in the recent past. In the future, forest loss is expected to continue, whether from commercial or residential development, energy production, or other pressures on forest lands. Impacts from forest habitat removal to individuals or colonies would be expected to range from minor (e.g., removal of a portion of foraging habitat in largely forested areas with no removal of roosts in areas with robust NLEB populations) to significant (e.g., removal of roosts, removal of a large percentage of summer home range, highly fragmented landscapes, areas with WNS impacts). In areas with little forest or highly fragmented forests (e.g., western U.S. and central Midwestern states), impacts would be more likely with a higher probability of removing roosts or causing loss of connectivity between roosting and foraging habitat.

Conservation Measures addressing winter roost loss and disturbance

Protecting these species from disturbance during winter is essential because any additional arousal from hibernation will require an increase in total energy expenditure at a time when food and water resources are scarce or unavailable. This is even more important for sites where a species is impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Willis and Boyles 2012, p. 96).

One method of reducing this disturbance is through installation of bat-friendly gates that allow passage of bats while reducing disturbance from human entry as well as changes to the cave microclimate from air restrictions (Kilpatrick et al. 2020, p. 6). Many state and Federal agencies, conservation organizations, and land trusts have installed bat-friendly gates to protect important hibernation sites. The National Park Service has proactively taken steps to minimize effects to underground bat habitat resulting from vandalism, recreational activities, and abandoned mine closures (Plumb and Budde 2011, unpublished data). Further, all known hibernacula within national grasslands and forestlands of the Rocky Mountain Region of the USFS are closed during the winter hibernation period, primarily due to the threat of WNS, although this will reduce disturbance to bats in general inhabiting these hibernacula (USFS 2013, unpaginated). Because of concern over the importance of bat roosts, including hibernacula, the American Society of Mammalogists developed guidelines for protection of roosts, many of which have been adopted by government agencies and special interest groups (Sheffield et al. 1992, p. 707). Also, regulations, such as the Federal Cave Resources Protection Act (16 U.S.C. 4301 *et seq.*), protects caves on Federal lands. Finally, many Indiana bat hibernacula have been gated and some have been permanently protected via acquisition or easement, which provides benefits to other bats that also use the sites.

Appendix 5. Supplemental Future Scenario Descriptions

A summary of the low and high impact scenarios is described below and summarized in Table A-5.1.

Table A-5.1. NLEB composite plausible future scenarios. Pd rate refers to whether % species composition was reduced following Pd arrival.

Plausible Scenario	**WNS Spread**	**WNS Duration**	**Wind Capacity**	**All-bat Fatality Rate**	**% Species Composition**	***Pd* rate**
Low impact	*Pd* occurrence model 1	15-yr species-specific survival rates	Lower build-out	Regional- specific	U.S. - combined, Canada - regional-specific	No
High impact	*Pd* occurrence model 2	40-yr species-specific survival rates	Higher build-out	Regional- specific	U.S. - combined, Canada - regional-specific	No

WNS

For current projections, we used the two *Pd* occurrence models (see Appendix 2) to assign a WNS stage to all known hibernacula. Table A-5.2 provides the current (2020) number of winter colonies in each of the five WNS stages.

Table A-5.2. Number of NLEB colonies in 2020 per WNS stage under Pd occurrence models 1 and 2.

Model	**Pre-arrival**	**Invasion**	**Epidemic**	**Established**	**Post-established**
Pd occurrence model 1	1 (0.1%)	0 (0%)	23 (3%)	320 (44%)	389 (53%)
Pd occurrence model 2	3 (0.4%)	11 (2%)	60 (8%)	140 (19%)	507 (69%)

The difference between the low and high impact scenarios is based on past year of arrival of *Pd* and future rate of *Pd* spread. We used *Pd* occurrence model 1 (Wiens et al. 2022, pp. 226–229) in our low impact scenario and *Pd* occurrence model 2 (Hefley et al. 2020, entire) in our high impact scenario. As *Pd* expands its range, we expect bat populations to be impacted similarly across the species' range. Thus, we apply the same WNS impacts schedule in low and high impact scenarios. Each hibernaculum's population abundance trajectory is divided into three segments with differing λ values: a pre-*Pd*-arrival λ typically ≳1, a *Pd*-arrival λ typically <1, and a post-established λ that can be less than, greater than, or approximately equal to 1. From years since arrival (YSA) 0 to 6, λ varied annually based on results of the status and trends model. We used site specific estimates to the extent possible, although relatively few colonies had sufficient data from counts more than 6 YSA. Therefore, for YSA >6, λ was estimated as the average predicted rate of change in that time period and is held constant through YSA=15 (low impact scenario) and through YSA=40 (high impact scenario). Based on current information, we do not foresee a scenario in which *Pd* is eradicated from sites, and we expect the fungus will continue to cause disease in populations even as some individuals exhibit resistance or tolerance to it. Thus, we set the duration of impacts under the high impact scenario to 40 years (i.e., the time throughout which WNS will affect survival in the population). To understand the sensitivity of the results to the duration of the disease dynamic and to fully capture the uncertainty, we used the shortest reasonable disease dynamic duration in the low impact scenario. Based on current data (i.e., data from hibernacula documented with WNS in 2008 continue to show impacts of disease through 2021, 14-years), 15 years is the shortest duration WNS would affect populations after *Pd* arrives. After YSA=15 *(*low impact) or YSA=40 (high impact), λ is assumed to return to pre-WNS rates (i.e., no further WNS impacts applied).

Wind

U.S. Current and Future Wind Capacity

We obtained current wind capacity data for the U.S. from the USWTDB (version 3.2) (Hoen et al. 2018) and corrected/incorporated curtailment information based on facility-specific, unpublished USFWS data. For future projections, we considered projections for 2030, 2040 and 2050 from four potential sources: (1) the U.S. Department of Energy (USDOE) April 2015 Wind Vision report (USDOE 2015, entire) & downloadable data for 2020; (2) the U.S. Energy Information Administration (USEIA) January 2020 Annual Energy Outlook (AEO) report (USEIA 2020, entire) and downloadable data; (3) the USFWS April 2016 Draft Midwest Wind Multi-Species Habitat Conservation Plan (USFWS 2016, Appendix B); and (4) the National Renewable Energy Laboratory (NREL)'s 2020 Standard Scenarios Report (Cole et al. 2020, entire) and downloadable data.

After exploring these data sets and their stated purposes and underlying assumptions and consulting with experts from the USEIA, USDOE, and NREL, we ultimately decided that the NREL Standard Scenarios would serve best for the purposes of our analysis. According to the Standard Scenarios report, it is "o*ne of a suite of National Renewable Energy Laboratory (NREL) products aiming to provide a consistent and timely set of technology cost and performance data and define a scenario framework that can be used in forward-looking electricity analyses by NREL and others. The long-term objective of this effort is to identify a range of possible futures for the U.S. electricity sector that illuminate specific energy system issues. This is done by defining a set of prospective scenarios that bound ranges of technology, market, and macroeconomic assumptions and by assessing these scenarios in NREL's market models to understand the range of resulting outcomes, including energy technology deployment and production, energy prices, and emissions*" (Cole et al. 2020, p. iii).

In addition to a Mid-case Scenario, which uses the reference, mid-level, or default assumptions for all scenario inputs, represents a reference case, and provides a useful baseline for comparing scenarios and evaluating trends, the NREL's 2020 report presents 46 power sector scenarios for the contiguous U.S. (CONUS) that consider the present day through 2050. The NREL report notes, "*the Standard Scenarios are not "forecasts," and we make no claims that our scenarios have been or will be more indicative of actual future power sector evolution than projections made by others*" (Cole et al. 2020, p. 1); however, our experts advised that although the NREL report doesn't calculate a level of probability associated with any given scenario, the Mid-case Scenario is a justifiably reasonable baseline scenario for future wind deployment to use in our analysis.

After further exploring the NREL Standard Scenarios data, we discussed with USDOE and NREL experts the option of using high and low deployment bounds rather than, or in addition to, a reasonable central projection (i.e., Mid-case Scenario). Our experts agreed that this approach would help to capture some of the uncertainty associated with modeled projections; however, we were cautioned not to simply use the lowest and highest deployment scenarios since some scenarios might best be thought of as edge cases intended to show the sensitivity of the model to tweaks in assumptions rather than realistic characterizations of future deployment. Instead, we were advised to use the High and Low Onshore Wind Cost Scenarios as a reasonable combination of scenarios for our SSA analysis, and ultimately decided to apply them as lower and upper bounds, respectively, for the U.S. projections.

The Mid-case, High Wind Cost, and Low Wind Cost Scenarios each implement a slightly different set of assumptions for electricity demand, fuel prices, electricity generation and technology costs, financing, resource and system conditions and more. Under the High Onshore Wind Cost Scenario (our lower bound or "Low Build-out Scenario"), other energy technologies become more cost competitive compared to new wind energy facilities or repowering existing sites. As wind turbines reach their end of life, more are retired than are replaced with newer machines, condensing where wind energy is deployed to only the most optimal sites that present the fewest barriers and the greatest return on investment (B. Straw 2021, personal communication). Therefore, under this scenario, the distribution of wind turbines across the species' range by 2050 is reduced compared to 2020 build-out and total wind capacity decreased for several regions (Table A-5.3), although total U.S. wind capacity is projected to increase slightly. Under the Low Onshore Wind Cost Scenario (our upper bound, or "High Build-out Scenario"), repowering existing wind energy facilities or installing new wind facilities is more cost competitive compared to other energy technologies, resulting in a broader future distribution of wind turbines across the U.S. and higher overall capacity compared to 2020 build-out (Table A-5.3, Figures 4.9–4.11). For a summary of input assumptions used in the Standard Scenarios see Appendix A.1 from the 2020 Standard Scenarios report (https://cambium.nrel.gov/). We assumed total curtailed MW per NREL grid cell would remain unchanged into the future unless MW capacity declined; in these cases, we reduced grid cell curtailment proportionally (e.g., if MW capacity is projected to decline from 10 to 1 MW and currently there is curtailment on 9 MW, there would be 0.9 MW with curtailment and 0.1 MW without curtailment; Udell et al. 2022, entire).

Canada Current and Future Wind Capacity

We obtained current wind capacity data for Canada from the Canadian Wind Turbine Database (CWTD) (Government of Canada 2020, entire). To obtain current and future wind capacity for

Canada, the SSA wind team considered current buildout and projections for 2030, 2040 and 2050 from two sources: (1) The Canadian Wind Energy Association (CanWEA) (CanWEA undated, entire); and (2) The Canada Energy Regulator (CER) Canada's Energy Future 2019 Report (CER 2019, entire). We decided that the CanWEA data would not serve well for our analysis because adequate projections were lacking through the future decades (2020–2050) for most provinces as well as the entire country.

The CER Canada's Energy Future 2019 (EF 19) report is an annual report published by the Government of Canada starting in 2013 and presents projections for wind energy buildout and future capacity through 2040 through updated baseline projections from previous years. According to the report "*the Energy Futures series explores how possible energy futures might unfold for Canadians over the long term. Energy Futures uses economic and energy models to make these projections. They are based on assumptions about future trends in technology, energy and climate policies, energy markets, human behavior and the structure of the economy*" (CER 2019, p. 1). The baseline projections EF 19 are based on one future projection scenario called the Reference Case. According to the report, the Reference Case is "*based on a current economic outlook, a moderate view of energy prices and technological improvements, and climate and energy policies announced and sufficiently detailed for modeling at the time of analysis*" (CER 2019, p. 1).

After we had selected the EF 2019 data for our analysis, the CER published an updated report (EF 20 report) in November 2020 (CER 2020, entire). Similar to previous reports, the EF 20 report presents projections for wind energy buildout and future capacity through updated baseline projections from previous years. Unlike its predecessors, the EF 20 projects buildout scenarios through 2050, 10 years longer than previous years. Additionally, unlike previous reports, the EF 20 Report analyzes two buildout scenarios rather than one: the Evolving Scenario and the Reference (baseline) Scenario. According to the report, the Evolving Scenario "*considers the impact of continuing the historical trend of increasing global action on climate change throughout the projection period. Globally, this implies lower demand for fossil fuels, which reduces international market prices. Advancements in low carbon technologies lead to improved efficiencies and lower costs. Within Canada, we assume a hypothetical suite of future domestic policy developments that build upon current climate and energy policies.*" (CER 2020, p. 4). The 2020 Reference Scenario "*provides an update to what has traditionally been the baseline projection in the Energy Futures series, the Reference Scenario. The scenario considers a future where action to reduce GHG emissions does not develop beyond measures currently in place. Globally, this implies stronger demand for fossil fuels, resulting in higher international market prices compared to the Evolving Scenario. Low carbon technologies with existing momentum continue to improve, but at a slower rate than in the Evolving Scenario*" (CER 2020, p. 4).

In addition to being more up-to-date than the 2019 data, the dual buildout scenarios included in the 2020 Update presented an opportunity to analyze a range of scenarios rather than a single projection and set of assumptions. Therefore, we assigned the Evolving Scenario as an upper bound buildout scenario and the Reference Scenario as a lower bound scenario for our analysis.

Table A-5.3. Wind capacity (MW) by USFWS Region and Canadian Province under 2020 and 2050 low and high scenario build-out.

Location	**Wind Capacity (MW)**		
	2020 Build-out	**2050 Low Build-out (% change)**	**2050 High Build-out (% change)**
Region 3	27,387	15,198 (-45%)	141,573 (+417%)
Region 6	21,280	40,944 (+92%)	83,033 (+290%)
Region 5	6,116	7,252 (+19%)	68,946 (+1027%)
Region 1	7,459	1,422 (-81%)	19,102 (+156%)
Region 8	2,466	1,414 (-43%)	20,624 (+736%)
Region 4	240	391 (+63%)	38,083 (+15768%)
Region 2	39,964	40,511 (+1%)	116,346 (+191%)
U.S. Total	104,912	107,132 (+2%)	487,707 (+365%)
Alberta	1,746	6,699 (+284%)	10,286 (+489%)
British Columbia	732	1,252 (+71%)	1,967 (+169%)
Manitoba	258	476 (+85%)	851 (+230%)
Ontario	5,436	5,646 (+4%)	12,300 (+126%)
Quebec	4,330	5,830 (+35%)	6,930 (+60%)
Atlantic Canada	873	1,408 (+61%)	2,394 (+174%)
Saskatchewan	221	3,256 (+1373%)	5,781 (+2516%)
Canada Total	13,597	24,569 (+81%)	40,510 (+198%)
U.S. + Canada	118,509	131,701 (+11%)	528,217 (+346%)