Species Status Assessment (SSA) Report

for the

Tricolored Bat

(*Perimyotis subflavus*)

Version 1.1

Tricolored bats (Photo credit: Tim Krynak, Cleveland Metroparks)

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[1] This SSA was developed in tandem with analyses and reports for the northern long-eared bat (NLEB; *Myotis septentrionalis*) and little brown bat (LBB; *Myotis lucifugus*).

EXECUTIVE SUMMARY

This report summarizes results of a species status assessment (SSA) conducted for the tricolored bat (TCB; *Perimyotis subflavus*). TCB is a widely distributed small insectivorous bat of eastern North America. Readily identifiable by its tricolored fur, TCB primarily roost in foliage of live and dead trees in the spring, summer, and fall, and hibernate in caves and other subterranean habitats during the winter.

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

For survival and reproduction at the individual level, TCB require suitable roosting and foraging habitat near abundant food and water resources in the spring, summer and fall; habitat with suitable microclimate conditions for prolonged bouts of torpor and shortened periods of arousal in the winter; and suitable habitat connectivity between summer and winter habitats. For TCB 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, TCB require resiliency (demographic, physically, and genetically healthy populations across a diversity of environmental conditions), representation (genetic and ecological diversity to maintain adaptive capacity), and redundancy (multiple and sufficient distribution of populations within areas of unique variation). Resiliency is the ability of the species to withstand environmental and demographic stochasticity and, in the case of TCB, is best measured by the number, distribution, and health of populations across the species' range. Redundancy is an indicator of the ability of a species to withstand catastrophic events by "spreading the risk" and can be measured through the duplication and distribution of resilient populations across the species' range. Representation is an indicator of the ability of a species to adapt to changing environmental conditions and can be measured by the number and distribution of healthy populations across areas of unique adaptive diversity. For TCB, we identified three representation units (RPUs).

The primary factors influencing TCB's viability which have led to its current condition include white-nose syndrome (WNS), wind related mortality, effects from climate change, habitat loss, and conservation efforts.

- WNS is a disease of bats that is caused by the fungal pathogen *Pseudogymnoascus destructans* (*Pd*). *Pd* invades the skin of bats, initiating a cascade of physiological and behavioral processes that often lead to mortality.

- Wind related mortality of TCB is also proving to be a consequential stressor at local and regional levels. TCB are killed at wind energy projects primarily through collisions with moving turbine blades.

- Loss of roosting, foraging, and commuting habitat may lead to minor or significant impacts to TCB depending on the timing, location, and extent of the removal. Loss or modification of winter habitats may also result in negative impacts to TCB, especially given the species' high site fidelity and narrow microclimate requirements for hibernation. Additionally, disturbance (e.g., human entry) during hibernation results in increased arousals in TCB, which leads to increased energy expenditure at a time when food and water resources are scarce or unavailable.

- Changing climatic variables including changes in temperature and precipitation influence TCB's resource needs, such as suitable summer and winter roosting habitat, foraging habitat, and prey availability. Although pervasive across TCB's range, the magnitude, direction, and seasonality of climate change will vary geographically (e.g., some regions will experience more frequent droughts which may lead to reduced TCB survival or reproductive success; alternatively, some regions will experience heavier and more frequent precipitation events that may lead to decreased foraging bouts and insect availability).

- Conservation efforts include multiple national and international initiatives underway in an attempt to reduce the impacts of WNS (to date, however, there are no proven measures to reduce the severity of impacts). Additionally, some wind facilities within TCB's range are implementing curtailment (e.g., feathering turbine blades during low wind periods) to reduce bat fatalities.

We used the best available data to assess TCB viability over time. Winter hibernacula counts provide the most consistent, long-term, reliable trend data, and provide the most direct measure of WNS impacts and thus were used to assess TCB current and future viability. The availability and quality of summer data substantially vary temporally and spatially but were useful for evaluating past population trends. We relied upon the data derived from North American Bat Monitoring Program (NABat) analyses of all available winter and summer data. Current demographic conditions based on past declines indicate TCB's rangewide winter abundance and number of extant winter colonies have declined by 52% and 29%, respectively. TCB winter abundance has declined across all RPUs but varies spatially (24–89%). Declining trends in TCB occurrence and abundance is also evident from summer data: 1) TCB rangewide occupancy declined 28% from 2010–2019; 2) mobile acoustic detections decreased 53% from 2009–2019; and 3) summer mist-net captures declined 12% compared to pre-WNS capture rates. Based on current conditions, future projections of TCB abundance, number of hibernacula, and spatial extent will continue to decline. By 2030, rangewide abundance declines by 89%, the number of winter colonies declines by 91%, and TCB's spatial extent declines by 65%. Projected declines in TCB's abundance, number of winter colonies, and spatial extent are widespread across all RPUs under current conditions.

To assess TCB'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: 1) WNS spread under Hefley et al. (2020, entire) model and lower wind energy capacity (low impact scenario) and 2) WNS spread under Wiens et al. (2022, pp. 215–248) model and higher wind energy capacity (high impact scenario). Using these scenarios and NABat data, we projected the species' abundance and distribution. We also qualitatively considered impacts from climate change, habitat loss, and conservation efforts. Under the future scenarios, TCB declines worsen precipitously, with rangewide and RPU-level declines predicted in abundance, number of winter colonies, and spatial extent.

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. Wind energy related mortality is also proving, especially in light of the steep declines stemming from WNS impacts, to be a pervasive and consequential driver to TCB's viability, with an estimated 3,327 TCB killed annually at wind facilities across the species' range. Although we consider habitat loss pervasive across TCB's range, severity has likely been low given historical abundance and spatial extent; however, as TCB's spatial extent is projected to decline in the future (i.e., consolidation into fewer winter and summer colonies) negative impacts (e.g., loss of a hibernaculum or maternity colony) may be significant. Lastly, although challenging to describe for such a wide-ranging species, climate change will continue and negative impacts are anticipated in the future.

In summary, TCB abundance has declined significantly and winter abundance, number of occupied hibernacula, spatial extent, and summer habitat occupancy are decreasing. Since the arrival of WNS, TCB abundance steeply declined. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, TCB's ability to recover from these low abundances is limited given their low reproduction output (two pups per year). Therefore, TCB's resiliency is greatly compromised in its current condition and is projected to worsen under future stressor conditions. Additionally, because TCB's spatial extent is projected to decline, TCB will become more vulnerable to catastrophic events. Lastly, the steep and continued declines in abundance have likely led to reductions in genetic diversity, and thereby reducing TCB's ability to adapt to changes in its biological and physical environments. 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 TCB'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 TCB'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, TCB's viability is likely to rapidly decline over the next 10 years.

There is currently uncertainty associated with progression of WNS within TCB winter colonies at road-associated culverts used as hibernacula in the southern U.S. No *Pd* has been detected at culverts in Louisiana and although *Pd* has been detected since 2014 at several culverts in Mississippi, no disease, mortality, or population impacts have been documented. Whether

environmental (e.g., shorter and milder winters) or biological factors (e.g., shorter torpor bouts, winter foraging opportunities) contribute to the differences observed at culverts is currently unknown.

Contents

ABBREVIATIONS 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 – Celcius
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
NABat – North American Bat Monitoring Program
NCSL – National Conference of State Legislatures
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

This report summarizes the results of a species status assessment (SSA) conducted for the tricolored bat (*Perimyotis subflavus*; TCB). It delivers the best available scientific and commercial information available on TCB 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 TCB'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 TCB. Chapter 4 describes the key drivers that led to TCB'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 TCB 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 and rainfall), periodic disturbances within the normal range of variation (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 (Mangel 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, pp. 2–3; Hendry et al. 2011, pp. 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, pp. 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, pp. 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, pp. 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, pp. 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, entire), it is lacking for most species. Fortunately, there are several proxies that collectively can serve as indicators of potentially underlying adaptive genetic variation: (1) phenotypic variation; (2) neutral genetic variation; and (3) disjunct or peripheral populations. 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, pp. 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 TCB 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.

Like 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 others) across the range to garner the best available and useable 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; accessed February 10, 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 TCB. Colony count data represents 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 (including extremely scarce information from Mexico and Central America; Reid 1997, p. 154; Medina-Fitoria et al. 2015, p. 49; Turcios-Casco et al. 2020, p. 532; Turcios-Casco et al. 2021, p. 10), 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, pp. 231–247) estimated the yearly population abundance (N) from 1990–2020. From these yearly abundances, λ was estimated over time for each hibernaculum. For sites with insufficient data-points, Wiens et al. (2022, pp. 231–247) applied λ values from the nearest neighbor. To capture uncertainty in the year of arrival of *Pseudogymnoascus destructans* (*Pd*), they calculated yearly abundance trajectories under two different *Pd* occurrence models (Wiens et al. 2022, pp. 226–229 and Hefley et al. 2020, entire).

Step 2. Describe Current Abundance, Health, and Distribution

To estimate current conditions, we relied upon analyses completed by Wiens et al. (2022, pp. 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 additional insights on population trends at regional scales (see Appendix 2A).

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 TCB's 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, pp. 28–35).

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 2A 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 beyond 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:

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

Where *Bfat* is the all-bat fatality rate per megawatt (MW) and *%Sp* is the species-specific percent composition of fatalities reported (see Appendix 2A 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 related mortality, we used the population model, BatTool (updated with TCB-specific demographic values). 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 TCB'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 TCB'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 events and epizootics. To assess TCB's ability to adapt to novel changes in its physical and biological environment, we characterized TCB's 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 2B).

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 and Figure 1.3, with more detail in Appendix 2. A conceptual model of the BatTool is provided in Figure 1.4. 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. 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

Citation	Data/Analyses	Step in Analytical Process	Chapter
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 1.4. 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

The tricolored bat (*Perimyotis subflavus*; TCB) was first described by Cuvier in 1832 from specimens collected from the eastern U.S., likely Georgia (Fujita and Kunz 1984, p. 1). Various common names have been applied to TCB, including Georgian bat, pigmy bat, southern pipistrel, and most commonly: eastern pipistrelle (Fujita and Kunz 1984, p. 4). In addition, this species has been identified by different scientific names: *Vespertilio subflavus*, *V. erythrodactylus*, *V. monticola*, *Vesperugo veraecrucis* and *Pipistrellus subflavus* (Fujita and Kunz 1984, p. 1). In 1897, Miller (pp. 90–95) placed TCB into genus *Pipistrellus* where it remained until recent phylogenetic analyses confirmed TCB do not share a recent common ancestor with other *Pipistrellus*-like bats and consequently belong in their own genus, *Perimyotis* (Hoofer and Van Den Bussche 2003, pp. 32–34; Hoofer et al. 2006, pp. 982–983). Davis (1959, entire) described four subspecies (*Pipistrellus subflavus subflavus*, *P. s. clarus*, *P. s. floridanus*, and *P. s. veraecrucis*) based on geographic variation in color, size, and cranial measurements (Figure 2.1); an analysis of TCB genetics across its entire range has not been conducted. Furthermore, when the genus reclassification from *Pipistrellus* to *Perimyoti*s was completed, no separate subspecies were proposed. Consequently, we find this point to be more persuasive than the morphological information provided in Davis 1959 (entire). Therefore, for the purposes of this SSA, we considered TCB a valid taxon and monotypic (Hoofer and Van Den Bussche 2003, entire).

As we mentioned above, TCB genetics information is limited. In one study, Martin (2014, entire) examined mitochondrial and microsatellite markers to assess genetic structure across TCB's eastern and midwestern range. Mitochondrial markers separated by large geographic distances were more genetically distinct[2] and suggest two subpopulations across the sampled range (Figure 2.1) (Martin 2014, pp. 20 and 39). Microsatellites, however, suggested very little genetic differentiation (Martin 2014, p. 21). Martin (2014, p. 24) postulated this observed pattern of significant structure in maternally inherited markers (i.e., mitochondrial DNA) with a lack of structure in nuclear markers (i.e., microsatellites) may be the result of male-biased dispersal, but additional analyses are required. Unfortunately, large portions of TCB's range were not sampled. We are unaware of additional genetic studies that analyze TCB population genetics in greater depth and we suggest more research is needed.

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[2] One exception was TCB samples collected from Vermillion, Indiana, which were significantly genetically distinct from all other sites; however, there were haplotypes shared with locations in the *West* population supporting some low but non-negligible intra-regional female dispersal (Martin 2014, pp. 25–26).

Figure 2.1. Mitochondrial markers suggest two subpopulations across the sampled range (from Martin 2014, p. 39).

Species Description

TCB (Figure 2.2) is one of the smallest bats in eastern North America and is distinguished by its unique tricolored fur that appears dark at the base, lighter in the middle, and dark at the tip (Barbour and Davis 1969, p. 115). TCB often appear yellowish (varying from pale yellow to nearly orange), but may also appear silvery-gray, chocolate brown, or black (Barbour and Davis 1969, p. 115). Males and females are colored alike, but females are consistently heavier than males (LaVal and LaVal 1980, p. 44). Newly volant young are much darker and grayer than adults (Allen 1921, p. 55). Other distinguishing characteristics include 34 teeth (compared with 38 teeth in eastern North American *Myotis* spp. for which it is sometimes confused), a calcar (i.e., spur of cartilage arising from the inner side of the ankle) with no keel, and only the anterior third of the uropatagium (i.e., the membrane that stretches between the legs) is furred (Barbour and Davis 1969, p. 115; Hamilton and Whitaker 1979, p. 85).

Figure 2.2. TCB bat with young (photo credit: Christopher E. Smith, Minnesota Department of Transportation).

Species Range

TCB are known from 39 States (Alabama, Arkansas, Colorado, Connecticut, Delaware, Florida, Georgia, Illinois, Indiana, Iowa, Kansas, Kentucky, Louisiana, Maine, Maryland, Massachusetts, Michigan, Minnesota, Mississippi, Missouri, Nebraska, New Hampshire, New Jersey, New Mexico, New York, North Carolina, Ohio, Oklahoma, Pennsylvania, Rhode Island, South Carolina, South Dakota, Tennessee, Texas, Vermont, Virginia, Wisconsin, West Virginia, Wyoming), Washington D.C., 4 Canadian Provinces (Ontario, Quebec, New Brunswick, Nova Scotia), and Guatemala, Honduras, Belize, Nicaragua and Mexico (Figure 2.3). The species current distribution in New Mexico, Colorado, Wyoming, South Dakota and Texas is the result of westward range expansion in recent decades (Geluso et al. 2005, p. 406; Adams et al. 2018, entire; Hanttula and Valdez 2021, p. 132) as well as into the Great Lakes basin (Kurta et al. 2007, p. 405; Slider and Kurta 2011, p. 380). This expansion is largely attributed to increases in trees along rivers and increases in suitable winter roosting sites, such as abandoned mines and other human-made structures (Benedict et al. 2000, p. 77; Geluso et al. 2005, p. 406; Slider and Kurta 2011, p. 380).

Figure 2.3. TCB range[3].

Individual-level Ecology and Needs

Below we describe the life history and ecological needs for TCB individuals to survive and reproduce; life history and ecological needs are summarized in Table 2.1 and Figure 2.5.

Longevity—The oldest TCB on record is a male captured in Illinois 14.8 years after it was originally banded (Walley and Jarvis 1972, p. 305). Paradiso and Greenhall (1967, pp. 251–252) reported an 11.2 year-old female in West Virginia and 4 additional TCB living at least 10 years. Based on monitoring recaptures over twelve years at two caves in West Virginia, Davis (1966, p. 389) suggested TCB survival is low in the first year, peaks in the third year, and then decreases as maximum life span is approached (Davis 1966, p. 389); however, this study did not account for the possibility that some individuals dispersed to different hibernacula.

Sheltering—During the spring, summer, and fall (i.e., non-hibernating seasons), TCB primarily roost among live and dead leaf clusters of live or recently dead deciduous hardwood trees (Veilleux et al. 2003, p. 1071; Perry and Thill 2007, pp. 976–977; Thames 2020, p. 32). In the southern and northern portions of the range, TCB will also roost in Spanish moss (*Tillandsia usneoides*) and *Usnea trichodea* lichen, respectively (Davis and Mumford 1962, p. 395; Poissant 2009, p. 36; Poissant et al. 2010, p. 374). In addition, TCB have been observed roosting during

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[3] Note map does not include single TCB record from northwestern Nicaragua (Medina-Fitoria et al. 2015, p. 49).

summer among pine needles (Perry and Thill 2007, p. 977), eastern red cedar (*Juniperus virginiana*) (Thames 2020, p. 32), within artificial roosts (e.g., barns, beneath porch roofs, bridges, concrete bunkers) (Jones and Pagels 1968, entire; Barbour and Davis 1969, p. 116; Jones and Suttkus 1973, entire; Hamilton and Whitaker 1979, p. 87; Mumford and Whitaker 1982, p. 169; Whitaker 1998, p. 652; Feldhamer et al. 2003, p. 109; Ferrara and Leberg 2005, p. 731; Smith 2020, pers. comm.), and rarely within caves (Humphrey et al. 1976, p. 367; Briggler and Prather 2003 p. 408; Damm and Geluso 2008, p. 384). Female TCB exhibit high site fidelity, returning year after year to the same summer roosting locations (Allen 1921, p. 54; Veilleux and Veilleux 2004a, p. 197). Female TCB form maternity colonies and switch roost trees regularly (e.g., between 1.2 days and 7 days at roost trees in Indiana) (Veilleux and Veilleux 2004a, p. 197; Quinn and Broders 2007, p. 19; Poissant et al. 2010, p. 374). Males roost singly (Perry and Thill 2007, p. 977; Poissant et al. 2010, p. 374).

During the winter, TCB hibernate (i.e., reduce their metabolic rates, body temperatures, and heart rate) in caves and mines, although in the southern U.S., where caves are sparse, TCB often hibernate in road-associated culverts (Sandel et al. 2001, p. 174; Katzenmeyer 2016, p. 32; Limon et al. 2018, entire; Bernard et al. 2019, p. 5; Lutsch 2019, p. 23; Meierhofer et al. 2019, p. 1276) and sometimes tree cavities (Newman 2020, p. 14) and abandoned water wells (Sasse et al. 2011, p. 126). TCB exhibit high site fidelity with many individuals returning year after year to the same hibernaculum (Davis 1966, p. 385; Jones and Pagels 1968, p. 137; Jones and Suttkus 1973, p. 964; Sandel et al. 2001, p. 175).

TCB are one of the first cave-hibernating species to enter hibernation in the fall and one of the last to leave in the spring in Missouri and Pennsylvania (LaVal and LaVal 1980, p. 29; Merritt 1987, p. 102). In the southern U.S., hibernation length is shorter compared to northern portions of the range and some TCB exhibit shorter torpor bouts and remain active and feed during the winter (Layne 1992, pp. 43–44; Grider et al. 2016, p. 8; Limon et al. 2018, p. 219; Newman 2020, pp. 13–17; Stevens et al. 2020, p. 528). The number of hibernating TCB does not peak at caves and mines until December or later, suggesting some bats stay on the landscape or in alternate hibernacula and only move in to caves and mines when it gets colder (Barbour and Davis 1969, p. 119; Vincent and Whitaker 2007, p. 61), although, in some cases, TCB may remain on the landscape and hibernate in rock shelters (e.g., fissures in sandstone and sedimentary rock) (Johnson 2021, pers. comm.).

TCB are often found hibernating at warmer locations within caves and mines compared to other cave-hibernating bat species within these locations (Barbour and Davis 1969, p. 119; Raesly and Gates 1987, p. 17). TCB was observed hibernating at a mean temperature of 51.6 degrees Fahrenheit (F; 10.9 degrees Celsius (C)) (range 50.5 – 52.5 degrees F (10.3–11.4 degrees C)) at caves and mines in Pennsylvania, Maryland, and West Virginia (Raesly and Gates 1987, p. 18). TCB are also found in areas of caves and mines with high humidity (e.g., 99%; Mohr 1976, p. 97) and were not observed in caves where relative humidity was below 80% (Ploskey and Sealander 1979, p. 72).

Hibernating TCB do not typically form large clusters; most commonly roost singly, but sometimes in pairs, or in small clusters of both sexes away from other bats (Hall 1962, p. 29; Barbour and Davis 1969, p. 117; Mumford and Whitaker 1982, p. 169; Raesly and Gates 1987,

p. 19; Briggler and Prather 2003, p. 408; Vincent and Whitaker 2007, p. 62). TCB roost on cave walls (more often) and ceilings and are rarely found in cave crevices (Mumford and Whitaker 1982, p. 169). TCB will shift roosts from one to another during the winter but arouse less frequently than other cave-hibernating bat species (Barbour and Davis 1969, p. 119; Mumford and Whitaker 1982, p. 169); consequently, sometimes water beads will collect on their fur making them appear almost white (Hamilton 1943, p. 86; Barbour and Davis 1969, p. 119). In road associated-culverts in the southern U.S., however, TCB exhibit shorter torpor bouts and move within and between culverts throughout the winter (Anderson et al. undated).

TCB hibernate in more caves and mines than any other cave-hibernating bat species in eastern North America (Sealander and Young 1955, pp. 23–24; Barbour and Davis 1969, p. 117; Brack et al. 2003, p. 65). TCB may use small caves and mines that are unsuitable to other cave-hibernating bat species (Barbour and Davis 1969, p. 117; Mumford and Whitaker 1982, p. 168; Hamilton and Whitaker 1979, p. 87); however, hibernating TCB have been observed in greater numbers in hibernacula with stable temperatures (Briggler and Prather 2003, p. 411). Raesly and Gates 1987 (p. 19) found TCB hibernating in 80% of the 50 locations surveyed in Pennsylvania versus little brown bats (*Myotis lucifugus*), Indiana bats (*M. sodalis*), northern long-eared bats (*M. septentrionalis*), and big brown bats (*Eptesicus fuscus*) which were found in 56%, 16%, 16%, and 34% of potential hibernacula, respectively. Almost every cave in Indiana has contained at least one TCB (Mumford and Whitaker 1982, pp. 167–168); and small numbers of TCB have likely occupied most of Missouri's 6,400 caves (Perry 2021, pers. comm.).

Prior to the arrival of WNS (see Chapter 4), hibernating TCB colonies varied between 1 and 5,300 individuals; however, 40% of hibernacula had between just 1 and 10 individuals (Figure 2.4). The largest TCB hibernating colony (n = 5,300) was observed in Georgia in 2010 (NABat 2021).
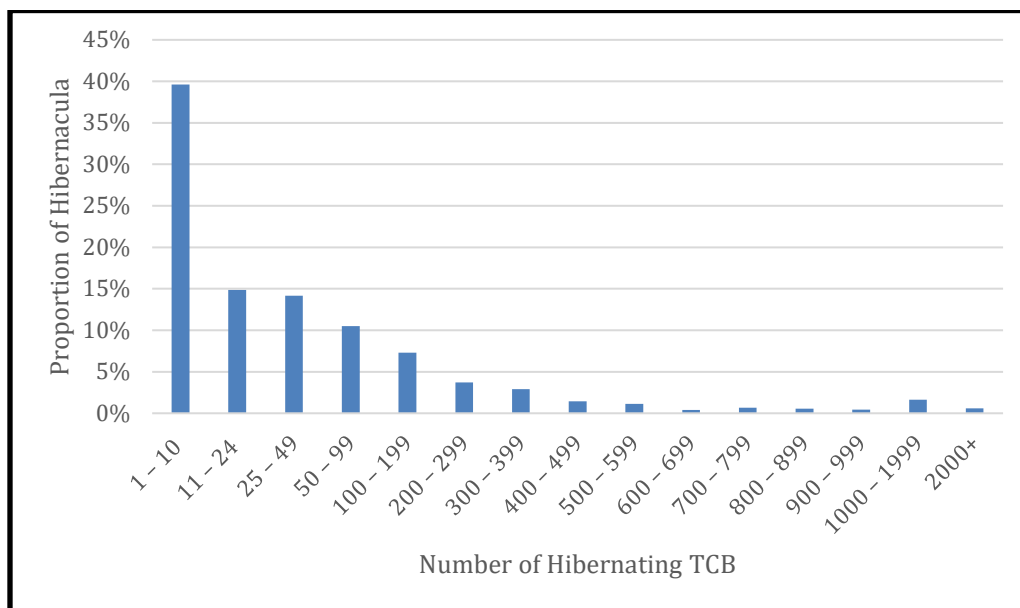

Figure 2.4. Total number of hibernating TCB observed during winter counts at hibernacula (n=1,236) prior to the arrival of white-nose syndrome (NABat 2021; accessed February 10, 2021).

Feeding— TCB are opportunistic feeders and consume small insects including caddisflies (Trichoptera), flying moths (Lepidoptera), small beetles (Coleoptera), small wasps and flying ants (Hymenoptera), true bugs (Homoptera), and flies (Diptera) (Whitaker 1972, p. 879; LaVal and LaVal 1980, p. 24; Griffith and Gates 1985, p. 453; Hanttula and Valdez 2021, p. 132). TCB emerge early in the evening and forage at treetop level or above (Davis and Mumford 1962, p. 397; Barbour and Davis 1969, p. 116) but may forage closer to ground later in the evening (Mumford and Whitaker 1982, p. 170). TCB exhibit slow, erratic, fluttery flight while foraging (Fujita and Kunz 1984, p. 4) and commonly forage with eastern red bats (*Lasiurus borealis*) and silver-haired bats (*Lasionycteris noctivagans*) (Davis and Mumford 1962, p. 397; Mumford and Whitaker 1982, p. 169). TCB forage most commonly over waterways and forest edges (Barbour and Davis 1969, p. 116; Mumford and Whitaker 1982, pp. 170–171; Hein et al. 2009, p. 1204). Maximal distance traveled from roost areas to foraging grounds was 4.3 kilometers (km; 2.7 miles) for reproductive (pregnant or lactating) adult females in Indiana (Veilleux et al. 2003, p. 1074) and 24.4 km (15.2 miles) (mean=11.4 km; 7.1 miles) for male TCB in Tennessee (Thames 2020, p. 61).

Reproduction— Male and female TCB converge at cave and mine entrances between mid-August and mid-October to swarm and mate. Adult females store sperm in their uterus during the winter and fertilization occurs soon after spring emergence from hibernation (Guthrie 1933, p. 209). Females typically give birth to two young, rarely one or three between May and July (Allen 1921, p. 55; Barbour and Davis 1969, p. 117; Cope and Humphrey 1972, p. 9). Young grow rapidly and begin to fly at 3 weeks of age and achieve adult-like flight and foraging ability at 4 weeks (Lane 1946, p. 59; Whitaker 1998, pp. 653–655). Adults often abandon maternity roosts soon after weaning, but young remain longer (Whitaker 1998, p. 653). TCB are considered juveniles (i.e., subadults) when entering their first hibernation and most probably do not mate their first fall (Fujita and Kunz 1984, p. 3).

Maternity colonies consist of 1 to 8 (mean = 4.4) females and pups at tree roosts in Indiana (Veilleux and Veilleux 2004b, p. 62). Perry and Thill 2007 (p. 977) observed an average of 6.9 adult females and pups per colony in Arkansas (range 3 to 13). Maternity colonies include up to 18 females in trees in Nova Scotia (Poissant et al. 2010, p. 374). Whitaker (1998, p. 652) found colonies in buildings averaged 15 adult females (range 7 to 29 adult females). Hoying and Kunz 1998 (p. 19) reported the largest colony on record in a Massachusetts barn (19 adult females and 37 young).

Movement/Dispersal— TCB disperse from winter hibernacula to summer roosting habitat in the spring. Fraser et al. 2012 (p. 5) concluded that at least some TCB engage in latitudinal migration that is more typically associated with hoary bats (*Lasiurus cinereus*), eastern red bats, and silver-haired bats, and this behavior is more common for males than for females. The maximum migration distance on record is a female TCB who migrated a straight-line distance of 243 km (151 miles) from her winter hibernaculum in southern Tennessee to a summer roost in Georgia (Samoray et al. 2019, p. 17). Other migration records between winter hibernacula and summer habitat include less than 80 km (50 miles) (Barbour and Davis 1969, p. 117), 44 km (27 miles) (Samoray et al. 2019, p. 18), and 137 km (85 miles) (Griffin 1940, p. 237). Hibernaculum to

hibernaculum movement up to 209 km (130 miles) has also been documented between two consecutive winters (Lutsch 2019, p. 38).

Figure 2.5. Generalized annual life history diagram for TCB (adapted from Silvis et al. 2016, p. 1).

Table 2.1. TCB individual-level needs for survival and reproduction.

LIFE STAGE	SEASON
Pups	<u>Summer</u> - roosting habitat with suitable conditions for lactating females and for pups to stay warm and protected from predators while adults are foraging.
Juveniles	<u>Summer</u> - other maternity colony members (colony dynamics, thermoregulation); suitable roosting and foraging habitat near abundant food and water resources. <u>Fall</u> - suitable roosting and foraging habitat near abundant food and water resources. <u>Winter</u> - habitat with suitable microclimate conditions.
Reproductive Females	<u>Summer</u> - other maternity colony members (colony dynamics); network of suitable roosts (i.e., multiple summer roosts in close proximity) near conspecifics and foraging habitat near abundant food and water resources.
All Adults	<u>Spring</u> - 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. <u>Summer</u> - roosts and foraging habitat near abundant food and water resources. <u>Fall</u> - 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. <u>Winter</u> - habitat with suitable microclimate conditions.

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 TCB population are discussed below and summarized in Figure 2.6 and Table 2.2.

Demography

For TCB populations to have a healthy demography, the population growth rate (lambda or λ) must be sufficient to withstand natural environmental fluctuations. At a minimum, λ must be at

least one for a population to remain stable over time. To maintain a healthy λ and *Ne*, TCB, are dependent on their ability to select environments with ample prey and appropriate conditions at summer and winter roosting habitat. For example, TCB winter roosts require stable microclimates within narrow temperature and humidity ranges, and low levels of disturbance. During favorable hibernating conditions, TCB survival and therefore reproductive rates are high (increasing λ); conversely, when environmental conditions are unfavorable, survival and reproductive rates are low (decreasing λ). Growth rates are not expected to vary across TCB's range and population numbers generally do not experience extreme variation from year-to-year or successive generations.

Habitat Quality and Quantity

To support a strong growth rate, TCB populations benefit from large population sizes (which helps maintain genetic health via large *Ne*) and sufficient quality and quantity of habitat to accommodate all life stages. The required habitat quality to support healthy demographic rates and physical health is described under *Individual-level Ecology and Needs*. The 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. Limited research suggest the minimum summer roost area (not including foraging area) for individual adult female TCB ranges between 0.1 and 2.2 hectare (ha) (0.25 and 5.4 acre [ac]) (Veilleux and Veilleux 2004a, p. 197). Mean foraging area for 7 adult male TCB was 2,350 ha (5,807 acres) (range 234–9,655 ha; 578–23,858 acres) and 364 ha (899 acres) for a single non-reproductive female in Tennessee (Thames 2020, p. 61). Although TCB hibernate in more caves and abandoned mines than any other cave-hibernating bat species in eastern North America (see *Individual-level Ecology and Needs*), higher numbers of TCB have been observed in caves with stable temperatures (Briggler and Prather 2003, p. 411). More research is needed on the specific optimal quality of TCB habitat.

Connectivity

To support all life stages, TCB populations require a matrix of interconnected habitats that support spring migration, summer maternity colony formation, 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. TCB migrate up to 243 km (151 miles) between winter hibernacula and summer roosting sites (Samoray et al. 2019, p. 17). TCB prefer landscapes with greater forest area, forest aggregation, and tree corridors and are less abundant among urban development (Duchamp and Swihart 2009, p. 855; Farrow and Broders 2011, p. 177). Thus, large stretches of urban development (i.e., less suitable habitat) may negatively influence connectivity between summer and winter habitats.

Figure 2.6. Conceptual model showing the connections between resource needs and the physiological needs and demographic rates of a TCB 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 TCB can withstand annual variation in its environment (resiliency), catastrophes (redundancy), and novel or extraordinary changes in its environment (representation). We describe TCB's requirements for resiliency, redundancy, and representation below, and summarize the key aspects in Table 2.3.

Resiliency

TCB's ability to withstand stochastic events requires maintaining healthy populations across spatially heterogeneous conditions. Healthy populations—demographically, genetically, or physically robust—are better able to withstand and recover from environmental and demographic variability and stochastic perturbations. The greater the number of healthy populations, the more likely TCB 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 TCB's resiliency by facilitating population-level heterozygosity via gene flow and demographic rescue following population decline or extinction due to stochastic events.

Redundancy

TCB'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 TCB'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

TCB's ability to withstand ongoing and future novel changes is influenced by its capacity to adapt (referred to as adaptive capacity). TCB 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 2B). Below we describe TCB's ability to colonize new areas and to alter its physical traits.

TCB's capacity to colonize new areas (or track suitable conditions) is a function of its physical capability and behavioral tendencies to disperse. TCB flight capabilities and behavior allows for TCB to shift their summer locales in response to local novel changes. Also, dispersal ability will hinge on the availability of suitable summer and winter roosting habitat. TCB primarily roost in foliage during the summer (see *Individual-level Ecology and Needs*), so TCB need landscapes with forest habitat and tree corridors likely promote movement between forested patches. TCB is found in more hibernacula than any other North American cave-hibernating bat species (see

Individual-level Ecology and Needs) and are able to exploit human-made structures (e.g., road-associated culverts). As previously discussed above (see *Species Range*), in recent decades, TCB distribution has expanded westward and into the Great Lakes basin. This expansion signifies TCB's ability to disperse when increases in suitable summer habitat (e.g., forested areas) and suitable winter roosting sites (e.g., human-made structures) are available. Maintaining suitable habitat within local home ranges and beyond likely enables TCB to shift their range to track suitable conditions. However, despite their capacity to fly long distances, females show limited capacity to make large, abrupt shifts. Their limits are likely owing to the energetic demands of migration and reproduction at a point when their fat reserves are at their lowest after hibernation and strong philopatry (i.e., tending to return to or remain near a particular area) to both winter and summer locales. Thus, TCB adapt to changing conditions via small, local shifts but are unlikely to possess the capacity for rapid, large shifts in response to broad-scale novel changes.

TCB'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 TCB'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 TCB 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 TCB'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 illustrated in Figure 2.7. Bailey's Eco-Divisions (Bailey 2016, entire) were overlaid 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 TCB (at a broad scale) is preserved.

Figure 2.7. Range of TCB organized into three Representation Units.

1. Eastern RPU:

   Eastern TCB are identified as a RPU because they contribute unique genetic variation and behavioral traits and occupy unique ecological conditions.

   Eastern TCB have unique mitochondrial genetic variability that differentiates this population from other populations (Martin 2014, entire). The Appalachian Mountain range separates the Eastern and Northern RPUs and may serve as a barrier (although permeable) to maternal gene flow (Martin 2014, p. 26; Miller-Butterworth 2014, p. 361). The more robust investigation of little brown bats supports this concept. Miller-Butterworth et al. (2014, entire) found that the Allegheny front (i.e., escarpment along the eastern edge of the Appalachian plateau) likely influences movements of female little brown bats in Pennsylvania and West Virginia. They found that mitochondrial DNA was significantly different in hibernating little brown bats at three of five hibernacula west of the Allegheny front versus those hibernating east of the front and concluded that topography plays an important role in limiting female movements and maternal gene flow (Miller-Butterworth 2014, p. 361). In addition, to determine whether this pattern of population substructure extended beyond Pennsylvania and West Virginia, Miller-Butterworth et al. (2014, p. 360) analyzed genetic samples from a hibernating little brown bat colony in Vermont. They found that despite the geographic distance (up to 840 km (522 miles)), mitochondrial DNA was not significantly different from the bat colony in Vermont and colonies east of the Allegheny front in Pennsylvania and West Virginia.

27

The pattern of WNS (see Chapter 4) spread also supports the observed maternal gene flow patterns reported in Martin (2014, entire) and Miller-Butterworth et al. (2014, entire). The pattern and timing of spread of WNS may be partially explained by female latitudinal movements that are unobstructed by landscape features (Miller-Butterworth et al. 2014, p. 362). The initial spread of WNS between 2007 and 2009 followed the Appalachian Mountains through Pennsylvania, Maryland, Virginia, and West Virginia (Figure 4.3). All the Pennsylvania hibernacula infected with WNS during or prior to 2009 were located to the east of the Allegheny front (see Figure 1 in Miller-Butterworth et al. 2014, p. 356). Consequently, it took another 1 to 2 years for WNS to spread to hibernacula on the western side of the Allegheny front (Miller-Butterworth et al. 2014, p. 355).

The southern reach of the Eastern RPU is predominantly marked by hot summers, cool winters, and deciduous forests. The northern reach is predominantly marked by warm summers, cold winters, and coniferous forests. During the summer, Eastern TCB predominantly roost in foliage of live or recently dead deciduous hardwood trees; however, TCB in Nova Scotia are unique in their exclusive selection of *Usnea trichodea* lichen as summer roosting habitat (Poissant et al. 2010, p. 374).

2. Northern RPU:
Northern TCB are identified as a RPU because they contribute unique genetic variation and behavioral traits and occupy unique ecological conditions.

Northern TCB have unique mitochondrial genetic variability that differentiates this population from other populations (Martin 2014, entire; see *Eastern Unit* above for further discussion). The Northern RPU is predominantly marked by hot summers, cool or cold winters, deciduous forests to the east, prairies to the west, and coniferous forests to the north. Cooler winters have led Northern TCB to exhibit longer hibernation periods. Northern TCB generally emerge from hibernation between April and May, compared to Southern TCB who emerge from hibernation as early as March (USFWS unpublished data).

3. Southern RPU:
Southern TCB are identified as a RPU because they contribute unique behavioral traits that include shorter hibernation duration, increased winter activity, and exploitation of road-associated culverts as hibernacula.

Southern TCB exhibit shorter hibernation lengths and some remain active and feed year round (Grider et al. 2016, p. 8; Newman 2020, pp. 13–17). The Southern RPU is predominantly marked by subtropical climate conditions, high humidity (especially in summer), and the absence of harsh cold winters. Southern TCB may benefit from reduced physiological pressures associated with maintaining torpor during long harsh winters and in turn have higher survival rate (Fraser et al. 2012, p. 6). Southern TCB are also unique in their frequent exploitation of road-associated culverts as winter hibernacula in the southern U.S. As discussed in *Individual-level Ecology and Needs*, culverts account for the majority of hibernacula documented in Mississippi, Georgia, and Louisiana (Limon et

al. 2018, entire; NABat 2021). Researchers have hypothesized that utilizing culverts coupled with sub-tropical climate conditions will lead to TCB exhibiting frequent arousal and foraging events during winter (Castleberry et al. 2019, p. 2). If TCB utilizing culverts are exhibiting increased winter activity related to foraging or otherwise, these euthermic bouts could significantly reduce their susceptibility to WNS (Cornelison et al. 2019, p. 3).

During the summer, Southern TCB predominantly roost in foliage of live or recently dead deciduous hardwood trees (see *Individual-level Ecology and Needs*); however, TCB will also roost in Spanish moss (Davis and Mumford 1962, p. 395). Note, TCB are considered rare and local in southeast Mexico and Central America (Reid 1997, p. 154; Medina-Fitoria et al. 2015, p. 49; Turcios-Casco et al. 2020, p. 532; Turcios-Casco et al. 2021, p. 10); consequently, given limited data from this region, we were unable to include Guatemala, Honduras, Belize, Nicaragua, and Mexico in our analysis.

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; Maintain evolutionary processes	Populations maintained across breadth of behavioral, physiological, ecological, and environmental diversity; Maintain evolutionary drivers--gene flow, natural selection--to mimic historical patterns

CHAPTER 3 – HISTORICAL CONDITION

This chapter describes the number, health, and distribution of TCB populations up to the present day. The historical condition provides the baseline condition from which we evaluated changes in TCB 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), TCB was highly abundant and widespread, with over 140,000 bats[4] observed hibernating in 1,951 known hibernacula spread across >1 billion acres in 34 states and 1 Canadian province (Figure 3.2, Table A-3A1). TCB numbers vary temporally and spatially, but abundance and occurrence on the landscape were generally stable (Cheng et al. 2022, pp. 204–205; Wiens et al. 2022, pp. 231–233). Although the majority of winter colony sizes were small (<100 individuals), the vast majority of individuals included in our dataset occupied a small subset of hibernacula; for example, in 2000, 32% (n=508) of the known winter colonies contained 90% of total known winter abundance (Figure 3.2).

Historically, of the known hibernacula, the Northern RPU contained approximately 58% of winter hibernacula (n=1,124) and 66% of the total TCB abundance. The Southern RPU contained approximately 32% of winter hibernacula (n=616) and 22% of the total abundance and the Eastern RPU was the smallest, comprising approximately 11% of winter hibernacula (n=211) and 11% of the total abundance (Table A-3A2).

TCB's range encompasses 39 states, 4 Canadian provinces, and Guatemala, Honduras, Belize, Nicaragua, and Mexico (Figure 2.3). In this SSA, we include occurrence records (i.e., TCB acoustic calls, captures, and hibernacula records) from 38 States, the District of Columbia, and 2 Canadian Provinces (Figure 3.3).

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[4] This number only represents TCB that were observed during internal winter hibernacula surveys submitted to NABat for use in this SSA; we acknowledge historical TCB abundance and the number of hibernacula were higher.

Figure 3.2. All known hibernacula and winter abundances for TCB in 2000. Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

Figure 3.3. Documented range of TCB (blue dots), as known from available records (acoustic calls, captures, and hibernacula records) 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 for TCB and does not replace the species range (Figure. 2.3).

31

CHAPTER 4 – PRIMARY INFLUENCES ON VIABILITY

Recognizing there are myriad influences operating on TCB, 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.

Threats

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 TCB. 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. 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 have 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 (Turner et al. 2011, p. 15; Reeder et al. 2012, p. 5; Carr et al. 2014, p. 21; 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; Francl et al. 2012, pp. 35–36; Meteyer et al. 2012, p. 3; Field et al. 2015, p. 20; Reynolds et al. 2016, pp. 199–200; Fuller et al. 2020, pp. 7–8). As of May 2021, WNS has been confirmed in 12 species in North America, including TCB, and numerous other species in Europe and Asia (www.whitenosesyndrome.org, accessed online May 13, 2021; Hoyt et al. 2021, Suppl. material).

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 online: 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. 2015a, p. 196; Cheng et al. 2021, p. 5). 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. 2015a, p. 196; Cheng et al. 2021, p. 5).

The effect of WNS on TCB 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 TCB experienced a 75% 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 TCB colony size. Most recently, Cheng et al. (2021, p. 7) used data from 27 states and 2 provinces to conclude WNS caused estimated population declines of 90–100% across 59% of TCB range. Although variation exists among sites, the arrival of *Pd* caused marked decreases in populations

during invasion, epidemic, and established stages of the disease (Figure 4.4), and lambda estimates less than 1 after the arrival of *Pd*, with few exceptions (Figure 4.5).

Figure 4.4. Percent change in TCB winter colony counts by disease stage relative to predicted median count prior to arrival of Pd (with 95% credible interval) (Cheng et al. 2022, Fig. D4).

Figure 4.5. Estimated TCB weighted lambda (function of growth rate and colony size) by year (left) and by year since arrival of Pd (right) (Cheng et al. 2022, Fig. D3).

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* occurrence to interpolate WNS occurrence to all documented hibernacula. The analysis predicted *Pd* is present at 85–100% of documented TCB hibernacula (see Table A-5.2 for current WNS stage by hibernacula). Although variation exists among sites, an overwhelming majority of hibernating colonies of TCB have developed WNS and experienced serious impacts within 2–3 years after the arrival of *Pd* (Cheng et al. 2021, p. 8; Wiens et al. 2022, pp. 231–247) (Figures 4.4 and 4.6).

With respect to road-associated culverts used as hibernacula in the southern U.S., there is uncertainty associated with progression of WNS within these TCB winter colonies (Sandel et al. 2001, p. 174; Katzenmeyer 2016, p. 32; Bernard et al. 2019, p. 5; Lutsch 2019, p. 23; Meierhofer et al. 2019, p. 1276). For example, *Pd* has been detected in several culverts that house overwintering TCB in Mississippi. Although *Pd* was first detected at these sites in 2014, no disease, mortality, or population impact has been documented (Cross 2019, entire). A variety of environmental and biological factors may contribute to the differences observed in culverts. 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). Winter length and climate may also affect the behavior and physiology of hibernating bats using culverts (e.g., shorter torpor bouts) or offer foraging opportunities that make it possible for

them to avoid more serious infections, but these mechanisms have not been tested (Hayman et al. 2016, p. 5). Regardless, the vast majority of TCB colonies exposed to *Pd* have developed and are expected to continue to develop WNS and experience impacts from the disease (Cheng et al. 2021, Appendix S3; Wiens et al. 2022, pp. 231–247) (Figure 4.6).

Caves and cave-like hibernacula in this region do not appear to have the same uncertainty as culverts, although winter length and foraging opportunities may be similar to those experienced by colonies in culverts. Where *Pd* has been detected in caves and tunnels in the Southern RPU, these colonies have exhibited declines more in line with those documented farther north. Black Diamond Tunnel in northern Georgia declined from a high count of over 5,000 TCB in 2013 to about 200 TCB 3 years later, after WNS was confirmed there. Carleton Cave in Alabama had a max count of 1,794 TCB in 2013 and declined to 54 in 2018. There is also evidence of declines that are not associated with known arrival of *Pd*. For example, in Florida, Smith et al. (2021, p. 21) found TCB declined 73.9% at caves between 2015 and 2020, even though *Pd* has not been detected. Whether these losses represent bats contracting WNS at swarming sites and dying elsewhere is unknown, but current evidence does not support that being the case and consequently, the reason behind these declines is currently unknown. It is also plausible that changes in these colonies are the result of bats relocating to other hibernacula, many of which may not be counted.

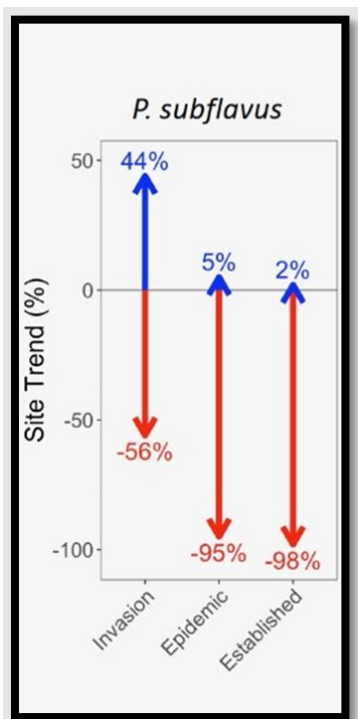*Figure 4.6. Percentage of winter colonies with increasing (blue) and decreasing (red), colony trend relative to WNS pre-arrival stage for invasion, epidemic, and established stages (Cheng et al. 2021, Appendix S3).*

There are multiple national and international initiatives underway in an attempt to reduce the impacts of WNS. To date, there are no proven measures to reduce the severity of impacts. See Appendix 4A 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 encompass tens or hundreds of turbines, 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).

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[5] (31.4 bats/MW and 31.7 bats/MW, respectively; Kerns and Kerlinger 2004, p. 15; Nicholson et al. 2005, p. 27). Bat fatalities continue to be documented at wind power installations across North America and Europe. We describe mechanisms leading to bat fatalities in Appendix 4B.

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 at wind facilities (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 TCB occurrences and wind facilities (Figure 4.7) and notable TCB mortality documented. Based on October 2020 installed MW capacity (Hoen et al. 2018, entire; USFWS unpublished data), we estimated 3,227 TCB are killed annually at wind facilities (Table 4.1; Figure A-2A6; Udell et al. 2022, pp. 265–266). 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 TCB. 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 19–21% by 2030 under the current wind scenario and up to 38% by 2060 under the future high impact wind scenario (Tables A-3D1 and A-3D2). Whitby et al. (2022, pp. 151–153) found a decline in the predicted relative abundance of TCB 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 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

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

effectiveness of feathering below various cut-in speeds (i.e., when turbine blades start rotating and generating power) 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 TCB, however, has not been documented. Hereafter, we refer to feathering below the manufacturer's cut-in speed or higher wind speeds collectively as curtailment.

Figure 4.7. Estimated total annual TCB 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 TCB 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 TCB mortality from wind facilities 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	912	301	929
Region 3	936	302	1,139
Region 4	159	48	197
Region 5	1,186	357	1,476
Region 6	30	12	31
Quebec	4	1	6
Total	**3,227**	**1,021**	**3,778**

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 4B 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 TCB's 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., average annual temperature has increased by 1.2 degrees F (0.7 degrees C) for the period of 1986–2016 relative to 1901–1960 (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). Average annual 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).

TCB 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) among populations may result in differing ability for TCB 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 TCB from a changing climate, overall negative impacts are anticipated. Although we lack species-specific observations for TCB, observed impacts to date for other insectivorous bats, such as 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) (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), which may be especially important for bats like TCB that roost in foliage rather than inside more protected shelters. Bats are likely to reduce their foraging bouts during heavy rain events and reduced reproduction of insectivorous bats has been observed during cooler, wetter springs (Grindal et al. 1992, pp. 342–343; Burles et al. 2009, p. 136). Responses will vary throughout TCB's range based on the extent of annual temperature rise in the future. For additional information on climate change see Appendix 4C.

Habitat Loss

Roosting/Foraging/Commuting Habitat Loss

As discussed in Chapter 2, TCB 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 TCB colonies.

We reviewed changes in various NLCD landcover classes within each RPU from 2006–2016 in the continental U.S. Deciduous forest landcover decreased across all RPUs by 768,903 ha (1,900,000 ac) for an average loss of 76,890 ha (190,000 ac) per year. Other cover types that provide foraging opportunities such as emergent wetland cover types decreased across all RPUs by 687,966 ha (1,700,000 ac). See Appendix 4D for additional information.

These changes in landcover may be associated with losses in suitable roosting or foraging habitat, longer flights between suitable roosting and foraging habitats due to habitat fragmentation, fragmentation of maternity colonies, and direct injury or mortality. While temporary or permanent habitat loss may occur throughout the species' range, impacts to TCB and its habitat typically occur at a more local-scale (i.e., individuals and potentially colonies). Impacts to TCB 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 largely forested landscapes with robust TCB populations) to significant (e.g., removal of roosting habitat in highly fragmented landscapes with small, disconnected populations). 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 4D for examples.

Winter Roost Loss and Disturbance

As discussed in Chapter 2, TCB require hibernation sites with specific microclimates and TCB 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 a bat's ability to access hibernacula (Spanjer and Fenton 2005, p. 1110) or can 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 TCB. 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 hibernacula where a species is impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Boyles and Willis 2010, p. 96).

There are many conservation efforts and protections (e.g., bat-friendly gates, closure of caves to exclude humans during hibernation, conservation easements) in place that attempt to reduce the risk of modifications to hibernacula and disturbance to overwintering bats. See Appendix 4D 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 TCB in parts of its range.

Federal, State, Provincial Protection

TCB was listed as endangered on Schedule 1 of Canada's Species at Risk Act in 2014. This provided the TCB protection from being killed, harmed, harassed, captured, or taken in Canada. Environment and Climate Change Canada finalized a recovery strategy for the little brown bat, northern long-eared bat, and TCB in 2018 (Environment and Climate Change Canada 2018, entire).

In addition, TCB receives varying degrees of protection through State laws as it is State-listed endangered in Connecticut, Indiana, Massachusetts, New Hampshire, Ohio, Pennsylvania, Vermont, and Virginia; State-threatened in Tennessee and Wisconsin; and special concern in Alabama, Georgia, Iowa, Maine, Michigan, Minnesota, Missouri, South Carolina, and West Virginia.

Synopsis of Current Threats

To provide a comparative and semi-qualitative 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 TCB (*High Impact*), with extreme population level declines (90–100%) over a large (59%) portion of its range (Cheng et al. 2021, p. 7). Wind energy related mortality has the next highest level of impact (*Medium Impact*), with moderate (19–21%, see Table A-3D1) population-level declines over a large (53%) portion of its range. Both habitat loss and climate change are pervasive across TCB's range, while severity of population level declines are considered slight; therefore, we assigned *Low Impact* level for both habitat loss and climate change given current state conditions. While confidence in impact to TCB from WNS and wind were "high" due to availability of quantitative data, our confidence in our impact analysis of habitat loss and climate change are "low" to "moderate" due to limited data. See Appendix 3D for additional details.

Table 4.2 Assessment of current impact to TCB 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	Large	Large	Pervasive	Pervasive
Severity	Extreme	Moderate	Slight	Slight
Impact	**High**	**Medium**	**Low**	**Low**
Confidence Level	High	High	Moderate	Low

Future Scenarios

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

To assess how TCB 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 are 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. TCB composite plausible future scenarios.

Plausible Scenario	*Pd* Occurrence Model	WNS Impact Duration	Wind Capacity	All-bat Fatality Rate	% Species Composition
Low impact	*Pd* Occurrence Model 1	15-yr species-specific survival rates	Lower build-out	Regional-specific	Regional-specific
High impact	*Pd* Occurrence Model 2	40-yr species-specific survival rates	Higher build-out	Regional-specific	Regional-specific

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 *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 TCB fatalities per MW using the species composition approach (see Chapter 1 methods and Appendix 2A for additional detail). We applied the reduced species composition rate observed after *Pd* arrival. 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 USDOE's Wind Energy Technology Office (Gilman 2020, pers. comm.). 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, entire) 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 (Cole 2020, pers. comm.).

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 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 NREL grid cell and Canadian future capacity by Province.

Table 4.4. Predicted mean annual TCB mortality[6] (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 Mortality		High Build-out Mortality	
Region 2	861	(284–877)	2,183	(720–2,233)
Region 3	458	(148–555)	9,546	(2,985–11,743)
Region 4	218	(66–269)	19,590	(5,924–24,332)
Region 5	1,681	(507–2,092)	25,699	(7,750–31,983)
Region 6	90	(37–93)	153	(62–158)
Quebec	4	(1–5)	20	(6–25)
Total	**3,312**	**(1,043–3,891)**	**57,191**	**(17,447–70,474)**

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[6] Mortality levels are based on pre and post *Pd* arrival % species composition estimates (see Appendix 2). It is likely that % 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 1B).

Climate Change

Over the next few decades, average annual temperature over the contiguous U.S. is projected to increase by about 2.2 degrees F (1.2 degrees C) relative to 1985–2015, 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 2.3–6.7 degrees F (1.3–3.7 degrees C) under RCP4.5 and 5.4–11.0 degrees F (3.0–6.1 degrees C) and 5.4–11.0 degrees F (3.0–6.1 degrees C) under RCP8.5, relative to 1986–2015 (Hayhoe et al. 2018, p. 86).

For the period of 2070–2099 relative to 1986–2015, 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).

TCB'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 4C). This includes reduced reproduction 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) or reduced reproduction during cooler, wetter spring 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. 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 2010–2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million ha (16–34 million ac 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 ha (9–21 million ac) (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). In addition, TCB is not uniformly distributed across the landscape. 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. In addition, forest lands also may remain in that classification (i.e., not converted to other land cover types) while roosts are annually

harvested. Loss of essential population needs of roosts and foraging and commuting habitat within TCB's home range where they remain is the issue. Furthermore, loss of summer roosting and foraging habitat and/or winter hibernacula compounds the impacts from WNS (see Appendix 4D).

Synopsis of Future Threats

Using the available data and information summarized above and in Chapters 5 and 6, for each of the primary influences, we assigned the scope, severity, and impact to TCB given the projected future state conditions (Table 4.5). WNS is predicted to continue to be the primary influence (*Very High Impact*), reaching 100% of TCB's range in the U.S. by 2025 (Wiens et al. 2022, pp. 226–229) and causing extreme population declines. Regardless of future low or high-build out, wind energy related mortality maintains the next highest level of impact (*Medium Impact* to *High Impact*) due to its large to pervasive scope (impacting 37–74% of TCB's range) and causing moderate to serious population declines up to 38% by 2060 (Table A-3D2). 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 increase from current state conditions. Given TCB'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 TCB (e.g., increased mortality, reduced reproductive success); therefore, our impact analysis predicts *Medium Impact* from climate change under future state conditions. While confidence in the level of impact to TCB from WNS and wind were "high" due to availability of quantitative data, our confidence in our impact analysis of habitat loss and climate change remain "low" to "moderate" due to limited data. See Appendix 3D for additional details.

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	Large-Pervasive	Pervasive	Pervasive
Severity	Extreme	Moderate-Serious	Slight-Extreme	Moderate
Impact	**Very High**	**Medium to High**	**Low to Very High**	**Medium**
Confidence Level	High	High	Moderate	Low

CHAPTER 5 – CURRENT CONDITION

Current viability is the ability of TCB to sustain healthy populations into the future given the current demographic condition of the species and the current state of the influences (Figure 5.1). To assess TCB current viability, we used the BatTool to project future abundance over time, which allows us to assess the future number, health, and distribution of TCB populations given *CURRENT* conditions, and hence, TCB's current viability. In this chapter, we describe the current demographic conditions and the projected number, health, and distribution of TCB populations given these current conditions (i.e., current abundance, growth rate, WNS occurrence, and installed wind energy capacity). We describe the viability implications for TCB in Chapter 7.

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

Current demographic conditions based on past declines - Available evidence indicates TCB 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 52% (Figure 5.2) and the number of extant winter colonies (populations) by 29% since 2000 (Table A-3B1). There has also been a noticeable shift towards smaller colony sizes (Figure 5.3). The magnitude of the winter declines, although widespread, varies spatially (Figure 5.4). Abundance has declined 89%, 57%, and 24% in the Eastern RPU, Northern RPU, and Southern RPU, respectively. The number of winter colonies (i.e., occupied hibernacula) have also decreased 46%, 24%, and 34% in the Eastern RPU, Northern RPU, and Southern RPU, respectively. Lastly, across all RPUs, the potential for population growth is currently undetectable, i.e., (λ) >1 is 0% (Table A-3B2).

Declining trends in TCB occurrence and abundance is also evident from summer data. Based on derived rangewide summaries from Stratton and Irvine (2022, pp. 99–108), for example, found rangewide occupancy has declined by 28% from 2010–2019 (Table A-3B4, Figure 5.7). Aggregated metrics of probability of occupancy declined in all RPUs (Table A-3B4). Similarly, Whitby et al. (2022, pp. 162–163) using data collected from mobile acoustic transects found a 53% decline in rangewide relative abundance from 2009–2019. They found measurable declines in the Northern RPU (86%), Southern RPU (65%), and Eastern RPU (38%) (Table A-3B4). Finally, Deeley and Ford (2022, entire) observed a significant decline in mean capture rates from 1999 to 2019 across the range. Estimates derived from their results correspond to a 12% decline in rangewide mist-net capture rates compared to pre-WNS capture rates. Capture rates decreased

19%, 16%, and 12%, in the Eastern RPU, Northern RPU, and Southern RPU, respectively (Table A-3B4).

Future projections based on current conditions - Collectively, these data indicate TCB 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 continued declines in rangewide abundance, number of hibernacula, and spatial extent in the future.

- By 2030 (~ 1 bat generation), rangewide abundance declines by 89% (CI 81–94%) (Figure 5.2; Table A-3B1).
- The number of winter colonies (i.e., occupied hibernacula) declines by 91%, with only 171 of 1,951 historical hibernacula occupied by 2030 (Figure 5.5) and only 49 extant hibernacula by 2040 (Table A-3B1).
- The colony size also declines, with the number of large hibernacula (≥100 bats) declining from 127 to 21 (83% decline) between 2020 and 2030 (Figure 5.3).
- Subsequent to declines in the number of hibernacula, TCB's range declines by 65% and winter occurrence becomes more concentrated (Table A-3B1), with 53 hibernacula containing 90% of individuals by 2030.

The projected declines are widespread across RPUs. Abundance declines in all RPUs through 2040, though afterward, there is a modest increase in TCB abundance projected in the Northern and Southern RPUs (Figure 5.4). Despite these projected increases in abundance, however, TCB's spatial extent and number of extant hibernacula will continue to decrease across all RPUs under the current scenario (Table A-3B1, Table A-3B2).

- In the Eastern RPU, abundance and the number of extant hibernacula decline by 99% by year 2030. Of the 211 historical sites, TCB will persist in only 3 hibernacula (Table A-3B2) and there is 0 probability that >500 bats will persist (Figure 5.6).
- In the Northern RPU, abundance declines by 94% and the number of extant hibernacula by 91% by year 2030. Of the 1,124 historical sites, TCB will persist in only 97 hibernacula (Table A-3B2) and only 7 will be large (≥100 individuals) (Figure 5.3).
- In the Southern RPU, abundance declines by 66% and the number of extant hibernacula by 88% by year 2030. Of the 616 historical sites, TCB will persist in 71 hibernacula (Table A-3B2) and only 14 will be large (≥100 individuals) (Figure 5.3).

Figure 5.2. Median projected rangewide abundance (black line) and 90% CI (gray shading) 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 and trend model informed by Pd occurrence model 1 (blue line) and c) status and 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 three 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. TCB extant hibernacula in 2000 (left) and projected 2030 (right) given CURRENT state conditions. Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

Figure 5.6. Probability of RPU-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 TCB 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 representation units. 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 TCB to sustain healthy populations into the future given its current demographic condition and future condition of the influences (Figure 6.1). To assess TCB future viability, we again used the BatTool to project hibernaculum abundance over time given projected *Pd* occurrence and wind energy build-out (see Chapter 4, *Future Scenarios*, for further description). Projection of future number, distribution, and health of populations is needed to understand TCB'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 TCB 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) the current step in our analytical framework.

Under the future scenarios, TCB declines worsen precipitously.
- Median rangewide abundance declines 93% (CI 91–94%) by 2030 and 95% by 2040 (Figure 6.2). Under the future scenarios, the decline trajectory halts after year 2040 and slowly grows (Table A-3C1).
- The number of extant hibernacula decline 94% by 2030 and decline 100% (CI 89–100%) by 2060, whereby, only 9 (CI 2–214) hibernacula remain (Figure 6.3, Table A-3C1).
- The colony size also declines, with the number of large hibernacula (≥100 bats) declining from 127 to 11 (91% decline) between 2020 and 2030 (Figure 6.4).
- Subsequent to declines in the number of hibernacula, TCB's range declines by 70% and winter occurrence becomes more concentrated (Table A-3C1), with 42 hibernacula containing 90% of individuals by 2030.

Similar to projections under current conditions, declines under the future scenarios are widespread across RPUs (Figure 6.5, Table A-3C2).
- In the Eastern RPU, median abundance and number of extant hibernacula both decline by 99% by year 2030. Of the 211 historical sites, TCB will persist in only 2 hibernacula (Table A-3C2) and there is 0 probability that >500 bats will persist (Figure 6.6).
- In the Northern RPU, median abundance and number of extant hibernacula decline by 95% and 93%, respectively, by year 2030. Of the remaining 81 extant hibernacula (out of 1,124 historical sites), only 7 will be large (≥100 individuals) by year 2030 (Figure 6.4).

- In the Southern RPU, median abundance and number of extant hibernacula decline by 84% and 93%, respectively, by year 2030. Of the remaining 41 extant hibernacula (out of 616 historical sites), only 14 will be large (≥100 individuals) by 2030 (Figure 6.4).

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

Figure 6.3. TCB extant hibernacula in 2000 (left) and projected 2030 (upper right) and 2060 (bottom right) given FUTURE state conditions. Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

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

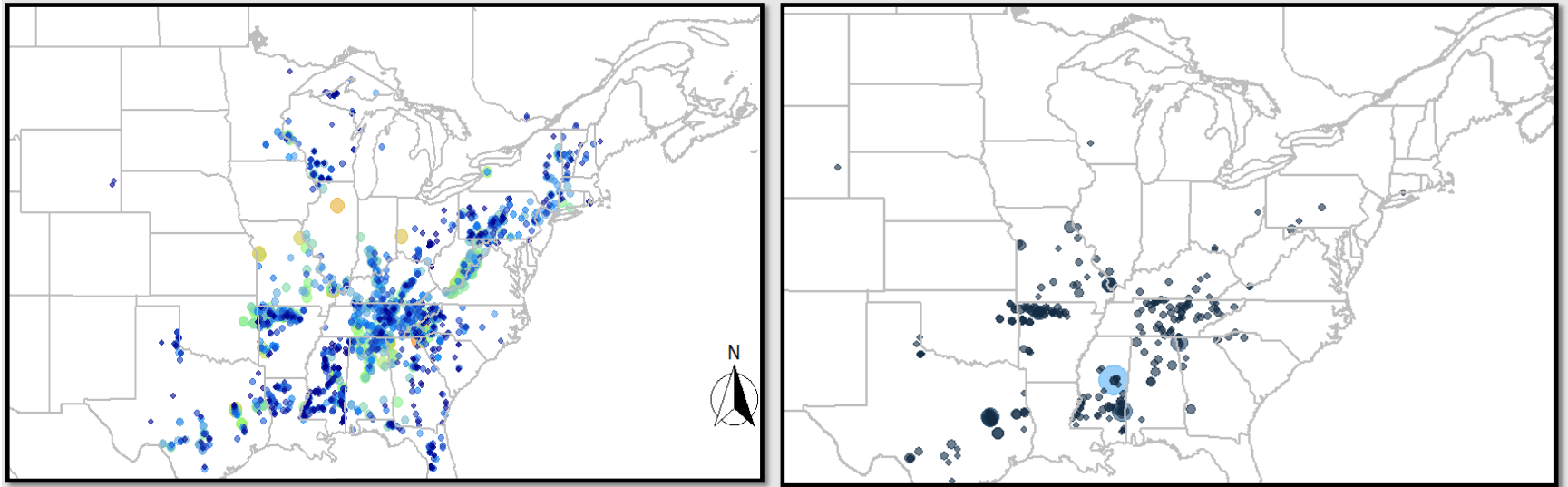Figure 6.5. Projected median (black line) and 90% CI (gray shading) for RPU abundance under FUTURE state conditions for the three RPUs. Abundance from 1990–2020 derived from raw data using a) constant interpolation (red line), b) status and trend model informed by Pd occurrence model 1 (green line) and c) status and trend model informed by Pd occurrence model 2 (blue line).
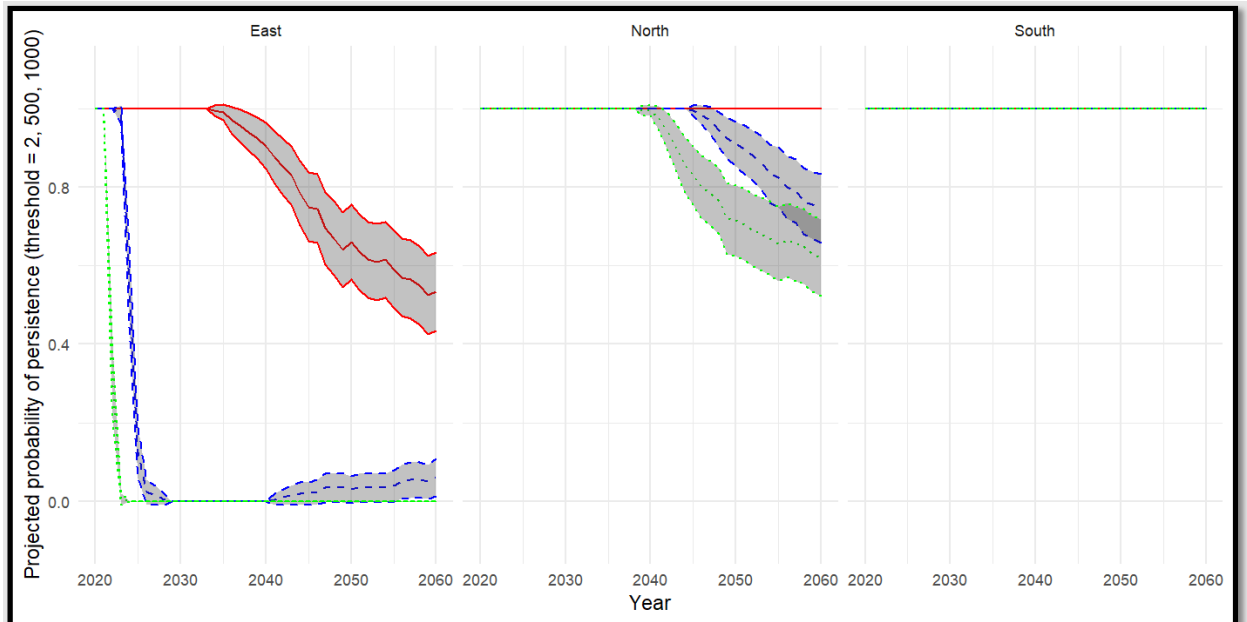.

61

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

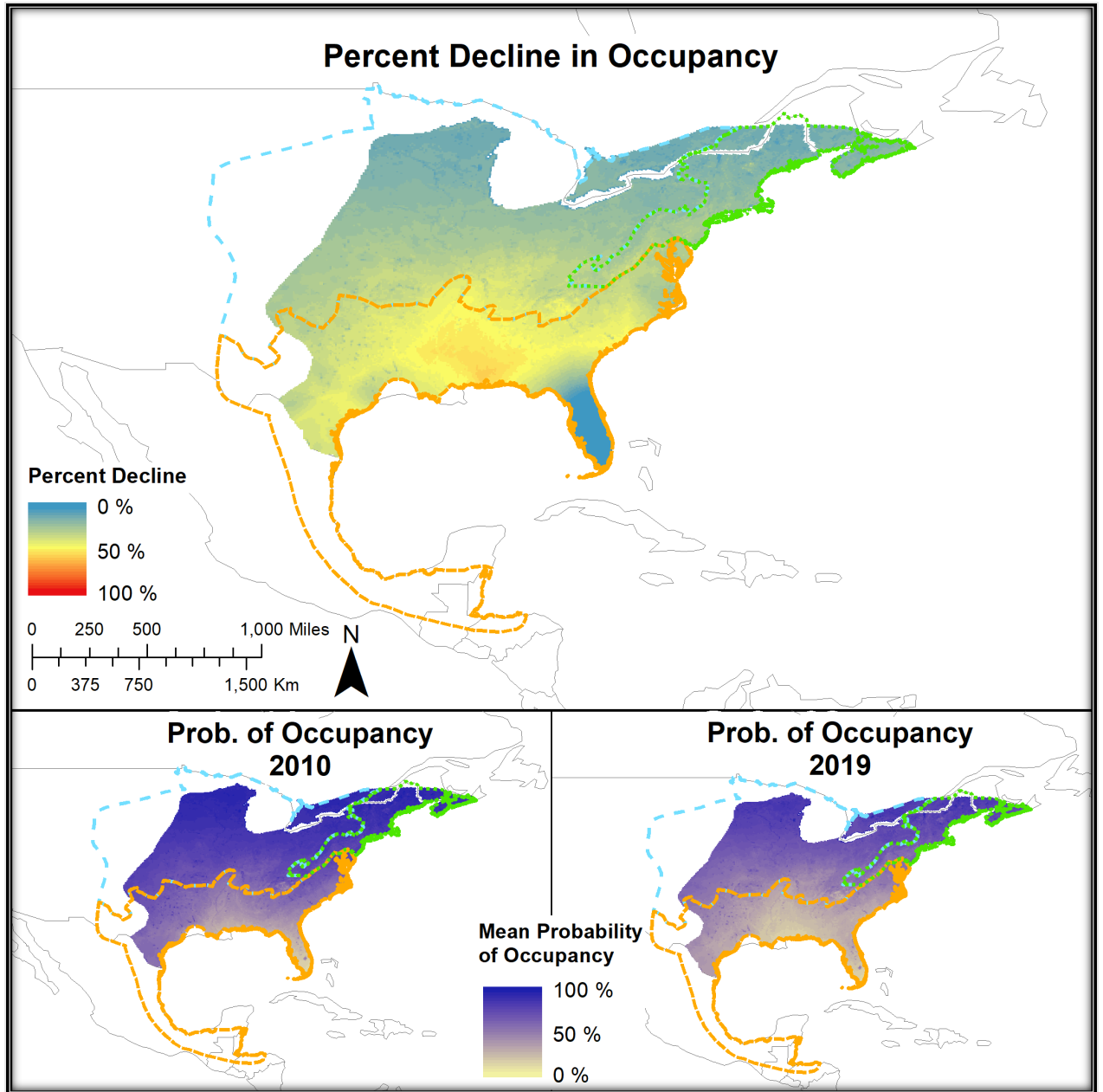## Habitat Loss and Climate Change

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 6.2–6.5). 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 the Appendix 4.

CHAPTER 7—SPECIES VIABILITY

This chapter synthesizes the results from our historical, current, and future analyses and discusses the consequences for TCB viability (Figure 7.1). TCB viability is influenced by the number, health, and distribution of populations. Across the range and within all RPUs, TCB abundance and distribution has decreased. Multiple data types and analyses indicate downward trends in TCB population abundance and distribution (Table 7.1), and we found little evidence to suggest that this downward trend will change in the future (Figure 7.2). Like all species status assessments we do not have perfect information on TCB'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 TCB 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).

Scale	Winter Colony	Summer Occupancy	Summer Capture	Summer Mobile Acoustic
Eastern	-89%	-17%	-19%	-38%
Northern	-57%	-17%	-16%	-86%
Southern	-24%	-37%	-12%	-65%
Rangewide	-52%	-28%	-12% to -19%	-53%

Figure 7.2. The projected TCB abundance over time given current (blue) Pd occurrence and installed wind capacity and plausible future scenarios (pink) for Pd occurrence and increased installed wind energy capacity. The solid, black lines represent the median abundance under current and future scenarios, respectively. Historical abundance from 1990–2020 derived from 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).

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 synchro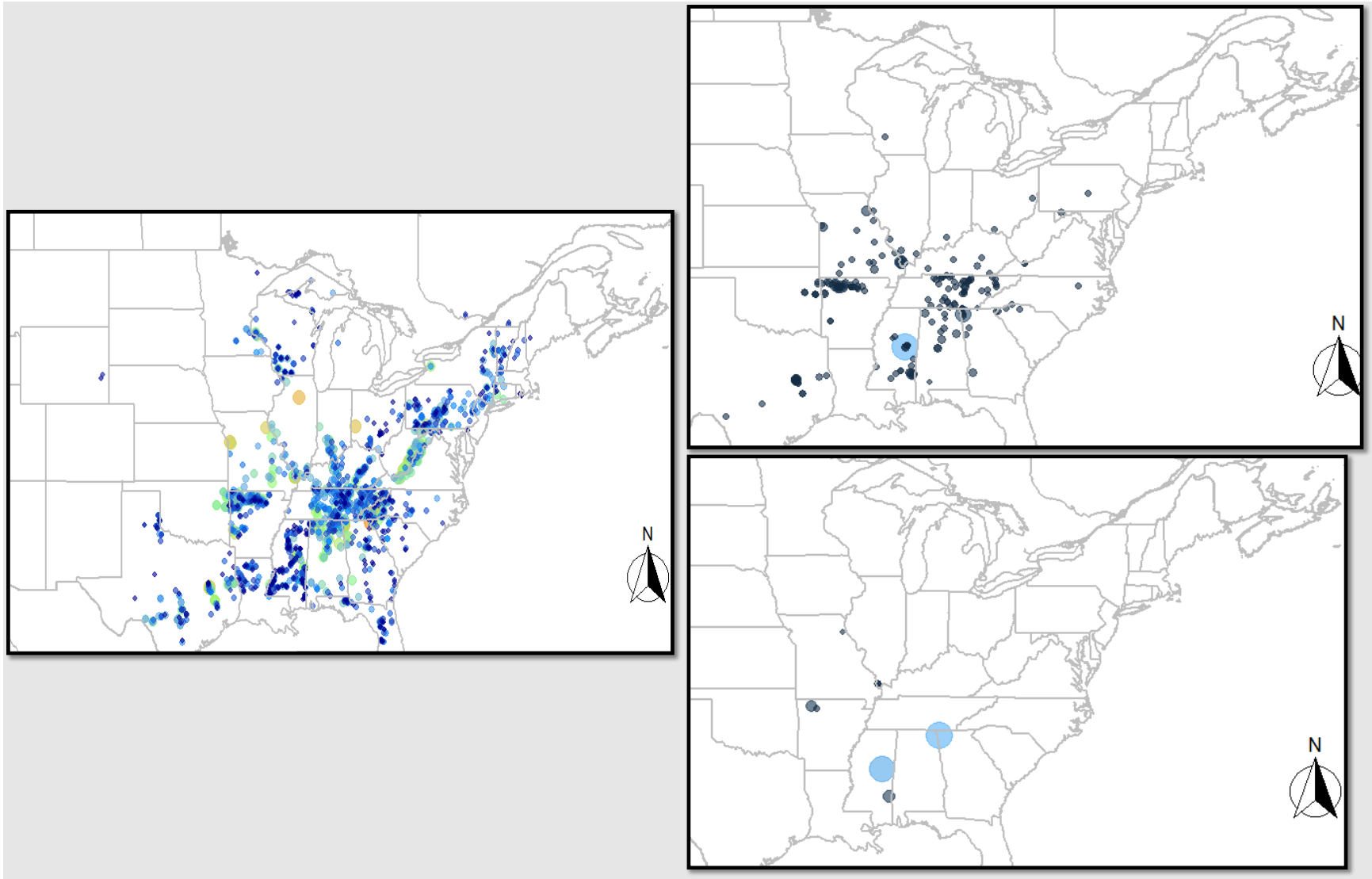nous population losses. Similarly, species with genetically healthy populations (large *Ne*, which begets genetic diversity) spread across the breadth of genetic and phenotypic diversity preserve a species' adaptive capacity, which is essential for adapting to their continuously changing environment (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 TCB'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 TCB viability has and will continue to steeply decline over time under the current and plausible future conditions.

64

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 TCB's range (Cheng et al. 2021, p. 7) and is estimated to be impacting 85–100% of hibernacula (Wiens et al. 2022, pp. 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; Figure A-2B4). Prior to WNS, TCB was abundant and widespread, and abundance and occupancy were generally stable (Cheng et al. 2022, p. 205). WNS impacts have resulted in most winter colonies experiencing a 90–100% decline in abundance compared to historical conditions (Cheng et al. 2021, p. 7).

Wind energy related mortality is proving to be a pervasive and consequential driver to TCB's viability (Table 7.2). Based on 2020 wind build-out, an estimated 1,021 to 3,778 (mean = 3,327) TCB are killed annually at wind facilities and annual mortality is projected to increase to 3,312 to 57,191 individuals by 2050 under the future low and high build-out scenarios, respectively (Figures 4.10 and 4.11). Wind related mortality is discernible, even with ongoing declines from WNS (Figure A-1B2; see also Whitby et al. 2022, pp. 151–153). TCB abundance is projected to decline between 19–21% by 2030 and up to 35% by 2060 from current wind related mortality alone and up to 38% under the future scenarios (Tables A-3D1 and A-3D2). Consequently, mortality from wind turbines will continue to cause detectable declines in TCB abundance.

Although we consider habitat loss pervasive across TCB's range, impacts to TCB 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 TCB was prior to the arrival of WNS, we assume the rangewide magnitude of impact from habitat loss was low. However, as TCB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer 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 TCB. The changing climate has had 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 TCB mortality and reduced reproductive success. As mentioned above, as TCB's spatial extent is projected to decline in the future (i.e., consolidation into fewer hibernacula and fewer summer colonies) and populations are anticipated to be less resilient, effects from climate change may be more impactful.

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	**High**	Medium	Low	Low
Low Impact	**Very High**	Medium	Low	Medium
High Impact	**Very High**	**High**	**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 TCB. TCB'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 2B, temperate zone insectivorous bats including TCB 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), concentration of individuals in both winter and summer, and specialized winter habitat requirements and summer roost needs. We have already observed the extremely limited ability for TCB to respond to the novel threat WNS. Most exposed to WNS have died and many individual bats that survive a year of exposure continue to return to infected hibernacula.

Viability under Current Conditions

Under current conditions, TCB 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, TCB abundance steeply declined, with most (93%) winter colonies having fewer than 100 individuals. At these low population sizes, colonies are vulnerable to extirpation from stochastic events. Furthermore, TCB's ability to recover from these low abundances is limited given their low reproduction output (two pups per year). Therefore, TCB's resiliency is greatly compromised in its current condition. Additionally, TCB's spatial extent is projected to decline, with 65% reduction by 2030. As TCB's abundance and spatial extent decline, TCB will also become more vulnerable to catastrophic events.

In addition to reduced redundancy and resiliency, TCB's representation has also been reduced. As explained above, TCB'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 TCB 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 TCB'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 TCB'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, TCB'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 100% of TCB's range (Wiens et al. 2022, pp. 226–229) and wind energy related mortality increases by 66% to more than 2000% (Udell et al. 2022, entire; see Table 4.4). By 2060, TCB abundance declines by 92% (Figure 7.1) and the number of extant hibernacula declines by 100% (CI 89–100%) (Figure 7.3). Under the future scenario, by 2060, 0 out of 211 hibernacula remain in the Eastern RPU and only 6 out of 1,124 and 3 out of 616 hibernacula remain in the Northern and Southern RPUs, respectively (Figure 7.3). Given the projected low abundance, the few number and restricted distribution of winter colonies, TCB's currently impaired ability to withstand stochasticity, catastrophic events, and novel changes will worsen under the range of plausible future scenarios.

Uncertainty remains with regards to WNS impacts to hibernating TCB at road-associated culverts in the southern U.S. As discussed in *Individual-level Ecology and Needs*, culverts account for the majority of hibernacula documented in Louisiana and Mississippi. No *Pd* has been detected at culverts in Louisiana (Limon et al. 2018; entire) and although *Pd* has been detected since 2014 at several culverts that house overwintering TCB in Mississippi, no disease, mortality, or population impacts have been documented (Cross 2019, entire). A variety of environmental and biological factors may contribute to the differences observed in culverts (e.g., year-round temperature profiles may affect the environmental reservoir of *Pd* and shorter winters and milder climates may affect TCB hibernating behavior and physiology). TCB in the Southern RPU exhibit shorter torpor bouts and remain active and feed during the winter (see Chapter 2), and TCB winter movements within and among culverts has been documented (Anderson et al. undated). Consequently, there is uncertainty associated with progression of WNS within these TCB winter colonies. Regardless, TCB summer occupancy, summer captures, and summer mobile acoustic detections have declined 37%, 12%, and 65% in the Southern RPU, respectively (Table 7.1). And, TCB winter colonies at caves and cave-like hibernacula in the Southern RPU have declined, regardless of similar winter length and climates shared with bats hibernating at culverts (Chapter 4).

Figure 7.3. Projected change in TCB winter distribution over time: 2000 (far left), 2030 under CURRENT state conditions (top right) and 2060 under FUTURE state conditions (bottom right). Point color and size corresponds to maximum number of TCB observed at a hibernaculum.

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APPENDICES

1. **Key Uncertainties, Wind Energy Mortality Sensitivity Analyses, and State-of-the-Knowledge**
   A. Key Uncertainties
   B. Wind Energy Mortality Sensitivity Analysis
   C. State-of-the-Knowledge
2. **Supplemental Methodology**
   A. Analytical Framework
   B. Adaptive Capacity Analysis
3. **Supplemental Results**
   A. Historical Condition
   B. Current Condition
   C. Future Condition
   D. Qualitative/Comparative Threat Analysis
4. **Supplemental Threat and Future Scenario Information**
   A. WNS
   B. Wind
   C. Climate Change
   D. Habitat Loss
5. **Supplemental Future Scenarios Descriptions**

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

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 are 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, TCB can be hidden in inaccessible locations within surveyed hibernacula and may be 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. 231–247a) to predict current abundance and population growth rate (trend) for each known hibernaculum. Cheng et al. (2021, entire) explained 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 counts 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-B.

The statistical models are constructed from the raw data available (i.e., 6,341 TCB winter observations). 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.

Regarding bats in general, estimating population abundance and trends is challenging due to their 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 (*Corynorhinus 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 TCB are available through a substantial portion of their range over recent decades. Winter survey efforts for TCB 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 TCB (**unknown hibernacula**).TCB hibernate in more caves (or cave-like subterranean habitats) than any other cave-hibernating bat species in eastern North America (Sealander and Young 1955, pp. 23–24; Barbour and Davis 1969, p. 117; Brack et al. 2003, p. 65). Almost every cave in Indiana has contained at least one TCB (Mumford and Whitaker 1982, pp. 167–168); and small numbers of TCB have likely occupied most of Missouri's 6,400 caves (Perry 2021, pers. comm.). Hibernating TCB do not typically form large clusters and most commonly roost singly (see *Individual-level Ecology and Needs*); therefore, many TCB (if not the majority) may be distributed in numerous small (and often unidentified) hibernacula during winter (Johnson 2021, pers. comm.; Perry 2021, pers. comm.).

Despite the expectation that many hibernating TCB remain unobserved during winter, abundance estimates based on winter counts represent a sound estimate of the site-specific abundances, relative abundances, or at least population trends. 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 are roosting together. Summer roost counts are possible but much less feasible for TCB 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. 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 TCB abundance, 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 population abundance, together they improve our understanding of TCB status.

White-nose Syndrome Impacts

To capture the uncertainty in the **rate of spread** of *Pd* we used 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 North America 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, 97% decline by the endemic stage, Cheng et al. 2021, entire), there is variability among sites. We identified sites that trended differently (i.e., better) than most and assumed they do not experience further WNS impacts. Wiens et al. (2022, pp. 231–235) used random draws from the impact distribution for each year (Appendix 2A). For all remaining sites, we assumed they would follow the empirically derived yearly impacts schedule.

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 2A). 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 (left), a 25-year impacts duration (middle), and a 40-year impacts duration (right). 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 and trend model informed by Pd occurrence model 2 (pink line).

Other sources of uncertainty include the **species' response** to WNS in areas where *WNS* has not yet been detected and our imperfect knowledge of every hibernacula throughout the range of the species (**unknown hibernacula**). This is particularly important in the southern portions of the range where TCB commonly overwinter in small colonies or individually, roost in numerous road-associated culverts, and may be active through much of the winter.

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 TCB colonies 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 TCB may experience less severe impacts than larger colonies during the initial stages of the disease. TCB colonies approached stabilization at low numbers (~ 6 bats) only after significant declines in larger colonies and not as a result of initial small colonies avoiding impacts (Langwig et al. 2012, p. 4). Similarly, Frick et al. (2015, p. 6) found that TCB colonies with smaller pre-WNS counts had a higher risk of local extinction due to WNS than larger ones.

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 bats in drier hibernacula have shown to have less severe impacts from WNS, but this pattern was not observed in TCB (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' range where bats hibernate for extended periods, regardless of whether these hibernacula are unknown or human inaccessible.

Wind Energy Related Mortality

We do not 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*. 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 postconstruction 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 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, 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 TCB's 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 the species.

Habitat Loss

We have high confidence of prior impacts to winter hibernacula and hibernating bats. We have high confidence that changes in vegetation cover types occur throughout TCB's range. 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 TCB occur. Therefore, we have uncertainty about which colonies (summer and winter) are at greatest risk of impacts associated with habitat loss.

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 to 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 surprising that the general trend in abundance is unaffected by wind energy mortality (Figure A-1B1). The additive effect of wind energy mortality is discernible as seen when controlling for WNS impacts and comparing no wind to wind scenarios under current wind conditions but not under future wind conditions (Figure A-1B2, see bar 1 vs 2 for current conditions and bar 3 vs 4 and 5 for future).

A likely explanation for this is as *Pd* spreads across the range over time, the results are less sensitive among the wind mortality scenarios because WNS impacts are severe and dominate the dynamics (Figure A-1B2, bars 3–5).

Figure A-1B1. TCB 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 and trend model informed by Pd occurrence model 2 (pink line).

Abundance under wind energy mortality levels

Figure A-1B2. TCB projected median 2060 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

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 6,341 records from 1,951 hibernacula (58% of the sites are from the Northern RPU).
- Based on these raw data:
  - Number of hibernacula with "Last observed = 0": 165 (1990–2020), 3 (2006–2009), 51 (2010–2015), 108 (2016–2020); the ratio (proportion) of extirpated to extant sites increased since WNS discovered in 2006 (Figure A-1C1)
  - Of the 1,786 potentially extant sites, 43 to 71% have uncertain status (768–1,267 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 number of hibernacula with 0 counts divided by the total number of hibernacula surveyed.

- As of May 2021, 580 counties across 40 states and 7 Canadian 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).

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

- The number of TCB hibernacula with suspected or confirmed WNS is not available; WNS has been confirmed in every RPU. Most of these are from northern portion of the range, and data are scant in non-cave hibernacula in the southern portion of the range.
  - As of May 2021, there are 148 TCB 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 (2%) hibernacula. On average, TCB colonies declined by mean 93% (95% CI 90-100%) 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 acoustic, stationary acoustic, and mist-net capture data), however, we incorporated all available data into the analyses.
  - Using mobile transect acoustic data, Whitby et al. (2022, entire) found that relative abundance declined 38% (Eastern RPU) to 86% (Northern RPU) from 2009 to 2019.
  - Using mist-net capture data, Deeley and Ford (2022, entire) found significant decreases in mean capture rate from 1999 to 2019. Estimates derived from their data found 12% (Southern RPU) to 19% (Eastern RPU) declines in mean capture rates post-WNS arrival.
  - Using all three data types (mobile transect acoustic, stationary acoustic, and mist-net capture), Stratton and Irvine (2022, entire) looked at changes in probability of occupancy across the range of the species. Their results showed a decline in TCB occupancy across all RPUs (Stratton and Irvine 2022, entire). Estimates derived from their results found declines in the probability of occupancy ranging from 17% (Eastern and Northern RPUs) and 37% (Southern RPU) from 2010 to 2019.

Appendix 2: **Supplemental 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 bat species, 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[7], 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 are aware of. Hibernacula counts were available for much of the range of TCB, although occurrence information is extremely scarce for the species in Mexico and Central America (Reid 1997, p. 154; Medina-Fitoria et al. 2015, p. 49; Turcios-Casco et al. 2020, p. 532; Turcios-Casco et al. 2021, p. 10). Consistent with the species' biology, we assumed that TCB 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).

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[7] Colony count data from North American Bat Monitoring Program Database v5.4.3: U.S. Geological Survey. Accessed 2021-02-10. NABat Request Number 12. batmonitoring.org / https://sciencebase.usgs.gov/nabat/#/home

Figure A-2A2. Documented range of TCB (blue dots), as known from available records (acoustic calls, mist-net captures, and hibernacula records) 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 for TCB and does not replace the species range (Figure. 2.3).

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 TCB *N* and *λ*, we relied upon analyses completed by Wiens et al. (2022, pp. 231–233). For TCB sites with more than 5 data-points (n=462; Table A-2A1), 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=1,489), 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* and surveillance effort 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.

Table A-2A1. Number of hibernacula by State/Province used to estimate historical TCB N and λ.

State/Province	# of hibernacula
Alabama	5
Arkansas	39
Connecticut	3
Delaware	1
Florida	9
Georgia	8
Illinois	13
Indiana	18
Iowa	1
Kentucky	40
Maryland	6
Massachusetts	4
Michigan	12
Minnesota	5
Mississippi	37
Missouri	2
New Hampshire	5
New Jersey	1
New York	16
North Carolina	27
Ohio	2
Oklahoma	11
Pennsylvania	41
Quebec	1
Rhode Island	1
South Carolina	3
Tennessee	44
Vermont	7
Virginia	13
West Virginia	33
Wisconsin	54
TOTAL	**462**

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 Current and Future Primary 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 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-2A3. 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.

Quantitative Threat Analysis – 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 TCB, and the fact that we had finite time and resources, we were unable to reliably quantify TCB's 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 TCB's range; however, given our finite time and resources we were unable to project rangewide future landcover changes or TCB's associated response in a manner that could be included in the BatTool (e.g., what specific landcover changes would result in changes to 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). 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-2A4. 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 1990–2020 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 (see 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), northern long-eared bay (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 and the proportion of curtailed turbines. We did, however, use different fatality rates for pre- and post-WNS as the data indicated different percent species compositions before and after WNS arrival.

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 TCB, little brown bat, and northern long-eared 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:

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

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 seven 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.

For the U.S., we assessed our species composition rates by USFWS Region. We had insufficient data to generate TCB percent composition rates for Regions 2, 4, and 6. We used American Wind Wildlife Institute's (AWWI) (2020, p. 19) TCB composition rates for Region 2 (AWWI 2020, p. 19), but they did not report values for Regions 4 and 6. For Region 4, along with the southern portion of Region 5, we observed significantly higher TCB fatalities compared to TCB

fatality rates in the northern portion of Region 5, likely due to higher TCB abundance in the southern extent of its range. Therefore, we calculated separate TCB composition rates for the northern and southern portions of Region 5, incorporating Region 4 data with Region 5 South. We combined Region 3 and Region 6 data to calculate a Region 3/6 percent species composition rate. For Canada, we used species composition rates (%*Sp*) reported in Bird Studies Canada et al. (2018, pp. 17–18). Additionally, we found differences in *%Sp* following *Pd* arrival, and thus, applied post-*Pd* arrival *%Sp* rates as suggested by the data.

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 facility-specific curtailment information (USFWS, unpublished 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) (CER 2020, pp. 5, 22–23, 56–57) Evolving and Reference (baseline) scenarios as our upper and lower bounds, respectively (see Appendix 4 for further description of future scenarios).

Lastly, to calculate hibernacula-specific wind 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 percent mortality reduction to turbines implementing any level of curtailment during the fall or summer seasons, per our 2020 data (USFWS unpublished data). We then multiplied this 50% mortality reduction by the relative proportion of all-bat mortality reported by

season in our post-construction mortality database (USFWS, unpublished data; Table A-2A2). Based on these proportions, we applied an overall mortality reduction of 50 percent to turbines curtailing in both summer and fall and a 34 percent reduction to turbines curtailing in fall only (Table A-2A3).

Table A-2A2. 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

Table A-2A3. 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-2A2).

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, updated with TCB-specific demographics (BatTool, Erickson et al. 2014). 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 representation unit-level (RPU, described in Chapter 2) metrics to describe the species' historical, current, and future number, health, and distribution of populations given current and future influences. Figure A-2A7 shows a simplified schematic of the purpose of the various models used and Figure A-2A8 provides the conceptual framework for the BatTool.

Figure A-2A7. Simplified schematic showing the role for each of the four mathematical models: two Pd prevalence models, Status and Trends model, and BatTool.

Figure A-2A8. 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 10,489 TCB in 3,290 sampling events in which 736 (7.0%) of 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 (model 1). Stratton and Irvine (2022, entire) assessed recent change in predicted summer occupancy using stationary and mobile acoustical

detector records and capture records across TCB'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 TCB's range. 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 TCB 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 populations at state and RPU scales.

B: Adaptive Capacity Analysis

To garner additional insights into the intrinsic (and historical) ability of these species 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 each species' 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 TCB adaptive capacity (from Thurman et al. 2020, Supporting Information).

Core Attribute	Relative Level	Evidence and Relevance
Extent of Occurrence	High	Broadly distributed (Davis 1959, entire; Geluso et al. 2005, p. 406; Kurta et al. 2007, p. 405; Slider and Kurta 2011, p. 380; Adams et al. 2018, entire; Hanttula and Valdez 2021, p. 132); typically, a broader distribution is expected to confer higher adaptive capacity.
Habitat Specialization	Low/ Moderate	Summer habitat: generalist; suitable roosting habitat includes live and dead leaf clusters of live or recently dead deciduous hardwood trees (Veilleux et al. 2003, p. 1071; Perry and Thill 2007, pp. 976–977; Thames 2020, p. 32); Spanish moss and *Usnea trichodea* lichen (Davis and Mumford 1962, p. 395; Poissant 2009, p. 36; Poissant et al. 2010, p. 374); pine needles (Perry and Thill 2007, p. 977); eastern red cedar (Thames 2020, p. 32); artificial roosts (e.g., barns, beneath porch roofs, bridges, concrete bunkers) (Jones and Pagels 1968, entire; Barbour and Davis 1969, p. 116; Jones and Suttkus 1973, entire; Hamilton and Whitaker 1979, p. 87; Mumford and Whitaker 1982, p. 169; Whitaker 1998, p. 652; Feldhamer et al. 2003, p. 109; Ferrara and Leberg 2005, p. 731; Smith 2020); and rarely within caves (Humphrey et al. 1976, p. 367; Briggler and Prather 2003 p. 408; Damm and Geluso 2008, p. 384). Specific roost requirements needed for successful pregnancy and recruitment likely include narrow temperature ranges. Exhibit high site fidelity.

Core Attribute	Relative Level	Evidence and Relevance
		<u>Winter habitat</u>: <u>specialist</u>; suitable hibernacula conducive to longer torpor bouts; hibernacula include caves and mines (Barbour and Davis 1969, p. 116); road-associated culverts (Sandel et al. 2001, p. 174; Katzenmeyer 2016, p. 32; Limon et al. 2018, entire; Bernard et al. 2019, p. 5; Lutsch 2019, p. 23; Meierhofer et al. 2019, p. 1276); tree cavities (Newman 2020, p. 14); abandoned water wells (Sasse et al. 2011, p. 126); rock shelters (e.g., fissures in sandstone and sedimentary rock) (Johnson 2021, pers. comm.). Exhibit high site fidelity.
Commensalism with Humans	Moderate	Broadly distributed across human-modified landscapes, but less tolerant when suitable roosting sites have been eliminated (e.g., urban and agricultural dominated landscapes) (Duchamp and Swihart 2009, p. 855; Farrow and Broders 2011, p. 177). Conversely, will utilize man-made infrastructure as hibernation sites (e.g., abandoned mines, tunnels, road-associated culverts) (see references above).
Genetic Diversity	Moderate	Martin (2014, entire) observed significantly distinct structure in maternally inherited mitochondrial DNA across the sampled range. Large portions of the range have not been sampled and we are unaware of additional genetic information.
Population Size	Low	Once common, populations have decreased significantly (Cheng et al. 2021, entire); adaptive capacity may decrease with smaller populations.
Dispersal Distance	Moderate/ High	Females migrate up to 243 km (151 miles) from winter to summer habitat (Samoray et al. 2019, entire); individuals have high site fidelity.
Climatic Niche Breadth	High	Broad climatic niche breadth across range (e.g., occur from Canada to Central America) (see references above); may indicate a broader tolerance to climate change because they currently encompass a broader array of climate conditions.
Physiological Tolerances	Moderate	If physiological tolerance reflects the degree to which a species is restricted to a narrow range of abiotic conditions, we assume TCB have at least a moderate level of physiological tolerance during the summer given they are found as far north as Canada and as far south as Central America. Physiological tolerance during hibernation, however, is narrower given requirement for suitable hibernacula conducive to longer torpor bouts.
Diet Breadth	High	Opportunistic feeders; small insects ranging from 4–10 mm in length (primarily Coleoptera, Diptera, Lepidoptera, and Trichoptera) (Ross 1967, p. 223; Whitaker 1972, p. 879; LaVal and LaVal 1980, p. 24; Griffith and Gates 1985, p. 453; Hanttula and Valdez 2021, p. 132). Hibernation period may decrease with warming temperatures, but insect hatches may occur earlier.
Reproductive Phenology	Low	TCB mate in the fall. Females store sperm in their uterus during the winter and fertilization occurs soon after spring emergence from hibernation (Guthrie 1933, p. 209). Females typically give birth to two young, rarely one or three between May and July (Allen 1921, p. 55; Barbour and Davis 1969, p. 117; Cope and Humphrey 1972, p. 9).

Core Attribute	Relative Level	Evidence and Relevance
Life Span	Moderate/ Low	The greatest longevity records are 14.8 years and 11.2 years for a male and female, respectively (Paradiso and Greenhall 1967, pp. 251–252; Walley and Jarvis 1972, p. 305).
Fecundity	Low	Litter size is usually two, rarely one or three, annually (see references above).

Appendix 3: Supplemental 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 known hibernacula	Abundance (max)
34/1	1.1 billion	1,951	140,547

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

RPU	# of known hibernacula	Abundance (max)
Eastern	211	16,576
Northern	1,124	95,906
Southern	616	32,433

B: Current Condition

Table A-3B1. Projected yearly rangewide number of states/provinces with 2 or more bats persisting, spatial extent (EOO in acres), number of hibernacula (90% CI), and median abundance (90% CI) under **current** conditions.

Year	# of States / Provinces	EOO (ac)	# of hibernacula		Abundance	
2020	29/1	929 million	1,378	(CI 1,317–1,378)	67,898	(CI 67,444–68,352)
2030	15/0	383 million	171	(CI 22–734)	15,661	(CI 8,312–26,690)
2040	8/0	262 million	49	(CI 4–464)	14,611	(CI 7,181–23,056)
2050	6/0	205 million	30	(CI 3–379)	16,557	(CI 12,368–22,444)
2060	5/0	136 million	23	(CI 3–340)	19,506	(CI 13,619–28,429)

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

RPU	Year	# of hibernacula	pPg
Eastern	2020	114	0%
	2030	3	10%
	2040	0	54%
	2050	0	45%
	2060	0	41%
Northern	2020	856	0%
	2030	97	0%
	2040	13	52%
	2050	7	66%

RPU	Year	# of hibernacula	pPg
	2060	5	98%
Southern	2020	408	0%
	2030	71	21%
	2040	36	63%
	2050	23	64%
	2060	18	58%

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

RPU	2020	2030	2040	2050	2060
Eastern	1,891 (CI 1,786–1,996)	103 (CI 32–257)	26 (CI 0–177)	14 (CI 0–249)	5 (CI 0–509)
Northern	41,448 (CI 41,428–41,468)	5,374 (CI 3,667–6,989)	2,733 (CI 1,208–5,437)	3,535 (CI 403–11,141)	3,864 (CI 202–17,433)
Southern	24,559 (CI 23,980–25,138)	11,042 (CI 3,860–20,086)	10,853 (CI 3,565–20,544)	11,810 (CI 5,065–20,665)	12,395 (CI 5,999–20,594)

Table A-3B4. Summary of recent TCB population trends from multiple data types and analyses. Winter Colony analysis – Wiens et al. (2022, entire); 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).

Scale	Winter Colony	Summer Occupancy	Summer Capture	Summer Mobile Acoustic
Eastern	-89%	-17%	-19%	-38%
Northern	-57%	-17%	-16%	-86%
Southern	-24%	-37%	-12%	-65%
Rangewide	-52%	-28%	-12% to -19%	-53%

C: Future Condition

*Table A-3C1. Projected rangewide number of states/provinces and hibernacula with 1 or more bats persisting, spatial extent (EOO), number of hibernacula (90% CI) and median abundance (90% CI) under **future** scenarios.*

Year	# of States / Provinces	EOO (ac)	# of hibernacula	Abundance
2020	29/1	929 million	1,378 (CI 1,317–1,378)	67,898 (CI 67,444–68,352)
2030	14/0	329 million	124 (CI 18–603)	10,138 (CI 8,053–12,519)
2040	7/0	169 million	18 (CI 3–324)	7,225 (CI 3,604–12,520)
2050	5/0	86 million	11 (CI 2–251)	8,495 (CI 2,524–19,690)
2060	5/0	77 million	9 (CI 2–214)	10,955 (CI 2,194–27,292)

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

RPU	Year	# of hibernacula	pPg
Eastern	2020	114	0%
	2030	2	12%
	2040	0	52%
	2050	0	50%
	2060	0	48%
Northern	2020	856	0%
	2030	81	1%
	2040	11	54%
	2050	7	72%
	2060	6	98%
Southern	2020	408	0%
	2030	41	11%
	2040	7	68%
	2050	4	72%
	2060	3	78%

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

RPU	2020	2030	2040	2050	2060
Eastern	1,891 (CI 1,786–1,996)	89 (CI 30–304)	29 (CI 0–183)	16 (CI 0–243)	12 (CI 0–467)
Northern	41,448 (CI 41,428–41,468)	4,860 (CI 3,608–6,541)	2,493 (CI 897–6,498)	2,863 (CI 271–10,955)	3,876 (CI 132–16,276)
Southern	24,559 (CI 23,980–25,138)	5,144 (CI 3,191–6,746)	4,553 (CI 1,639–7,654)	5,684 (CI 1,561–9,974)	6,020 (CI 1,579–12,855)

D: **Qualitative/Comparative Threat Analysis**

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

1. Buffer extant hibernacula by avg. migration distance (126 km)
2. Buffer summer points by avg. migration distance (126 km)
3. Merge & dissolve buffered hibernacula and summer shapefiles into a "TCB occupied" area, clip TCB range by contiguous U.S. border for "TCB U.S. range", and clip TCB occupied area by TCB U.S. range.
4. Buffer and dissolve current turbines (Hoen et al. 2018, entire) by avg. migration distance for "wind threat" area

5. Clip wind threat area by TCB occupied area for "TCB wind risk" area
6. Compare TCB wind risk area with range area in U.S.: range area (4,605,467 km^2) and 2020 wind risk area: 2,449,924 km^2 (53% of U.S. range) (Figure A-3D1)

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

To estimate the proportion of TCB'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 Onshore Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance (126 km) for "future wind threat: area. Note: Future MW summed by 11x11-km NREL grid so does not capture actual distribution of turbines on landscape.
    b. Clip wind threat area by TCB occupied areas for TCB 2050 low wind risk" area (U.S.)
    c. Compare TCB 2050 low wind risk areas with range area in U.S: range area (4,605,467 km^2) and 2050 low wind risk areas (1,720,963 km^2) (37% of U.S. range) (Figure A-3D2)
2. 2050 High Build-out Scenario:
    a. Buffer & dissolve 2050 Low Onshore Wind Cost Scenario NREL data (Cole et al. 2020, entire) by avg. migration distance (126 km) for "future wind threat" area. Note: Future MW summed by 11x11-km NREL grid so does not capture actual distribution of turbines on landscape.

b. Clip wind threat area by TCB occupied areas for "TCB 2050 high wind risk" area (U.S.)
c. Compare TCB 2050 high wind risk areas with range area in U.S.: range area (4,605,467 km^2) and 2050 high wind risk areas (3,414,613 km^2)(74% of U.S. range) (Figure A-3D3)

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

Figure A-3D3. Estimated extent of TCB'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. Results are presented in Tables A-3D1 and A-3D2.

Table A-3D1 Projected median rangewide abundance given CURRENT wind energy mortality under 4 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 ,and 4) Pd model 2 and no wind energy related mortality (all values derived from Wiens et al. 2022, entire).

	Median Rangewide Abundance			
Scenario	**2030**	**2040**	**2050**	**2060**
Pd Model 1 – Current mortality	10,623	10,355	17,379	23,657
Pd Model 1 – No mortality	13,416	13,823	25,032	36,388
% change	-21%	-25%	-31%	-35%
Pd Model 2 – Current mortality	24,195	19,599	17,809	17,252
Pd Model 2 – No mortality	29,846	24,245	22,629	21,955
% change	-19%	-19%	-21%	-21%

Table A-3D2. Projected median rangewide abundance given FUTURE wind energy mortality under 4 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, 4) Pd model 2 and no wind energy related mortality (all values derived from Wiens et al. 2022, entire).

	Median Rangewide Abundance			
Scenario	**2030**	**2040**	**2050**	**2060**
Pd Model 1 – low impact mortality	10,493	10,320	16,521	22,553
Pd Model 1 – future no mortality	13,789	14,226	24,766	36,277
% change	-24%	-27%	-33%	-38%
Pd Model 2 – high impact mortality	10,214	5,415	4,665	4,339
Pd Model 2 – future no mortality	14,797	7,397	5,719	4,811
% change	-31%	-27%	-18%	-10%

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 May 2021, *Pd* and WNS have spread to 40 states and 7 provinces, with lesions indicative of disease confirmed in 12 species of North America bats, including TCB (Figure A-4A1, www.whitenosesyndrome.org; accessed online 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, 2021)

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, mainly subterranean hibernacula (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 degrees C to 16.7 degrees C for TCB) (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 other areas, 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 including TCB (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. 2015b, 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. 2015b, 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 nonreproductive 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 which are correlated with the patterns and timing of *Pd* spread (Hallam and Federico, 2012, p. 270; Maher et al. 2012, p. 3; Alves et al. 2014, p. 2; Hefley 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 to them either via dispersing bats or via inadvertent transport by humans, it has led to disease and continued spread of the fungus (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. 2015a, p. 196; Cheng et al. 2021, p. 5). 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 *Pd*. 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. 2015a, p. 196; Cheng et al. 2022, entire). 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. (2015b, 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 TCB (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 (Osborn 2021, pers. comm.). 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 bat, although northern long-eared bat 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. 277; Hoyt et al. 2021, pp. 6–7). For example, TCB in the coastal and far southern portions of its range may use shallower torpor or engage in periodic foraging through the winter, thus avoiding severe disease (Bernard et al. 2017, p. 8; Newman, 2020, pp. 21–22). However, Sirajuddin (2018, p. 19) found that skin temperatures of TCB in the south does fall within the optimal range of growth for *Pd* during winter. Notably, *Pd* has been detected on bats overwintering in culverts in Mississippi and WNS has not manifested in the colony (Cross 2019, entire). 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 predictable based on geographic or biological features.

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 75% for TCB at WNS infected winter colonies 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 TCB was extirpated from 10% of historical hibernacula (Frick et al. 2015, p. 5). Hoyt et al. (2021, p. 7) summarized overall TCB declines from WNS to be 95% in the Northeast and 99% the Midwest. Using data from 27 states and 2 provinces, the most complete dataset available at the time, Cheng et al. (2021, p. 7) reported similar patterns. They estimated that WNS has caused a 90–100% decline in TCB across 59% of the range (Figure 4.4.; Cheng et al. 2021, p. 7). Although there are ecological and environmental differences across the currently affected regions of North America, WNS has consistently caused significant declines in TCB populations (Figure 4.6), with very few examples of colonies that are avoiding impacts (Figure 4.6).

Conservation Measures Associated with WNS

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.

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). 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 the states in TCB'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 (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).

Wind related mortality of TCB, 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). Grodsky et al. (2011, p. 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, and 46). Based on our analysis, 6.5%, 25.5%, and 68% 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 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

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

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).

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 most of all known bat fatalities at wind facilities (e.g., 74–90%). The disproportionate amount of fatalities involving these species has resulted in less attention and concern for other bat species. However, there is notable spatial overlap between TCB occurrences and wind facilities and notable TCB mortality documented (Figure 4.7). Based on October 2020 installed MW capacity (Hoen et al. 2018, entire; USFWS unpublished data), we estimated 3,227 TCB are killed annually at wind facilities (Table 4.1; Figure A-2A6; Udell et al. 2022, pp. 265–266). Similarly, Whitby et al. (2022, entire) analyses suggest that the impact of wind related mortality is discernible in the ongoing decline of TCB. Comparing a no wind (and no WNS) baseline scenario to current and future wind (and no WNS) scenarios, the percent change in abundance relative to the baseline no wind scenario ranges from a 19–21% decrease by 2030 under the current wind scenario up to a 38% decrease by 2060 under the future high impact wind scenario (Tables A-3D1 and A-3D2). Whitby et al. (2022, pp. 151–153) found a decline in the predicted relative abundance of TCB 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 TCB 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 A-2A). 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 pursuant to the Endangered Species Act and detailing methods to avoid incidental take of Indiana bat; however, these areas with risk to Indiana bat do not fully overlap with those where TCB and other species may be susceptible to mortality.

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 (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 TCB, 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 Plains 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 individual bats are exposed to *Pd* and effects from 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 in Spring (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, which is particularly important during the fall when bats are trying to accumulate extra body fat for winter hibernation.

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 1977, 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 Texas, external temperature had a greater influence on microclimate temperatures in culverts than in caves, likely as a result of culvert design (generally being straight, with two entrances to allow for air flow) and maintenance (clearing of brush around entrances), whereas not all caves have multiple entrances or may have additional barriers (vegetation) to reduce airflow (Leivers et al. 2019, p. 5). Overall, culverts and caves/mines with little complexity have greatest potential for being impacted by increasing external temperatures.

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 TCB 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 (Johnson et al. 2021, entire). 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., increased temperatures in the 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 bats and forest communities, and increases in 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 likely continue to expand. 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).

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).
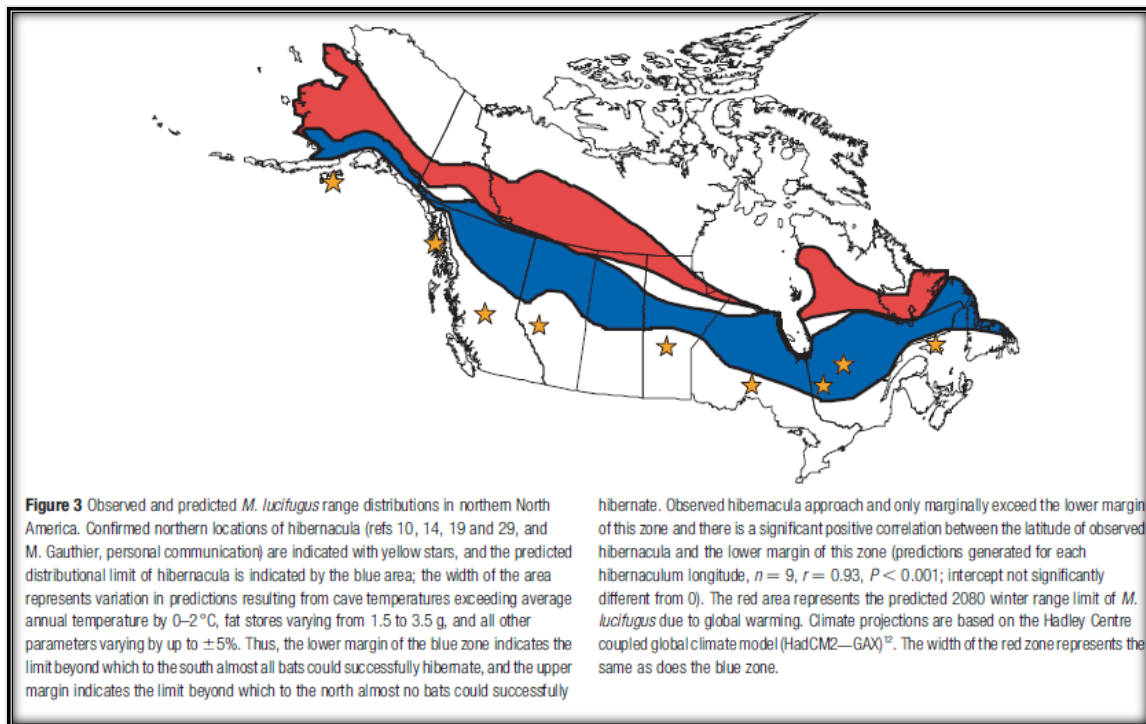
Figure 3 Observed and predicted *M. lucifugus* range distributions in northern North America. Confirmed northern locations of hibernacula (refs 10, 14, 19 and 29, and M. Gauthier, personal communication) are indicated with yellow stars, and the predicted distributional limit of hibernacula is indicated 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 TCB 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 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-

131

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 Hawaii) divided by the average for the first half of the century (from 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, for example, 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 nonreproductive females (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 nonreproductive females captured from 1996 through 2008 and (a) mean high temperature, (b) mean precipitation, and (c) stream discharge rate (from 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 survival rates for the little brown bat (Frick et al. 2010, pp. 131–133). Frick et al. (2010, p. 133) 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. Drying summer conditions may negatively impact aquatic insect prey and therefore, 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 1960–2009. 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 the 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 female 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 pups 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 nonvolant 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, pp. 2–3) and have wing damage (Meteyer et al. 2009, p. entire; Reichard and Kunz 2009, entire) that makes flight (migration and foraging) more challenging while also bringing the energetic cost of healing (Davy et al. 2017, p. 705; Fuller et al. 2020, p. 8; Meierhofer et al. 2018, p. 487).

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 1985–2015 regardless of future scenario (Hayhoe et al. 2018, p. 86; Figure A-4C4). Larger increases are projected by late century of 2.3–6.7 degrees F (1.3–3.7 degrees C) under RCP4.5 and 5.4–11.0 degrees F (3.0–6.1 degrees C) and 5.4 to 11.0 degrees F (3.0–6.1 degrees C) under RCP8.5, relative to 1986–2015 (Hayhoe et al. 2018, p. 86). For the period of 2070–2099 relative to 1986–2015, precipitation increases of up to 20–30% are projected in winter and spring for northcentral 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).

TCB's response to these changes are expected to be similar to what has already been observed in North American insectivorous bats, such as the little brown bat (see above). This includes reduced reproduction 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 TCB's range and 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).

Climate change may additionally impact TCB 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 Northeast and western Gulf of Mexico (Hayhoe et al. 2018, p. 99), which may reduce access to cave roosts 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 TCB's 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, TCB require suitable forest habitat for roosting, foraging, and commuting between those habitats during spring, summer, and fall. Wetlands and water features are also important for foraging and serve as drinking water sources. There are a variety of reasons for roosting, foraging, and commuting habitat loss within the range of TCB. Hammerson et al. (2017, entire) assessed the scope and severity of threats to bats and determined the highest projected threats included: 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 forest conversion. Tree cutting and wetland loss can occur from a variety of sources (e.g., development, energy production and transmission, transportation projects). These activities leading to the loss of roosting and foraging habitat are increasing across TCB's range (USFWS 2015, p. 17991; Oswalt et al. 2019, p. 17) and may result in impacts to TCB.

Past and Current Habitat Loss

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 1987–2012, but since 2012 appears to have reached a plateau (2019, p. 4).

In addition to reviewing these reports, we examined more recent (2006–2016) changes in various NLCD landcover classes within each RPU in the continental U.S. Forest landcover increased overall (primarily based on increases in coniferous forest in the Southern RPU). However,

deciduous forest landcover decreased across all RPUs by 768,903 ha (1,900,000 ac) for an average loss of 76,890 ha (190,000 ac) per year and coniferous forest decreased in both the Northern and Eastern RPUs (Table A-4D1). Other cover types that provide foraging opportunities such as emergent wetland cover types also decreased across all RPUs by 687,966 ha (1,700,000 ac).

Table A-4D1. Changes in land cover types in acres (NLCD 2006-2016) by TCB RPU.

NLCD Lower 48 2006-2016	TCB Representation Units – Δ Acres			
Land Cover Type	**Northern**	**Eastern**	**Southern**	**All Units**
No Data	0	0	0	0
Open Water	403201	-54478	158744	507467
Developed, Open Space	227144	47603	328783	603530
Developed, Low Intensity	305072	80143	493413	878628
Developed, Medium Intensity	397935	106263	580635	1084833
Developed, High Intensity	176968	43008	223158	443134
Barren Land	28274	-6241	-2119	19913
Deciduous Forest	-299557	-464127	-1195633	-1959317
Evergreen Forest	-151629	-18069	1778105	1608407
Mixed Forest	335074	-15987	140023	459110
Shrub/Scrub	490801	386026	-1212135	-335309
Grassland/Herbaceous	-3632147	25255	-543963	-4150855
Pasture/Hay	-3257867	-253408	-1459287	-4970562
Cultivated Crops	5070264	90132	848862	6009257
Woody Wetlands	777936	74636	645345	1497918
Emergent Herbaceous Wetlands	-871468	-40755	-783931	-1696154
Forest change over 10 years	661824	-423547	1367840	1606117

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 do not carry the same level of regulatory certainty as do Federal 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).

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. This assessment was influenced by a set of future scenarios with varying assumptions with regard to global and U.S. population, economic growth, climate change, wood energy consumption, and land use change from 2010–2060 (USFS 2012, p. xiii). The 2010 Assessment projected (2010–2060) forest losses of 6.5–13.8 million ha (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 ha (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 approximately 2022 and then leveling off or declines of forest lands (USFS 2016, p. 8-7). However, TCB are found roosting in mature forest stands significantly more often than in younger stands given that regenerating young forests lack the structural diversity preferred by roosting TCB (e.g., broken tree branches with dead leaf clusters) (Veilleux et al. 2003, p. 1072; Perry and Thill 2007, p. 978; Thames 2020, pp. 32–34). In addition, where roosting and foraging habitat is removed, impacts are greater to the species where the species has been impacted by WNS.

Impacts to bats

Forest removal may result in the following impacts to TCB: loss of suitable roosting or foraging habitat, longer flights between suitable roosting and foraging due to habitat fragmentation of remaining forest patches, fragmentation of maternity colonies due to removal of travel corridors, and direct injury or mortality (during active season tree removal).

Loss of roosts →death or injury
TCB may be directly affected by forest habitat loss by removal of occupied roost trees (Belwood 2002, p. 193; McAlpine et al. 2021, p. 2) 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 is likely to result in direct injury or mortality to some bats (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.

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

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). TCB are known to switch roosts (Veilleux and Veilleux 2004a, p. 197; Quinn and Broders 2007, p. 19; Poissant et al. 2010, p. 374); therefore, TCB likely can tolerate some loss of roosts, provided suitable alternative roosts are available (see Chapter 2). However, loss of central or important roosts has caused colony fragmentation in the

northern long-eared bat. For example, Silvis et al. (2015, pp. 6–12) found a loss of approximately 17% of roosts may begin to cause colony fragmentation; however, we have no additional information specific to TCB. One of the most prominent advantages of colonial roosting is the thermoregulatory benefit (Humphrey et al. 1977, pp. 343–344; Kurta et al. 1996, entire). Therefore, smaller colonies are expected to provide fewer thermoregulatory benefits for adults in cool spring temperatures and for nonvolant 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 (Lacki et al. 2007, p. 89) and therefore, reproductive success. TCB 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 with strong summer site fidelity (Allen 1921, p. 54; Veilleux and Veilleux 2004a, p. 197). Reproduction is one of the most energetically demanding periods for temperate-zone bats (Broders et al. 2013, p. 1174). Female TCB produce a maximum of two pups per year; therefore, loss of just two pups 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 TCB by maintaining sufficient quality and quantity of swarming habitat. Some states conduct some of their own forest management activities in the winter within known federally listed endangered and/or threatened bat home ranges, as a measure that would protect maternity colonies and nonvolant 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: rangewide transportation consultation for Indiana bats and northern long-eared bat[9], NiSource Habitat Conservation Plan[10], and rangewide in-lieu fee program for Indiana bats. Many of the beneficial actions associated with these and similar efforts may benefit TCB 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) forest habitat appears to benefit North American tree roosting bats (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 young (Perry and Thill 2007, p. 224). Preserving mature forest habitats should allow for increased roosting opportunities (Veilleux et al. 2003, p. 1072; Perry and Thill 2007, p. 978; Thames 2020, pp. 32–34) which may increase survival or reproductive success. Consequently, we should continue to pursue tried and true management approaches, such as providing for the continual recruitment of mature forest in landscapes with a variety of well-connected forested habitat types.

Summary

In summary, U.S. forest area trends have remained relatively stable with some geographic regions experiencing more forest 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. Impacts from forest habitat removal to individuals or colonies would be expected to range from minor (e.g., removal of a small portion of foraging habitat in largely forested landscapes with robust TCB populations) to significant (e.g., removal of roosting habitat in highly fragmented landscapes with small, disconnected populations). In areas with little forest or highly fragmented forests (e.g., western U.S. and central Midwestern states), impacts from forest removal would likely be higher given decreased roosting opportunities and potential loss of connectivity between roosting and foraging habitat.

Winter Roost Loss and Disturbance

As discussed in Chapter 2, TCB require hibernation sites with specific microclimates and TCB 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. 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.

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[9] https://www.fws.gov/midwest/endangered/section7/fhwa/index.html
[10] https://www.fws.gov/midwest/endangered/permits/hcp/nisource/

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 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 TCB. 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.

Conservation Measures Addressing Hibernacula Loss and Disturbance

Protecting TCB from disturbance during winter is essential because increased arousals from hibernation require greater energy expenditures at a time when food and water resources are scarce or unavailable. This is even more important for hibernacula impacted by WNS because more frequent arousals from torpor increases the probability of mortality in bats with limited fat stores (Boyles and Willis 2010, p. 96).

One method of reducing disturbance at bat hibernacula is through installation of bat-friendly gates that allow passage of bats while reducing disturbance from human entry as well as avoiding 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, the USFS has closed hibernacula 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 use these sites for hibernation.

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. TCB composite plausible future scenarios.

Plausible Scenario	Pd Occurrence Model	WNS Impact Duration	Wind Capacity	All-bat Fatality Rate	% Species Composition
Low impact	Pd Occurrence Model 1	15-yr species-specific survival rates	Lower build-out	Regional-specific	Regional-specific
High impact	Pd Occurrence Model 2	40-yr species-specific survival rates	Higher build-out	Regional-specific	Regional-specific

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 TCB 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	0 (0%)	32 (2%)	421 (22%)	756 (39%)	738 (38%)
Pd occurrence model 2	286 (15%)	63 (3%)	324 (17%)	271 (14%)	997 (51%)

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) 0–6, 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, entire) and corrected/incorporated curtailment information based on facility-specific, unpublished USFWS data. For future projections, we considered projections for 2030, 2040 and 2050 from 4 potential sources: (1) the U.S. Department of Energy (USDOE) April 2015 Wind Vision report (USDOE 2015) and downloadable data; (2) the U.S. Energy Information Administration (USEIA) January 2020 Annual Energy Outlook (AEO) report (USEIA 2020) and downloadable data; (3) the USFWS April 2016 Draft Midwest Wind Multi-Species Habitat Conservation Plan (USFWS 2016); 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 "*one 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 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 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 (Straw 2021, pers. comm.). 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 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 -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 Canada Wind Turbine Database (CWTD). 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); and (2) The Canada Energy Regulator (CER) Canada's Energy Future 2019 Report (CER 2019). 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.*" 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). 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, ten 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*". 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%)

Location	Wind Capacity (MW)		
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%)

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