Effects of Climate Change on Midwestern Ecosystems: Appalachian-Interior-Northeast Mesic Forest

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Foreword

Series Overview

The following is the second chapter of a multi-part series examining the vulnerability of natural habitats in the Midwest region to the impacts of climate change. This work was conceived as part of an interagency effort to facilitate the inclusion of climate change considerations into 2025 revisions of State Wildlife Action Plans (SWAPs). Although tailored to the needs of state agencies within the Midwest region, these assessments focus on habitats and species found throughout North America and are thus broadly relevant to regions and considerations beyond that of the Midwest.

This work was developed as a collaboration between the U.S. Geological Survey (USGS) Midwest Climate Adaptation Science Center (MW CASC) and natural resource agencies from Minnesota, Wisconsin, Michigan, Ohio, Indiana, Illinois, Iowa, and Missouri. During discussions on the needs and challenges of the 2025 SWAP revision process, collaborators identified habitat climate vulnerability as a key area requiring further exploration and deeper understanding.

Process

To assess the impact of climate change on habitats in the Midwest, the MW CASC developed a process comprised of the following steps: 1) potential habitat identification and delineation; 2) summarizing exposure to key climate change stressors; and 3) reviewing and synthesizing relevant scientific literature into qualitative summaries of climate change impacts to habitats.

Habitat Selection

We identified and defined habitats using the U.S. National Vegetation Classification database (USNVC, 2020), which provides a common lexicon of hierarchically classified and spatially referenced vegetation communities throughout the United States. Within this hierarchy, the Group level represents the finest resolution classification tier that is coupled with spatially referenced distributions. Hereafter, discrete “habitats” are synonymous with USNVC Groups and their corresponding lower-level community classifications (Alliance and Association). To enhance relevancy for the Midwest region, the lowest-level classifications for each group (Association) not occurring within the Midwest were excluded from this assessment.

Climate Exposure

To summarize climate exposure, we utilized a combination of Level II and Level III ecoregions (U.S. Environmental Protection Agency, 2013) to identify 12 ecologically relevant subregions of the Midwest that served as the spatial basis for summarizing climate exposure (Figure i). Climate projections are derived from the Coupled Model Intercomparison Project Phase 5 (CMIP5) and were statistically downscaled using the MACAv2-METDATA dataset, which applies the Multivariate Adaptive Constructed Analogs (MACA) method (Abatzoglou, 2013; van Vuuren and others, 2011). Projections are presented under two greenhouse gas concentration pathways: RCP4.5, an emissions scenario assuming global mitigation efforts that lead to greenhouse gas emissions peaking mid-century and stabilizing by 2100, and RCP8.5, a high-emissions scenario that assumes continued growth in greenhouse gas emissions with

limited mitigation. These scenarios represent a range of plausible future climate conditions and are commonly used to bracket moderate and high-end climate change trajectories. In-text references to climate projections primarily detail RCP8.5 estimates to highlight particularly severe outcomes, although all tables and figures report both RCP4.5 and RCP8.5 estimates. To capture variation across CMIP5 climate models, models with the lowest (minimum model) and highest (maximum model) projected values under the RCP8.5 scenario are presented in addition to the mean of all models. Appendix 2 includes a list of all climate models as well as the minimum and maximum model selected for each region and climate variable.

Figure i. Distribution and names of 12 ecoregions within the larger Midwest region. Ecoregions represent a mixture of Level II and Level III EPA ecoregions (U.S. Environmental Protection Agency, 2013) and have been renamed following the Midwest Conservation Blueprint Ecoregions (Wright and others, 2022).

Habitat Prioritization

Seventy-seven natural Groups (i.e., habitats) occur in states comprising the Midwest region (Minnesota, Wisconsin, Michigan, Ohio, Indiana, Illinois, Iowa, and Missouri). To focus this assessment on a subset of these habitats, representatives from each state identified two or more habitats for each ecoregion

present in their respective state. We reconciled conflicting priorities to identify two habitats per ecoregion (hereafter identified as "priority habitats") for a total of 22 priority habitats that served as the focus of our analysis (Table i).

Table i. Priority habitats for each ecoregion. Analogous Level III EPA ecoregion numbers (U.S. Environmental Protection Agency, 2013) and U.S. National Vegetation Classification database Group numbers (USNVC, 2020) are provided for each ecoregion and priority habitat.

Ecoregion	Priority habitat
Appalachian Forest (8.4.3)	Appalachian-Central Interior Mesic Forest (G020)
	Central Appalachian-Northeast Oak Forest and Woodland (G650)
Central Irregular Plains (9.2.4)	Central Midwest Oak Openings and Barrens (G181)
	South Central-Appalachian-Northeast Floodplain Forest Group (G673)
Central Plains (8.2.1–8.2.4)	Midwest Floodplain Forest (G652)
	Central Interior-Great Lakes Flatwoods and Swamp Forest (G917)
Driftless Area (8.1.5)	North-Central Beech- Maple- Basswood Forest (G021)
	North-Central Oak- Hickory Forest and Woodland (G649)
Erie Drift Plains (8.1.1, 8.1.10)	North-Central Beech- Maple- Basswood Forest (G021)
	Appalachian-Northeast Mesic Forest (G742)
Interior Plateau (8.3.2, 8.3.3, 8.3.6, 8.5.2)	Bald-cypress- Tupelo Floodplain Forest (G033)
	Interior Low Plateau Oak Forest, Woodland and Savanna (G950)
Mixed Wood Plains (8.1.4)	Laurentian-Acadian Alkaline Swamp (G046)
	Eastern North American Freshwater Marsh (G125)
Northern Hardwood Forest (5.2.1, 5.2.2)	Laurentian Subboreal Mesic Balsam Fir- Spruce- Hardwood Forest (G048)
	Eastern North American Boreal-Subboreal Bog and Acidic Fen (G1172)
Northern Plains (9.2.1, 9.2.2)	Northern Tallgrass Prairie (G075)
	Midwest Wet Prairie, Wet Meadow, and Shrub Swamp (G770)
Ozarks (8.4.5)	Central Midwest Oak Openings and Barrens (G178)
	Oak- Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna (G949)
Temperate Prairie (9.2.3)	North-Central Oak-Hickory Forest and Woodland (G059)
	Interior Low Plateau Oak Woodland Forest (G333)
Great Lakes Plains (8.1.6)	Midwest Prairie Alkaline Fen (G183)

Focal Wildlife Species

We include focal wildlife species to demonstrate how habitat impacts may influence wildlife responses to climate change. For each priority habitat (Table i), we review climate change impacts on one focal wildlife species, with specific attention paid to habitat considerations. We selected focal wildlife species based on consultation with state agencies, with the aim of choosing species listed by one or more state agency as a Species of Greatest Conservation Need (SGCN).

Focal Aquatic Systems

Because the USNVC is predominantly focused on terrestrial habitat types, we also include sections summarizing the impact of climate change on select focal aquatic systems. Unlike the focal wildlife species sections, which are associated with specific priority habitats (Table i), we selected one focal aquatic system for each ecoregion (Figure i). These selections were based on consultation with state agencies, with the aim of discussing a diverse set of aquatic system types. While focused on specific geographic areas, each section includes climate change impacts that are relevant to the broader aquatic system type.

Literature Review

For each priority habitat, we reviewed and synthesized available scientific literature. We used USNVC habitat descriptions and lists of dominant plant species provided for each Group and relevant sub-level to characterize each habitat and focus our literature search (Appendix 1). We searched for relevant literature related to these habitat characteristics and dominant species that either explicitly mentioned climate change or indirectly discussed climate-related topics. Lastly, to narrow our focus, we identified two particularly relevant climate-related stressors for each habitat to hone our literature search.

Taxonomic Conventions

Wherever possible, we follow the Integrated Taxonomic Information System database (Integrated Taxonomic Information System (ITIS), 2025) for common names, Latin names, and taxonomic authorities. Because many species have multiple common names, we defer to the PLANTS Database (USDA, NRCS, 2025) to select a single, consistent common name for each plant species. In cases where taxonomic resolution is lacking, such as for certain pests or pathogens not included in ITIS, we use the most widely accepted names from the primary scientific literature. To enhance clarity and usability, especially for readers consulting individual sections as standalone references, we define Latin names at first mention for each common name in each chapter subsection.

Chapter Organization

Each chapter in this series focuses on a subset of ecologically related priority habitats (Table ii). Because climate change impacts are frequently shared between similar habitat types, we first provide an overview of climate impacts to the broader system type before summarizing climate impacts for each specific habitat. The higher-level USNVC classification that links ecologically similar habitats (typically at the Macrogroup level and hereafter referred to as “ecosystem”) determines each broader system type into which chapters are organized.

Table ii. Series chapters and priority habitats. Habitats are organized by the series chapter in which they occur, with their corresponding U.S. National Vegetation Classification database Group numbers (USNVC, 2020).

Series chapter	Priority habitat
1. Temperate Flooded and Swamp Forest	Midwest Floodplain Forest (G652)
	South Central-Appalachian-Northeast Floodplain Forest Group (G673)
	Bald-cypress- Tupelo Floodplain Forest (G033)
	Central Interior-Great Lakes Flatwoods and Swamp Forest (G917)
	Laurentian-Acadian Alkaline Swamp (G046)
2. Appalachian-Interior-Northeast Mesic Forest	Appalachian-Central Interior Mesic Forest (G020)
	Appalachian-Northeast Mesic Forest (G742)
	North-Central Beech- Maple- Basswood Forest (G021)
3. North American Bog and Fen	Midwest Prairie Alkaline Fen (G183)
	Eastern North American Boreal-Subboreal Bog and Acidic Fen (G1172)
4. Eastern North American Temperate Freshwater Marsh, Wet Meadow and Shrubland	Eastern North American Freshwater Marsh (G125)
	Midwest Wet Prairie, Wet Meadow and Shrub Swamp (G770)
5. Central and Eastern North American Grassland and Shrubland	Central Tallgrass Prairie (G333)
	Northern Tallgrass Prairie (G075)
	Central Interior Acidic Open Glade and Barrens (G178)
	Eastern North American Ruderal Meadow and Shrubland (G059)
6. Central Interior Oak Forest, Woodland and Savanna	Interior Low Plateau Oak Forest, Woodland and Savanna (G950)
	Oak- Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna (G949)
	Central Midwest Oak Openings and Barrens (G181)
	North-Central Oak- Hickory Forest and Woodland (G649)
	Central Appalachian-Northeast Oak Forest and Woodland (G650)
7. Laurentian-Acadian Mesic Hardwood- Conifer Forest	Laurentian Subboreal Mesic Balsam Fir- Spruce- Hardwood Forest (G048)

Executive Summary

The Appalachian-Interior-Northeast Mesic Forest ecosystem, historically buffered by cool, moist conditions, may experience significant stress under future climate change, particularly due to intensifying droughts and milder winters in the midwestern United States. Droughts are expected to intensify in frequency and severity, depleting soil moisture, increasing tree mortality, and reshaping species composition. Increasing aridity and disrupted hydrologic cycles will likely accelerate soil erosion, deplete nutrients, and heighten wildfire risk. Meanwhile, milder winters may reduce snowpack insulation, increase freeze-thaw cycles, and alter growing seasons, potentially amplifying cold stress, disrupting phenology, and contributing to shifts in habitat structure and community composition. While easing winter severity may temporarily boost plant productivity and facilitate species migration into and throughout the Midwest, it can also increase the risk of frost damage for early-leafing trees and disrupt ecological relationships, such as plant-pollinator interactions.

Together, these stressors may drive fundamental shifts in habitat structure and community composition, favoring drought-, fire-, and cold-tolerant species, while historically dominant, moisture-dependent species decline. Species with limited drought resistance, such as those with shallow roots or low water-use efficiency, may be especially vulnerable, while drought-adapted taxa could gain a competitive advantage. This shift could trigger a departure from over a century of mesophication in the Appalachian-Interior-Northeast Mesic Forest ecosystem, which has favored shade-loving, moisture-dependent species in fire-suppressed landscapes. As a result, these forests may be particularly ill-equipped to withstand the novel environmental conditions imposed by intensifying droughts and milder winters. The Appalachian-Northeast Mesic Forest habitat, dominated by eastern hemlock (*Tsuga canadensis*) and eastern white pine (*Pinus strobus*), is likely particularly vulnerable, as both dominant species are projected to decline due to increasing drought stress and shifting competitive dynamics. Western portions of the North-Central Beech - Maple - Basswood Forest habitat, found in the Driftless Area of Wisconsin, Minnesota, and Iowa, may be more vulnerable to climate change than more eastern portions of the habitat due to its already drier conditions.

Species interactions, including invasive species, pests, and herbivory, are also likely to be reshaped by climate change, compounding stress on habitats throughout the Appalachian-Interior-Northeast Mesic Forest ecosystem. Warmer winters and increased disturbance may facilitate the expansion of invasive species, which outcompete native vegetation and alter ecosystem dynamics. At the same time, pests and pathogens are likely to become more destructive, as milder winters enhance their survival and spread and drought weakens tree defenses. Additionally, rising white-tailed deer (*Odocoileus virginianus*) populations, supported by warmer winters, may shift forest regeneration patterns by selectively browsing on sensitive seedlings and saplings, limiting the recruitment of historically dominant tree species while favoring browse-resistant plants. Collectively, these pressures can drive significant and ongoing ecological transformation in the Appalachian-Interior-Northeast Mesic Forest, highlighting the need for adaptive management strategies to sustain biodiversity and ecosystem function.

Appalachian-Interior-Northeast Mesic Forest

Key Climate Change Stressors

The Appalachian-Interior-Northeast Mesic Forest ecosystem is a broadly distributed temperate forest ecosystem that supports a high diversity of hardwood tree species and spring ephemerals. In the Midwestern United States, this ecosystem typically occurs in locally cool, moist areas like north-facing slopes and coves, where microclimatic conditions may partially offset some near-term climate change impacts. However, the adaptive capacity provided by these refugia may be increasingly challenged by projected shifts in drought frequency and intensity and by milder winters (Ackerly and others, 2020).

Drought

Drought is a novel climate stressor for ecosystems historically governed by ample precipitation. Severe droughts were historically rare in the Appalachian-Interior-Northeast Mesic Forest ecosystem (Clark and others, 2016; Cook and others, 1999; Cook and others, 2010; Pederson and others, 2014; Vose and others, 2019). While the eastern United States has become wetter over the past several decades, a trend expected to continue, droughts are also projected to become more frequent and intense (Zhao and others, 2020; Ford and others, 2021; Chen and Ford, 2023; Zhao and others, 2023). This shift is driven by increasing seasonal variability in precipitation, more extreme precipitation events, and higher temperatures that raise aridity and evaporative demand (Byun and Hamlet, 2018; Ficklin and others, 2015; Zhao and others, 2020). As a result, the Midwest may face both short-term droughts and prolonged seasonal droughts with increasing frequency and severity, nearing levels currently observed in the drier western United States (Will and others, 2013; Ford and others, 2021; Costanza and others, 2023). Projections based on the Standardized Precipitation-Evapotranspiration Index (SPEI) indicate higher incidences of extreme drought by the end of the century (Figure 1, Figure 2).

As droughts become more frequent and severe, wildfire risk may also increase. Historically, wildfire was an infrequent but ecologically important disturbance in the Appalachian-Interior-Northeast Mesic Forest ecosystem (Arthur and others, 2021; Brown and others, 2021; Cohen, 2005; Frelich and others, 2021; Gao and others, 2021). While catastrophic fires were rare, low-intensity fires played a key role in shaping species distributions (Arthur and others, 2021; Cohen, 2005). Modern fire suppression, combined with increasing precipitation, has largely eliminated fire from these systems, leading to widespread mesophication (i.e., the increase of fire-intolerant species) (Arthur and others, 2021; Dey and others, 2019; Nowacki and Abrams, 2008). However, rising temperatures and more frequent droughts have already contributed to increasing fire activity in parts of the Midwest (Apfelbaum and others, 2023; Donovan and others, 2023; Vose and others, 2019). Fire frequency and severity could rise by as much as 100% in the coming decades as extremely dry conditions become more common (Anderegg, 2015; Gao and others, 2021; Kerr and others, 2018; Ma and others, 2016; Stanturf and others, 2024).

Mild Winters

Mild winters are another influential climate change stressor, reshaping ecological processes in the Appalachian-Interior-Northeast Mesic Forest ecosystem. Winter conditions characteristic of the Midwest, such as shortened growing seasons, prolonged snow cover, and harsh minimum temperatures,

strongly shape the composition and functioning of many ecosystems (Henne and others, 2007; Kreyling, 2019; Rittenhouse and Rissman, 2015). However, over the past century, winters have become milder, marked by fewer snow-covered days, reduced snow depth, moderate temperature extremes, and an increase in freeze-thaw cycles (Contosta and others, 2019; Contosta and others, 2020; Ford and others, 2021b; Ford and others, 2022; Sinha and Cherkauer, 2010). These trends are expected to intensify in the coming decades, with projections indicating longer growing seasons, further reductions in snowfall, and continued winter warming (Figure 3) (Byun and others, 2019; Demaria and others, 2016). Across much of the Midwest, snowpack is expected to decline, with some regions becoming nearly snow-free by the end of the century (Demaria and others, 2016). While lake-effect snowfall may temporarily increase due to reduced ice cover on the Great Lakes, the overall trend will be toward less frequent and shallower snow cover, earlier snowmelt, and more mid-winter thaws (Ford and others, 2021b; Ford and others, 2022; Wright and others, 2013). Counterintuitively, however, declining snowpack reduces thermal insulation, exposing overwintering species to more extreme cold events, while fewer snow-covered days and earlier springs can lead to more frequent freeze-thaw cycles, further increasing cold stress (Contosta and others, 2019; Contosta and others, 2020; Contosta and others, 2024; Dey and others, 2019; Sanders-DeMott and others, 2018).

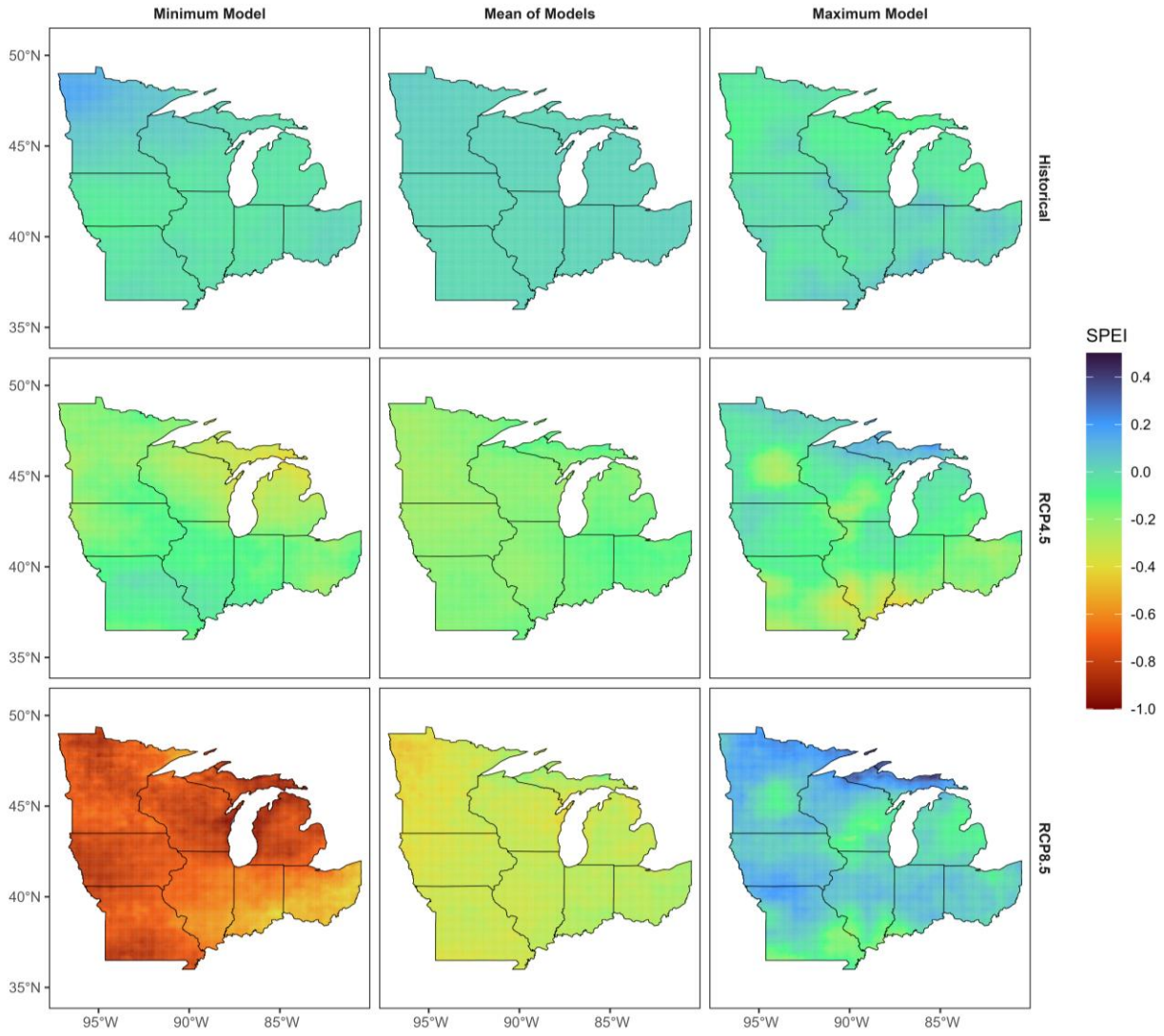

Figure 1. Projected changes in Standardized Precipitation-Evapotranspiration Index in the Midwest. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). SPEI values are standardized to a mean of zero and a standard deviation of one, with positive values indicating wetter conditions and negative values indicating drier conditions relative to a reference period (1981–2020) under RCP 4.5 (Beguería and others, 2017; Vicente-Serrano and others, 2010). Values show average monthly historical (1971–2000, top row) and projected (2070–2099) SPEI for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected SPEI under RCP 8.5 are presented in addition to the mean of all models (middle column).

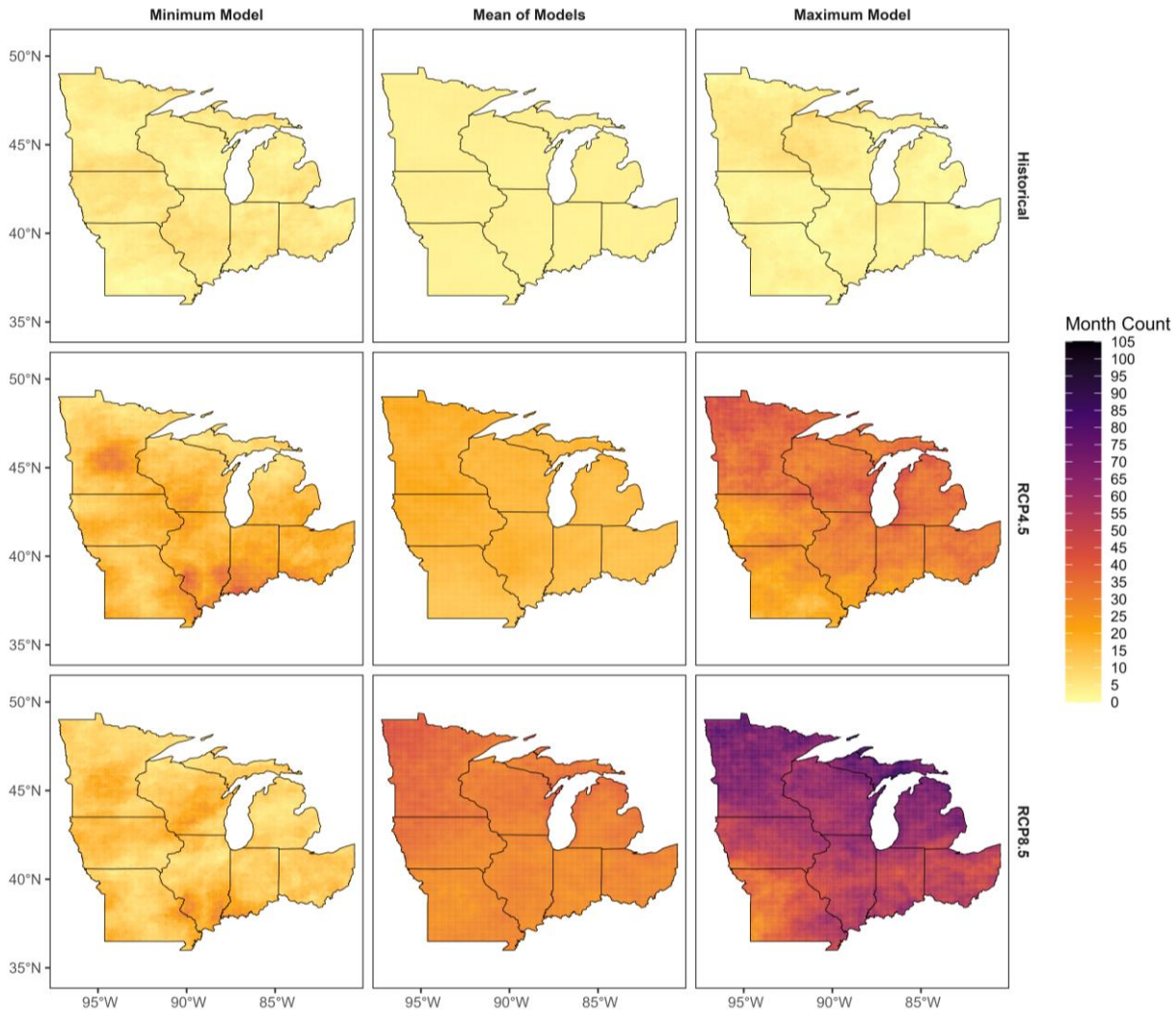

Figure 2. Projected changes in 30-year extreme drought incidence in the Midwest. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000, top row) and projected (2070–2099) 30-year extreme drought incidence (months) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models (middle column).

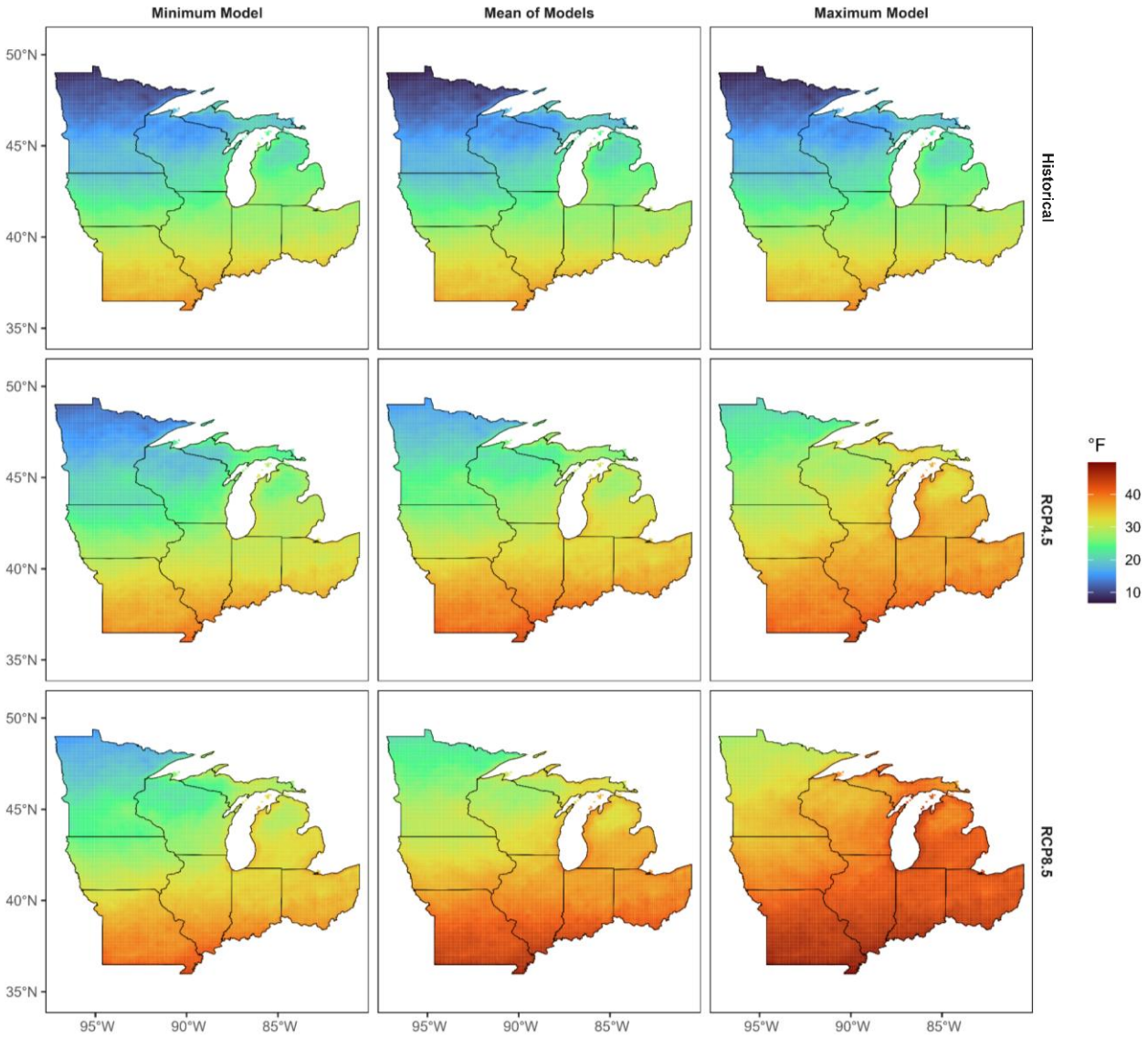

Figure 3. Projected changes in winter mean temperature in the Midwest. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000, top row) and projected (2070–2099) winter (December–February) mean temperature (°F) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected winter mean temperature under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

The Appalachian-Interior-Northeast Mesic Forest ecosystem has historically served as a climate and fire refugium for some species and may continue to do so (Bellemare and Moeller, 2014; Butler and others, 2015; Swanston and others, 2018). However, increasing drought and milder winters are expected to drive substantial changes in this ecosystem, altering habitat structure, community composition, and species interaction.

Habitat Structure

Abiotic responses to drought and mild winters. Drought and mild winters can alter the physical structure and abiotic processes of the Appalachian-Interior-Northeast Mesic Forest ecosystem, particularly soil biogeochemistry and erosion dynamics. Reduced vegetative cover due to prolonged drought weakens soil cohesion and increases susceptibility to erosion (Vose and others, 2016). Concurrently, declining snowpack and earlier snowmelt contribute to drier soils and greater variability in runoff, further increasing erosion risk (Gaertner and others, 2020; Melton and others, 2016; Wieder and others, 2022). As snowfall declines and more precipitation falls as rain, higher water infiltration rates may accelerate soil mineral leaching, altering nutrient availability (Campbell and others, 2009). These changes, combined with shifts in precipitation patterns and heightened fire risk, threaten to accelerate soil degradation and nutrient loss (Butler and others, 2015; Contosta and others, 2019).

Soil biogeochemistry and forest community composition are closely linked, so as climate change alters species composition, it can also indirectly reshape soil dynamics. For example, forests dominated by maples (*Acer* spp.) tend to maintain cooler, wetter soils with lower carbon-to-nitrogen (C:N) ratios and higher nitrogen immobilization compared to forests dominated by oaks (*Quercus* spp.) (Alexander and Arthur, 2010; Lovett and others, 2004; Palus and others, 2018; Woodbridge and others, 2022). Similarly, litter from dogwoods (*Cornus* spp.) contributes to higher soil fertility and higher soil pH (Small and Chamberlain, 2015). However, increasing drought and fire incidence may reduce the abundance of these species, weakening their influence on soil properties. Milder winters further drive biogeochemical shifts by promoting the expansion of invasive species that alter soil chemistry. For instance, tree-of-heaven (*Ailanthus altissima*) and Nepalese browntop (*Microstegium vimineum*), both of which can modify soil nutrient cycling, may become more abundant as winter severity declines (Coville and others, 2021; Isler and others, 2023; McGrath and Binkley, 2009; Medina-Villar and others, 2016).

Biotic habitat responses to drought. Rising temperatures and evaporative demand are already slowing plant growth and reducing community resilience throughout the Appalachian-Interior-Northeast Mesic Forest ecosystem, leading to increased mortality (Allen and others, 2010; D'Orangeville and others, 2018; Forzieri and others, 2022; Lucash and others, 2018; Quetin and others, 2023). As these conditions intensify and precipitation patterns change, drought further stresses forest species by reducing their water flow efficiency and limiting photosynthetic capacity, making them more vulnerable to mortality and secondary stressors such as pests and pathogens (Choat and others, 2018; Fusco and others, 2024; Novick and others, 2022; Wang and others, 2012). Species-specific traits may increase individual species' vulnerability. For example, the shallow roots of sugar maple (*Acer saccharum*) limit access to groundwater (Boakye and others, 2023), and the seeds of eastern hemlock (*Tsuga canadensis*) are highly susceptible to desiccation (Evans and others, 2022; Godman and Lancaster, 1990; Rollinson and others, 2021; Vose and others, 2016). As rates of mortality and regeneration failure increase, expanding canopy gaps will drive structural changes, enhancing light penetration and thermal exposure in the understory

(Pederson and others, 2014; Roberts, 2004), which can promote recruitment, likely favoring those better adapted to drought conditions (Vose and others, 2016). However, while canopy gaps may promote recruitment of drought-adapted saplings, they can also increase drought and thermal stress for herbaceous understory species such as Canadian wild ginger (*Asarum canadense*) and pale touch-me-not (*Impatiens pallida*) (Augspurger, 2017; Deng and others, 2023; Elliott, 2014; Landuyt and others, 2019).

In the Appalachian-Interior-Northeast Mesic Forest ecosystem, as climate change increases drought intensity, fire is expected to become a more significant disturbance in this system (Butler and others, 2015; DeSantis and Moser, 2016; Frelich and others, 2021; Hanberry, 2024). Many species vulnerable to drought are also highly susceptible to fire. Severe burns cause high mortality in species like flowering dogwood (*Cornus florida*) and red maple (*Acer rubrum*), whereas more fire-adapted species, such as white oak (*Quercus alba*), experience significantly lower rates of mortality (Arthur and others, 2021).

Biotic habitat responses to mild winters. Declining snowpack and earlier snowmelt will likely increase frost susceptibility, shift spring soil moisture availability, and heighten physiological stress for mesic forest species (Boakye and others, 2023; Contosta and others, 2019; Sanders-DeMott and others, 2018; Weigel and others, 2021). Similarly, warmer winters and longer growing seasons may advance phenology, such as budburst and flowering timing. Such phenological shifts may increase vulnerability to late spring frosts, damaging leaves and reproductive structures, limiting growth, and impairing reproduction (Calinger and others, 2013; Chamberlain and Wolkovich, 2021; Dey and others, 2019). Additionally, future warming may disrupt unique physiological requirements for plants, such as overwinter chilling requirements; consequently, leaf-out may be delayed and freeze tolerance decreased, compounding the risks of late-season frosts (Chamberlain and Wolkovich, 2021). These changes threaten not only mature trees but also regeneration, as seedlings and saplings face greater exposure to freezing temperatures (Dey and others, 2019; Walck and others, 2011).

Shifting winter conditions may also disrupt ecological interactions in the Appalachian-Interior-Northeast Mesic Forest ecosystem. For example, understory herbaceous species such as dogtooth violet (*Erythronium americanum*) and Virginia springbeauty (*Claytonia virginica*) are now flowering earlier, potentially creating phenological asynchrony with their pollinators and disrupting other key ecological relationships (Augspurger and Zaya, 2020; Bertin, 2017; Calinger and others, 2013; Chamberlain and Wolkovich, 2021; Dey and others, 2019; Miller and Stuble, 2024; Mura and others, 2022; Tessier, 2022). While some species may benefit from longer growing seasons, the combined effects of increased frost damage, disrupted plant-pollinator relationships, and shifting competition dynamics may outweigh gains in productivity (Calinger and others, 2013; Contosta and others, 2019; Landuyt and others, 2019; Swanston and others, 2018; Tessier, 2022).

Community Composition

Community responses to drought. Currently, mesic species dominate most existing regeneration cohorts, and wetter annual conditions further promote mesic community assemblages (Novick and others, 2022; Vose and others, 2016; Woodbridge and others, 2022). In turn, as these tree species exhibit limited adaptive capacity to fire and drought, this current trend may further increase the vulnerability of these systems to future drought (Dey and others, 2019). Eventually, projected increases in drought frequency and severity may counteract wetter conditions overall and drive shifts in community composition in the Appalachian-Interior-Northeast Mesic Forest ecosystem (Hwang and others, 2020; Maxwell and others, 2024; Quetin and others, 2023). Potential shifts toward xerophytic assemblages would then represent a

reversal of mesophication trends that have shaped the Appalachian-Interior-Northeast Mesic Forest ecosystem over the past century (Abrams and others, 2021; Frelich and others, 2017; Novick and others, 2022; Woodbridge and others, 2022). Projected increases in drought frequency and severity will likely favor drought- and fire-tolerant species, such as oaks (*Quercus* spp.), while reducing the abundance of moisture-dependent taxa, including eastern white pine (*Pinus strobus*), sugar maple (*Acer saccharum*), and American beech (*Fagus grandifolia*) (Asbjornsen and others, 2021; Bapikee, 2013; Batllori and others, 2020; Butler and others, 2015; DeSantis and Moser, 2016; Frelich and others, 2017; Joyce and Rehfeldt, 2013; Knoot and others, 2015; Vose and others, 2016; Wuebbles and others, 2021). Similarly, following canopy loss within communities dominated by the highly drought-sensitive eastern hemlock (*Tsuga canadensis*), forest composition may transition into stands dominated by oak (*Quercus* spp.), birch (*Betula* spp.), maple (*Acer* spp.), and tuliptree (*Liriodendron tulipifera*), or become overrun by rhododendron (*Rhododendron* spp.) thickets, which can suppress tree regeneration (Small and Chamberlain, 2015). The loss of dense canopy cover, driven by drought- and fire-induced mortality and shifts in canopy composition, will reduce shading and moisture retention, making understory species more vulnerable to severe warming and drought conditions (Augspurger, 2017; Landuyt and others, 2019; Sanczuk and others, 2023; Wang and others, 2023). Additionally, due to their limited dispersal capacity, understory herbaceous species may be particularly susceptible to these changes.

Community responses to mild winters. Milder winters, leading to reduced snow accumulation, warmer temperatures, and altered phenology, are expected to drive community composition shifts in the Appalachian-Interior-Northeast Mesic Forest ecosystem. In the northeastern Midwest, lake-effect snowfall strongly influences mesic forest species assemblages, with sugar maple (*Acer saccharum*) dominating snowbelt regions southeast of the Great Lakes, while eastern hemlock (*Tsuga canadensis*) is more prevalent in areas with lower snowfall (Henne and others, 2007). With mild winters, species previously limited by extreme cold and high snowpack may expand northward and into snowbelt regions. However, changing freeze-thaw cycles and increased late spring frosts may impose new constraints on both resident and arriving species (Aerts and others, 2006; Boisvert-Marsh and de Blois, 2021; Hanberry, 2024; Harrison and others, 2020; Kreyling, 2010; Ma and others, 2019; Osland and others, 2021; Sakai and Weiser, 1973). These conditions may disadvantage species that rely on snowpack insulation and favor frost-resilient and phenologically flexible species (Chu and others, 2023; Contosta and others, 2019; Lindback and others, 2023; Reinmann and others, 2019). However, if canopy species accelerate their growth in response to early spring warming faster than understory plants, many native wildflowers that rely on early-spring light availability may struggle to survive (Miller and others, 2023).

Invasive Species

Invasive species responses to drought and mild winters. Increasing drought and milder winters are likely to accelerate the expansion of invasive species, reducing biodiversity and reshaping the Appalachian-Interior-Northeast Mesic Forest ecosystem (Small and Chamberlain, 2015). While drought may directly suppress some drought-intolerant invasive species, such as garlic mustard (*Alliaria petiolata*), it can also drive long-term increases in fire severity and frequency, which may ultimately favor many invasive species (Arthur and others, 2021; Black and others, 2018; Byers and Quinn, 1998; Hanberry, 2022). Milder winters may enable the expansion of invasive species currently limited by cold temperatures and prolonged snow cover, including Japanese barberry (*Berberis thunbergii*), kudzu (*Pueraria montana*), Oriental bittersweet (*Celastrus orbiculatus*), and tree-of-heaven (*Ailanthus altissima*) (Bradley and others, 2010; Butler and others, 2015; USDA Forest Service, USA and others, 2019; Vose and others, 2013a; Wavrek and others, 2017). Additionally, as novel winter-to-spring phenology disrupts

germination of native species with narrow physiological requirements, invasive species with broad germination niches may gain a competitive advantage (Hicks and others, 2019). As the combined effects of milder winters and increasing drought drive native species mortality and regeneration failure, creating canopy gaps, they may also indirectly favor shade-intolerant invaders that can readily exploit these newly available openings (Anderegg, 2015; Burnham and Lee, 2010; Olson and others, 2021; USDA Forest Service, USA and others, 2019; Vose and others, 2019).

Pests and Pathogens

Pest responses to drought. Drought will likely weaken tree defenses and increase susceptibility to pests, driving substantial compositional changes in the Appalachian-Interior-Northeast Mesic Forest ecosystem (Dey and others, 2019; Dukes and others, 2009; Lucash and others, 2018; Stanturf and others, 2024). Droughts can trigger physiological stress, weakening tree defenses and making them more vulnerable to pest outbreaks (Anderegg, 2015; Choat and others, 2012; Jactel and others, 2012; Kolb and others, 2016a; Rauschendorfer and others, 2022; Stanturf and others, 2024). For example, cambium-feeding insects such as the emerald ash borer (*Agrilus planipennis*) and hemlock woolly adelgid (*Adelges tsugae*), which have already decimated ash (*Fraxinus* spp.) and eastern hemlock (*Tsuga canadensis*) populations, thrive on drought-stressed hosts (Butler and others, 2015; Chakraborty and others, 2014; Jactel and others, 2012; Ramsfield and others, 2016; Vose and others, 2019). Periodic oak (*Quercus* spp.) mortality events are also linked to drought, as seen in past outbreaks of red oak borer (*Enaphalodes rufulus*) (Fierke and others, 2007; Haavik and others, 2015; Stump and others, 2024). Drought-stressed trees are also more vulnerable to defoliating insects, such as forest tent caterpillar (*Malacosoma disstria*), which can cause extensive defoliation (Stump and others, 2024). Drought can also diminish natural pest suppression by other organisms. For example, the fungal pathogen *Entomophaga maimaiga*, which helps control spongy moth (*Lymantria dispar*) populations, is less effective under dry conditions (Reilly and others, 2014).

Pest responses to mild winters. Mild winters may also intensify pest impacts in the Appalachian-Interior-Northeast Mesic Forest ecosystem, increasing outbreak potential and weakening tree defenses (Canelles and others, 2021; Flower and Gonzalez-Meler, 2015; Pureswaran and others, 2018). Warmer winter temperatures may enhance overwinter survival rates for many pest species, including bark beetles (Scolytinae), hemlock woolly adelgid (*Adelges tsugae*), southern pine beetle (*Dendroctonus frontalis*), and spongy moth (*Lymantria dispar*), allowing populations to expand into areas of the Midwest where winter conditions once limited their range (Bentz and Jönsson, 2015; Contosta and others, 2019; Dukes and others, 2009; Hatfield and others, 2015; Pureswaran and others, 2018). By the end of the century, hemlock woolly adelgid and southern pine beetle are projected to expand their range northward to encompass nearly all of the Midwest (Cheah, 2017; Fitzpatrick and others, 2019; Lesk and others, 2017; Paradis and others, 2008a; Pureswaran and others, 2018). Other pests already present in the region, such as the emerald ash borer (*Agrilus planipennis*) and ambrosia beetles (Scolytinae and Platypodinae), may proliferate as lower winter mortality fosters higher population densities (Pureswaran and others, 2018; Ranger and others, 2015; Stanturf and others, 2024). Because most pests are very responsive to ambient temperature, warming may accelerate their development and increase the number of generations per growing season (Anderegg and others, 2022; Bentz and Jönsson, 2015; Flower and Gonzalez-Meler, 2015; Hamann and others, 2021; Pureswaran and others, 2018; Stanturf and others, 2024). Additionally, earlier growing seasons and increased plant productivity may benefit pests by providing greater foliage availability (Pureswaran and others, 2018). While warmer conditions may increase pest abundance, they may also strengthen natural pest control mechanisms, such as predation and parasitism, provided these organisms can track and respond to climatic changes (Furlong and

Zalucki, 2017; Hunter and others, 2014; Jeffs and Lewis, 2013; Jing and others, 2017; Pureswaran and others, 2018; Tougeron and others, 2020).

Pathogen responses to drought. Many pathogens, including root rot (*Armillaria* spp.), Hypoxylon canker (*Hypoxylon* spp.), and oak wilt (*Bretziella fagacearum*), preferentially infect stressed hosts and often become more virulent following drought events (Dukes and others, 2009; Jactel and others, 2012; Janowiak and others, 2014; Moser and others, 2020; Sturrock and others, 2011; Vose and others, 2016; Wood and others, 2018). Similarly, oak decline, a widespread disease complex devastating oak (*Quercus* spp.) stands across North America, is exacerbated by drought, as multiple pests and pathogens take advantage of weakened trees (Conrad and others, 2020; Contreras-Cornejo and others, 2023; Sturrock and others, 2011; Wood and others, 2018). Additionally, pathogen infections may reduce tree resilience to subsequent drought events, creating a feedback loop of weakening defenses and increasing vulnerability (Jactel and others, 2012). However, drought may not benefit all pathogens, as many require moist conditions for infection and spread. Reduced moisture availability could inhibit fungal pathogens that thrive in wet environments, potentially making some diseases, such as white-pine blister rust (*Cronartium ribicola*), less prevalent (Kinloch, 2003; Kolb and others, 2016b).

Pathogen responses to mild winters. By reducing cold-induced mortality and extending the duration of mild temperatures conducive to infection, mild winters enhance pathogen survival and virulence (Balla and others, 2021; Dukes and others, 2009; Sturrock and others, 2011; Velásquez and others, 2018). For example, shorter winters may lengthen the period of soil activity for fungal rhizomorphs, such as those of root rot (*Armillaria* spp.) (La Porta and others, 2008). Additionally, higher average winter temperatures increase the overwintering survival of white pine needle damage pathogens (including *Bifusella linearis*, *Lecanosticta acicula*, *Lophophacidium dooksii*, and *Septorioides strobi*), leading to more frequent mass defoliation events in eastern white pine (*Pinus strobus*) (Wyka and others, 2018). Historically, cold winter temperatures have limited the geographic distribution of many forest pathogens (Kliejunas, 2011). However, as winter severity declines, several water molds (e.g., *Phytophthora* spp.) that afflict oaks (*Quercus* spp.), maples (*Acer* spp.), and pines (*Pinus* spp.) may expand their ranges into the Midwest (Burgess and others, 2017; Contreras-Cornejo and others, 2023; Sturrock and others, 2011; Teshome and others, 2020). Perhaps most significantly, milder winters may facilitate the survival and spread of previously unknown or novel pathogens (Nnadi and Carter, 2021).

Herbivory

Herbivory responses to climate change. In the Midwest, abundant white-tailed deer (*Odocoileus virginianus*) may increase further as mild winters enhance survival and improve access to winter forage (Côté and others, 2004; Felton and others, 2024; MDNR, 2016; Patton and others, 2021; Weiskopf and others, 2019). Selective browsing can suppress the regeneration of preferred species, including sugar maple (*Acer saccharum*), eastern hemlock (*Tsuga canadensis*), eastern white pine (*Pinus strobus*), and most oaks (*Quercus* spp.) (Henry and Walters, 2023; Patton and others, 2021; Sample and others, 2023). This shift favors less palatable, browsing-resistant species, such as American beech (F*agus grandifolia*) and hophornbeam (*Ostrya virginiana*), ultimately leading to more uniform, less diverse forests over time (Champagne and others, 2021; Dobor and others, 2024; White, 2012). Additionally, deer browsing has contributed to losses of palatable understory species (Koon and others, 2024; Sabo and others, 2023; Small and Chamberlain, 2015). Reduced snowpack and earlier spring snowmelt may intensify herbivory on understory herbs, particularly slow-growing spring ephemerals such as white trillium (*Trillium grandiflorum*) and Jack-in-the-pulpit (*Arisaema triphyllum*) (Royo and others, 2010; Sabo and others, 2023), as well as on tree seedlings and saplings, including sugar maple (*Acer saccharum*) (Henry and

Walters, 2023; Sanders-DeMott and others, 2018). Hotter and drier summers may also influence browsing dynamics. Summer drought conditions alter forage availability and nutritional quality, potentially increasing both overall forage consumption and browse selectivity (Deyoung and others, 2019; Dykes and others, 2020; Felton and others, 2024; Folks and others, 2014; Lashley and Harper, 2012). As deer browse fewer plant species, droughts indirectly amplify browse pressure on species that are selected, such as blackgum (*Nyssa sylvatica*), and weaken browse pressure on unselected species, such as flowering dogwood (*Cornus florida*) (Lashley and Harper, 2012).

Climate Change Impacts to Habitat Groups

Appalachian-Central Interior Mesic Forest

In the Midwest, the Appalachian-Central Interior Mesic Forest habitat is found in the southern portions of Ohio, Indiana, and Illinois (Figure 4). In this region, these forests typically occur on cool, moist slopes, steep ravines, coves, and valley bottoms, often in sheltered positions with north- to east-facing aspects that reduce solar radiation and evapotranspiration and enhance soil moisture retention (Hack and Goodlett, 1960; Parker, 1987; Pyne and Gawler, 2015; USNVC, 2025). Community composition is primarily shaped by site-specific soil characteristics, such as chemistry, acidity, substrate depth, and drainage capacity, along with elevation and exposure (Butler and others, 2015; Pyne and Gawler, 2015; Sampson, 1930). Poorly drained sites are often dominated by beech (*Fagus* spp.) and maple (*Acer* spp.), drier sites tend to be dominated by oak (*Quercus* spp.), and nutrient-poor, acidic soils are more likely to be dominated by conifers (gymnosperms) (Butler and others, 2015; Parker, 1987; Pyne and Gawler, 2015; Sampson, 1930). Forest stands are predominantly tall and closed, with a diverse canopy and subcanopy layer, while the low-light understory supports a diverse and often lush assemblage of shade-tolerant forbs and shrubs (Pyne and Gawler, 2015; USNVC, 2025). These forests are distinguished by forty-one dominant plant species (Table 1), though other non-dominant species also contribute to overall habitat diversity and function.

These communities rarely experience stand-wide disturbances. For example, fire is rare in these sheltered forests, typically occurring only during extreme dry spells (Butler and others, 2015; Faber-Langendoen and others, 2020; USNVC, 2025). Instead, gap-phase regeneration is primarily driven by individual treefall from wind or natural mortality, resulting in variable age-class structure (Butler and others, 2015; Faber-Langendoen and others, 2020; Parker, 1987; USNVC, 2025). However, due to favorable site conditions for tree growth, these forests are highly prone to logging, which has likely promoted more even-aged stands and greater dominance by deciduous species (Braun, 1967; Butler and others, 2015). Additionally, drier sites have also likely undergone mesophication, in which increasingly wet conditions favor shade-tolerant maples over oaks, shifting forest composition over time (Nowacki and Abrams, 2008; Rudolph and others, 2024; Woodbridge and others, 2022).

These forests face growing threats from increased wind disturbances, intensified drought stress, and more frequent wildfires (Anderegg and others, 2022; Brown and others, 2021; Butler and others, 2015; Forzieri and others, 2021; Stanturf and others, 2024). Additionally, pests such as the emerald ash borer (*Agrilus planipennis*) and hemlock woolly adelgid (*Adelges tsugae*) have already impacted large areas of the Appalachian-Central Interior Mesic Forest habitat in the Midwest, altering successional dynamics and accelerating canopy turnover (Butler and others, 2015; Klooster and others, 2018; Parker, 1987; Vose and others, 2013b). These forests are also increasingly threatened by the spread of shade-tolerant invasive species, including garlic mustard (*Alliaria petiolata*), honeysuckle (*Lonicera* spp.), rhododendron (*Rhododendron* spp.), and tree-of-heaven (*Ailanthus altissima*), that disrupt native plant communities and regeneration processes (Ao and others, 2025; Braun, 1967; Butler and others, 2015; Elliott and Miniat, 2018; Faber-Langendoen and others, 2020; Pyne and Gawler, 2015; USNVC, 2025).

Table 1. Plant species common to the Appalachian-Central Interior Mesic Forest habitat. Taxonomic authority, common name, and growth habit are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Adiantum pedatum (Rupr.) Paris	Aleutian maidenhair	Fern
Botrychium virginianum (L.) Sw.	Rattlesnake fern	Fern
Dennstaedtia punctilobula (Michx.) T. Moore	Eastern hayscented fern	Fern
Polystichum acrostichoides (Michx.) Schott	Christmas fern	Fern
Actaea racemosa L.	Black baneberry	Forb/herb
Amphicarpaea bracteate (L.) Fernald	American hogpeanut	Forb/herb, vine
Arisaema triphyllum (L.) Schott	Jack in the pulpit	Forb/herb
Asarum canadense L.	Canadian wildginger	Forb/herb
Caulophyllum thalictroides (L.) Michx.	Blue cohosh	Forb/herb
Claytonia virginica L.	Virginia springbeauty	Forb/herb
Erythronium americanum Ker Gawl.	Dogtooth violet	Forb/herb
Eurybia divaricate (L.) G.L. Nesom	White wood aster	Forb/herb
Galium triflorum Michx.	Fragrant bedstraw	Forb/herb, vine
Geranium maculatum L.	Spotted geranium	Forb/herb
Hydrophyllum canadense L.	Bluntleaf waterleaf	Forb/herb
Hydrophyllum virginianum L.	Eastern waterleaf	Forb/herb
Impatiens pallida Nutt.	Pale touch-me-not	Forb/herb
Laportea canadensis (L.) Weddell	Canadian woodnettle	Forb/herb
Osmorhiza claytonia (Michx.) C.B. Clarke	Clayton's sweetroot	Forb/herb
Prosartes lanuginose (Michx.) D. Don	Yellow fairybells	Forb/herb
Sanguinaria canadensis L.	Bloodroot	Forb/herb
Sedum ternatum Michx.	Woodland stonecrop	Forb/herb
Tiarella cordifolia L.	Heartleaf foamflower	Forb/herb
Trillium erectum L.	Red trillium	Forb/herb
Trillium grandiflorum (Michx.) Salisb.	White trillium	Forb/herb
Viola canadensis L.	Canadian white violet	Forb/herb
Acer saccharum Marshall	Sugar maple	Shrub, tree
Aesculus flava Aiton	Yellow buckeye	Shrub, tree
Cornus alternifolia L. f.	Alternateleaf dogwood	Shrub, tree
Cornus florida L.	Flowering dogwood	Shrub, tree
Prunus serotina Ehrh.	Black cherry	Shrub, tree
Acer rubrum L.	Red maple	Tree
Fagus grandifolia Ehrh.	American beech	Tree
Fraxinus americana L.	White ash	Tree
Liriodendron tulipifera L.	Tuliptree	Tree
Pinus strobus L.	Eastern white pine	Tree

Species	Common name	Growth habit
Quercus alba L.	White oak	Tree
Quercus rubra L.	Northern red oak	Tree
Quercus velutina Lam.	Black oak	Tree
Tilia americana L.	American basswood	Tree
Tsuga canadensis (L.) Carrière	Eastern hemlock	Tree

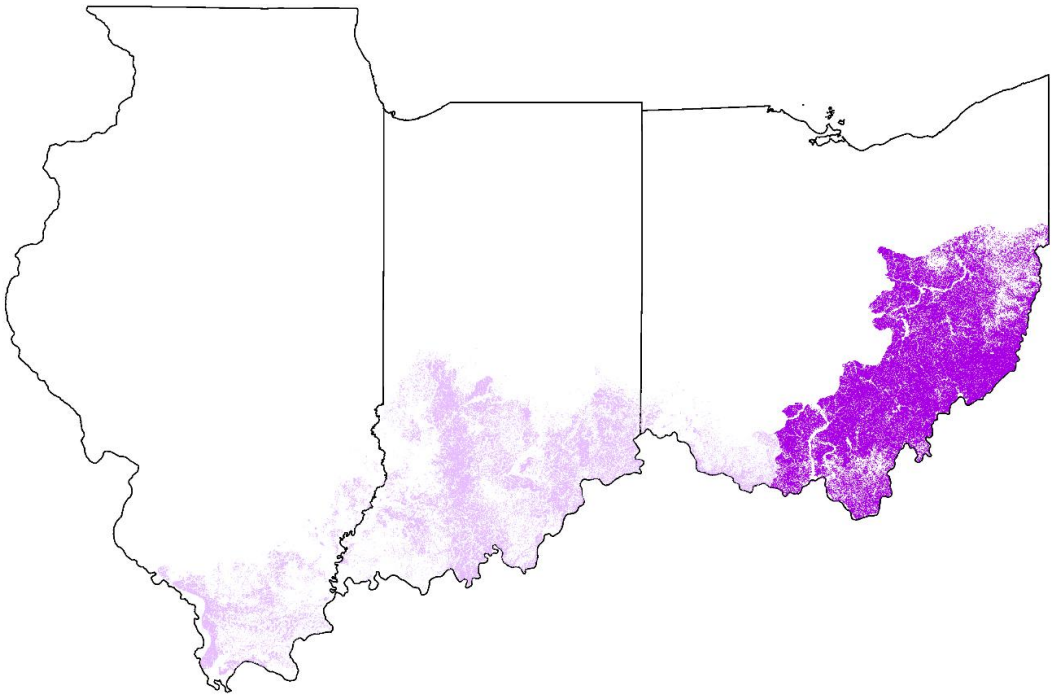

Figure 4. Appalachian-Central Interior Mesic Forest habitat distribution. Geographic distribution of the Appalachian-Central Interior Mesic Forest habitat (Group Code: G020) in the Midwest (USNVC, 2025). Dark purple areas indicate where the Appalachian-Central Interior Mesic Forest habitat is found in the Appalachian Forest ecoregion, while light purple areas mark additional locations throughout the Midwest. Polygon boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The Appalachian-Central Interior Mesic Forest habitat is a priority within the Appalachian Forest ecoregion. Both average and extreme temperatures are expected to rise throughout the coming century, with model projections indicating a 9.3°F increase by the end of the century under the high emissions scenario (RCP 8.5; Table 2) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). Most models also project an increase in precipitation (Table 3) (Marvel and others, 2023). However, precipitation patterns in the Midwest are expected to become more variable and seasonally stratified, with drier summers (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 2. Projected changes in annual temperature in the Appalachian Forest ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual mean temperature (°F) by mid- (2040–2069) and end of the century (2070–2099), as well as the absolute change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected temperature under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	51.8°F	53.7°F (1.9)	54.7°F (2.9)	54.3°F (2.5)	56.9°F (5.1)
Mean of models	52.0°F	56.3°F (4.4)	57.8°F (5.8)	57.2°F (5.3)	61.3°F (9.3)
Maximum model	51.7°F	57.8°F (6.1)	60.0°F (8.3)	59.8°F (8.1)	65.5°F (13.7)

Table 3. Projected changes in annual precipitation in the Appalachian Forest ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual precipitation (inches) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected precipitation under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	41.6 in	42.7 in (3%)	41.9 in (1%)	43.5 in (4%)	41.3 in (-1%)
Mean of models	41.4 in	43.5 in (5%)	43.8 in (6%)	43.3 in (4%)	44.7 in (8%)
Maximum model	40.4 in	47.6 in (18%)	43.7 in (8%)	47.9 in (19%)	47.9 in (19%)

Key Climate Change Stressors

Drought and mild winter. Across the Appalachian Forest ecoregion, 30-year extreme drought incidence is projected to increase by approximately 758% compared to historical conditions by the end of the century, with 29.6 months of extreme drought expected within a 30-year period (Table 4). This trend remains relatively consistent throughout the region (Figure 5). This dramatic increase in extreme drought frequency reflects a severe moisture deficit, which may heighten wildfire risk and intensify drought stress. Average winter minimum temperature is projected to increase by approximately 42% by the end of the century, approaching above-freezing levels at 31.6°F (Table 5). Slightly higher temperatures are expected near the southern and eastern borders of Ohio (Figure 6). Warmer winters may increase some aspects of cold exposure for overwintering species, as declining snowpack reduces insulation and fewer snow-covered days lead to earlier springs and more frequent freeze-thaw cycles.

Table 4. Projected changes in 30-year extreme drought incidence in the Appalachian Forest ecoregion. Projections use Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000) and projected 30-year extreme drought incidence (months) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	3.6	4.7 (32%)	8.8 (146%)	13.8 (284%)	11.4 (219%)
Mean of models	3.5	11.7 (238%)	15.8 (357%)	15.0 (335%)	29.6 (758%)
Maximum model	1.7	18.4 (1014%)	24.0 (1355%)	32.8 (1887%)	50.9 (2981%)

Table 5. Projected changes in winter minimum temperature in the Appalachian Forest ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected winter (December–February) minimum temperature (°F) by mid- (2040–2069) and end of the century (2070–2099), as well as the absolute change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected winter minimum temperature under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	22.6°F	25.1°F (2.5)	26.3°F (3.7)	25.5°F (3.0)	28.1°F (5.5)
Mean of models	22.2°F	27.0°F (4.8)	28.2°F (5.9)	27.8°F (5.5)	31.6°F (9.3)
Maximum model	21.7°F	28.6°F (6.9)	30.4°F (8.7)	29.5°F (7.9)	35.8°F (14.1)

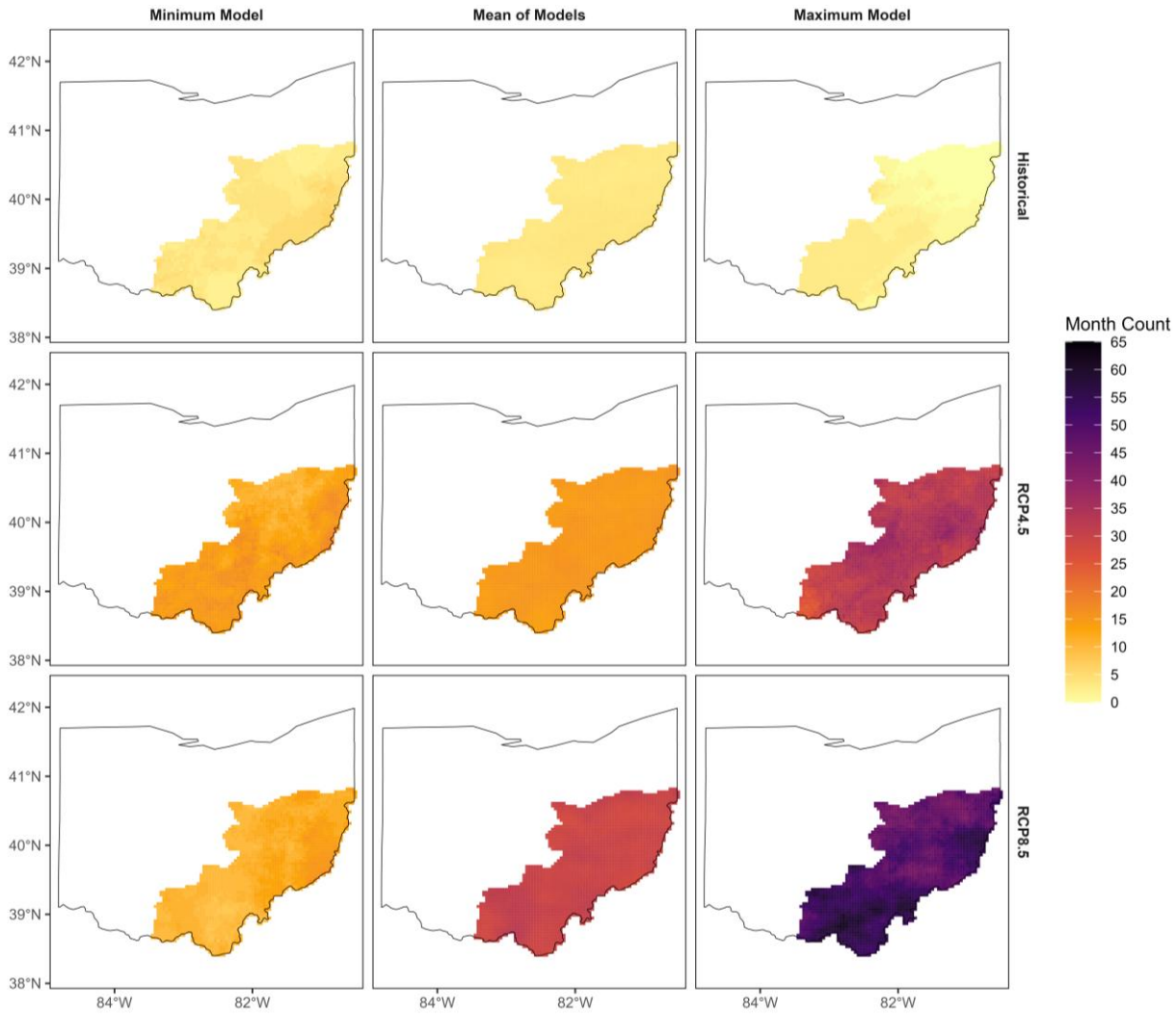

Figure 5. Projected changes in 30-year extreme drought incidence in the Appalachian Forest habitat. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000, top row) and projected (2070–2099) 30-year extreme drought incidence (months) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models (middle column).

Figure 6. Projected changes in winter minimum temperature in the Appalachian Forest habitat. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000, top row) and projected (2070–2099) winter (December–February) minimum temperature (°F) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected winter minimum temperature under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

This section summarizes how climate change is impacting the Appalachian-Central Interior Mesic Forest. It covers both the overall influence on species distributions, adaptive capacity, and vulnerability (Table 6) and the specific effects of drought and fire and mild winters (Table 7). The bullet points below provide a concise overview of these impacts on habitat structure, community composition, invasive species, pests and pathogens, and herbivory.

Habitat structure:

- Because these forests are often found on slopes and hillsides, they are vulnerable to erosion and nutrient loss, which may be exacerbated by drought and mild winters as altered vegetative and snow cover destabilize soils (Contosta and others, 2019; Vose and others, 2016; Wieder and others, 2022).
- Drought and mild winters may also alter community composition, driving shifts in soil biogeochemistry (e.g., mineralization rates, soil pH, decomposition, nutrient availability), which in turn further influences community composition due to its strong dependence on soil characteristics (Campbell and others, 2009; Palus and others, 2018; Small and Chamberlain, 2015; Tong and others, 2024; Woodbridge and others, 2022).

Community composition:

- Adaptations to cooler, wetter conditions may predispose these forests to climate vulnerability if climate change surpasses their buffering capacity, with the greatest impacts likely in forests extending beyond thermally sheltered areas.
- While some species may gain suitable habitat under climate change, several are likely to experience habitat loss (e.g., *Pinus strobus*, *Quercus rubra*, *Tsuga canadensis*) (Table 6).
- While limited adaptive capacity and heightened vulnerability may hinder the persistence of some species, many exhibit high adaptability and low vulnerability (Table 6).
- Drought may increase mortality and regeneration failure and reduce productivity, shifting canopy community composition toward drought-tolerant species (e.g., *Quercus* spp., *Pinus strobus*) (Table 7).
- Canopy coverage buffers understory herbs and saplings from drought, but increased drought-induced mortality may weaken this protection, negatively impacting drought-sensitive understory species (e.g., *Adiantum pedatum*, *Arisaema triphyllum*, *Botrychium virginianum*, *Caulophyllum thalictroides*, *Geranium maculatum*, *Impatiens pallida*) (Table 7) (Deng and others, 2023; Landuyt and others, 2019; Wang and others, 2023).
- Drought is expected to shorten fire return intervals and increase fire severity, causing higher mortality in fire-intolerant species that have proliferated through mesophication (e.g., *Acer* spp.), while favoring the persistence and expansion of fire-adapted species (e.g., *Quercus* spp., *Pinus strobus*) (Table 7).
- Mild winters may reduce snowpack insulation, increase freeze-thaw cycles, and heighten the risk of freeze and frost damage, leading to regeneration failure, mortality, and potential shifts in community composition, particularly for cold-sensitive species (e.g., *Aesculus flava*, *Fraxinus americana*, *Prunus serotina*, *Tilia americana*) (Table 7).
- While mild winters and longer growing seasons may increase productivity, many species require a minimum number of chilling days to break dormancy (e.g., *Fagus grandifolia*, *Liriodendron tulipifera*, *Tsuga canadensis*), and failing to meet this requirement could disrupt phenology, negatively impacting growth and reproduction (Table 7) (Giambuzzi and Richardson, 2024; Ma and others, 2021; Man and others, 2021; Nanninga and others, 2017).
- Some species may adapt their phenology to mild winters and shifting seasonal timing, but those that respond inappropriately could face regeneration failure, reduced competitive capacity, or trophic mismatches, such as early-season wildflowers being shaded out by asynchronous canopy

leaf-out (e.g., *Caulophyllum thalictroides*, *Osmorhiza claytonia*, *Sedum ternatum*) (Table 7) (Lee and others, 2024).

Invasive species:

- More frequent and severe droughts, coupled with increased fire severity, may facilitate invasive species colonization, particularly shade-intolerant and fire-tolerant species (e.g., *Ailanthus altissima*), by reducing vegetative cover and creating open habitat space (Arthur and others, 2021; Black and others, 2018; Elliott, 2014; Evans and others, 2016; Hanberry, 2022; Small and Chamberlain, 2015; Vose and others, 2019).
- Mild winters may allow invasive species at the northern edge of their range to expand (e.g., *Berberis thunbergii*) while facilitating the establishment of others previously limited by winter severity (e.g., *Pueraria montana*), increasing invasive species pressure and abundance (Bradley and others, 2010; O'Uhuru and others, 2024; Small and Chamberlain, 2015; USDA Forest Service, USA and others, 2019; Vose and others, 2019).
- Shifting and more variable seasonal timing from mild winters may favor invasive species over natives, as their broader germination niches and greater phenological plasticity give them a competitive advantage in earlier and longer growing seasons (Hicks and others, 2019; Wolkovich and others, 2013; Zettlemoyer and others, 2019).

Pests and pathogens:

- Climate change is expected to amplify pest and pathogen pressure by directly increasing their populations and indirectly heightening host susceptibility (Dukes and others, 2009; Jactel and others, 2012; Stanturf and others, 2024; Sturrock and others, 2011; Teshome and others, 2020; Vose and others, 2013a; Weed and others, 2013).
- Drought and fire can heighten host susceptibility, leading to increased outbreaks of pests such as emerald ash borer (*Agrilus planipennis*) and red oak borer (*Enaphalodes rufulus*), and pathogens like cankers (e.g., *Hypoxylon* spp., *Neonectria* spp.) and root rot (*Armillaria* spp.), resulting in higher mortality for drought-intolerant species (e.g., *Tsuga canadensis*) and placing otherwise drought-tolerant species (e.g., *Quercus* spp.) under amplified stress and potential decline (Canelles and others, 2021; Haavik and others, 2015; Hamann and others, 2021; Hubbart and others, 2016; Jactel and others, 2012; Novick and others, 2022; Ramsfield and others, 2016; Stump and others, 2024; Sturrock and others, 2011; Wood and others, 2018; Wyka and others, 2018).
- Mild winters are likely to increase the overwinter survival of pest and pathogens such as beech bark disease (*Cryptococcus fagisuga* and *Neonectria* spp.) and forest tent caterpillar (*Malacosoma disstria*); facilitate northward expansion of species already in the Midwest, such as the Asian long-horned beetle (*Anoplophora glabripennis*) and spongy moth (*Lymantria dispar*); and introduce novel species to the region, such as hemlock woolly adelgid (*Adelges tsugae*), southern pine beetle (*Dendroctonus frontalis*), and certain water molds (e.g., *Phytophthora* spp.) (Birch and others, 2023) (Burgess and others, 2017; Conrad and others, 2020; Contosta and others, 2019; Contreras-Cornejo and others, 2023; Ellison and others, 2018; Flower and Gonzalez-Meler, 2015; Kasson and Livingston, 2012; Lesk and others, 2017; Paradis and others, 2008a; Srivastava and others, 2020; Stephanson and Ribarik Coe, 2017; Zhou and others, 2021).
- Some aspects of climate change may be detrimental to certain pest and pathogens, as more frequent heatwaves and emerging phenological asynchrony with host species could reduce the impact of pests such as forest tent caterpillar (*Malacosoma disstria*) and spongy moth (*Lymantria*

dispar) (Abarca and Lill, 2015; Banahene and others, 2018; Garrett and others, 2016; Schwartzberg and others, 2014).

Herbivory:

- Mild winters may intensify browsing pressure across the Midwest by increasing white-tailed deer (*Odocoileus virginianus*) survival, population growth, and foliage accessibility (Champagne and others, 2021; Felton and others, 2024; Sabo and others, 2023; Sanders-DeMott and others, 2018; Weiskopf and others, 2019).
- As browse-preferred species decline (e.g., *Acer rubrum*, *Acer saccharum*, *Pinus strobus*, *Prunus serotina*, *Quercus* spp., *Trillium grandiflorum*, *Tilia americana*, *Tsuga canadensis*), browsing pressure may shift community composition toward browse-resistant species (e.g., *Arisaema triphyllum*, *Fagus grandifolia*, *Microstegium vimineum*) (Champagne and others, 2021; Dahlberg and Guettinger, 1956; Farinosi and Walters, 2023; Flinn and others, 2019; Frerker and others, 2014; Koon and others, 2024; Patton and others, 2021; Sabo and others, 2023; Sample and others, 2023; Wakeland and Swihart, 2009; Webster and others, 2008; White, 2012).
- Selective deer browsing may exacerbate climate-driven regeneration failure by suppressing climate-resilient species assemblages and indirectly facilitating recruitment of rarely browsed, drought- and fire-tolerant species (e.g., *Arisaema triphyllum*) or those sensitive to milder winter conditions (e.g., *Fagus grandifolia*) (Champagne and others, 2021; Frerker and others, 2014; Henry and Walters, 2023; Novick and others, 2022; Patton and others, 2021).

Table 6. Summary of the expected general influence of climate change on Appalachian-Central Interior Mesic Forest habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory columns (e.g., showing both negative and positive influences on distribution), reflecting conflicting findings in the literature.

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and/or vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity/ high vulnerability (–)	High adaptive capacity/ low vulnerability (+)
Acer rubrum[1]	*Acer rubrum*[5]	*Acer saccharum*[11]	*Acer rubrum*[13,17–19]
**Ailanthus altissima*[2]	*Acer saccharum*[9]	*Arisaema triphyllum*[12]	*Acer saccharum*[1,20]
Cornus florida[3]	**Alliaria petiolata*[10]	*Asarum canadense*[11]	*Adiantum pedatum*[17]
Fagus grandifolia[4]	*Cornus alternifolia*[3]	*Fagus grandifolia*[12]	*Aesculus flava*[21]
Pinus strobus[1,5,6]	*Fagus grandifolia*[1]	*Fraxinus americana*[12,13]	*Arisaema triphyllum*[17]
Prunus serotina[1,5]	**Lonicera maackii*[2]	*Liriodendron tulipifera*[14]	*Caulophyllum thalictroides*[17]
Quercus alba[3,5]	*Quercus alba*[1]	*Tilia americana*[15]	*Dennstaedtia punctilobula*[16]
Quercus rubra[1,5,7]	*Quercus velutina*[1,7]	*Trillium erectum*[16]	*Fagus grandifolia*[18]
Tilia americana[4]	*Tilia americana*[1,9]	*Tsuga canadensis*[12,13]	*Fraxinus americana*[21]
Tsuga canadensis[5,8]		*Viola canadensis*[16]	*Liriodendron tulipifera*[15]
			Pinus strobus[20]
			Quercus alba[21]

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and/or vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity/ high vulnerability (–)	High adaptive capacity/ low vulnerability (+)
			Quercus rubra[1,17,21,22]
			Quercus velutina[21]
			Tilia americana[17,21]
			Tsuga canadensis[12,13,15]

[1]Peters and others (2020); [2]Evans and others (2024); [3]Hanberry (2024); [4]Philips and others (2020); [5]Prasad and others (2020); [6]Joyce and Rehfeldt (2013); [7]Rauschendorfer and others (2022); [8]Dunckel and others (2017); [9]Knoot and others (2015); [10]Bradley and others (2010); [11]GLIFWC (2023); [12]Sansom (2020); [13]Handler and others (2014b); [14]LeBlanc and others (2020); [15]Potter and others (2017); [16]Molano-Flores and others (2019); [17]Inter-Tribal Council of Michigan (2016); [18]Royer-Tardif and others (2021); [19]Mortsch and others (2006); [20]Brinker and others (2018); [21]Rogers and others (2016); [22]Etterson and others (2020)

Table 7. Summary of impacts from key climate stressors on Appalachian-Central Interior Mesic Forest habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory columns (e.g., showing both negative and positive drought impacts), reflecting conflicting findings in the literature.

Stressor	Impact	Effected species
Drought	When drought tolerance is low, heightened susceptibility to disturbances, competitive exclusion, and mortality	*Acer rubrum*[1–7]
		Acer saccharum[2,5,6,8–17]
		Adiantum pedatum[7]
		Aesculus flava[7]
		**Alliaria petiolate*[18]
		Arisaema triphyllum[7]
		Asarum canadense[19]
		Botrychium virginianum[7]
		Caulophyllum thalictroides[7]
		Cornus alternifolia[7]
		Cornus florida[14,20]
		Fagus grandifolia[3,21–26]
		Fraxinus americana[7, 27]
		Geranium maculatum[28]
		Impatiens pallida[7]
		Liriodendron tulipifera[13]
		Pinus strobus[5,7,29]
		Prunus serotina[4,30]
		Quercus alba[31–36]
		Quercus rubra[5,6,12,17,34,35]

Stressor	Impact	Effected species
		Quercus velutina[7,35,37]
		Tilia americana[7]
		Tsuga canadensis[3,7,14,38–43]
	When drought tolerance is high, competitive advantage and increased abundance despite added stress	*Acer rubrum*[44]
		**Ailanthus altissima*[79]
		Fagus grandifolia[7]
		Pinus strobus[45]
		Prunus serotina[7,43]
		Quercus alba[2,13,35,45–51]
		Quercus rubra[2,51–54]
		Quercus velutina[51,54]
	When fire tolerance is low, heightened vulnerability to other stressors and mortality	*Acer rubrum*[7,55–57]
		Acer saccharum[56,57]
		Aesculus flava[43]
		Arisaema triphyllum[7]
		Caulophyllum thalictroides[7]
		Cornus alternifolia[7]
		Cornus florida[7,20]
		Fagus grandifolia[56–59]
		Fraxinus americana[43,57,60]
		Liriodendron tulipifera[56,57]
		Pinus strobus[7,56,61]
		Prunus serotina[7,43,56,62]
		Quercus rubra[52,57]
		Quercus velutina[7,63]
		Tilia americana[7,57,63]
		Tsuga canadensis[57,58]
	When fire tolerance is high, competitive advantage and increased abundance despite added stress	**Ailanthus altissima*[79]
		Botrychium virginianum[7]
		Fagus grandifolia[7]
		Pinus strobus[57]
		Prunus serotina[7]
		Quercus alba[35,56,57,65]
		Quercus rubra[7,35,43,56]
		Quercus velutina[57]
		Tilia americana[7]
Mild winters	When sensitivity to freeze and frost stress is high, heightened susceptibility to	*Acer rubrum*[66]
		Acer saccharum[66–71]

Stressor	Impact	Effected species
	damage, disturbances, reduced productivity, and regeneration failure	*Aesculus flava*[67]
		Fagus grandifolia[59]
		Fraxinus americana[43]
		Prunus serotina[43]
		Quercus alba[43]
		Quercus rubra[67,73]
		Tilia americana[74]
	When chilling requirements are unmet or phenological responses are inadequate, higher rates of trophic mismatches, competitive exclusion, and regeneration failure	*Acer saccharum*[7,75]
		Aesculus flava[7]
		Caulophyllum thalictroides[7]
		Cornus alternifolia[7]
		Cornus florida[7]
		Eurybia divaricate[76]
		Fagus grandifolia[7]
		Fraxinus americana[7]
		Laportea canadensis[77]
		Liriodendron tulipifera[7]
		Osmorhiza claytonii[78]
		Pinus strobus[7]
		Prunus serotina[7]
		Quercus rubra[7,12]
		Quercus velutina[7]
		Sedum ternatum[78]
		Tilia americana[7]
		Tsuga canadensis[7]

[1]Ibanez and others (2017); [2]McEwan and others (2011); [3]Niinemets and Vallardares (2006); [4]Ramage and others (2023); [5]Reich and others (2022); [6]Wang and others (2023); [7]USDA, NRCS (2025); [8]Au and others (2020); [9]Bapikee (2013); [10]Boakye and others (2023); [11]Fisichelli and others (2014); [12]Lee and Ibanez (2021); [13]Maxwell and others (2024); [14]Peters and others (2015); [15]Randall and Walters (2019); [16]Rudolph and LeBlanc (2020); [17]Stangler and others (2016); [18]Anderson and others (2021); [19]Augspurger (2017); [20]McLemore (1990); [21]Cohen (2005); [22]Caspersen and Kobe (2001); [23]Seeley and others (2019); [24] Stephanson and Ribarik Coe (2017); [25]U.S. Department of Agriculture, U.S. Forest Service (1965); [26]Booth and others (2012); [27]Marchin and others (2008); [28]Šajbidorová and others (2019); [29]Apfelbaum and others (2023); [30]Chhin (2018); [31]Dannenberg and others (2019); [32]Kellner and Swihart (2016); [33]LeBlanc and Berland (2019); [34]Lockwood and others (2023); [35]Novick and others (2022); [36]Au and Maxwell (2022); [37]Asbjornsen and others (2021); [38]Shuman and others (2019); [39]Parshall (1995); [40]Goerlich and Nyland (1980); [41]Godman and Lancaster (1990) ; [42]U.S. Department of Agriculture, Forest Service (1965); [43]Peters and others (2020); [44]Vaughn and others (2021); [45]Gustafson and Sturtevant (2013); [46]Duan and others (2022); [47]Hayford and Chin (2022); [48]Novick and others (2022); [49]Clark and others (2014); [50]Rogers (1990) [51]Brzostek and others (2014); [52]Sander (1990b); [53]Walters and others (2023); [54]Rauschendorfer and others (2022); [55]Walters and Yawney (1990); [56]Arthur and others (2021); [57]Hoving and others (2021); [58]Godman and Lancaster (1990); [59]Tubbs and Houston (1990); [60]Handler and others (2014a); [61]Wendel and Smith (1990); [62]Marquis (1990); [63]Sander (1990a);

[65]Hanberry and others (2012); [66]Sanders-DeMott and others (2018); [67]Augspurger (2009); [68]Comerford and others (2013); [69]Henne and others (2007); [70]Reinmann and others (2019); [71]Tatina (2015); [73]Lindback and others (2023); [74]Crow (1990); [75]Solarik and others (2016); [76]Bertin and others (2017); [77]Augspurger and Zaya (2020)2; [78]Abu-Asab and others (2001); [79]Evans and others (2016)

Focal Wildlife Species: Allegheny Woodrat

The Allegheny woodrat (*Neotoma magister*) is a rodent native to North America that inhabits rocky slopes, cliffs, bluffs, and outcrops at higher elevations. Its diet primarily consists of acorns and seeds from masting trees such as honeylocust (*Gleditsia triacanthos*) and eastern hemlock (*Tsuga canadensis*), along with leaves, twigs, fruits, and various other seeds (NatureServe, 2025). Historically, this species ranged from Connecticut in the north to Alabama in the south, but northern populations have declined rapidly, including extirpation from Connecticut. In most states where it is found, the Allegheny woodrat is listed as endangered or threatened, and in the Midwest states of Ohio and Indiana, it is listed as endangered due to declining and extirpated populations.

The decline of the Allegheny woodrat has been linked to raccoon roundworm (*Baylisascaris procyonis*) (Powers and others, 2025; Wolfkill and others, 2021); competition for food with growing populations of white-tailed deer (*Odocoileus virginianus*), black bears (*Ursus americanus*), and wild turkeys (*Meleagris gallopavo*) (Mengak and Castleberry, 2008); pest outbreaks such as spongy moth (*Lymantria dispar*); and mast failures (LoGiudice, 2008; Mengak and Castleberry, 2008). Climate change may further influence these threats by increasing the abundance and range of generalist species, including common raccoons (*Procyon lotor*), white-tailed deer, and black bears, which could intensify disease transmission and food competition (Kurth and others, 2024; Weiskopf and others, 2019). This pressure may be exacerbated if food sources become scarcer due to climate-driven changes in forest composition. For example, spongy moth outbreaks, which have already been implicated in woodrat declines, are expected to increase with climate change, as the biocontrol fungal disease that helps regulate spongy moth populations requires cool, moist conditions that may become less frequent (Liu and others, 2025). As eastern hemlock, a mast-producing species, faces increased vulnerability to climate change, its declines could impact Allegheny woodrat populations similarly to the past loss of American chestnut (*Castanea dentata*). Additionally, climate change could disrupt the reproductive synchrony of masting tree species. While this could initially benefit woodrats by increasing short-term seed availability, it may come at an overall reproductive cost to the trees, potentially altering forest dynamics and leading to unpredictable food shortages (Bogdziewicz and others, 2021). Adaptive management for Allegheny woodrats will likely include climate-adaptive forestry practices, mitigation of disease and pest impacts, and strategies to address food competition and shifting forest composition due to climate change.

Appalachian-Northeast Mesic Forest

The Appalachian-Northeast Mesic Forest habitat occurs across the northern central Appalachian Mountains, the High Allegheny Plateau, and portions of the northeastern United States, including the eastern margin of Ohio (Figure 7) (Anderson, 1982; Braun, 1967). In Ohio, these forests represent the northwestern edge of their range and are limited to a single representative plant association, the Central Appalachian White Pine - Eastern Hemlock Forest Association (USNVC, 2025). These forests typically occur on mesic to dry-mesic sites, occupying middle slopes, ravines, well-drained valley bottoms, and flat till and lacustrine deposits (Anderson 1982; NatureServe). Soils are generally acidic to neutral, infertile, and well-drained, often with a rocky or sandy substrate that retains moisture, except during severe droughts (Anderson, 1982; Braun, 1967; Faber-Langendoen and others, 2015; Faber-Langendoen and others, 2020). Forest canopies are closed and mixed-aged, with a sparse understory (Braun, 1967; Faber-Langendoen and others, 2015). Canopies are dominated almost exclusively by two species: eastern hemlock (*Tsuga canadensis*) and eastern white pine (*Pinus strobus*) (Table 8). However, other non-dominant species, including American beech (*Fagus grandifolia*), birch (*Betula* spp.), maples (*Acer* spp.), and oaks (*Quercus* spp.), also contribute to overall habitat diversity and function (Anderson, 1982; USNVC, 2025).

Natural disturbance processes are relatively uncommon and are mostly limited to windthrow, winter storm damage, and pest outbreaks, which drive gap-phase regeneration dynamics (Faber-Langendoen and others, 2015). However, these forests are increasingly impacted by pests and pathogens, particularly elongate hemlock scale (*Fiorinia externa*), hemlock woolly adelgid (*Adelges tsugae*), and white-pine blister rust (*Cronartium ribicola*), which are causing widespread mortality of dominant tree species and may be driving shifts in community composition (Braun, 1967; Faber-Langendoen and others, 2015; Fleming and others, 2021). These forests are also threatened by invasive species such as garlic mustard (*Alliaria petiolata*), and Japanese barberry (*Berberis thunbergii*).

Table 8. Plant species common to the Appalachian-Northeast Mesic Forest habitat. Taxonomic authority, common name, and growth habit are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Pinus strobus L.	Eastern white pine	Tree
Tsuga canadensis (L.) Carrière	Eastern hemlock	Tree

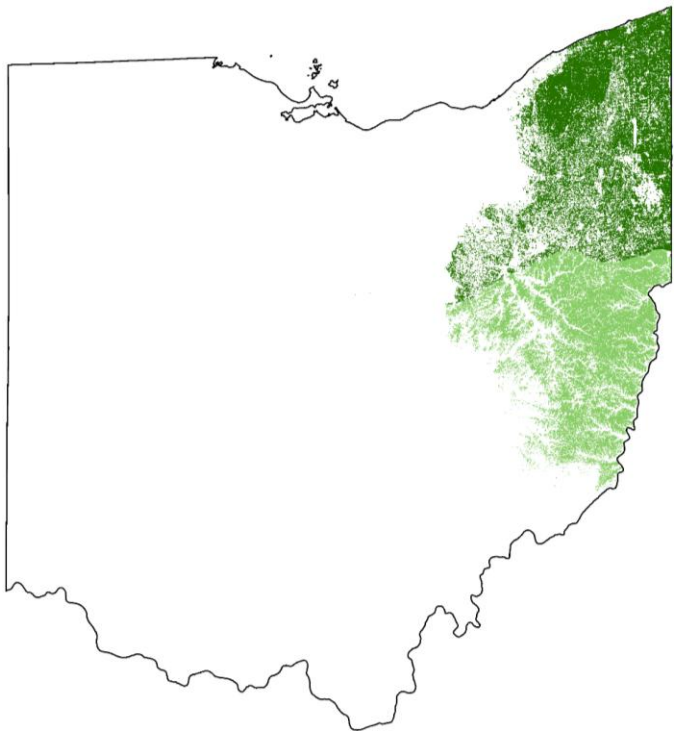

Figure 7. Appalachian-Northeast Mesic Forest habitat distribution. Geographic distribution of the Appalachian-Northeast Mesic Forest habitat (Group Code: G742) in the Midwest (USNVC, 2025). Dark green areas indicate where the Appalachian-Northeast Mesic Forest habitat is found in the Erie Drift Plains ecoregion, and light green areas mark additional locations throughout the Midwest. Polygon boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The Appalachian-Northeast Mesic Forest habitat is a priority within the Erie Drift Plains ecoregion. Both average and extreme temperatures are expected to rise throughout the coming century, with model projections indicating a 9.7°F increase by the end of the century under the high emissions scenario (RCP 8.5; Table 9) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). All models also project an increase in precipitation (Table 10) (Marvel and others, 2023). However, precipitation patterns in the Midwest are expected to become more variable and seasonally stratified, with drier summers (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 9. Projected changes in annual temperature in the Erie Drift Plains ecoregion ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual mean temperature (°F) by mid- (2040–2069) and end of the century (2070–2099), as well as the absolute change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected temperature under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	49.7°F	52.1°F (2.4)	53.2°F (3.4)	52.8°F (3.1)	55.6°F (5.8)
Mean of models	49.9°F	54.4°F (4.6)	55.9°F (6.1)	55.4°F (5.5)	59.6°F (9.7)
Maximum model	49.6°F	55.9°F (6.3)	58.1°F (8.4)	58.1°F (8.4)	63.8°F (14.2)

Table 10. Projected changes in annual precipitation in the Erie Drift Plains ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual precipitation (inches) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected precipitation under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	40.2 in	42.3 in (5%)	40.9 in (2%)	43.2 in (8%)	40.6 in (1%)
Mean of models	40.5 in	42.9 in (6%)	43.2 in (7%)	42.8 in (6%)	44.3 in (9%)
Maximum model	39.6 in	47.6 in (20%)	43.8 in (11%)	47.3 in (20%)	48.6 in (23%)

Key Climate Change Stressors

Drought and mild winters. Across the Erie Drift Plains ecoregion, 30-year extreme drought incidence is projected to increase by approximately 841% compared to historical conditions by the end of the century, with 31.1 months of extreme drought expected within a 30-year period (Table 11). This trend remains relatively consistent throughout the region (Figure 8). This dramatic increase in extreme drought frequency reflects a severe moisture deficit, which may heighten wildfire risk and intensify drought stress. Winter minimum temperature is projected to increase by approximately 50% by the end of the century, approaching above-freezing levels at 31.1°F (Table 12). Slightly higher temperatures are expected near the northern border of Ohio (Figure 9). Somewhat counterintuitively, warmer winters may increase cold exposure for overwintering species, as declining snowpack reduces insulation and fewer snow-covered days lead to earlier springs and more frequent freeze-thaw cycles.

Table 11. Projected changes in 30-year extreme drought incidence in the Erie Drift Plains ecoregion. Projections use Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000) and projected 30-year extreme drought incidence (months) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	3.4	14.1 (312%)	11.4 (232%)	19.3 (464%)	12.6 (266%)
Mean of models	3.3	11.5 (249%)	15.9 (380%)	15.5 (369%)	31.1 (841%)
Maximum model	0.9	14.7 (1628%)	19.4 (2171%)	33.6 (3838%)	51.1 (5894%)

Table 12. Projected changes in winter minimum temperature in the Erie Drift Plains ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected winter (December–February) minimum temperature (°F) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected winter minimum temperature under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	21.0°F	23.8°F (2.9)	25.0°F (4.1)	24.1°F (3.2)	26.9°F (6.0)
Mean of models	20.7°F	26.1°F (5.3)	27.4°F (6.6)	27.0°F (6.2)	31.1°F (10.4)
Maximum model	20.3°F	27.7°F (7.4)	29.7°F (9.5)	29.1°F (8.9)	36.0°F (15.7)

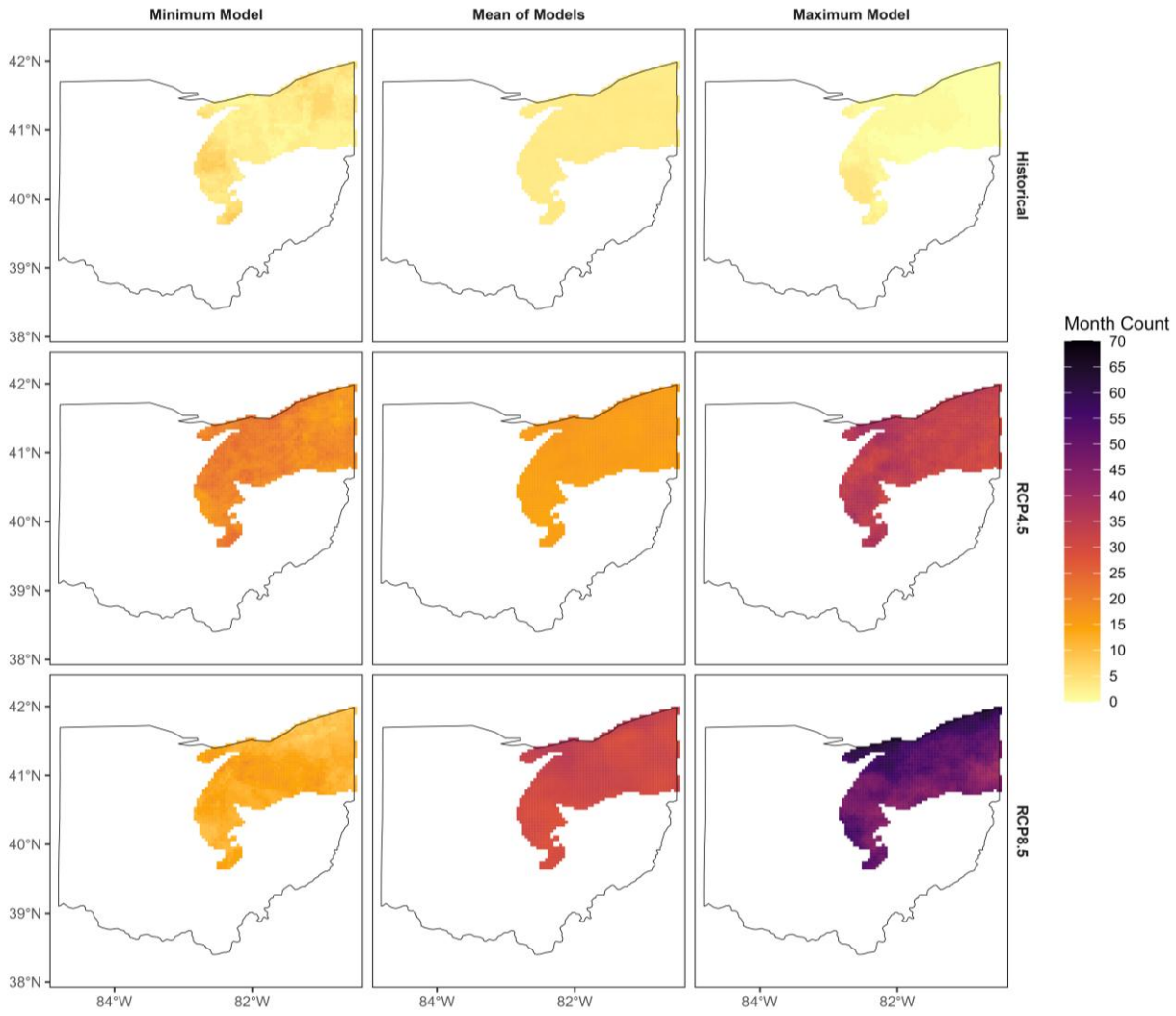

Figure 8. Projected changes in 30-year extreme drought incidence in the Erie Drift Plains ecoregion. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000, top row) and projected (2070–2099) 30-year extreme drought incidence (months) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models (middle column).

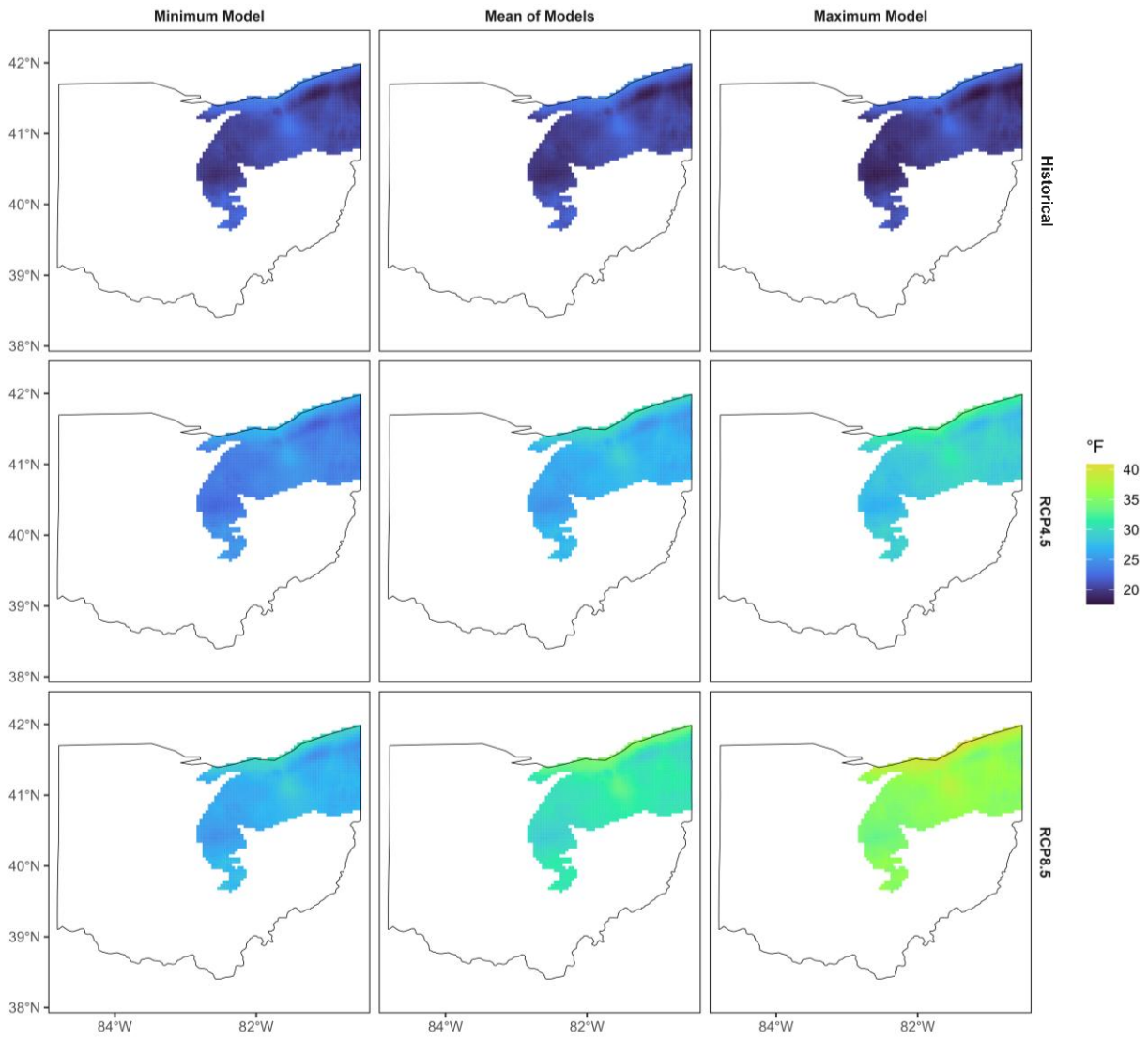

Figure 9. Projected changes in winter minimum temperature in the Erie Drift Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000, top row) and projected (2070–2099) winter (December–February) minimum temperature (°F) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected winter minimum temperature under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

This section summarizes how climate change is impacting the Appalachian-Northeast Mesic Forest habitat. It covers both the overall influence on species distributions, adaptive capacity, and vulnerability (Table 13) and the specific effects of drought and fire and mild winters (Table 14). The bullet points below provide a concise overview of these impacts on habitat structure, community composition, invasive species, pests and pathogens, and herbivory.

Habitat structure:

- Because these forests are often found on slopes and hillsides, they are vulnerable to erosion and nutrient loss, which may be exacerbated by drought and mild winters as altered vegetative and snow cover destabilize soils (Contosta and others, 2019; Vose and others, 2016; Wieder and others, 2022).
- Drought and mild winters may also alter community composition, driving shifts in soil biogeochemistry (e.g., mineralization rates, soil pH, decomposition, nutrient availability), which in turn further influence community composition in these forests adapted to acidic and nutrient-poor soils (Campbell and others, 2009; Palus and others, 2018; Small and Chamberlain, 2015; Tong and others, 2024; Woodbridge and others, 2022).

Community composition:

- Adaptations to cooler, wetter conditions may predispose these forests to climate vulnerability if climate change surpasses their buffering capacity, with the greatest impacts likely in forests extending beyond thermally sheltered areas.
- Both eastern hemlock (*Tsuga canadensis*) and eastern white pine (*Pinus strobus*), the two dominant species in the Appalachian-Northeast Mesic Forest, are likely to experience habitat loss under climate change (Table 13).
- While limited adaptive capacity and heightened vulnerability may hinder the persistence of eastern hemlock (*Tsuga canadensis*), eastern white pine (*Pinus strobus*) exhibits high adaptability and low vulnerability (Table 13).
- Drought may increase mortality and regeneration failure and reduce productivity, shifting canopy community composition toward drought-tolerant species (e.g., *Pinus strobus*) over drought-intolerant ones (e.g., *Tsuga canadensis*) (Table 14).
- Canopy gaps may increase thermal and drought stress for understory species, leading to decreased abundances of drought-intolerant species (Deng and others, 2023; Landuyt and others, 2019; Wang and others, 2023).
- While mild winters and longer growing seasons may increase productivity, both eastern hemlock (*Tsuga canadensis*) and eastern white pine (*Pinus strobus*) require a minimum number of chilling days to break dormancy, and failing to meet this requirement could disrupt phenology, negatively impacting growth and reproduction (Table 14) (Giambuzzi and Richardson, 2024; Ma and others, 2021; Man and others, 2021; Nanninga and others, 2017).
- Altered canopy dynamics and the loss of eastern hemlock (*Tsuga canadensis*) have already driven transitions into deciduous hardwood forests and rhododendron thickets in the eastern range of this habitat group, signaling a significant threat to its persistence at its western margin in the Midwest as climate change intensifies (Small and Chamberlain, 2015).

Invasive species:

- More frequent and severe droughts, coupled with increased fire severity, may facilitate invasive species colonization, particularly shade-intolerant and fire-tolerant species (e.g., *Ailanthus altissima*), by reducing vegetative cover and creating open habitat space (Arthur and others, 2021; Black and others, 2018; Elliott, 2014; Hanberry, 2022; Small and Chamberlain, 2015; Vose and others, 2019).

- Mild winters may allow invasive species at the northern edge of their range to expand (e.g., *Berberis thunbergii*) while facilitating the establishment of others previously limited by winter severity (e.g., *Pueraria montana*), increasing invasive species pressure and abundance (Bradley and others, 2010; Small and Chamberlain, 2015; USDA Forest Service, USA and others, 2019; Vose and others, 2019).
- Shifting and more variable seasonal timing from mild winters may favor invasive species over natives, as their broader germination niches and greater phenological plasticity give them a competitive advantage in earlier and longer growing seasons (Hicks and others, 2019; Wolkovich and others, 2013; Zettlemoyer and others, 2019).

Pests and pathogens:

- Climate change is expected to amplify pest and pathogen pressure by directly increasing their populations and indirectly heightening host susceptibility (Dukes and others, 2009; Jactel and others, 2012; Stanturf and others, 2024; Sturrock and others, 2011; Teshome and others, 2020; Vose and others, 2013a; Weed and others, 2013).
- Drought and fire can heighten host susceptibility, leading to increased outbreaks of pests such as hemlock woolly adelgid (*Adelges tsugae*) and pathogens like cankers (e.g., *Hypoxylon* spp., *Neonectria* spp.) and needle blight (*Dothistroma septosporum* and *D. pini*), resulting in higher mortality for drought-intolerant species (e.g., *Tsuga canadensis*) and creating a feedback loop where infestations further weaken trees, reducing their resistance to future water stress (Abella, 2018; Canelles and others, 2021; Domec and others, 2013; Hamann and others, 2021; Hickin and Preisser, 2015; Hubbart and others, 2016; Jactel and others, 2012; Ramsfield and others, 2016; Sturrock and others, 2011; Wood and others, 2018; Wyka and others, 2018).
- Mild winters are likely to increase the overwinter survival of pests and pathogens such as the pathogen complex that causes white pine needle damage (including *Bifusella linearis*, *Lecanosticta acicula*, *Lophophacidium dooksii*, and *Septorioides strobi*), facilitate northward expansion of species already in the Midwest, and introduce novel species to the region such as hemlock woolly adelgid (*Adelges tsugae*) (Cheah, 2017; Costanza and others, 2018; Dey and others, 2019; Ellison and others, 2018; Kasson and Livingston, 2012; Paradis and others, 2008b; Pureswaran and others, 2018; Stanturf and others, 2024; Stephanson and Ribarik Coe, 2017; Wyka and others, 2017; Wyka and others, 2018).
- Some aspects of climate change may be detrimental to certain pests and pathogens, as more frequent heatwaves and emerging phenological asynchrony with host species could reduce outbreak potential of pests, such as some pine-defoliating moths (e.g., *Panolis flammea*, *Dendrolimus pini*), even as others benefit from warming (Hanberry and others, 2024; Haynes and others, 2014; Pureswaran and others, 2018; Teshome and others, 2020).

Herbivory:

- Mild winters may intensify browsing pressure across the Midwest by increasing white-tailed deer (*Odocoileus virginianus*) survival, population growth, and foliage accessibility (Champagne and others, 2021; Felton and others, 2024; Sabo and others, 2023; Sanders-DeMott and others, 2018; Weiskopf and others, 2019).
- Both eastern hemlock (*Tsuga canadensis*) and eastern white pine (*Pinus strobus*), the two dominant species in the Appalachian-Northeast Mesic Forest, tend to be highly browse-preferred

(Dahlberg and Guettinger, 1956; Flinn and others, 2019; Frerker and others, 2014; Patton and others, 2021; White, 2012).

- Selective deer browsing may exacerbate climate-driven regeneration failure, promoting species assemblages poorly suited to future climate conditions (Champagne and others, 2021; Frerker and others, 2014; Henry and Walters, 2023; Patton and others, 2021).

Table 13. Summary of the expected general influence of climate change on Appalachian-Northeast Mesic Forest habitat species. Some species appear in contradictory columns (e.g., showing both negative and positive influences on distribution), reflecting conflicting findings in the literature.

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and/or vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity/ high vulnerability (–)	High adaptive capacity/ low vulnerability (+)
Pinus strobus[1–3]		*Tsuga canadensis*[5,6]	*Pinus strobus*[7]
Tsuga canadensis[2,4]			*Tsuga canadensis*[8]

[1]Peters and others (2020); [2]Prasad and others (2020); [3]Joyce and Rehfeldt (2013); [4]Dunckel and others (2017); [5]Handler and others (2014b); [6]Sansom (2020); [7]Brinker and others (2018); [8]Potter and others (2017)

Table 14. Summary of impacts from key climate stressors on Appalachian-Northeast Mesic Forest habitat species. Some species appear in contradictory columns (e.g., showing both negative and positive drought impacts), reflecting conflicting findings in the literature.

Stressor	Impact	Effected species
Drought	When drought tolerance is low, heightened susceptibility to disturbances, competitive exclusion, and mortality	*Pinus strobus*[1–4] *Tsuga canadensis*[3,5–12]
	When drought tolerance is high, competitive advantage and increased abundance despite added stress	*Pinus strobus*[13]
	When fire tolerance is low, heightened vulnerability to other stressors and mortality	*Pinus strobus*[3,14–16] *Tsuga canadensis*[17,10]
	When fire tolerance is high, competitive advantage and increased abundance despite added stress	*Pinus strobus*[17]
Mild winters	When sensitivity to freeze and frost stress is high, heightened susceptibility to damage, disturbances, reduced productivity, and regeneration failure	*Pinus strobus*[3] *Tsuga canadensis*[3]

[1]Apfelbaum and others (2023); [2]Reich and others (2022); [3]USDA, NRCS (2025); [4]Livingston and Kenefic (2018); [5]Niinemets and Vallardares (2006); [6]Peters and others (2015); [7]Shuman and others (2019); [8]Shuman and others

(2019); [9]Goerlich and Nyland (2000); [10]Goldman and Lancaster (1990); [11]Fowells (1965); [12]Peters and others (2020); [13]Gustafson and Sturtevant (2013); [14]Wendel and Smith (1990); [15]Arthur and others (2021); [16]Varner (2018); [17]Hoving and others (2021)

Focal Wildlife Species: American Woodcock

The American woodcock (*Scolopax minor*) is a migratory shorebird. Although inclement weather may increase migratory risk, including mortality due to building collisions (Loss and others, 2020), woodcocks exhibit considerable plasticity in migratory behavior, using local weather conditions to modify migratory timing (Meunier and others, 2008). The American woodcock is also a popular game bird, with 150,000–200,000 birds harvested each year (Seamans and Rau, 2023). Phenological shifts for many migratory birds have important implications for harvest opportunity. Under the current regulatory framework, harvest opportunity and migratory timing remain aligned in northern and southern wintering areas. In mid-latitudes, however, a shift in season opening could expand harvest opportunity for American woodcocks (Fish and others, 2024).

The species occupies young forests across the eastern U.S., which comprise both the breeding and nonbreeding range (McAuley and others, 2020). Soil type and condition (e.g., moisture) may be an important habitat feature, reflecting the abundance of earthworms (Oligochaeta), their primary food source (~75–80% of their diet) (McAuley and others, 2020). While drought and flooding may have negative effects on reproduction and survival, climate change favors expansion of the earthworm range, presenting new opportunities for woodcocks (Mathieu and others, 2022; Singh and others, 2019). In response to population declines, there are extensive efforts to expand woodcock habitat (Kelley and others, 2008). American woodcocks' observed capability to rapidly utilize and proliferate in recently created young forest stands demonstrates their overall high adaptive capacity (McAuley and others, 2020). Given the potential for amplified forest successional processes and higher rates of forest turnover due to climate change, American woodcocks likely have a low vulnerability to climate change within Appalachian-Northeast Mesic Forest habitats.

Focal Aquatic System: Grand River (Ohio)

The Grand River, a major tributary of Lake Erie in northeastern Ohio, spans approximately 102 miles and flows through a diverse watershed composed of forests, wetlands, agricultural lands, and urban areas (Ohio EPA, 2024). Designated as Ohio's second Wild and Scenic River in 1974, its hydrology is influenced by glacial physiography, groundwater-fed tributaries, and seasonal precipitation, all of which help regulate stream temperatures and maintain water quality (Ohio EPA, 2009; Ohio EPA, 2024). The upper reaches of the river feature low-gradient, groundwater-fed streams, supporting cold-water species such as river darter (*Percina shumardi*) and brook trout (*Salvelinus fontinalis*) (Ohio EPA, 2009). Downstream from Mechanicsville, the river transitions through swamp-wetland habitats into a shale gorge, providing habitat for a diverse fish community, including trout-perch (*Percopsis omiscomaycus*) and sunfish (*Lepomis* spp.) (Ohio EPA, 2006). This section is also home to the only self-sustaining population of Great Lakes muskellunge (*Esox masquinongy*) among Ohio's Lake Erie tributaries, alongside other top predators, such as walleye (*Sander vitreus*) and northern pike (*Esox lucius*) (Ohio EPA, 2006). The upper portion of the Grand River features habitats ranging from fine sediments in the headwaters to varied bedrock and glacial till formations downstream, with high water quality and natural barriers that support some of Ohio's highest quality aquatic habitats (Ohio EPA, 2006). Climate change, however, poses a significant threat to the river's aquatic habitats and the species that rely on them.

Climate change is expected to impact the Grand River through rising water temperatures, altered streamflow patterns, and increased flood and drought variability (Allan and others, 2021; Byun and others, 2019; Scholl and others, 2025). Projected increases in extreme precipitation events will likely exacerbate erosion, nutrient runoff, and habitat instability, particularly in the lower portion of the Grand River, where urbanization amplifies the impact of stormwater (Byun and others, 2019; Ohio EPA, 2006; Scholl and others, 2025). More frequent summer droughts may reduce habitat availability for cold-water and flow-sensitive species, leading to localized declines in thermally sensitive populations (Scholl and others, 2025; Sievert and others, 2022). Winter hydrology is also expected to change significantly, with reduced ice cover, earlier snowmelt, and increased winter streamflow (Table 15) (Byun and others, 2019; Lamichhane and others, 2022; Scholl and others, 2025). These changes could alter sediment transport, disrupt spawning cycles, and increase stress on fish populations (Byun and others, 2019; Costa and others, 2021).

Table 15. Projected changes in streamflow for Grand River. Projections use gridded simulations of daily runoff based on CMIP5 climate models under the high-emissions (RCP 8.5) scenario. Values show average historical (1971–2000) and projected (2070–2099) streamflow metrics, as well as the percent change over this timeframe. Data and metric definitions are provided by the U.S Stream Flow Metric Dataset (Luce and others, 2025; Wenger et al. 2010; U.S. Department of the Interior, Bureau of Reclamation 2014; USDA Forest Service Office of Sustainability and Climate 2022).

Streamflow metric	1977–2006	2070–2099	Percent change
Mean Winter (December–February) Flow	644 ft^3/s	735 ft^3/s	14%
Mean Spring (March–May) Flow	730 ft^3/s	822.8 ft^3/s	13%
Mean Summer (June–August) Flow	288 ft^3/s	258 ft^3/s	-10%
Mean Fall (September–November) Flow	319 ft^3/s	331 ft^3/s	4%
Mean Annual Flow	495 ft^3/s	536 ft^3/s	8%
1.5-year Flood (33rd percentile)	1705 ft^3/s	1993 ft^3/s	17%
10-year Flood (90th percentile)	3766 ft^3/s	4349 ft^3/s	15%
25-year Flood (96th percentile)	4537 ft^3/s	5013 ft^3/s	10%
Number of Winter Floods	10.3	9.7	-6%
Average Lowest 7-day Flow	136 ft^3/s	119 ft^3/s	-13%
Center of Flow Mass Timing	March 18	March 15	-2%

North-Central Beech - Maple - Basswood Forest

The North-Central Beech - Maple - Basswood Forest habitat occurs throughout the Midwest, extending from eastern Ohio and southern Michigan to northern Illinois, southern Wisconsin, and into parts of Iowa, Missouri, and Minnesota (Figure 10) (Braun, 1967; Menard, 2015; USNVC, 2025). This habitat group is most commonly found on well-drained mesic-moist to mesic-dry sites along rolling uplands, valley slopes, and bottomlands, often with northern or eastern aspects, and fertile, nutrient-rich loamy soils (Dodge and Harman, 1985; Faber-Langendoen and others, 2020; Kost and others, 2007; Menard, 2015; Minnesota Natural Heritage Program (MNNHP), 1993). These cooler, moist microclimates distinguish these forests from often adjacent oak-hickory communities (Kost and others, 2007; Menard, 2015). The forest canopy is typically dense to moderately dense and is dominated by a mix of deciduous species that vary regionally, influenced by factors such as soil fertility, drainage, disturbance regimes, and climate (Butler and others, 2015; Faber-Langendoen and others, 2020; Kost and others, 2007; Menard, 2015). The light-limited understory typically has a sparse shrub layer but supports a well-developed and diverse herbaceous layer, rich in spring ephemerals that take advantage of seasonal light availability (Busing, 1998; Cohen, 2004; Curtis, 1959; Kost and others, 2007; Menard, 2015). These forests are typically characterized by fifteen dominant plant species (Table 16), though other non-dominant species contribute significantly to overall diversity and function.

These communities rarely experience stand-wide disturbances. Instead, small-scale windthrow, winter storms, and gap-phase regeneration are the primary natural disturbance mechanisms, creating mixed-aged patches (Busing, 1998; Canham and Loucks, 1984; Cohen, 2004; Kost and others, 2007; Runkle, 1991). Larger windstorms, ice storms, and occasional fires contributed to forest dynamics in the past but were less frequent compared to forests dominated by oaks (*Quercus* spp.) (Daubenmire, 1936; Nowacki and others, 1990). Most dominant species are fire-intolerant, and fire suppression over the past century has facilitated mesophication, leading some oak-dominated stands to transition into mesic forests resembling the North-Central Beech - Maple - Basswood Forest habitat, particularly in the western regions of the Midwest (Butler and others, 2015; Faber-Langendoen and others, 2020; Menard, 2015; Nowacki and others, 1990).

Human activities, including agriculture, logging, and urban development, have dramatically reduced the extent of the North-Central Beech - Maple - Basswood Forest habitat, fragmenting once-extensive areas into small remnant patches (Butler and others, 2015; Faber-Langendoen and others, 2020; Menard, 2015). Additionally, a variety of invasive plant species, such as common buckthorn (*Rhamnus cathartica*), garlic mustard (*Alliaria petiolata*), Japanese barberry (*Berberis thunbergii*), and tree-of-heaven (*Ailanthus altissima*), have altered understory composition and hindered native regeneration (New York Natural Heritage Program, 2025). The future of these forests will also likely be increasingly shaped by herbivory patterns and a growing number of pests and pathogens (Augustine and Frelich, 1998; Brandt and others, 2014; Butler and others, 2015; Cohen, 2004; Kost and others, 2007).

Table 16. Plant species common to the North-Central Beech - Maple - Basswood Forest habitat. Taxonomic authority, common name, and growth habit are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Acer saccharum Marshall	Sugar maple	Shrub, tree
Ostrya virginiana (Mill.) K. Koch	Hophornbeam	Shrub, tree
Prunus serotina Ehrh.	Black cherry	Shrub, tree

Species	Common name	Growth habit
Prunus virginiana L.	Chokecherry	Shrub, tree
Quercus macrocarpa Michx.	Bur oak	Shrub, tree
Fagus grandifolia Ehrh.	American beech	Tree
Liriodendron tulipifera L.	Tuliptree	Tree
Quercus rubra L.	Northern red oak	Tree
Tilia americana L.	American basswood	Tree

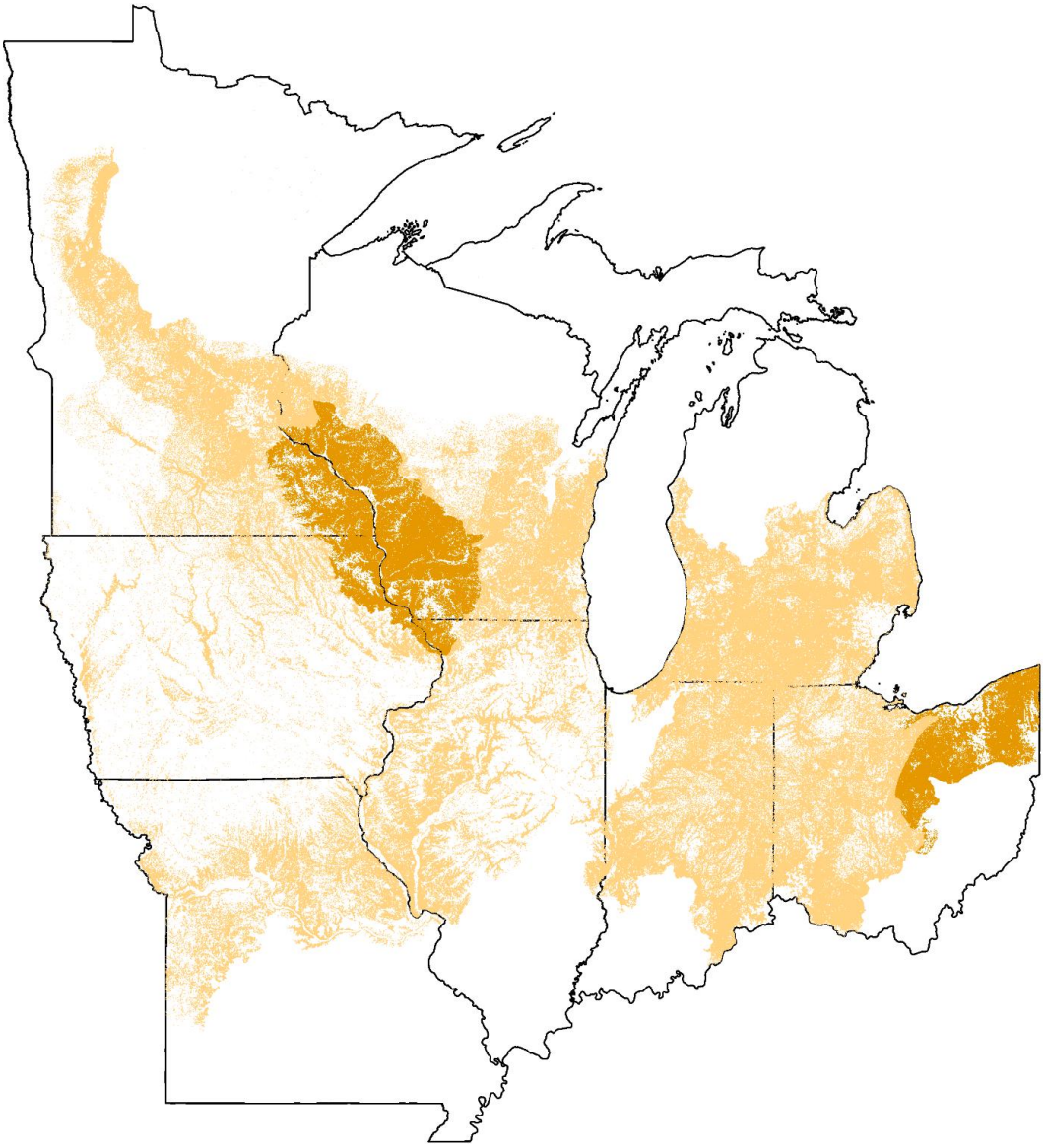

Figure 10. North-Central Beech-Maple-Basswood Forest habitat distribution. Geographic distribution of the North-Central Beech - Maple - Basswood Forest habitat (Group Code: G021) in the Midwest (USNVC, 2025). Dark orange areas indicate where North-Central Beech - Maple - Basswood Forest habitat is found in the Erie Drift Plains and Driftless Area ecoregions, and light orange areas mark additional locations throughout the Midwest. Polygon boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The North-Central Beech - Maple - Basswood Forest habitat is a priority within the Driftless Area and Erie Drift Plains ecoregions. In the Driftless Area ecoregion, both average and extreme temperatures are expected to rise throughout the coming century, with model projections indicating a 10.7°F increase by the end of the century under the high emissions scenario (RCP 8.5; Table 17) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). All models also project an increase in precipitation (Table 18) (Marvel and others, 2023). In the Erie Drift Plains ecoregion, both average and extreme temperatures are again expected to rise throughout the coming century, with model projections indicating a 9.7°F increase by the end of the century under the high emissions scenario (Table 19) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). Most models also project an increase in precipitation (Table 20) (Marvel and others, 2023). However, in both ecoregions, precipitation patterns in the Midwest are expected to become more variable and seasonally stratified, with drier summers (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2022).

Table 17. Projected changes in annual temperature in the Driftless Area ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual mean temperature (°F) by mid- (2040–2069) and end of the century (2070–2099), as well as the absolute change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected temperature under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	45.9°F	48.2°F (2.4)	49.2°F (3.4)	49.1°F (3.2)	51.8°F (6.0)
Mean of models	45.9°F	51.0°F (5.1)	52.6°F (6.6)	52.2°F (6.2)	56.6°F (10.7)
Maximum model	45.8°F	52.5°F (6.6)	54.5°F (8.7)	54.9°F (9.0)	60.9°F (15.1)

Table 18. Projected changes in annual precipitation in the Driftless Area ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual precipitation (inches) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected precipitation under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	34.5 in	35.4 in (2%)	34.5 in (0%)	37.9 in (10%)	32.6 in (-6%)
Mean of models	34.3 in	36.3 in (6%)	36.5 in (6%)	36.0 in (5%)	37.2 in (8%)
Maximum model	34.6 in	36.2 in (5%)	35.3 in (2%)	38.3 in (11%)	40.4 in (17%)

Table 19. Projected changes in annual temperature in the Erie Drift Plains ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual mean temperature (°F) by mid- (2040–2069) and end of the century (2070–2099), as well as the absolute change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected temperature under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	49.7°F	52.1°F (2.4)	53.2°F (3.4)	52.8°F (3.1)	55.6°F (5.8)
Mean of models	49.9°F	54.4°F (4.6)	55.9°F (6.1)	55.4°F (5.5)	59.6°F (9.7)
Maximum model	49.6°F	55.9°F (6.3)	58.1°F (8.4)	58.1°F (8.4)	63.8°F (14.2)

Table 20. Projected changes in annual precipitation in the Erie Drift Plains ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected annual precipitation (inches) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected precipitation under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	40.2 in	42.3 in (5%)	40.9 in (2%)	43.2 in (8%)	40.6 in (1%)
Mean of models	40.5 in	42.9 in (6%)	43.2 in (7%)	42.8 in (6%)	44.3 in (9%)
Maximum model	39.6 in	47.6 in (20%)	43.8 in (11%)	47.3 in (20%)	48.6 in (23%)

Key Climate Change Stressors

Drought and mild winters. Across the Driftless Area ecoregion, 30-year extreme drought incidence in projected to increase by approximately 847% compared to historical conditions by the end of the century, with 32.4 months of extreme drought expected within a 30-year period (Table 21). This trend remains relatively consistent throughout the region (Figure 11). This dramatic increase in extreme drought frequency reflects a severe moisture deficit, which may heighten wildfire risk and intensify drought stress. Winter minimum temperature is projected to increase by approximately 119% by the end of the century, with temperatures reaching 24.1°F (Table 22). Slightly higher temperatures are expected near the border region where Wisconsin, Illinois, and Iowa converge (Figure 12). Somewhat counterintuitively, warmer winters may increase cold exposure for overwintering species, as declining snowpack reduces insulation and fewer snow-covered days lead to earlier springs and more frequent freeze-thaw cycles.

In the Erie Drift Plains ecoregion, 30-year extreme drought incidence is projected to increase by approximately 841% compared to historical conditions by the end of the century, with 31.1 months of extreme drought expected within a 30-year period (Table 23). This trend remains relatively consistent throughout the region (Figure 13). Like with the Driftless Area ecoregion, this dramatic increase in extreme drought frequency reflects a severe moisture deficit, which may heighten wildfire risk and intensify drought stress. The growing season is projected to lengthen by approximately 25% by the end of the century, extending by 46 days (Table 24). Slightly longer growing seasons are expected near the northern border of Ohio (Figure 14). The extended growing season is likely to alter species phenology, with consequences for survival, reproduction, and community composition.

Table 21. Projected changes in 30-year extreme drought incidence in the Driftless Area ecoregion. Projections use Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000) and projected 30-year extreme drought incidence (months) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	3.6	6.9 (90%)	3.9 (7%)	10.6 (192%)	13.7 (279%)
Mean of models	3.4	12.8 (274%)	16.9 (396%)	17.6 (416%)	32.4 (847%)
Maximum model	5.0	14.8 (194%)	19.1 (279%)	15.3 (205%)	59.2 (1076%)

Table 22. Projected changes in winter minimum temperature in the Driftless Area ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected winter (December–February) minimum temperature (°F) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected winter minimum temperature under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	11.2°F	15.2°F (35%)	16.8°F (49%)	15.6°F (39%)	18.8°F (68%)
Mean of models	11.0°F	17.8°F (62%)	19.4°F (76%)	18.8°F (71%)	24.1°F (119%)
Maximum model	10.9°F	19.3°F (77%)	21.2°F (94%)	20.4°F (87%)	29.0°F (167%)

Table 23. Projected changes in 30-year extreme drought incidence in the Erie Drift Plains ecoregion. Projections use Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000) and projected 30-year extreme drought incidence (months) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	3.4	14.1 (312%)	11.4 (232%)	19.3 (464%)	12.6 (266%)
Mean of models	3.3	11.5 (249%)	15.9 (380%)	15.5 (369%)	31.1 (841%)
Maximum model	0.9	14.7 (1628%)	19.4 (2171%)	33.6 (3838%)	51.1 (5894%)

Table 24. Projected changes in growing season length in the Erie Drift Plains ecoregion. Projections use data obtained via the Climate Toolbox Climate Mapper (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000) and projected growing season length (consecutive days with minimum temperatures above freezing) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected growing season length under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	177.7 d	190.8 d (7%)	201.2 d (13%)	188.2 d (6%)	205.3 d (16%)
Mean of models	180.9 d	204.7 d (13%)	213.8 d (18%)	201.7 d (11%)	226.9 d (25%)
Maximum model	184.8 d	225.7 d (22%)	231.1 d (25%)	222.8 d (21%)	253.2 d (37%)

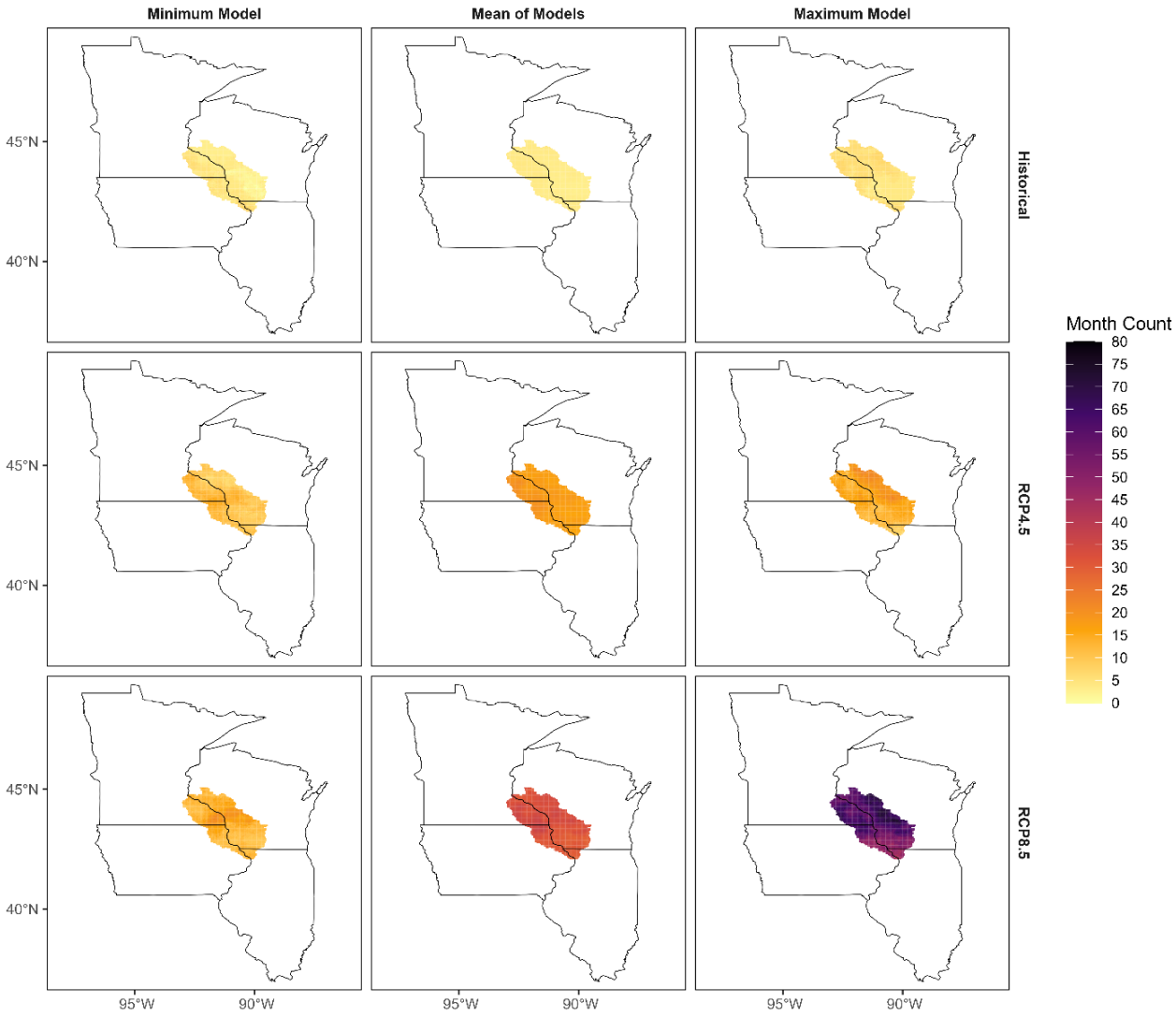

Figure 11. Projected changes in 30-year extreme drought incidence in the Driftless Area ecoregion. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000, top row) and projected (2070–2099) 30-year extreme drought incidence (months) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models (middle column).

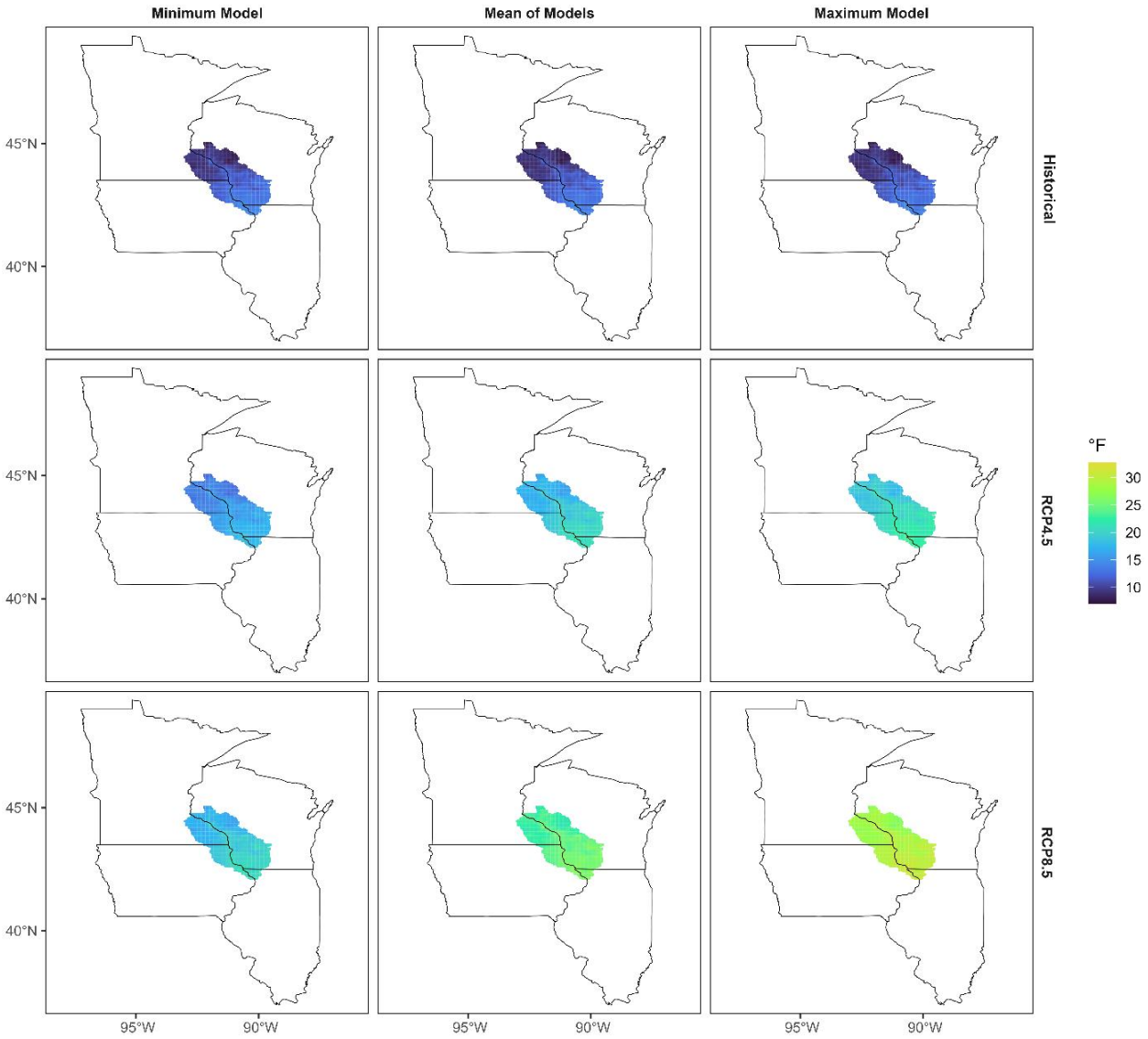

Figure 12. Projected changes in minimum winter temperature in the Driftless Area ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou and Brown 2012; Abatzoglou 2013). Values show average historical (1971–2000, top row) and projected (2070–2099) winter (December–February) minimum temperature (°F) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected winter minimum temperature under RCP 8.5 are presented in addition to the mean of all models (middle column).

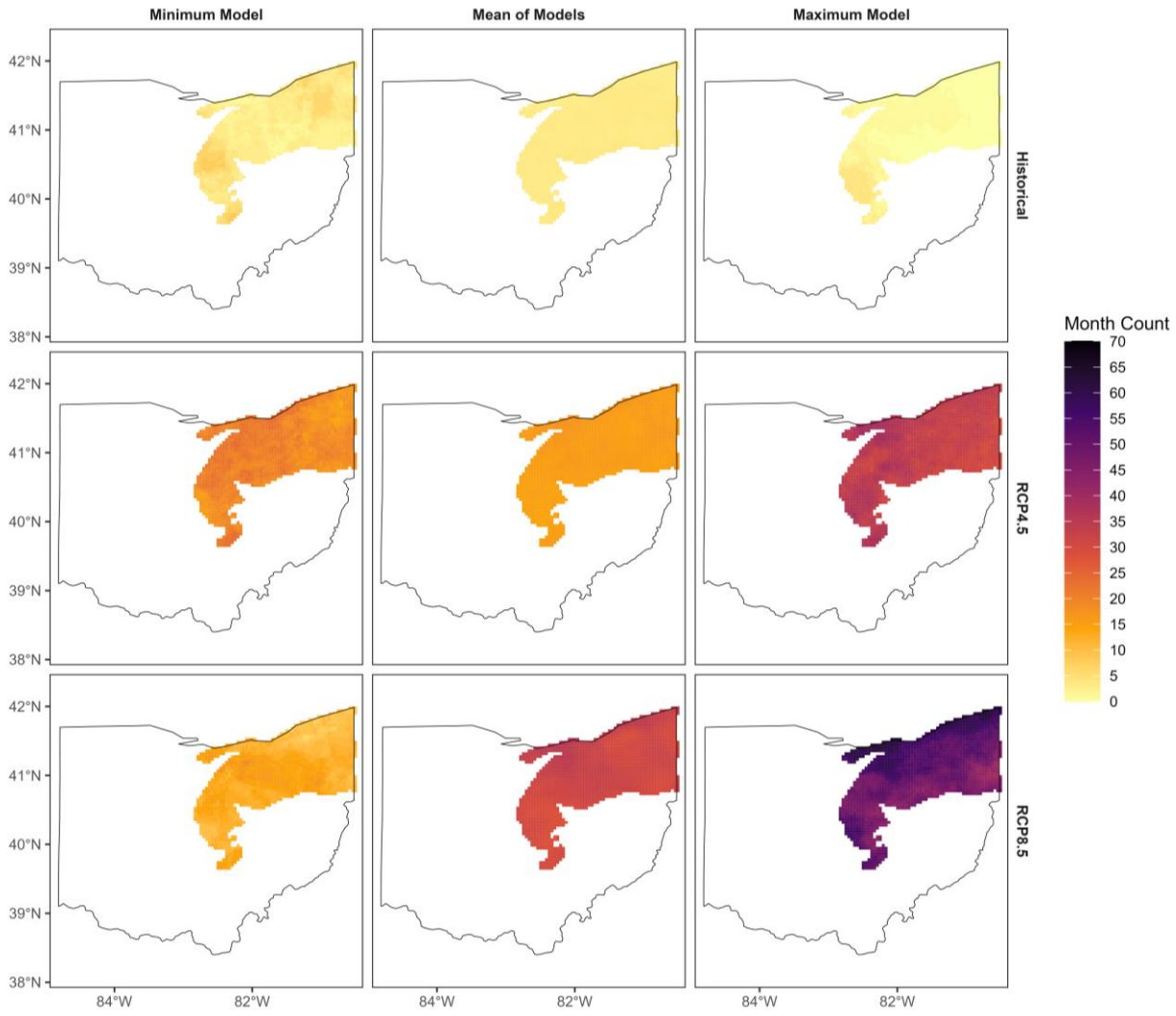

Figure 13. Projected changes in 30-year extreme drought incidence in the Erie Drift Plains ecoregion. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Thirty-year extreme drought incidence is defined as the number of months within a 30-year period with an SPEI value of ≤ -2, corresponding to the driest 2.5% of months relative to a reference period (1981–2020) under RCP 4.5 (Li and others, 2015; McKee and others, 1993). Values show average historical (1971–2000, top row) and projected (2070–2099) 30-year extreme drought incidence (months) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models (middle column).

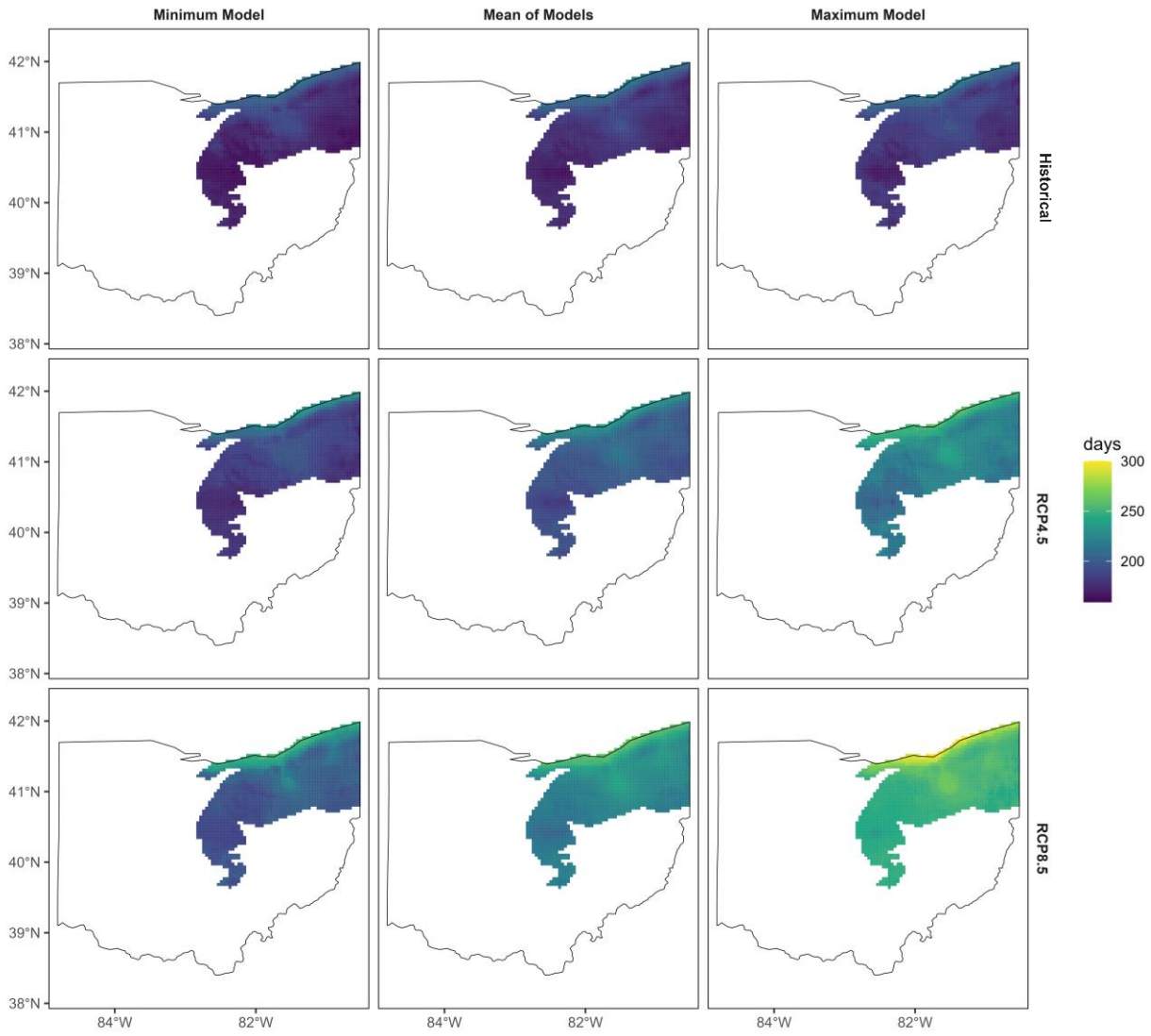

Figure 14. Projected changes in annual growing season length in the Erie Drift Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000, top row) and projected (2070–2099) annual growing season length (days) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected annual growing season length under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

This section summarizes how climate change is impacting the North-Central Beech - Maple - Basswood Forest habitat. It covers both the overall influence on species distributions, adaptive capacity, and vulnerability (Table 25) and the specific effects of drought and fire and mild winters (Table 26). The bullet points below provide a concise overview of these impacts on habitat structure, community composition, invasive species, pests and pathogens, and herbivory.

Habitat structure:

- Because these forests are often found on slopes and hillsides, they are vulnerable to erosion and nutrient loss, which may be exacerbated by drought and mild winters as altered vegetative and snow cover destabilize soils (Contosta and others, 2019; Vose and others, 2016; Wieder and others, 2022).
- Drought and mild winters may also alter community composition, driving shifts in soil biogeochemistry (e.g., mineralization rates, soil pH, decomposition, nutrient availability), which in turn further influences community composition due to its strong dependence on soil characteristics (Campbell and others, 2009; Palus and others, 2018; Small and Chamberlain, 2015; Tong and others, 2024; Woodbridge and others, 2022).

Community composition:

- Adaptations to cooler, wetter conditions may predispose these forests to climate vulnerability if climate change surpasses their buffering capacity, with the greatest impacts likely in forests extending beyond thermally sheltered areas.
- The Driftless Area, already drier than other parts of this habitat group, may be particularly vulnerable to climate change, whereas eastern regions like the Erie Drift Plains may experience short-term buffering from increased lake-effect snowfall due to declining Great Lakes ice cover (Swanston and others, 2018; Wright and others, 2013).
- While some species may gain suitable habitat under climate change, many are likely to experience habitat loss (e.g., *Liriodendron tulipifera*, *Ostrya virginiana*, *Tilia americana*) (Table 25).
- While limited adaptive capacity and heightened vulnerability may hinder the persistence of some species, many exhibit high adaptability and low vulnerability (Table 25).
- Drought may increase mortality and regeneration failure and reduce productivity, shifting canopy community composition toward drought-tolerant species (e.g., *Quercus* spp.) (Table 26).
- Canopy coverage buffers understory herbs and saplings from drought, but increased drought-induced mortality may weaken this protection, negatively impacting drought-sensitive understory species (Table 26) (Deng and others, 2023; Landuyt and others, 2019; Wang and others, 2023).
- Drought is expected to shorten fire return intervals and increase fire severity, causing higher mortality in fire-intolerant species that have proliferated through mesophication (e.g., *Acer saccharum*), while favoring the persistence and expansion of fire-adapted species (e.g., *Quercus macrocarpa*) (Table 26).
- Mild winters may reduce snowpack insulation, increase freeze-thaw cycles, and heighten the risk of freeze and frost damage, leading to regeneration failure, mortality, and potential shifts in community composition, particularly for cold-sensitive species (e.g., *Acer saccharum*) (Table 26).
- While mild winters and longer growing seasons may increase productivity, many species require a minimum number of chilling days to break dormancy (e.g., *Fagus grandifolia*), and failing to meet this requirement could disrupt phenology, negatively impacting growth and reproduction (Table 26) (Giambuzzi and Richardson, 2024; Ma and others, 2021; Man and others, 2021; Nanninga and others, 2017).
- Some species may adapt their phenology to mild winters and shifting seasonal timing, but those that respond inappropriately could face regeneration failure, reduced competitive capacity, or

trophic mismatches, such as early-season wildflowers being shaded out by asynchronous canopy leaf-out (Lee and others, 2024).

Invasive species:

- More frequent and severe droughts, coupled with increased fire severity, may facilitate invasive species colonization, particularly shade-intolerant and fire-tolerant species (e.g., *Ailanthus altissima*), by reducing vegetative cover and creating open habitat space (Arthur and others, 2021; Black and others, 2018; Elliott, 2014; Hanberry, 2022; Small and Chamberlain, 2015; Vose and others, 2019).
- Mild winters may allow invasive species at the northern edge of their range to expand while facilitating the establishment of others previously limited by winter severity (e.g., *Pueraria montana*), increasing invasive species pressure and abundance (Bradley and others, 2010; Small and Chamberlain, 2015; USDA Forest Service, USA and others, 2019; Vose and others, 2019).
- Shifting and more variable seasonal timing from mild winters may favor invasive species over natives, as their broader germination niches and greater phenological plasticity give them a competitive advantage in earlier and longer growing seasons (Hicks and others, 2019; Wolkovich and others, 2013; Zettlemoyer and others, 2019).

Pests and pathogens:

- Climate change is expected to amplify pest and pathogen pressure by directly increasing their populations and indirectly heightening host susceptibility (Dukes and others, 2009; Jactel and others, 2012; Stanturf and others, 2024; Sturrock and others, 2011; Teshome and others, 2020; Vose and others, 2013a; Weed and others, 2013).
- Drought and fire can heighten host susceptibility, leading to increased outbreaks of pests such as emerald ash borer (*Agrilus planipennis*) and red oak borer (*Enaphalodes rufulus*), and pathogens, such as cankers (e.g., *Hypoxylon* spp., *Neonectria* spp.) and root rot (*Armillaria* spp.), placing otherwise drought-tolerant species (e.g., *Quercus* spp.) under amplified stress and potential decline (Canelles and others, 2021; Haavik and others, 2015; Hamann and others, 2021; Hubbart and others, 2016; Jactel and others, 2012; Novick and others, 2022; Ramsfield and others, 2016; Stump and others, 2024; Sturrock and others, 2011; Wood and others, 2018).
- Mild winters are likely to increase the overwinter survival of pests and pathogens such as beech bark disease (*Cryptococcus fagisuga* and *Neonectria* spp.) and forest tent caterpillar (*Malacosoma disstria*), facilitate northward expansion of species already in the Midwest like Asian long-horned beetle (*Anoplophora glabripennis*) and spongy moth (*Lymantria dispar*), and introduce novel species to the region such as southern pine beetle (*Dendroctonus frontalis*) and certain water molds (e.g., *Phytophthora* spp.) (Birch and others, 2023; Burgess and others, 2017; Conrad and others, 2020; Contosta and others, 2019; Contreras-Cornejo and others, 2023; Dey and others, 2019; Flower and Gonzalez-Meler, 2015; Kasson and Livingston, 2012; Lesk and others, 2017; Pureswaran and others, 2018; Srivastava and others, 2020; Stanturf and others, 2024; Stephanson and Ribarik Coe, 2017; Zhou and others, 2021).
- Some aspects of climate change may be detrimental to certain pests and pathogens, as more frequent heatwaves and emerging phenological asynchrony with host species could reduce the impact of pests such as forest tent caterpillar (*Malacosoma disstria*) and spongy moth (*Lymantria dispar*) (Abarca and Lill, 2015; Banahene and others, 2018; Garrett and others, 2016; Schwartzberg and others, 2014).

Herbivory:

- Mild winters may intensify browsing pressure across the Midwest by increasing white-tailed deer (*Odocoileus virginianus*) survival, population growth, and foliage accessibility (Champagne and others, 2021; Felton and others, 2024; Sabo and others, 2023; Sanders-DeMott and others, 2018; Weiskopf and others, 2019).
- As browse-preferred species decline (e.g., *Acer saccharum*, *Pinus strobus*, *Prunus serotina*, *Quercus* spp., *Tilia americana*), browsing pressure may shift community composition toward browse-resistant species (e.g., *Fagus grandifolia*, *Ostrya virginiana*) (Champagne and others, 2021; Dahlberg and Guettinger, 1956; Farinosi and Walters, 2023; Flinn and others, 2019; Frerker and others, 2014; Patton and others, 2021; Sample and others, 2023; Wakeland and Swihart, 2009; White, 2012).
- Selective deer browsing may exacerbate climate-driven regeneration failure by suppressing climate-resilient species assemblages and indirectly facilitating recruitment of rarely browsed, drought- and fire-tolerant species (e.g., *Ostrya virginiana*) or those sensitive to milder winter conditions (e.g., *Fagus grandifolia*) (Champagne and others, 2021; Frerker and others, 2014; Henry and Walters, 2023; Novick and others, 2022; Patton and others, 2021).

Table 25. Summary of the expected general influence of climate change on North-Central Beech - Maple - Basswood Forest habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory columns (e.g., showing both negative and positive influences on distribution), reflecting conflicting findings in the literature.

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity or high vulnerability (–)	High adaptive capacity or low vulnerability (+)
**Ailanthus altissima*[1]	*Acer saccharum*[8]	*Acer saccharum*[10]	*Acer saccharum*[6,9]
**Berberis thunbergii*[2]	**Alliaria petiolata*[9]	*Fagus grandifolia*[11]	*Fagus grandifolia*[13]
Fagus grandifolia[3]	*Fagus grandifolia*[6]	*Tilia americana*[12]	*Ostrya virginiana*[6,14,15]
Liriodendron tulipifera[4]	*Quercus macrocarpa*[5,6]		*Quercus macrocarpa*[14,16]
Ostrya virginiana[5]	*Tilia americana*[6,8]		*Quercus rubra*[6,15–17]
Prunus serotina[5,6]			*Tilia americana*[15,17]
Quercus rubra[5–7]			
**Rhamnus cathartica*[1,2]			
Tilia americana[3]			

[1]Bradley and others (2010); [2]Evans and others (2024); [3]Philips and others (2020); [4]Hanberry (2024); [5]Prasad and others (2020); [6]Peters and others (2020); [7]Rauschendorfer and others (2022); [8]Knoot and others (2015); [9]Brinker and others (2018); [10]GLIFWC (2023); [11]Sansom (2020); [12]Potter and others (2017); [13]Royer-Tardif and others (2021); [14]Handler and others (2014b); [15]Inter-Tribal Council of Michigan (2016); [16]Etterson and others (2020); [17]Rogers and others (2016)

Table 26. Summary of impacts from key climate stressors on North-Central Beech - Maple - Basswood Forest habitat species. Some species appear in contradictory columns (e.g., showing both negative and positive drought impacts), reflecting conflicting findings in the literature.

Stressor	Impact	Effected species
Drought	When drought tolerance is low, heightened susceptibility to disturbances, competitive exclusion, and mortality	*Acer saccharum*[1–13] *Fagus grandifolia*[14–20] *Liriodendron tulipifera*[21,22] *Prunus serotina*[23,24] *Quercus macrocarpa*[25] *Quercus rubra*[5,10,12,13,26,27] *Tilia americana*[28]
	When drought tolerance is high, competitive advantage and increased abundance despite added stress	*Fagus grandifolia*[28] *Prunus serotina*[28,29] *Quercus macrocarpa*[8,17,28,30] *Quercus rubra*[7,21,31–34]
	When fire tolerance is low, heightened vulnerability to other stressors and mortality	*Acer saccharum*[35–37] *Fagus grandifolia*[30,35–38] *Liriodendron tulipifera*[30,39] *Ostrya virginiana*[37,40] *Prunus serotina*[28,29,35,36,41] *Quercus rubra*[33,37] *Tilia americana*[28,37,42]
	When fire tolerance is high, competitive advantage and increased abundance despite added stress	*Fagus grandifolia*[28] *Prunus serotina*[28] *Prunus virginiana*[28,37,43] *Quercus macrocarpa*[28,35–37,44] *Quercus rubra*[27–29,35,36] *Tilia americana*[28]
Mild winters	When sensitivity to freeze and frost stress is high, heightened susceptibility to damage, disturbances, reduced productivity, and regeneration failure	*Acer saccharum*[45–50] *Fagus grandifolia*[51] *Liriodendron tulipifera*[29] *Prunus serotina*[29] *Quercus macrocarpa*[45] *Quercus rubra*[45,51] *Tilia americana*[52]
	When chilling requirements are unmet or phenological responses are inadequate, higher rates of trophic	*Acer saccharum*[28,53] *Fagus grandifolia*[28] *Ostrya virginiana*[28]

Stressor	Impact	Effected species
	mismatches, competitive exclusion, and regeneration failure	*Prunus serotina*[28]
		Prunus virginiana[28]
		Quercus macrocarpa[28]
		Quercus rubra[5,28]
		Tilia americana[28]

[1]Au and others (2020); [2]Bapikee (2013); [3]Boakye and others (2023); [4]Fisichelli and others (2014); [5]Lee and Ibanez (2021); [6]Maxwell and others (2024); [7]McEwan and others (2011); [8]Peters and others (2015); [9]Randall and Walters (2019); [10]Reich and others (2022); [11]Rudolph and LeBlanc (2020); [12] Stangler and others (2016); [13]Wang and others (2023); [14]Booth and others (2012); [15]Cohen (2005); [16]Barnes (1991); [17]Niinemets and Vallardares (2006); [18]Seeley and others (2019); [19]Stephanson and Ribarik Coe (2017); [20]U.S. Department of Agriculture, U.S. Forest Service (1965); [21]Brzostek and others (2014); [22]Leblanc and others (2020) [33]LeBlanc and Berland (2019); [23]Chhin (2018); [24]Ramage and others (2023); [25 31]Dannenberg and others (2019); [26]Lockwood and others (2023); [27]Novick and others (2022); [28]USDA, NRCS (2025); [29] Peters and others (2020); [30]Tubbs and Houston (1990); [31]Gustafson and Sturtevant (2013); [32]Rauschendorfer and others (2022); [33]Sander (1990b); [34]Walters and others (2023); [3556]Arthur and others (2021)); [36]Varner (2018); [37]Hoving and others (2021); [38]Tubbs and Houston (1990); [39]Beck (1990); [40]Coladonato ((1992); [41]Marquis (1990); [42]Curtis (1959); [43]Johnson (2000); [44]Deitschmann (1965); [45]Augspurger (2009); [46]Comerford and others (2013); [47]Henne and others (2007); [48]Reinmann and others (2019); [49]Sanders-DeMott and others (2018); [50]Tatina (2015); [51]Lindback and others (2023); [52]Crow (1990); [53]Solarik and others (2016)

Focal Wildlife Species: Cerulean Warbler

The cerulean warbler (*Setophaga cerulea*) is a small, Nearctic-Neotropical migratory songbird considered to be a Bird of Conservation Concern (Buehler and others, 2020; U.S. Fish and Wildlife Service, 2021). Like many long-distance migratory birds, cerulean warblers demonstrate a limited capacity to advance migratory arrival in response to advancing spring conditions and may be vulnerable to phenological asynchrony (i.e., peak nesting and insect availability) (Connare and Islam, 2022). This is of special concern given cerulean warblers' heavy reliance on Lepidoptera (Auer and others, 2016) and the vulnerability of insects to climate change (Edwards and others, 2025). The species is associated with mature, deciduous forests with canopy gaps (Buehler and others, 2020). In the breeding range, they demonstrate preference for white oak (*Quercus alba*) and bitternut hickory (*Carya cordiformis*), avoiding northern red oak (*Quercus rubra*) and red maple (*Acer rubrum*) (Buehler and others, 2020), potentially reflecting higher lepidopteran abundance in forests dominated by these tree types (Connare and Islam, 2022). Novel interspecific interactions are also of concern. For example, as the northern parula (*Setophaga americana*) breeding range expands northward, hybridization with cerulean warbler may become more common (Trimbath and others, 2019).

In the overwintering range, cerulean warblers may be found in old-growth forests, shade coffee plantations, and low- to mid-elevation forests of the Andes (Buehler and others, 2020). The U.S. Gulf Coast and Mississippi Valley provide important stopover habitat to meet migratory demands of trans-Gulf crossings (Raybuck and others, 2022). Climate change threatens suitable habitats across the nonbreeding range. In the Andes, studies predict a moderate contraction (19–43%) in suitable range, driven by climate change, and an upslope elevational shift (Brodie et al. 2024). Both climate-driven flood hazard and human alterations to the Mississippi River system threaten the viability of bottomland hardwood forests (Munoz and others, 2018). Targeted forest management and restoration may be viable strategies to offset declines in the species, provided it increases stand structural diversity, creates

small canopy openings, and promotes preferential tree species (e.g., white oak) (Kellner and others, 2016; Nareff and others, 2019).

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Appendix 1

Table ii. Woody plant species common to the Appalachian-Interior-Northeast Mesic Forest ecosystem. Key species were identified using USNVC (USNVC, 2025) floristic descriptions, filtered by associations occurring within the Midwest that collectively comprise the Appalachian-Interior-Northeast Mesic Forest ecosystem. Taxonomic authority, common name, and growth habit are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit	Appalachian-Central Interior Mesic Forest	Appalachian-Northeast Mesic Forest	North-Central Beech - Maple - Basswood Forest
Acer saccharum Marshall	Sugar maple	Shrub, tree	×		×
Aesculus flava Aiton	Yellow buckeye	Shrub, tree	×		
Cornus alternifolia L. f.	Alternateleaf dogwood	Shrub, tree	×		
Cornus florida L.	Flowering dogwood	Shrub, tree	×		
Ostrya virginiana (Mill.) K. Koch	Hophornbeam	Shrub, tree			×
Prunus serotina Ehrh.	Black cherry	Shrub, tree	×		×
Prunus virginiana L.	Chokecherry	Shrub, tree			×
Quercus macrocarpa Michx.	Bur oak	Shrub, tree			×
Acer rubrum L.	Red maple	Tree	×		
Fagus grandifolia Ehrh.	American beech	Tree	×		×
Fraxinus americana L.	White ash	Tree	×		
Liriodendron tulipifera L.	Tuliptree	Tree	×		×
Pinus strobus L.	Eastern white pine	Tree	×	×	
Quercus alba L.	White oak	Tree	×		
Quercus rubra L.	Northern red oak	Tree	×		×
Quercus velutina Lam.	Black oak	Tree	×		
Tilia americana L.	American basswood	Tree	×		×
Tsuga canadensis (L.) Carrière	Eastern hemlock	Tree	×	×	

Table iii. Non-woody plant species common to the Appalachian-Interior-Northeast Mesic Forest ecosystem,. Key species were identified using USNVC (USNVC, 2025) floristic descriptions, filtered by associations occurring within the Midwest that collectively comprise the Appalachian-Interior-Northeast Mesic Forest ecosystem. Taxonomic authority, common name, and growth habit are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit	Appalachian-Central Interior Mesic Forest	Appalachian-Northeast Mesic Forest	North-Central Beech - Maple - Basswood Forest
Adiantum pedatum (Rupr.) Paris	Aleutian maidenhair	Fern	×		
Botrychium virginianum (L.) Sw.	Rattlesnake fern	Fern	×		
Dennstaedtia punctilobula (Michx.) T. Moore	Eastern hayscented fern	Fern	×		
Polystichum acrostichoides (Michx.) Schott	Christmas fern	Fern	×		
Actaea racemosa L.	Black baneberry	Forb/herb	×		
Amphicarpaea bracteate (L.) Fernald	American hogpeanut	Forb/herb, vine	×		
Arisaema triphyllum (L.) Schott	Jack in the pulpit	Forb/herb	×		
Asarum canadense L.	Canadian wildginger	Forb/herb	×		
Caulophyllum thalictroides (L.) Michx.	Blue cohosh	Forb/herb	×		
Claytonia virginica L.	Virginia springbeauty	Forb/herb	×		
Erythronium americanum Ker Gawl.	Dogtooth violet	Forb/herb	×		
Eurybia divaricate (L.) G.L. Nesom	White wood aster	Forb/herb	×		
Galium triflorum Michx.	Fragrant bedstraw	Forb/herb, vine	×		
Geranium maculatum L.	Spotted geranium	Forb/herb	×		
Hydrophyllum canadense L.	Bluntleaf waterleaf	Forb/herb	×		
Hydrophyllum virginianum L.	Eastern waterleaf	Forb/herb	×		
Impatiens pallida Nutt.	Pale touch-me-not	Forb/herb	×		
Laportea canadensis (L.) Weddell	Canadian woodnettle	Forb/herb	×		
Osmorhiza claytonia (Michx.) C.B. Clarke	Clayton's sweetroot	Forb/herb	×		
Prosartes lanuginose (Michx.) D. Don	Yellow fairybells	Forb/herb	×		
Sanguinaria canadensis L.	Bloodroot	Forb/herb	×		
Sedum ternatum Michx.	Woodland stonecrop	Forb/herb	×		

Species	Common name	Growth habit	Appalachian-Central Interior Mesic Forest	Appalachian-Northeast Mesic Forest	North-Central Beech - Maple - Basswood Forest
Tiarella cordifolia L.	Heartleaf foamflower	Forb/herb	×		
Trillium erectum L.	Red trillium	Forb/herb	×		
Trillium grandiflorum (Michx.) Salisb.	White trillium	Forb/herb	×		
Viola canadensis L.	Canadian white violet	Forb/herb	×		

Appendix 2

Climate projections are derived from the Coupled Model Intercomparison Project Phase 5 (CMIP5) and include model outputs from 20 unique models.

CMIP5 Models:

- bcc-csm1-1
- bcc-csm1-1-m
- CNRM-CM5
- CSIRO-Mk3-6-0
- GFDL-ESM2M
- HadGEM2-CC365
- HadGEM2-ES365
- inmcm4
- IPSL-CM5A-LR
- PSL-CM5A-MR
- MIROC5
- MIROC-ESM
- MIROC-ESM-CHEM
- MRI-CGCM3
- NorESM1-M
- IPSL-CM5B-LR
- CanESM2
- CCSM4
- GFDL-ESM2G
- BNU-ESM

To capture variation across CMIP5 climate models, models with the lowest (minimum model) and highest (maximum model) projected values under the RCP8.5 scenario are presented in addition to the mean of all models. Table vi includes a list of all minimum and maximum models selected for each region and climate variable.

Table vi. Climate Projection Minimum and Maximum Models. For each region and climate variable, minimum and maximum models were determined by averaging the each climate variable projection for RCP8.5 end of the century (2070–2099) scenario across each region.

Ecoregion	Climate Variable	Season	Minimum Model	Maximum Model
Erie Drift Plains	Extreme Dry Months	Annual	MRI-CGCM3_r1i1p1	HadGEM2-CC365_r1i1p1
Midwest Region	Extreme Dry Months	Annual	MRI-CGCM3_r1i1p1	HadGEM2-CC365_r1i1p1
Appalachian Forest	Extreme Dry Months	Annual	inmcm4_r1i1p1	HadGEM2-CC365_r1i1p1
Driftless Area	Extreme Dry Months	Annual	inmcm4_r1i1p1	IPSL-CM5A-MR_r1i1p1
Erie Drift Plains	Growing Season Length	Annual	bcc-csm1-1-m	MIROC-ESM-CHEM
Midwest Region	Mean SPEI	Annual	IPSL-CM5A-MR_r1i1p1	MRI-CGCM3_r1i1p1
Appalachian Forest	Precipitation	Annual	IPSL-CM5A-MR	CSIRO-Mk3-6-0
Driftless Area	Precipitation	Annual	IPSL-CM5A-MR	CSIRO-Mk3-6-0
Erie Drift Plains	Precipitation	Annual	IPSL-CM5A-MR	CSIRO-Mk3-6-0
Appalachian Forest	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Driftless Area	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Erie Drift Plains	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Appalachian Forest	Mean Minimum Temperature	Winter	GFDL-ESM2M	HadGEM2-CC365
Driftless Area	Mean Minimum Temperature	Winter	GFDL-ESM2M	HadGEM2-CC365
Erie Drift Plains	Mean Minimum Temperature	Winter	GFDL-ESM2M	HadGEM2-CC365
Midwest Region	Mean Minimum Temperature	Winter	GFDL-ESM2M	HadGEM2-CC365