Effects of Climate Change on Midwestern Ecosystems: Central Interior Oak Forest, Woodland and Savanna

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Foreword

Series Overview

The following is the sixth 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 Central Interior Oak Forest, Woodland and Savanna ecosystem is a drought- and fire-adapted ecosystem that historically persisted as an intermediate between grassland and closed-canopy forest across the Midwest. Prioritized habitats from this ecosystem consist of the Interior Low Plateau Oak Forest, Woodland and Savanna; Ozark-Ouachita Oak - Shortleaf Pine Woodland and Savanna; Central Midwest Oak Openings and Barrens; North-Central Oak - Hickory Forest and Woodland; and Central Appalachian-Northeast Oak Forest and Woodland habitats. These oak (*Quercus* spp.)-dominated habitats historically occupied sites that were drier and more fire-prone than those occupied by closed forests, yet more moist and less frequently burned than those occupied by grasslands. This balance between moisture availability and recurring fire maintains the open-canopy structure characteristic of this ecosystem. Climate change may alter this balance, increasing drought frequency and severity and reshaping fire regimes. Rising vapor pressure deficit (VPD) and more variable precipitation may intensify drought stress and amplify moisture variability. Furthermore, shifts in fire weather and fuel dynamics can increase wildfire risk and constrain the safe use of prescribed fire to maintain frequent, low-intensity burns. Because this ecosystem is already degraded by fire suppression and mesophication (the process by which shade-tolerant, fire-sensitive trees expand, reduce canopy openness, and dampen future fire spread), climate change may restore historical moisture and fire conditions that once maintained this system.

Drought and fire are expected to impact future habitat structure and community composition. Drought may intensify soil drying, reduce herbaceous cover, and increase erosion risk, especially on thin-soiled, nutrient-poor, or sloped sites. In the overstory, mesophytic trees such as maples (*Acer* spp.), beeches (*Fagus* spp.), and birches (*Betula* spp.) are vulnerable to recruitment failure and canopy decline; in contrast, drought- and fire-adapted oaks and hickories (*Carya* spp.) may expand under moderate drought conditions. Frequent, low-intensity fire promotes open-canopy, oak- and grass-dominated assemblages, but severe or frequent fire can exceed tolerance thresholds, even for pyrophytic species. Drought and post-disturbance nutrient pulses can also favor fire-promoting invasive grasses such as cogongrass (*Imperata cylindrica*), Nepalese browntop (*Microstegium vimineum*), and smooth brome (*Bromus inermis*). In contrast, both native and invasive shrubs and encroaching trees, including common buckthorn (*Rhamnus cathartica*), eastern red cedar (*Juniperus virginiana*), honeysuckles (*Lonicera* spp.), and roughleaf dogwood (*Cornus drummondii*), often persist through fire.

Among habitats, the Interior Low Plateau Oak Forest, Woodland and Savanna, the Ozark-Ouachita Oak - Shortleaf Pine Woodland and Savanna, and the Central Midwest Oak Openings and Barrens habitats are projected to experience some of the highest increases in VPD and fire danger days. Combined with their typical placement on drier upland sites and shorter historical fire return intervals, these conditions place them at greater risk of transitions into grassland. In contrast, the North-Central Oak - Hickory Forest and Woodland habitat and the Central Appalachian-Northeast Oak Forest and Woodland habitat are likely to experience only moderate increases in VPD and fire danger, and together with their historically longer fire return intervals, are more vulnerable to transitions toward shrubland or closed-canopy forest.

Additional pressures from pests, pathogens, and herbivory may compound these risks. Drought-stressed oaks are particularly vulnerable to pests such as the two-lined chestnut borer (*Agrilus bilineatus*) and to stem- and root-infecting pathogens that thrive under fluctuating moisture regimes. Meanwhile, warming winters are expected to boost white-tailed deer (*Odocoileus virginianus*) populations, intensifying browsing pressure on oak seedlings and native forbs and further undermining regeneration.

Adaptation and management strategies may help address these interacting stressors. Restoring frequent, low-intensity fire can support open-canopy conditions and favor fire-adapted oaks and herbaceous diversity, though managers may also need to adjust burn timing and intensity to accommodate shifting fire weather. Efforts such as targeted invasive species control, protecting regeneration from herbivory, and incorporating drought- and fire-adapted native species into restoration plantings may also provide pathways for adaptation under climate change.

Central Interior Oak Forest, Woodland and Savanna

Key Climate Change Stressors

Climate change is altering the disturbance regimes that maintain the Central Interior Oak Forest, Woodland and Savanna ecosystem across the Midwest. This oak (*Quercus* spp.)-dominated, open-canopied system historically persisted as an intermediate state between grassland and closed forest, with its structure shaped by the opposing influences of moisture availability, which favors forest development, and recurring fire, which favors grasslands. Today, increasing drought and fire activity are pushing this system beyond its historical range of variability, challenging the processes that once stabilized it.

Drought

Drought has long been a defining stressor for the Central Interior Oak Forest, Woodland and Savanna ecosystem, shaping both its historical distribution and contemporary dynamics. Over the past millennium, the geographic boundary between forests and grasslands shifted partly in response to changes in drought frequency and intensity. Grasslands experienced more frequent and intense drought than neighboring forests; in the transitional zone, the Central Interior Oak Forest, Woodland and Savanna ecosystem persisted (Anderson and Bowles, 1999; Barros and others, 2018; Briere and Gajewski, 2023; Changnon and others, 2002; Danz and others, 2013; Samson and others, 2004; Umbanhowar and others, 2006). Oaks *(Quercus* spp.) withstood recurrent moisture deficits through physiological tolerance and by benefitting from fire, which reduced competition from more moisture-demanding tree species (Hayford and Chhin, 2020; Nowacki and Abrams, 2008).

In the 20th century, increased precipitation across the Midwest reduced the influence of drought as a natural disturbance (Hanberry and others, 2018). Combined with widespread fire suppression, this shift favored the establishment of mesic tree species and drove habitat mesophication (the process by which shade-tolerant, fire-sensitive trees expand, reduce canopy openness, and dampen future fire spread in formerly open oak systems) (Alexander and others, 2021; Barros and others, 2018; Ford and others, 2021; Hoell and others, 2021; Nowacki and Abrams, 2008). Over the past few decades, however, precipitation has become more variable, with rapid transitions between heavy rainfall and drought, including longer dry periods between events (Ford and others, 2021; Hoell and others, 2021; Marvel and others, 2023).

Future climate projections suggest drought will become both more frequent and more severe (Figure 1), driven by higher evaporative demand and greater variability in seasonal precipitation (Chen and Ford, 2023; Ford and others, 2021; Marvel and others, 2023; Zhao and others, 2020; Zhao and others, 2023). Both chronic seasonal drought (i.e., occurring over a span of months to years) and flash drought (i.e., rapid onset over days to weeks) are expected to become more common (Christian and others, 2021; Koster and others, 2019; Lesinger and Tian, 2022; Pendergrass and others, 2020), with some projections suggesting the area of oak forest exposed to severe or extreme drought could triple by mid-century (Costanza and others, 2023). Although some models indicate modest increases in annual precipitation, these gains are unlikely to offset warming-induced drying, particularly as rising vapor pressure deficit (VPD) accelerates soil moisture loss and plant stress (Gamelin and others, 2022; Iglesias and others, 2022; Novick and others, 2022; Will and others, 2013; Zhao and others, 2020; Zhuang and others, 2024).

Fire

Fire is another critical driver of structure and composition in the Central Interior Oak Forest, Woodland and Savanna ecosystem. Recurrent surface fires historically limited shade-tolerant, fire-sensitive trees and shrubs, reinforced oak (*Quercus* spp.) dominance, and maintained diverse herbaceous understories (Abrams and others, 2021; Alexander and others, 2021; Brose and others, 2014; Frelich and others, 2015; Ma and others, 2016; Ratajczak and others, 2014). Paleoecological records and historical reconstructions consistently link oak prevalence to periods of warming, drying, and widespread fire (Abrams, 1992; Alexander and others, 2021; Camill and others, 2003; Hanberry and others, 2012; Patterson III, 2006). Typical fire return intervals varied throughout the region but often ranged from less than 10 years to no more than 50, frequent enough to maintain open canopies, but not so frequent as to eliminate trees entirely (Considine and others, 2013; Faber-Langendoen, 1995; Fralish, 1997; Frelich and others, 2015; Hanberry and Abrams, 2018; Lettow and others, 2014).

As fire use shifted from regular burning under Indigenous land management practices to widespread suppression following Euro-American settlement, fire return intervals lengthened or disappeared, resulting in a dramatic decline of the Central Interior Oak Forest, Woodland and Savanna ecosystem across the Midwest and transitions toward closed-canopy mesic forests (Abrams and others, 2021; Alexander and others, 2021; Briggs and others, 2002; Dey and Kabrick, 2015; Hanberry and others, 2014; Hanberry and others, 2020; Hutchinson and others, 2024; McClain and others, 2021; Nowacki and Abrams, 2008; Nuzzo, 1986). Throughout much of the 20th century, wildfires tended to be small, with suppression reducing their average size (Hai and others, 2023; Nowacki and Abrams, 2008; Parks and others, 2025). More recently, large wildfires are increasing in frequency and area burned throughout much of the eastern temperate forest region (Donovan and others, 2023; Iglesias and others, 2022; Jones and others, 2022).

Climate change is projected to alter fuel loads and weather conditions (Figure 2), likely increasing fire activity across the Midwest. Warmer, wetter winters and springs can amplify fine-fuel production, while hotter, drier summers enhance fuel drying and flammability (Briske, 2017; Gao and others, 2021; Gedalof, 2011; Hanberry and others, 2024; Jones and others, 2022; Stanturf and others, 2024). Wildfire weather indices are also projected to rise, with more high-risk days, longer high-risk periods, earlier fire season onset, and an overall lengthening of the fire season (Brown and others, 2021; Gao and others, 2021; Hanberry and others, 2024; Jones and others, 2022; Kerr and others, 2018; Ma and others, 2016; Tang and others, 2015). These changes may shorten fire return intervals—promoting frequent low-intensity surface fires (Ma and others, 2016; Vose and others, 2021; Vose and Elliott, 2016) but also increasing the likelihood of extreme wildfire events (Donovan and others, 2023; Fan and others, 2023; Krueger and others, 2022; Littell and others, 2016). Moreover, as climate change also reduces the number of safe burn windows, especially in spring, prescribed burns may become more difficult to conduct (Frelich and others, 2015; Meunier and others, 2021).

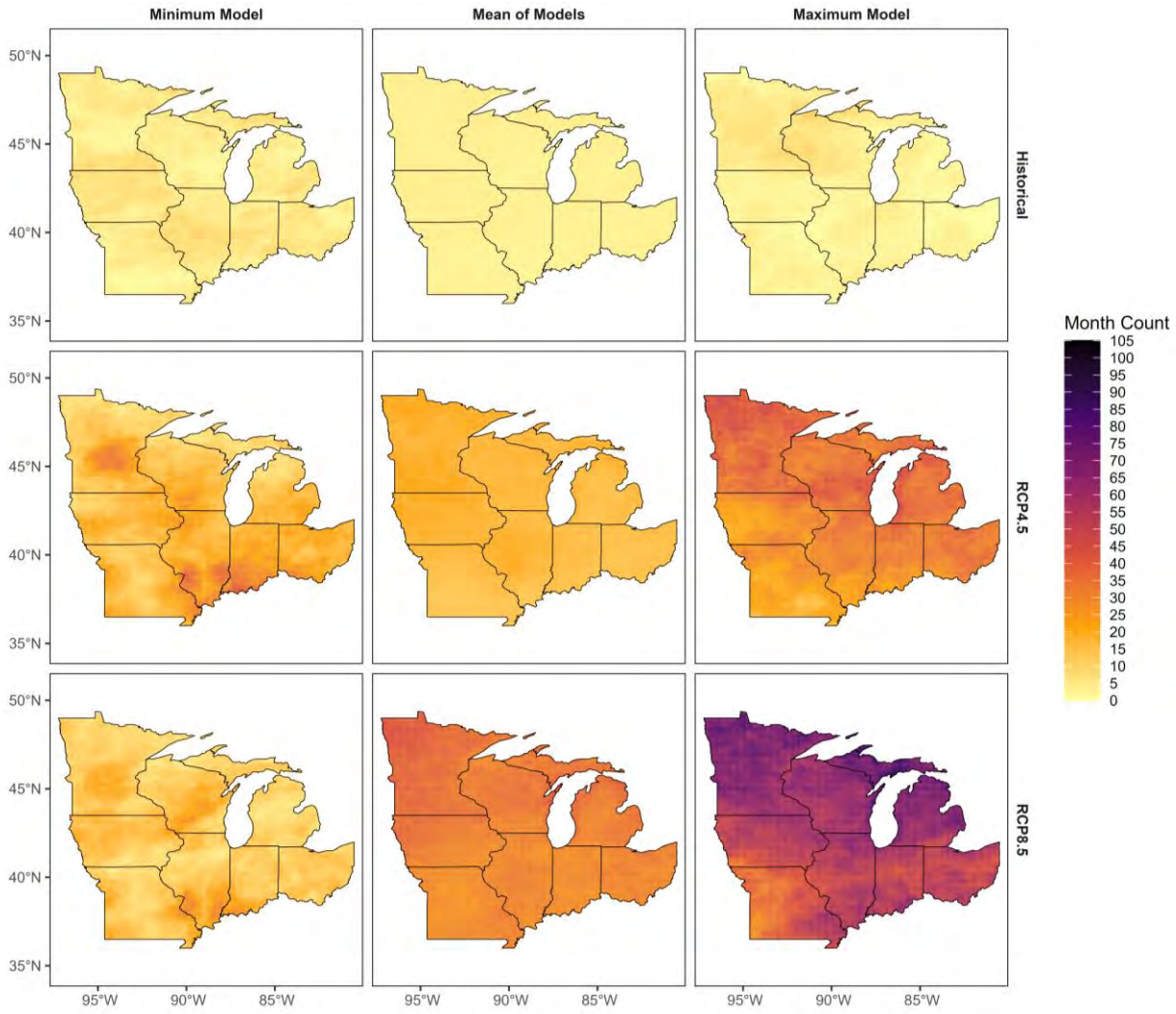

Figure 1. Projected changes in 30-year extreme drought incidence in the Midwest. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (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).

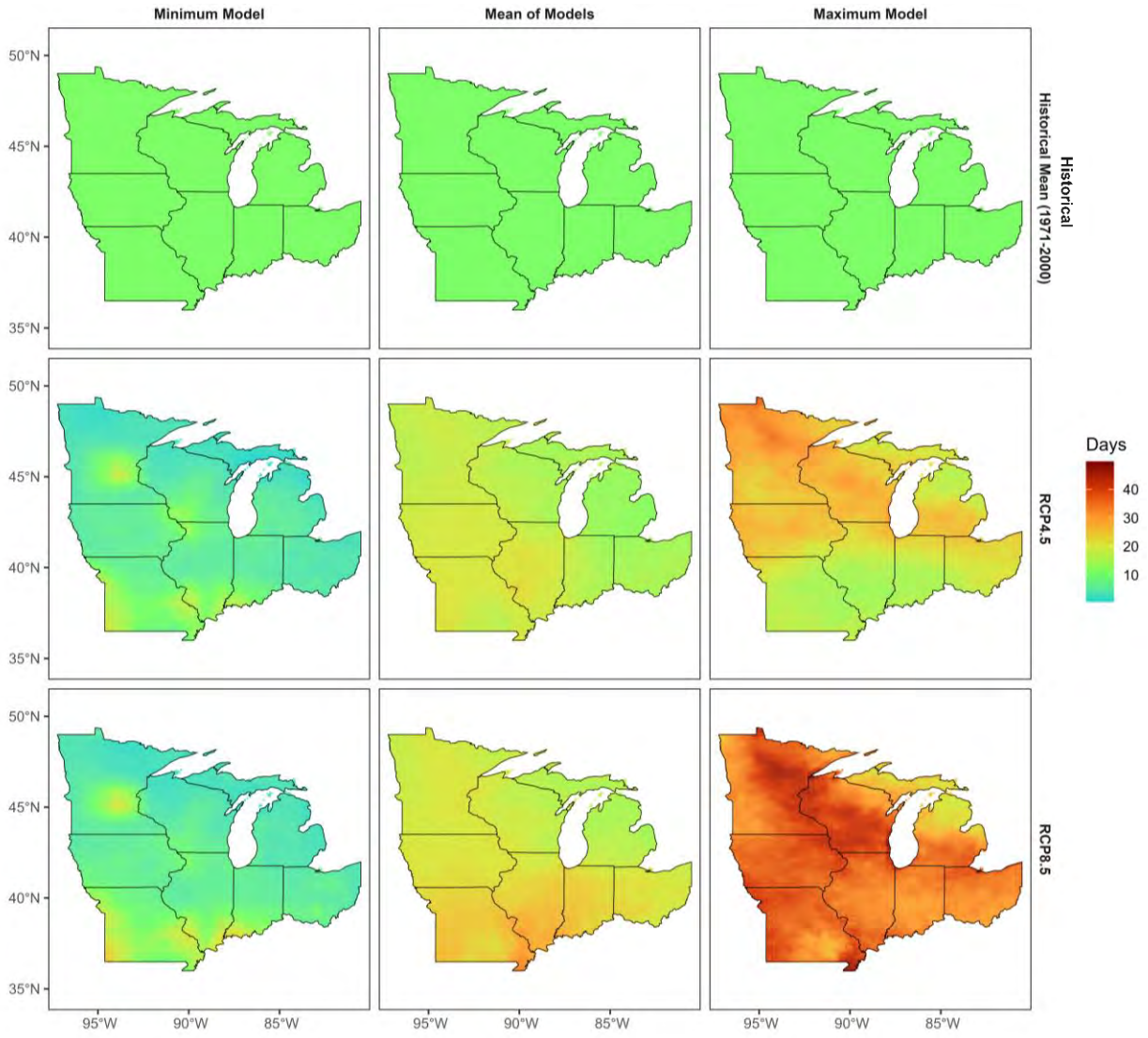

Figure 2. Projected changes in annual extreme fire danger in the Midwest. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000, top row) and projected (2040–2069) annual extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) 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 extreme fire danger under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

Climate change is altering the conditions that maintain the structure and function of the Central Interior Oak Forest, Woodland and Savanna ecosystem. Increasing drought and changing fire activity are likely to drive significant changes in habitat structure, community composition, and species interactions, as even modest disruptions to these disturbance processes can have cascading effects.

Habitat Structure

Abiotic responses to drought. Drought is expected to intensify soil drying in the Central Interior Oak Forest, Woodland and Savanna ecosystem. Because these open systems often have limited water storage capacity, interannual variation in rainfall can cause large fluctuations in surface soil moisture (Fay and others, 2011; O'Keefe and others, 2024; Räsänen and others, 2020; Smith and others, 2016; Ziegler and others, 2008). As soils dry, protective herbaceous cover can be lost (Ebel and others, 2022; Vose and others, 2016; Ziegler and others, 2008). Without this cover, soils are more exposed to heat and wind, which accelerates moisture loss (Dai, 2013; Mukherjee and Mishra, 2021; Werner and others, 2020). Bare soils are also more susceptible to erosion, especially when heavy rainfall follows drought, a pattern projected to become more common with climate change (Breshears and others, 2016; Butler and others, 2015; Swanston and others, 2018; Vose and others, 2016; Ziegler and others, 2008). Drought also modifies nutrient dynamics. Suppressed plant growth and uptake allow inorganic nitrogen, soil carbon, and other nutrients to accumulate in the soil (Hartmann and others, 2013; He and Dijkstra, 2014; Hoover and Rogers, 2016; Schlesinger and others, 2016; Vázquez and others, 2025). When rainfall returns, these accumulated nutrients can be rapidly released or leached out of the system (Burt and others, 2015; Homyak and others, 2017; Leizeaga and others, 2021; Reinsch and others, 2024; Vose and others, 2016).

Abiotic responses to fire. Fire alters habitat structure in the Central Interior Oak Forest, Woodland and Savanna ecosystem by reshaping vegetation, soils, and nutrient dynamics. Low-intensity surface fires remove accumulated litter and detritus that can suppress herbaceous vegetation, especially where fire has been historically excluded (Fowler and Beckage, 2019; Schlesinger and others, 2016; Stambaugh and others, 2021). High-intensity fires, which are projected to become more common with climate change, can reduce the midstory and occasionally reduce the overstory cover, increasing light availability to the understory (Arthur and others, 2021; Beebe and others, 2024; Dey and others, 2017; Lettow and others, 2014). Low-intensity fires generally have minimal effect on soil structure, but higher-severity fires can reduce porosity, decrease infiltration rates, and consume organic matter (Hanberry and others, 2017; Singh and others, 2025; Stavi, 2019; Wells, 1979). Combustion of organic material may also expose the underlying inorganic soil layers, increasing erosion risk on slopes or when heavy rainfall follows fire (Fernández and others, 2006; Roberts, 2004; Schlesinger and others, 2016). Fire also initiates short-term nutrient pulses and longer-term nutrient redistribution. Although combustion releases nitrogen and carbon into the atmosphere, it can simultaneously increase concentrations of other soil nutrients through ash inputs and organic matter oxidation (González-Pérez and others, 2004; Knicker, 2007; Pile Knapp and others, 2024; Raison and others, 2009; Stavi, 2019). This process produces a short-term nutrient pulse that enhances postfire plant growth (González-Pérez and others, 2004; Knicker, 2007; Pile Knapp and others, 2024). However, over the longer-term, repeated burning may contribute to nutrient losses and shifts in soil carbon and mineral pools (Knicker, 2007; Pellegrini and others, 2020; Singh and others, 2025; Stavi, 2019).

Biotic responses to drought. By reducing soil moisture and increasing vapor pressure deficit (VPD), drought can limit plant growth, survival, and reproduction in the Central Interior Oak Forest, Woodland and Savanna ecosystem. Water deficits constrain photosynthesis and nutrient uptake (D'Orangeville and others, 2018; Felton and others, 2020; Marchin and others, 2010; Rauschendorfer and others, 2022; Stanturf and others, 2024; Vose and others, 2016), slowing growth in both trees and herbaceous species, particularly when drought occurs early in the growing season (Brzostek and others, 2014; D'Orangeville and others, 2018; Druckenbrod and others, 2019; Knapp and others, 2024; Matheus, 2021; Smith and others, 2024; Vose and others, 2016). These effects can result from both short-term (seasonal) and long-term (multi-year) droughts, and are often most severe during consecutive drought years, under high temperatures that increase evaporative demand, or on shallow or south-facing sites (Allen and others, 2015; Bauweraerts and others, 2014; Kabrick and others, 2008; Stanturf and others, 2024; Teskey and others, 2015). Intensifying VPD may already be pushing some species toward hydraulic failure, contributing to increased mortality (Allen and others, 2010; Batllori and others, 2020; Fusco and others, 2024; Pederson and others, 2014; Will and others, 2013).

Plant responses to drought vary widely by life history traits. Warm-season (C_4) grasses, for example, generally have high water-use efficiency and may tolerate moderate drought, whereas cool-season (C_3) grasses are more sensitive but can avoid drought by growing earlier in the season (Havrilla and others, 2023; Knapp and others, 2024; Taylor and others, 2011; Volder and others, 2010; Wilcox and others, 2023). In grass-dominated understories, during key phenological windows, drought can reduce flowering stalk density and reproductive output among dominant warm-season (C_4) grasses (Lemoine and others, 2017). During short or moderate droughts, deep-rooted shrubs and trees can access deep water reserves and maintain physiological function (Archer and others, 2017; Lambers and Oliveira, 2019; Vose and others, 2016; Wedel, 2024), whereas shallow-rooted herbaceous plants, woody seedlings, and species reliant on surface moisture may desiccate rapidly under high evaporative demand (Canham and Murphy, 2017; Clark and others, 2016; Fisichelli and others, 2014; Polley and others, 2013b; Vose and others, 2016). Early successional trees, adapted to warmer and drier soils, are often more tolerant to drought than late-successional species like sugar maple (*Acer saccharum*) or yellow birch (*Betula alleghaniensis*) (Bazzaz, 1979; Beloiu Schwenke and others, 2024; Dreyer and others, 1993; Niinemets and Valladares, 2006).

Compared to many co-occurring mesic species, oaks (*Quercus* spp.) are also broadly considered relatively drought-tolerant (Abrams, 1990; Au and others, 2020; Bernal-Escobar and others, 2022; Boakye and others, 2023; Kannenberg and others, 2019a; Maxwell and others, 2024; Rauschendorfer and others, 2022; Swanston and others, 2018; Yi and others, 2019). However, the broad assumption of superior drought tolerance in oaks may be overstated and may be context dependent. During drought, oak growth advantages are more evident in the xeric rather than mesic portions of their range, whereas during consecutive or exceptionally severe droughts, oak mortality may equal or exceed that of mesic associates (Asbjornsen and others, 2021; Au and Maxwell, 2022; Bose and others, 2024; D'Orangeville and others, 2018; LeBlanc and Terrell, 2011; Martin-Benito and Pederson, 2015; Novick and others, 2022). This apparent contradiction may reflect their anisohydric water-use strategy; under moderate droughts, oaks sustain growth by keeping stomata open, but when droughts are particularly severe, oaks, and other plants with conservative water-use strategies, become more vulnerable to damage and death (Bose and others, 2024; Elliott and others, 2015; Kannenberg and others, 2019b; Lockwood and others, 2023; Martínez-Vilalta and Garcia-Forner, 2017; McDowell and others, 2008; Novick and others, 2022). Among oaks, the potential for drought-induced mortality is particularly high for red oaks (e.g., *Q. rubra, Q. velutina, Q. coccinea*) (Druckenbrod and others, 2019; Fan and others, 2008; Kabrick and others, 2008; Novick and others, 2022).

For tree species, early life stages, including seed production, germination, and establishment, are especially sensitive to soil moisture; thus, more frequent and severe droughts may lead to regeneration failure (Canham and Murphy, 2017; Clark and others, 2016; Cowden and others, 2014; Dey and others, 2019; Fisichelli and others, 2014; Ibáñez and others, 2017). Canopy loss during drought may intensify understory stress by increasing solar radiation and local VPD (Wang and others, 2023). Additionally, overstory mortality may also create canopy openings that enhance light availability and allow seedlings and saplings to recover once moisture returns (Pederson and others, 2014; Vose and others, 2016).

Finally, soil microbes, which drive decomposition and nutrient cycling, are highly sensitive to moisture stress. Once soil moisture falls below critical thresholds, microbial activity declines, particularly during hot summer months (Leizeaga and others, 2021; Lellei-Kovács and others, 2011; Lemoine and others, 2024; Reinsch and others, 2024). These conditions favor drought-tolerant microbes over moisture-dependent taxa, shifting belowground community composition (Baldrian and others, 2023; Lemoine and others, 2024; Schlesinger and others, 2016).

Biotic responses to fire. Fire has historically been a defining process in the Central Interior Oak Forest, Woodland and Savanna ecosystem. Frequent, low-intensity surface fires maintained open canopies by suppressing shade-tolerant, fire-sensitive mesic species while promoting oak recruitment and herbaceous diversity and resetting successional trajectories (Abrams, 1992; Anderson and Bowles, 1999; Batllori and others, 2015; Dey and Fan, 2009; Dey and Kabrick, 2015; Fowler and Beckage, 2019; Frelich and others, 2015; Lettow and others, 2014; Pile Knapp and others, 2024; Taft, 1997; Urza and others, 2017; Wolf, 2004). Many dominant species, including oaks and grasses, possess traits (e.g., resprouting) that promote survival and regrowth after disturbance (Arthur and others, 2021; Dey and Kabrick, 2015; Higgins and others, 2000; Peterson and Reich, 2001; Refsland and Fraterrigo, 2018; Simpson and others, 2021). Some oaks (e.g., *Q. laevis, Q. stellata, Q. marilandica*) further reinforce fire cycles by producing highly flammable litter, while grasses provide fine fuels that enable fire spread (Briske, 2017; Frelich and others, 2015; Gold and others, 2023; Simpson and others, 2022).

With widespread fire suppression, mesic species, such as maples (*Acer* spp.) and beeches (*Fagus* spp.), expanded into the canopy, creating shadier, cooler, and moister understories with reduced herbaceous cover (Alexander and others, 2021; Arthur and others, 2021; Nowacki and Abrams, 2008; Palus and others, 2018). These conditions accelerate litter decomposition and reduce fuel accumulation—dampening future fire spread and reinforcing “mesophication” (Alexander and others, 2021; Babl-Plauche and others, 2022; Gold and others, 2023; Nowacki and Abrams, 2008; Palus and others, 2018). As climate change increases the likelihood of more frequent and severe fires, these new fire regimes, although still limited by fire suppression and landscape fragmentation, may more closely resemble pre-European settlement times and become influential in restoring and maintaining habitat structure (Frelich and others, 2017; Hutchinson and others, 2024; Lenihan and others, 2008; Vander Yacht and others, 2019; Vose and Elliott, 2016).

However, emerging fire regimes may not have a wholly positive impact on habitat structure – the frequency and severity of fire determines its influence on plant survival and recruitment (Hutchinson and others, 2024; Roberts, 2004; Stambaugh and others, 2021; Stavi, 2019; Vose and Elliott, 2016). Fire that is too frequent or too severe can exceed tolerance thresholds, even for fire-adapted species (Ajao and others, 2024; Grau-Andrés and others, 2024; Hutchinson and others, 2024; Peterson and Reich, 2001). Oak seedlings, for example, often require 10–30 fire-free years to grow beyond the height susceptible to fire-induced dieback (Arthur and others, 2021; Brose and others, 2006; Dey and others,

2017; Frelich and others, 2015; Kabrick and others, 2014). If fires recur too often, saplings may be lost and recruitment into the canopy can fail, favoring only the most pyrophytic species and reducing diversity (Ajao and others, 2024; Burton and others, 2010; Dey and others, 2017; Frelich and others, 2015; Grau-Andrés and others, 2024; Peterson and Reich, 2001). Further, while surface fires seldom kill adult trees, higher intensity fires capable of causing crown and cambial injury could significantly increase mortality risk in adult stands (Ajao and others, 2024; Anderegg and others, 2022; Lucash and others, 2018; Stambaugh and others, 2021).

The biotic effects of fire are also shaped by its interactions with drought. Drought can increase fuel loads and heighten fire severity by desiccating litter layers and increasing dead woody material, thus increasing the likelihood of fire-related tree mortality (Cansler and others, 2024; Robbins and others, 2024; Vose and others, 2016; Vose and Elliott, 2016; Woodall and others, 2007). Fire may also exacerbate drought stress by removing thermally buffering vegetation, exposing soils and regrowing vegetation to higher evaporative demand, and potentially negating its positive influence on oak recruitment (Refsland and Fraterrigo, 2018; Taft, 2020).

Community Composition

Community responses to drought. Increasing drought frequency and severity is expected to reshape species composition across the Central Interior Oak Forest, Woodland and Savanna ecosystem. Although these systems are generally drought-adapted, future moisture stress may intensify selection for xeric species. In the overstory, mesic trees such as maple (*Acer* spp.), beech (*Fagus* spp.), and birch (*Betula* spp.) are particularly vulnerable to recruitment failure and dieback (Abrams and Nowacki, 2016; Asbjornsen and others, 2019; Canham and Murphy, 2017; Dey and others, 2019; Fisichelli and others, 2014). Drought-related declines have already been observed in sugar maple (*Acer saccharum*) and are projected for red maple (*Acer rubrum*) and yellow birch (*Betula alleghaniensis*) (Allen and others, 2015; Bapikee, 2013; Boakye and others, 2023; Broderick, 2018; Coble and others, 2017; Dey, 2014; Dey and Kabrick, 2015; Moore and others, 2016). Even where mature trees persist, repeated establishment failure could gradually decrease mesic species abundance. Oaks (*Quercus* spp.), with generally higher drought tolerance, may increase in abundance as mesic competitors decline (Frelich and others, 2017; McEwan and others, 2011; Rauschendorfer and others, 2022; Volder and others, 2010; Vose and others, 2016). Species such as black oak (*Q. velutina*), blackjack oak (*Q. marilandica*), and post oak (*Q. stellata*) are especially well-adapted to dry, resource-poor sites and often benefit when disturbance from drought reduces competition from mesic trees (Abrams, 1990; Alexander and others, 2021; Dey and others, 2017; Nowacki and Abrams, 1992; Prolic and Goldblum, 2016; Spetich and others, 2022). On dry upland sites, as droughts intensify, oaks and other xeric species like pines (*Pinus* spp.) are particularly likely to displace mesic taxa (Clark and others, 2016; Handler and others, 2014a; Iverson and others, 2008b; Toot and others, 2020; Vose and others, 2016).

However, increased oak dominance under future drought is not assured. Severe drought can drive oak mortality and inhibit regeneration (Asbjornsen and others, 2021; Au and Maxwell, 2022; Camarero and others, 2021; Druckenbrod and others, 2019; Novick and others, 2022; Palus and others, 2018), particularly in drier regions where oaks may already be at their hydrologic limits (Au and Maxwell, 2022; Prolic and Goldblum, 2016; Swanston and others, 2018). Outcomes also depend on disturbance context. If drought co-occurs with other stressors, oaks and other xeric species like hickory (*Carya* spp.) may be at a competitive disadvantage to mesic species that recover more quickly and capitalize on post-disturbance resource availability (Arthur and others, 2012; Batllori and others, 2020; Cowden and others, 2014; Palus and others, 2018; Volder and others, 2010). Compared to their mesic associates, some oak

species like white oak (*Quercus alba*) and post oak (*Quercus stellata*) more frequently exhibit asymmetric growth responses to precipitation variability (Dannenberg and others, 2019; Lockwood and others, 2023; Maxwell and others, 2024).

Shifts in overstory dominance may favor shrubs able to withstand prolonged moisture deficits (Abella and others, 2020; Archer and others, 2017; Batllori and others, 2020; Breshears and others, 2016). Species such as eastern red cedar (*Juniperus virginiana*), roughleaf dogwood (*Cornus drummondii*), and smooth sumac (*Rhus glabra*) are highly drought-tolerant, and when tree and herbaceous competition is weakened during dry years, can expand in these open-canopied conditions (Ansley and others, 2023; Ganguli and others, 2016; Hanberry, 2022; O'Keefe and others, 2024; Volder and others, 2010; Wedel and others, 2025). Due to its high tolerance of hydraulic stress and competitive advantage in disturbed sites, eastern red cedar may outperform oaks under drought (Ganguli and others, 2016; Hanberry and others, 2012; Volder and others, 2010). Once established, shrubs can reinforce their dominance by shading the understory and altering soil moisture, potentially suppressing the recruitment of trees and herbs in favor of their own seedlings (Hou and others, 2025; Londe and others, 2022; Ratajczak and others, 2011; Riddle and others, 2014).

Understory communities are also likely to shift under increasing drought. In some instances, drought may reduce the competitiveness of grasses compared to trees, such as when deeper rooting depths allow trees to more readily survive drought periods (Holdo and Nippert, 2023; Polley and others, 2013a; Volder and others, 2010). In other instances, drought may promote grass abundance – e.g., when increasingly variable precipitation patterns, particularly heavy precipitation events, accelerate transitions from drought to wet conditions, grasses and other species with rapid water use strategies may more effectively utilize resource pulses to quickly recover from droughts (Davis and others, 1999; Frelich and others, 2017; Holdo and Nippert, 2023; Peterson and Reich, 2001; Wigley and others, 2024). Overall, drought is expected to drive "xerophication" of the understory, increasing the abundance of drought-adapted grasses and prairie species (Batllori and others, 2020; Cavender-Bares and Reich, 2012; Galatowitsch and others, 2009; Nippert and others, 2009; Randall and Walters, 2019). Warm-season (C_4) grasses such as big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and Indiangrass (*Sorghastrum nutans*) are especially likely to benefit under moderate drought; however, more extreme drought may instead favor deep-rooting forbs that can access soil water at greater depths or cool-season (C_3) grasses that avoid late-summer drought stress by growing earlier in the season (Havrilla and others, 2023; Knapp and others, 2024; Taylor and others, 2011; Volder and others, 2010).

At the landscape scale, cumulative drought effects may contribute to broader ecological transformations. In fire-suppressed or fragmented landscapes, repeated drought could catalyze transitions toward shrublands (Breshears and others, 2016; Smith and Goetz, 2021; Toot and others, 2020). More broadly, increasing drought may drive long-term shifts in the prairie-forest boundary, with oak woodlands and savannas occupying an increasingly narrow band between grassland and closed-canopy forest (Barros and others, 2018; Breshears and others, 2016; Danz and others, 2013; Fei and others, 2017; Nelson and others, 2006; Toot and others, 2020; Wang, 2020).

Community responses to fire. Changing fire regimes under climate change are also expected to alter species composition in the Central Interior Oak Forest, Woodland and Savanna ecosystem, particularly where historical suppression has allowed fire-sensitive species to expand (Batllori and others, 2020; Nowacki and Abrams, 2008; Peterson and Reich, 2001; Stambaugh and others, 2021; Vander Yacht and others, 2019). The return of frequent, low-intensity surface fires is likely to promote more fire-adapted assemblages by maintaining open canopy structures and balancing herbaceous and woody cover (Fowler

and Beckage, 2019; Frelich and others, 2017; Gold and others, 2023; Lettow and others, 2014; Peterson and Reich, 2001; Stephen and others, 2024). Fire-adapted trees such as white oak (*Quercus alba*), black oak (*Q. velutina*), shagbark hickory (*Carya ovata*), and shortleaf pine (*Pinus echinata*), typically resist fire-induced stem mortality and recover following fire, enabling them to retain or expand canopy dominance at the expense of fire-sensitive mesic species like sugar maple *(Acer saccharum*) (Arthur and others, 2021; Bataineh and others, 2022; Iverson and others, 2017; Schweitzer and others, 2023).

In areas where fire has been excluded, drought-tolerant shrubs and encroaching trees such as eastern red cedar (*Juniperus virginiana*), roughleaf dogwood (*Cornus drummondii*), and smooth sumac (*Rhus glabra*) have expanded (Hanberry and others, 2012; Meneguzzo and Liknes, 2015; O'Keefe and others, 2024; Ratajczak and others, 2016). More frequent and intense fires, especially when coupled with drought, may suppress some of these woody species (Breshears and others, 2016; Dey and others, 2019; Grau-Andrés and others, 2024; Meunier and others, 2021; Ratajczak and others, 2014; Ratajczak and others, 2019; Stambaugh and others, 2021; Twidwell and others, 2016). However, where shrubs have become well established, without additional disturbance or management interventions, more frequent fire may be insufficient to reverse encroachment (Keen and others, 2024a; Meunier and others, 2021; O'Keefe and others, 2024; Ratajczak and others, 2011; Ratajczak and others, 2014). Shrubs may reinforce their own dominance by reducing fine fuel availability and altering microclimate, limiting fire spread and severity (Capozzelli and others, 2020; Dey and Kabrick, 2015; Hanberry and others, 2012; Keen and others, 2024a; Li and others, 2022; Meunier and others, 2021).

The abundance and diversity of most herbaceous species, by contrast, tend to benefit from fire (Arthur and others, 2021; Beebe and others, 2024; Gold and others, 2023; Grau-Andrés and others, 2024; Peterson and Reich, 2008). Warm-season (C_4) grasses such as big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*) resprout quickly after fire and may become dominant in the understory—benefiting from canopy gaps and increased resource availability (Brooks and others, 2004; Peterson and Reich, 2001; Steinberg, 2002; Uchytil, 1988).

The impact of fire on community composition is inherently complex, with multiple, context-dependent factors producing a range of possible outcomes. At the landscape scale, changes in fire regimes may drive compositional transformation between ecosystem types. In forested systems, where fire reduces mesophyte dominance, it may facilitate habitat "savannafication" by favoring open-canopy, fire-adapted species assemblages (Brandt and others, 2014; Frelich and others, 2015; Gustafson and Sturtevant, 2013; Handler and others, 2014b). Conversely, where fires fail to overcome local site conditions created by mesic species that limit its spread and severity, continued mesophytic dominance and poor oak recruitment may promote closed canopies (Arthur and others, 2012; Bassett and others, 2020; Brose and others, 2006; Brudvig and Asbjornsen, 2008; Frelich and others, 2015; Gold and others, 2023; Hutchinson and others, 2024; Nowacki and Abrams, 2008; Palus and others, 2018; Pellegrini and others, 2021b; Peterson and Reich, 2008). In more open systems, however, increasingly frequent or severe fire may suppress tree regeneration altogether, favoring a transition toward grass-dominated communities (Frelich and others, 2017; Gold and others, 2023; Hanberry and Noss, 2022; Mandle and others, 2011; Peterson and Reich, 2001; Thom, 2023). These outcomes are likely to vary across gradients of soil moisture, topography, and fire history, and may interact with concurrent climate stressors like drought to push systems toward alternative stable states, potentially shifting the distribution of these habitats across the broader Midwest landscape (Breshears and others, 2016; Changnon and others, 2002; Collins, 2000; Galatowitsch and others, 2009; Thom, 2023; Toot and others, 2020).

Invasive Species

Invasive species responses to drought. Drought is likely to exacerbate invasion pressure in the Central Interior Oak Forest, Woodland and Savanna ecosystem. Many invasive species have traits, such as broad environmental tolerance, rapid growth, or fast recovery after disturbance, that allow them to gain a competitive edge over native species during and after drought (Clements and Jones, 2021; Duell and others, 2016; Duell and others, 2021; Kaushik and others, 2022; Poland and others, 2021; Sobuj and others, 2024; Zhang and others, 2023). Invasion risk is especially high in disturbed or nutrient-rich soils, where post-drought pulses of nitrogen can enhance invasive species recovery (Schrama and Bardgett, 2016; Zhang and others, 2023). Grasses such as cogongrass (*Imperata cylindrica*), Nepalese browntop (*Microstegium vimineum*), and smooth brome (*Bromus inermis*) are expected to remain competitive under drought and may expand under future climate conditions, often at the expense of regenerating native trees (Howard, 2005; Kirwan and others, 2021; Kroeger and Otfinowski, 2025; Touchette and Romanello, 2010; Warren and others, 2011). Invasive woody shrubs, such as common buckthorn (*Rhamnus cathartica*), may also accelerate their spread through both direct drought tolerance and enhanced dispersal and regeneration following extreme events (Batllori and others, 2020; Clements and Jones, 2021; Knight and others, 2007). Further, warming may enable species currently limited to the southern Midwest, such as kudzu (*Pueraria montana*), to expand northward, and, because kudzu is highly drought-tolerant, its spread could further increase invasion pressure (Bradley and others, 2010; Butler and others, 2015; Munger, 2002b).

Invasive species responses to fire. More frequent and intense fire may affect invasive species in complex and sometimes opposing ways in the Central Interior Oak Forest, Woodland and Savanna. In some cases, fire can control or suppress woody invaders, particularly when fires occur late in the growing season when topkill is most effective (Beebe and others, 2024; Meunier and others, 2021; Vander Yacht and others, 2017) or if fires are especially frequent or severe (DiTomaso and Johnson, 2006; Zouhar and others, 2008). However, fire rarely eliminates invasive species altogether, as many resprout vigorously or persist through soil seed banks (Brooks and others, 2009; Mandle and others, 2011; Zouhar and others, 2008). For instance, common buckthorn (*Rhamnus cathartica*) and invasive honeysuckles (*Lonicera* spp.) often survive topkill and resprout prolifically after fire (Illinois Department of Natural Resources, 2017), and pyrophytic shrubs such as autumn olive (*Elaeagnus umbellata*) may even increase in abundance under frequent or severe fire (Culbert and others, 2025; Stavi, 2019).

Invasive species can also alter fire behavior. Highly flammable invasive grasses such as cogongrass (*Imperata cylindrica*), Nepalese browntop (*Microstegium vimineum*), and smooth brome (*Bromus inermis*) add fine fuels that promote fire spread and intensity (Fusco and others, 2019; Kerns and others, 2020; Salemme and Fraterrigo, 2021; Stambaugh and others, 2021). Following fires, these grasses rapidly colonize burned areas, displacing native understory plants and reinforcing feedbacks that sustain their dominance (Donovan and others, 2023; Fusco and others, 2019; Kerns and others, 2020). Conversely, some woody invaders such as privet (*Ligustrum* spp.) and common buckthorn may suppress fire by shading out herbaceous fuels and shifting regimes toward less frequent but more intense crown fires (Brooks and others, 2004; Dibble and others, 2008; Mandle and others, 2011; Wang and others, 2016; Zouhar, 2011).

Invasives that tolerate both fire and drought, such as tree-of-heaven (*Ailanthus altissima*) and invasive honeysuckles, may be particularly problematic under future climate scenarios, where interacting disturbances are likely to amplify their spread (Butler and others, 2015; Dey and others, 2017; Stambaugh and others, 2021).

Pests and Pathogens

General pest and pathogen responses to climate change. Through shifts in temperature and host susceptibility, climate change is expected to alter pest and pathogen dynamics in the Central Interior Oak Forest, Woodland and Savanna ecosystem. Warmer conditions can accelerate insect development and reproduction, increasing the potential for more frequent or severe outbreaks, particularly among butterflies and moths (Lepidoptera spp.) and bark beetles (Scolytidae spp.) (Dukes and others, 2009; Pureswaran and others, 2018; Stanturf and others, 2024). Depending on their life history constraints, extreme events or altered seasonality may also limit some pest species. For example, heatwaves can reduce moth larvae survival; additionally, phenological desynchronization between forest tent caterpillars (*Malacosoma disstria*) and birch (*Betula* spp.) hosts can reduce caterpillar populations (Bracalini and others, 2024; Flower and Gonzalez-Meler, 2015; Harvey and others, 2020; Lehmann and others, 2020; Mironidis and Savopoulou-Soultani, 2010; Pureswaran and others, 2018; Schwartzberg and others, 2014). Similarly, some soil-borne pathogens may also become more virulent under elevated temperatures, while others (e.g., *Phytophthora* spp., *Ceratocystis* spp.) may face thermal limits that constrain their spread (Stanturf and others, 2024; Tooley and others, 2008). Climate change can also affect pests and pathogens indirectly by weakening host defenses. Shifts in host physiology and nutrient allocation under stress may weaken plant defenses (Cole and others, 2021; Dukes and others, 2009), while disruption of synchrony between pests and their competitors may exacerbate their impacts (Pureswaran and others, 2018). Pathogens like root rots (*Armillaria* spp.) and bur oak blight (*Tubakia iowensis*) may become especially damaging where climate stress both favors the disease and reduces host resistance to disease (Harrington and McNew, 2016; Vose and others, 2016; Dukes and others, 2009).

Pest responses to drought. Drought may intensify pest pressure across the Central Interior Oak Forest, Woodland and Savanna ecosystem by weakening host defenses. Water stress can compromise tree resistance mechanisms (Canelles and others, 2021; Dey and others, 2019; Dukes and others, 2009; Jactel and others, 2012; Kolb and others, 2016; Pureswaran and others, 2018), increasing vulnerability to pests such as borers and defoliators that target weakened hosts (Butler and others, 2015; Haynes and others, 2014; Jactel and others, 2012; Kolb and others, 2016; Stanturf and others, 2024). Several cambium-feeding insects, including the emerald ash borer (*Agrilus planipennis*) and two-lined chestnut borer (*Agrilus bilineatus*), exploit drought-stressed trees and can cause extensive mortality in susceptible species (Anderegg, 2015; Fierke and others, 2007; Haavik and others, 2015; Rauschendorfer and others, 2022; Stanturf and others, 2024; Stump and others, 2024; White, 2015). Drought has also been linked to periodic outbreaks of forest tent caterpillar (*Malacosoma disstria*) and other native defoliators, which can cause widespread canopy damage and reduced growth in already stressed stands (Bragard and others, 2022; Fierke and others, 2007; Haavik and others, 2015; Kolb and others, 2016; Stump and others, 2024). In addition, the fungus *Entomophaga maimaiga*, which normally helps limit outbreaks of spongy moth (*Lymantria dispar*), is less effective under dry conditions, potentially increasing defoliation (Reilly and others, 2014).

Oaks (*Quercus* spp.) are among the most vulnerable to pests under prolonged or repeated drought. In recent decades, severe drought events have been implicated in oak decline, mortality, and regeneration failure caused by insect pests and secondary pathogens (Ansley and others, 2023; Dey and others, 2019; Haavik and others, 2015; Swanston and others, 2018; Wood and others, 2018). Water deficits not only impair oak defense systems but can elicit physiological cues that attract wood-boring beetles (Canelles and others, 2021; Kolb and others, 2016). As a result, pests such as the red oak borer (*Enaphalodes*

rufulus) and two-lined chestnut borer can cause widespread damage and mortality in drought-stressed red (*Q. rubra*) and black oaks (*Q. velutina*) (Fierke and others, 2007; Haavik and others, 2012; Haavik and others, 2015; Stump and others, 2024; USDA Forest Service, USA, 2002).

However, not all drought-pest interactions are amplifying. By lowering forage quality and quantity, drought may also inhibit certain pest populations (Jactel and others, 2012; Kolb and others, 2016; Pureswaran and others, 2018; Stanturf and others, 2024). Water-stressed plants often produce tougher leaves with lower nitrogen content, reducing insect growth, reproduction, and survival (Anderegg, 2015; Canelles and others, 2021; Huberty and Denno, 2004; Jactel and others, 2012; Kolb and others, 2016). Overall, the net effect of drought on pest dynamics will depend on complex interactions between tree condition, pest life history, and the timing and duration of dry periods.

Pathogen responses to drought. Many pathogens also show complex and often contrasting responses to drought in the Central Interior Oak Forest, Woodland and Savanna ecosystem. As with pests, water stress can compromise resistance mechanisms, including immune function, making trees more susceptible to fungal and bacterial infection (Ansley and others, 2023; Dey and others, 2019; Dukes and others, 2009; Francioli and others, 2020; Kolb and others, 2016; Kumar and Mukhopadhyay, 2025; Pozner and others, 2022; Stanturf and others, 2024; Sturrock and others, 2011). Root rots (*Armillaria* spp.) and water molds (*Phytophthora* spp.) can become particularly problematic under fluctuating soil moisture regimes, where cycles of drought and rewetting promote infection and spread (Haavik and others, 2015; Kolb and others, 2016; Kubiak and others, 2017; Lockman and Kearns, 2016; Sturrock and others, 2011; Wood and others, 2018). Drought can also intensify foliar diseases, with powdery mildew (*Erysiphales* spp.) and leaf spot diseases (e.g., *Taphrina* spp.) able to gain a foothold in stressed plants (Gosling and others, 2024; Oliva and others, 2014; Sturrock and others, 2011; Warren and Mordecai, 2010).

Similarly to pest dynamics, drought-stressed oaks (*Quercus* spp.) are especially vulnerable to pathogens. Hypoxylon canker (*Biscogniauxia atropunctata*), for example, benefits from warm, dry conditions, remaining latent in healthy oaks but causing disease and dieback when drought weakens host defenses (Ansley and others, 2023; Wood and others, 2018, p. 2). Such interactions highlight how more frequent droughts, amplifying both pest and pathogen impacts, may undermine long-term oak dominance. However, by disrupting the moisture conditions required for survival, reproduction, or dispersal, drought can also suppress some pathogens (Delavaux and others, 2021; Jactel and others, 2012; Kolb and others, 2016; Vose and others, 2016), interrupting pathogen life cycles or delaying outbreaks.

Pest and pathogen responses to fire. As with drought, fire can influence pest and pathogen dynamics in the Central Interior Oak Forest, Woodland and Savanna ecosystem in variable ways. On one hand, fire reduces forest density and alters age structure, creating younger stands that may be less susceptible to certain insect outbreaks (Fettig and others, 2022; Holzmueller and others, 2008; Lombardero and Ayres, 2011; McCullough and others, 1998; Parker and others, 2006). Fire may also hinder pathogen spread by thinning host populations and altering microclimate conditions (Beals and others, 2022; Hart and others, 2005; Parker and others, 2006; Pellegrini and others, 2021a; Simler-Williamson and others, 2019). However, fire can also create opportunities for pests and pathogens. Fire-damaged trees may be more vulnerable to colonization, as injury can weaken defenses and release compounds that attract bark beetles and other insects (Bär and others, 2019; Battisti and Larsson, 2023; Canelles and others, 2021; Catry and others, 2017; Fettig and others, 2022). Post-fire, dead or dying trees may also attract insect pests and serve as breeding substrates for wood-boring insects (Bell, 2023; Fettig and others, 2022; Hjältén and others, 2018; New, 2014; Swengel, 2001; Wikars, 1997). Insect outbreaks themselves can influence fire dynamics. High pest-related mortality can increase canopy and surface fuel loads, raising

fire risk in some conditions (Anderegg and others, 2022; Canelles and others, 2021; Clark and others, 2017; Fettig and others, 2022; Shaw and others, 2022).

Herbivory

General herbivory responses to climate change. Climate change may amplify herbivory pressure in the Central Interior Oak Forest, Woodland and Savanna ecosystem, particularly from white-tailed deer (*Odocoileus virginianus*). Warming winters and reduced snowpack are expected to improve deer survival, contributing to higher populations across much of the Midwest (Côté and others, 2004; Felton and others, 2024; MDNR, 2016; Patton and others, 2021; Weiskopf and others, 2019). At high densities, deer can suppress regeneration of preferred browse species, particularly oaks (*Quercus* spp.), whose seedlings and saplings are preferentially browsed during these vulnerable stages (Blossey and others, 2019; Hopkins-Vanzant and Miyanishi, 1993; Miller and others, 2023; Patton and others, 2021; Sample and others, 2023). Selective browsing also reduces the abundance and reproductive output of native forbs, facilitating the persistence of introduced perennial grasses (Anderson and others, 2001; Batzli and Dejaco, 2013; DiTommaso and others, 2014; Nisi and others, 2015; Pruszenski and Hernández, 2020).

Herbivory responses to drought and fire. While ungulates can sometimes help reduce fuel loads or limit woody encroachment in some fire-prone systems (Hanberry and Faison, 2023; Hobbs, 1996; Pruszenski and Hernández, 2020; Rouet-Leduc and others, 2021), their long-term impact on vegetation structure and diversity in the Central Interior Oak Forest, Woodland and Savanna may depend on how browsing pressure interacts with climate and fire. Drought stress reduces plant investment in chemical defenses and narrows the range of species that meet deer's nutritional needs, making palatable and nutritious species even more susceptible to browsing (Duell and others, 2024; Laoué and others, 2024; Lashley and Harper, 2012). Deer browsing may also diminish beneficial impacts from fire on oak (*Quercus* spp.) regeneration—limiting the resprouting potential of oak seedlings exposed to chronic browsing before fire and by counteracting the competitive advantage of newly sprouting oaks in post-fire conditions (Albrecht and McCarthy, 2006; Collins and Carson, 2003; Johnson and others, 2019; Nuttle and others, 2013; Perles and others, 2021). Deer may also preferentially browse in burned sites, further reducing recruitment post-fire (Calhoun and others, 2024; Masters and others, 1993; Royo and others, 2010).

Climate Change Impacts by Habitat

Interior Low Plateau Oak Forest, Woodland and Savanna

The Interior Low Plateau Oak Forest, Woodland and Savanna habitat occurs across southern Indiana, Illinois, and Ohio, a few small areas of Missouri, and into Kentucky (Figure 3), typically on dry ridges and uplands with thin, rocky, and low-fertility soils (Faber-Langendoen and others, 2020; USNVC, 2025). Communities are dominated by oaks (*Quercus* spp.) and hickories (*Carya* spp.), accompanied by a rich ground flora of prairie and woodland species (Table 1) (Dey and Kabrick, 2015; Fralish, 2004; Paciorek and others, 2021; The Nature Conservancy, 1995; USNVC, 2025). Commonly occurring summer droughts, particularly on shallow soils and together with nutrient limitations, influence community composition (Au and Maxwell, 2022; Brandt and others, 2014; Clark and others, 2016; Sutherland and Hutchinson, 2003a; USNVC, 2025).

Canopy structure varies along gradients of fire history and soil moisture. Savannas historically maintained sparse tree cover (10–30%) and supported a rich herbaceous layer, while woodlands had greater canopy cover (up to 80%) but still allowed sufficient light for diverse understories (Fowler and Beckage, 2019; Homoya and Abrekk, 1983; Law and others, 1994; Paciorek and others, 2021; Taft, 1997). Fire, recurring approximately every 3–12 years, played a central role in limiting woody encroachment, maintaining open canopies, and creating favorable conditions for oak regeneration (Guyette and others, 2003; Hutchinson and others, 2008; McClain and others, 2010; Peterson and Reich, 2001; Stambaugh and others, 2020; Stambaugh and others, 2021).

Over the past century, fire suppression, land conversion, and altered disturbance regimes have driven widespread canopy closure and compositional change. Reduced light availability has favored mesic, fire-intolerant species like maple (*Acer* spp.), beech (*Fagus* spp.), and eastern red cedar *(Juniperus virginiana*) (Abrams and others, 2021; Alexander and others, 2021; Darling, 2021; Nowacki and Abrams, 2008; Sutherland and Hutchinson, 2003a; USNVC, 2025). Consequently, more than 98% of historical Interior Low Plateau Oak Forest, Woodland and Savanna habitat area has been lost from the region, and many remaining areas are at risk of transforming into closed-canopy forest (Dey and Kabrick, 2015; Fowler and Beckage, 2019; Fralish, 2004; Hanberry and others, 2017; Nuzzo, 1986; Pile Knapp and others, 2024).

Table 1. Plant species common to the Interior Low Plateau Oak Forest, Woodland and Savanna habitat. Asterisks (*) denote species introduced to the region. Taxonomic authority, common name, growth habit, and native status are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Carya tomentosa (Lam.) Nutt.	Mockernut hickory	Tree
Quercus alba L.	White oak	Tree
Quercus coccinea Münchh.	Scarlet oak	Tree
Quercus falcata Michx.	Southern red oak	Tree
Quercus montana Willd.	Chestnut oak	Tree
Quercus rubra L.	Northern red oak	Tree
Quercus stellata Wangenh.	Post oak	Tree
Quercus velutina Lam.	Black oak	Tree

Species	Common name	Growth habit
Carex picta Steud.	Boott's sedge	Graminoid
**Lonicera japonica* Thunb.	Japanese honeysuckle	Vine
Polystichum acrostichoides (Michx.) Schott	Christmas fern	Forb/herb

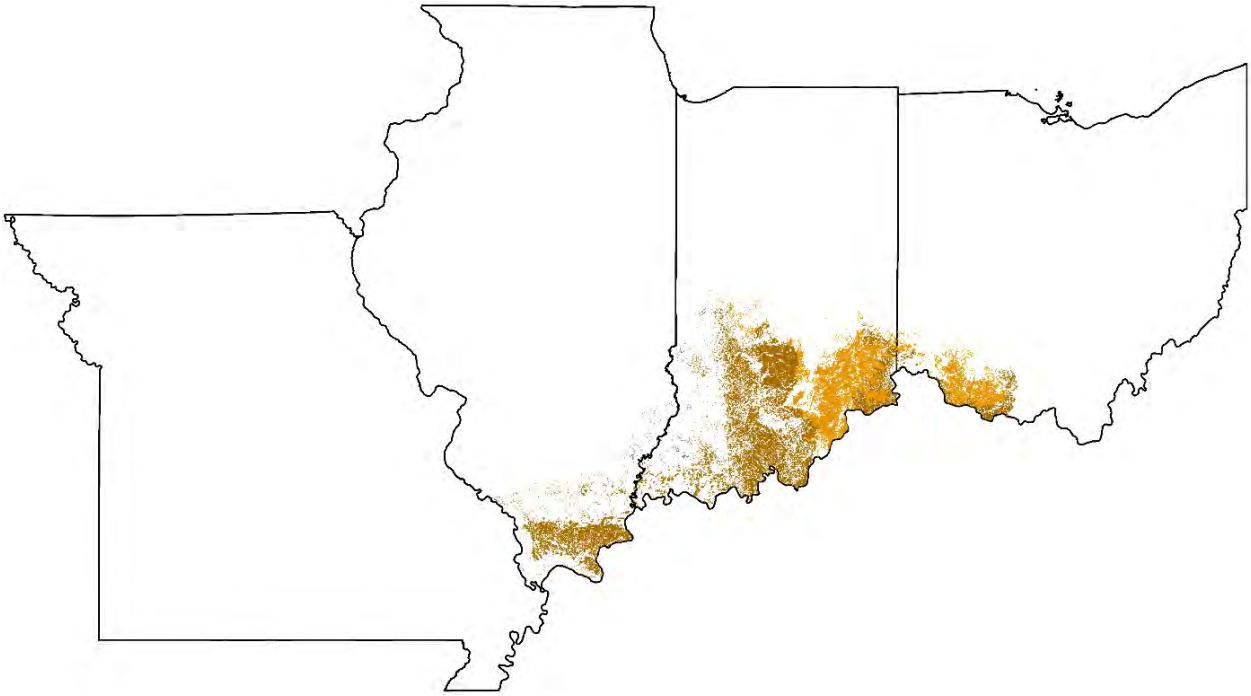

Figure 3. Interior Low Plateau Oak Forest, Woodland and Savanna habitat distribution. Geographic distribution of the Interior Low Plateau Oak Forest, Woodland and Savanna habitat (Group Code: G950) in the Midwest (USNVC, 2020). Dark orange areas indicate Interior Low Plateau Oak Forest, Woodland and Savanna habitat in the Interior Plateau ecoregion, while light orange areas mark additional locations throughout the Midwest. Habitat boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The Interior Low Plateau Oak Forest, Woodland and Savanna habitat is prioritized for the Interior Plateau ecoregion (Table i), and climate projections for this area largely reflect regional trends. Both average and extreme temperatures are expected to rise throughout the coming century, with model projections indicating a 9.6°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 annual precipitation, averaging 2.8 additional inches per year, although some indicate a slight decline (Table 3) (Marvel and others, 2023). However, precipitation is expected to become more variable, with more extreme events and notably drier summers despite the overall increase (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 2. Projected changes in annual temperature in the Interior Plateau 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	54.5°F	56.7°F (2.2)	57.8°F (3.3)	57.3°F (2.8)	60.0°F (5.4)
Mean of models	54.6°F	59.1°F (4.5)	60.6°F (6.0)	60.0°F (5.4)	64.2°F (9.6)
Maximum model	54.4°F	60.6°F (6.1)	62.5°F (8.0)	62.6°F (8.2)	68.1°F (13.6)

Table 3. Projected changes in annual precipitation in the Interior Plateau 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	42.1 in	43.7 in (4%)	39.7 in (-6%)	45.6 in (8%)	39.4 in (-6%)
Mean of models	42.6 in	44.7 in (5%)	44.8 in (5%)	44.3 in (4%)	45.4 in (7%)
Maximum model	42.5 in	44.3 in (4%)	46.3 in (9%)	47.4 in (11%)	48.9 in (15%)

Key Climate Change Stressors

Drought and fire. Across the Interior Plateau ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 85% above historical conditions by the end of the century, equating to more than 1 kPa of additional VPD (RCP 8.5; Table 4). The highest values are expected in the southern part of the region, peaking in southeastern Missouri (Figure 4). Higher VPD means more rapid evaporation and water loss from plants and soils, which can intensify drought conditions. Annual extreme fire danger is also projected to increase by approximately 132% by mid-century under the high emissions scenario, adding more than 14 days of elevated risk and bringing the total to nearly a month each year (Table 5). As with VPD, the greatest fire risk is projected in the southern part of the region, with the maximum model indicating portions of southern Missouri, Illinois, and Indiana could experience more than 80 days of extreme fire risk annually (Figure 5). Rising fire risk could increase the frequency of wildfire and push this habitat toward more historically frequent fire intervals, but it also raises the likelihood of extreme events, including crown fires, that exceed historical norms and constrain opportunities to safely apply prescribed fire.

Table 4. Projected changes in summer vapor pressure deficit in the Interior Plateau 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

summer vapor pressure deficit (VPD; kilopascals) 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 VPD 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	1.18 kPa	1.21 kPa (3%)	1.32 kPa (12%)	1.29 kPa (10%)	1.39 kPa (18%)
Mean of models	1.18 kPa	1.56 kPa (32%)	1.75 kPa (48%)	1.66 kPa (41%)	2.19 kPa (85%)
Maximum model	1.19 kPa	1.85 kPa (56%)	2.07 kPa (74%)	2.27 kPa (91%)	3.42 kPa (187%)

Table 5. Projected changes in annual extreme fire danger in the Interior Plateau 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 extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) by mid-century (2040–2069), 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 annual extreme fire danger under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069	
	Historical	RCP 4.5	RCP 8.5
Minimum model	11.0	9.8 (-11%)	10.5 (-5%)
Mean of models	11.0	19.2 (75%)	25.5 (132%)
Maximum model	11.0	21.9 (100%)	53.4 (387%)

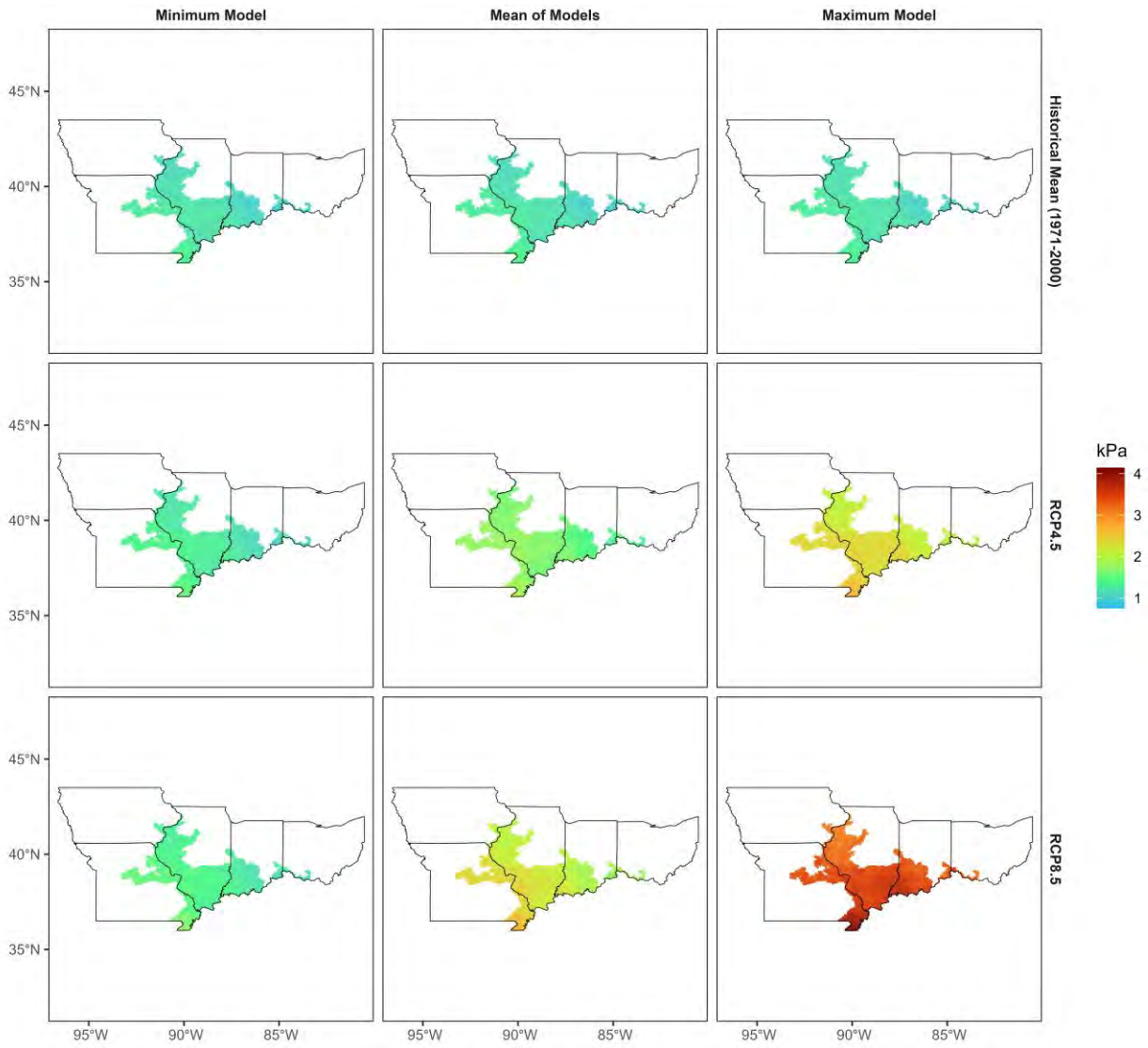

Figure 4. Projected changes in summer vapor pressure deficit in the Interior Plateau ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for 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) summer vapor pressure deficit (VPD; kilopascals) 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 VPD under RCP 8.5 are presented in addition to the mean of all models (middle column).

Figure 5. Projected changes in annual extreme fire danger in the Interior Plateau 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 (2040–2069) annual extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) 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 extreme fire danger under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

Climate change is affecting the Interior Low Plateau Oak Forest, Woodland and Savanna habitat through its overall influence on species distributions, adaptive capacity, and vulnerability, as well as through the specific impacts of drought and fire (Table 6). 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:

- Drought can intensify soil drying, reduce herbaceous cover, and increase erosion risk, particularly when followed by heavy rainfall on steep, rocky slopes (Breshears and others, 2016; Vose and others, 2016; Ziegler and others, 2008).
- Drought may alter nutrient dynamics by reducing plant uptake, triggering post-drought nutrient flushes, and destabilizing nutrient availability (Hartmann and others, 2013; Homyak and others, 2017; Reinsch and others, 2024; Schlesinger and others, 2016).
- Low-intensity fire maintains herbaceous layers and promotes nutrient cycling through ash deposition, while high-intensity fire can expose mineral soil, degrade vegetation structure, and increase erosion risk (Arthur and others, 2021; Fernandez and others, 2006; Hanberry and others, 2017; Knicker, 2007; Pellegrini and others, 2020; Singh and others, 2025).
- Nutrient availability often increases immediately following fire, but repeated fire may gradually deplete soil carbon and nitrogen pools in the long term (Knicker, 2007; Pellegrini and others, 2020; Singh and others, 2025).

Community composition:

- Drought may shift communities toward xeric-adapted species, particularly drought-tolerant warm-season (C_4) grasses such as big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*), while reducing drought-sensitive forbs and trees (Batllori and others, 2020; Galatowitsch and others, 2009; Havrilla and others, 2023; Volder and others, 2010).
- Mesophytic species like sugar maple (*Acer saccharum*) are likely to decline as drought increases recruitment failure and canopy mortality (Boakye and others, 2023; Dey and Kabrick, 2015).
- Oaks (e.g., *Quercus alba*, *Q. velutina*, *Q. stellata*) and hickories (e.g., *Carya ovata*) may maintain or expand dominance under moderate drought due to deep rooting, though this advantage may decline under severe or prolonged drought (Au and Maxwell, 2022; Dey and others, 2017; Novick and others, 2022).
- In the absence of fire, drought may favor the expansion of shrubs and encroaching trees such as eastern red cedar (*Juniperus virginiana*), roughleaf dogwood (*Cornus drummondii*), and smooth sumac (*Rhus glabra*), which compete with herbaceous species and suppress future fire spread (Hanberry and others, 2012; O'Keefe and others, 2024; Volder and others, 2010).
- Interactions among repeated drought and fire may catalyze transitions toward either grassland, shrubland, or closed-canopy forest, depending on disturbance history and site context (Breshears and others, 2016; Smith and Goetz, 2021; Toot and others, 2020).

Invasive species:

- Drought can amplify invasion pressure by weakening native vegetation and creating open microsites where invasive grasses and forbs such as smooth brome (*Bromus inermis*) and Nepalese browntop (*Microstegium vimineum*) rapidly establish and outcompete natives (Duell and others, 2021; Fusco and others, 2019; Zhang and others, 2023).
- Woody invaders like common buckthorn (*Rhamnus cathartica*) and invasive honeysuckles (*Lonicera* spp.) tolerate drought and recover quickly after disturbance, posing a long-term threat to native regeneration and diversity (Batllori and others, 2020; Knight and others, 2007).
- Fire may suppress some invasive shrubs if applied frequently or during the late growing season, but it can also favor pyrophytic invaders such as autumn olive (*Elaeagnus umbellata*) (Culbert and others, 2025; Vander Yacht and others, 2017).
- Flammable invasive grasses like cogongrass (*Imperata cylindrica*) and smooth brome (*Bromus inermis*) can reinforce feedbacks that increase fire intensity and frequency, perpetuating their dominance in burned areas (Fusco and others, 2019; Kerns and others, 2020).

Pests and pathogens:

- Drought weakens host defenses, increasing vulnerability to pests such as twolined chestnut borer (*Agrilus bilineatus*) and forest tent caterpillar (*Malacosoma disstria*), particularly in red oak (*Quercus rubra*) and black oak (*Q. velutina*) (Haavik and others, 2015; Stump and others, 2024).
- Root- and stem-infecting pathogens, including Hypoxylon canker (*Biscogniauxia atropunctata*), root rots (*Armillaria* spp.), and water molds (*Phytophthora* spp.), become more damaging under fluctuating moisture regimes and added host stress, potentially accelerating oak decline (Wood and others, 2018; Kubiak and others, 2017).
- Drought may inhibit some pests and pathogens by lowering forage quality or disrupting life cycles (Delavaux and others, 2021; Kolb and others, 2016).
- Fire may indirectly reduce pest and pathogen pressure by thinning host populations, but fire-damaged trees can also attract bark beetles and wood borers (Barkley, 2006; Canelles and others, 2021; Jenkins and others, 2014; Progar and others, 2014).

Herbivory:

- Warmer winters may increase white-tailed deer (*Odocoileus virginianus*) populations, intensifying browsing pressure (Champagne and others, 2021; Felton and others, 2024; Frelich and others, 2015; Hanberry and Abrams, 2018; Weiskopf and others, 2019).
- Higher deer densities may suppress regeneration of preferred browse species such as oaks (*Quercus* spp.) and native forbs, especially in drought-stressed or severely burned habitats where defenses are weakened (Champagne and others, 2021; Duell and others, 2024).
- Selective browsing can reduce native forb diversity and facilitate the dominance of introduced perennial grasses, altering understory composition and fire regimes (DiTommaso and others, 2014; Urbanek and others, 2012).

Table 6. Summary of impacts from key climate change stressors on Interior Low Plateau Oak Forest, Woodland and Savanna habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory impacts (e.g., showing both high and low tolerance to drought), reflecting conflicting findings in the literature.

Stressor	Impact	Affected species
Drought	When tolerance of droughts is high, competitive advantage and/or increased abundance	*Polystichum acrostichoides*[1]
		Quercus alba[2–10]
		Quercus coccinea[10–12]
		Quercus falcata[10,13]
		Quercus rubra[2,4,7,12,14,15]
		Quercus stellata[10,13,16–19]
		Quercus velutina[12,16]
		**Rhamnus cathartica*[20]
	When tolerance of droughts is low, reduced growth and/or abundance	**Berberis thunbergii*[13]
		Carya tomentosa[21]
		**Lonicera japonica*[22]
		**Lonicera maackii*[13]
		Quercus alba[21,23–25]
		Quercus coccinea[21]
		Quercus rubra[8,21,26–30]
		Quercus stellata[8,31]
		Quercus velutina[8,13,32]
Fire	When tolerance of fire is high, competitive advantage and/or increased abundance	*Carya tomentosa*[33]
		**Lonicera japonica*[13]
		**Lonicera maackii*[13]
		**Lonicera tatarica*[13]
		Quercus alba[5,8,33–38]
		Quercus coccinea[13,33–35]
		Quercus falcata[33–35]
		Quercus rubra[8,13,33–35,39]
		Quercus stellata[33–35]
		Quercus velutina[33,37]
	When tolerance of fire is low, increased mortality or/and reduced abundance	**Ailanthus altissima*[13]
		**Berberis thunbergii*[13]
		Carya tomentosa[40]
		**Lonicera japonica*[22]
		Quercus coccinea[16]
		Quercus falcata[13,16]
		Quercus rubra[14,37]
		Quercus stellata[16]
		Quercus velutina[13,41]

[1]Anderson (2023); [2]Brzostek and others (2014); [3]Duan and others (2022); [4]Gustafson and Sturtevant (2013); [5]Hayford and Chin (2020); [6]Maxwell (2024); [7]McEwan and others (2011); [8]Novick and others (2022); [9]Rogers (1990); [10]Niinements and Vallardes (2006); [11]Carey (1992b); [12]Rauschendorfer and others (2022); [13]Koeser (2015); [14]Sander (1990b); [15]Walters and others (2023); [16]Carey (1992e); [17]Ni and Pallardy (1991); [18]Ni and Pallardy (1992); [19]Pallardy and Rhoads (1993); [20]Fisichelli and others (2014); [21]Peters and others (2015); [22]Munger (2002a); [23]Au and Maxwell (2022); [24]Kellner and Swihart (2016); [25]LeBlanc and Berland (2019); [26]Lee and Ibanez (2021); [27]Lockwood and others (2023); [28]Reich and others (2022); [29]Stangler and others (2016); [30]Wang and others (2023); [31]Prolic and Goldblum (2016); [32]Asbjornsen and others (2021); [33]Nowacki and Abrams (2015a); [34]Arthur and others (2021); [35]Varner (2018); [36]Hanberry and others (2012); [37]Hoving and others (2021); [38]Tirmenstein (1991d); [39]Peters and others (2020); [40]Coladonato (1992b); [41]Sander (1990a)

Focal Wildlife Species: Gray Bat

The gray bat (*Myotis grisescens*) is a species of bat found only in limestone karst areas in the southeastern and midwestern United States. This species roosts in very specific caves – the majority of gray bats hibernate in just 15 caves in the United States (Holliday and others, 2023). Because up to 95% of the population hibernates in only a few caves, extreme stochastic events like flooding could impact a substantial amount of the population (Harriman, 2003; Tuttle, 1979).

Though planning documents frequently list climate change as a concern for the gray bat, there is little information specifically on gray bat responses to changing climate conditions. However, research and modeling for other bat species' response to climate change have identified the potential for these highly mobile species to shift their distributions with changing environmental conditions (Festa and others, 2023). Gray bats are strong fliers, covering hundreds of kilometers during migration, but their reliance on specific caves and evidence for site fidelity may not allow for distribution shifts (Holliday and others, 2023). Bats may also adjust the timing of events like migrations and torpor in response to changing climate. This could result in phenological mismatch, particularly if bats emerge at times when food is not available. Bats can suffer mortality events or fail to reproduce during droughts (Adams and Hayes, 2021). Gray bats can fly up to 20 miles to locate water. However, traveling longer distances for food and water is correlated with longer generation time and slower offspring development in gray bats (Adams and Hayes, 2021).

Gray bats do not appear to be as susceptible to white-nose syndrome as other bats in the Midwest (U. S. Fish & Wildlife Service, 2021). However, their gregarious nature could cause concern for future pathogen spread, especially if physiologically stressed in a drier or warmer climate. Management for gray bats requires protecting caves from human disturbance (U. S. Fish & Wildlife Service, 2021) and likely protecting movement corridors from certain types of development (Holliday and others, 2023).

Ozark-Ouachita Oak- Shortleaf Pine Woodland and Savanna

The Ozark-Ouachita Oak - Shortleaf Pine Woodland and Savanna habitat spans the Interior Highlands of Missouri, Arkansas, and eastern Oklahoma, covering the Ozark and Ouachita plateaus and associated uplands (Figure 6) (USNVC, 2025). These communities typically occur on slopes and ridgetops, especially on south- and west-facing aspects where sun exposure is greatest and soils are driest (Anderson, 2020; Nelson, 1985; The Nature Conservancy, 2003; USNVC, 2025). Soils range from very shallow (0–40 cm) to moderately deep (up to 100 cm) and frequently contain coarse substrates that contribute to rapid drainage and summer drought stress (Kabrick and others, 2007; Nelson, 1985; The Nature Conservancy, 2003; USNVC, 2025).

Canopies are open and variable in height, typically dominated by oaks (*Quercus* spp.) and shortleaf pine (*Pinus echinata*), while the ground layer features a diverse, grass-dominated herbaceous community of warm-season (C_4) grasses and a variety of forbs where light availability remains high (Table 7) (Brandt and others, 2014; Hanberry and Nowacki, 2016; Nelson, 1985; The Nature Conservancy, 2003; USDA Forest Service, USA, 1999; USNVC, 2025). Understory density, species composition, and canopy openness vary with soil depth, topographic position, and disturbance history.

Historically, these habitats were maintained by frequent low- to moderate-intensity fires, wind disturbance, and periodic drought (Anderson, 2020; Cutter and Guyette, 1994; Guyette and others, 2007; Hanberry and others, 2012; Nelson, 1985; USNVC, 2025). Natural fire return intervals likely ranged from 4 to 10 years, with shorter intervals documented before Euro-American settlement (Anderson, 2020; Brose and others, 2014; Cutter and Guyette, 1994; Smith and others, 2008; USDA Forest Service, USA, 1999). Recurring fire limited the establishment of fire-intolerant, mesic species and maintained open canopy structure favorable to herbaceous diversity and regeneration of pyrophytic oaks and shortleaf pine (Arthur and others, 2012; Brose and others, 2014; Hutchinson and others, 2024; Nelson, 1985; Smith and others, 2008).

Over the past century, fire suppression, land use change, and climate stress have driven extensive canopy closure, reduced graminoid and forb diversity, and allowed woody encroachment to expand across the region (Abrams and others, 2021; Arthur and others, 2021; Hanberry and others, 2020; Hutchinson and others, 2008; Nelson, 1985; Nowacki and Abrams, 2008). Today, these habitats also face increasing oak decline, typically associated with recurrent drought, aging stand structure, and interactions with secondary pests and pathogens including red oak borer (*Enaphalodes rufulus*) and root rots (*Armillaria* spp.) (Fan and others, 2008; Gosling and others, 2024; Haavik and others, 2012; Haavik and others, 2015; Kabrick and others, 2008; Kubiak and others, 2017; The Nature Conservancy, 2003).

Table 7. Plant species common to the Ozark-Ouachita Oak- Shortleaf Pine Woodland and Savanna habitat. Daggers (†) denote some species within the genus may be introduced to the region, though most are native. Taxonomic authority, common name, growth habit, and native status are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Carya texana Buckley	Black hickory	Tree
Carya tomentosa (Lam.) Nutt.	Mockernut hickory	Tree
†*Crataegus* spp. L.	Hawthorn	Shrub, tree

Species	Common name	Growth habit
Juniperus virginiana L.	Eastern redcedar	Tree
Pinus echinata Mill.	Shortleaf pine	Tree
Quercus alba L.	White oak	Tree
Quercus coccinea Münchh.	Scarlet oak	Tree
Quercus marilandica Münchh.	Blackjack oak	Shrub, tree
Quercus rubra L.	Northern red oak	Tree
Quercus stellata Wangenh.	Post oak	Tree
Quercus velutina Lam.	Black oak	Tree
Vaccinium pallidum Aiton	Blue Ridge blueberry	Shrub
Amphicarpaea bracteata (L.) Fernald	American hogpeanut	Forb/herb, vine
Desmodium glutinosum (Muhl. ex Willd.) Alph. Wood	Pointedleaf ticktrefoil	Forb/herb
Desmodium nudiflorum (L.) DC.	Nakedflower ticktrefoil	Forb/herb
Dichanthelium boscii (Poir.) Gould & C.A. Clark	Bosc's panicgrass	Graminoid
Monarda bradburiana Beck	Eastern beebalm	Forb/herb
Parthenocissus quinquefolia (L.) Planch.	Virginia creeper	Vine
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid
Toxicodendron radicans (L.) Kuntze	Eastern poison ivy	Forb/herb, vine

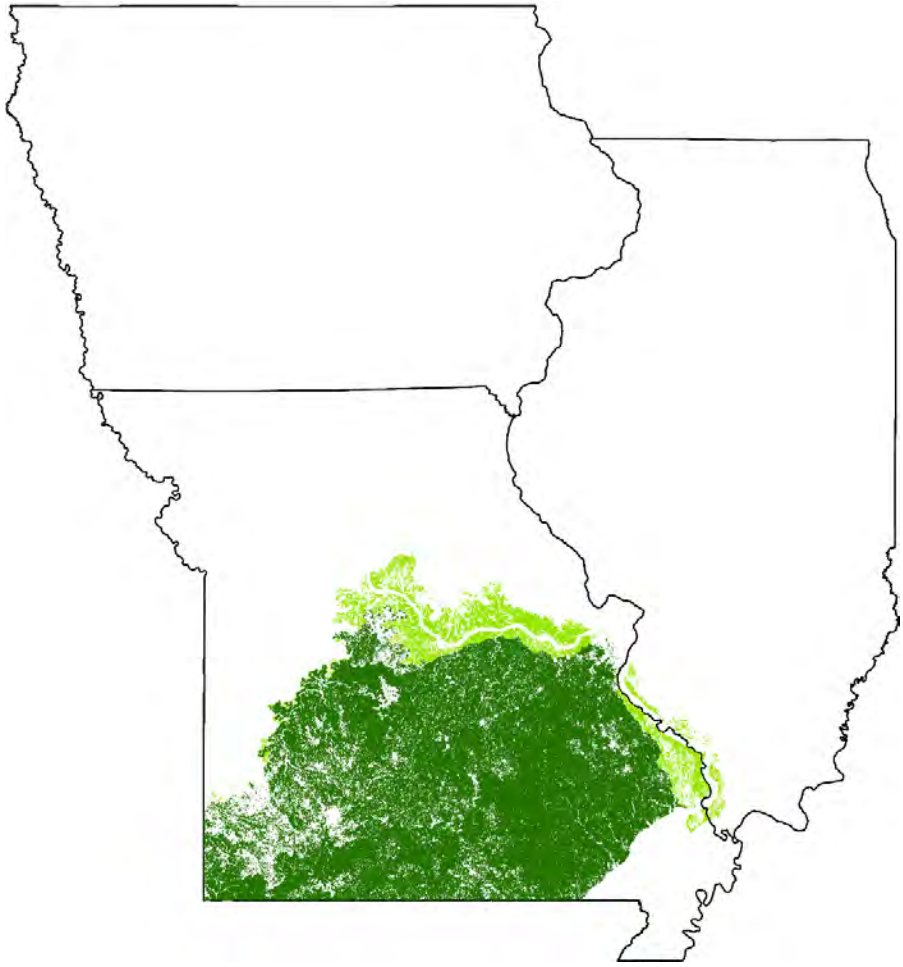

Figure 6. Ozark-Ouachita Oak- Shortleaf Pine Woodland and Savanna habitat distribution. Geographic distribution of the Ozark-Ouachita Oak-Shortleaf Pine Woodland and Savanna habitat (Group Code: G949) in the Midwest (USNVC, 2020). Dark green areas indicate Ozark-Ouachita Oak-Shortleaf Pine Woodland and Savanna habitat in the Ozarks ecoregion, while light green areas mark additional locations throughout the Midwest. Habitat boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The Ozark-Ouachita Oak - Shortleaf Pine Woodland and Savanna habitat is prioritized for the Ozarks ecoregion (Table i), and climate projections for this area largely reflect regional trends. Both average and extreme temperatures are expected to rise throughout the coming century, with model projections indicating a 9.4°F increase by the end of the century under the high emissions scenario (RCP 8.5; Table 8) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). Most models also project an increase in annual precipitation, averaging 1.8 additional inches per year, although some indicate a slight decline (Table 9) (Marvel and others, 2023). However, precipitation is expected to become more variable, with more extreme events and notably drier summers despite the overall increase (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 8. Projected changes in annual temperature in the Ozarks 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	55.8°F	57.8°F (2.0)	58.8°F (3.0)	58.4°F (2.6)	61.0°F (5.2)
Mean of models	55.9°F	60.2°F (4.3)	61.8°F (5.9)	61.1°F (5.2)	65.3°F (9.4)
Maximum model	56.0°F	62.3°F (6.3)	63.9°F (7.9)	62.3°F (6.3)	68.7°F (12.7)

Table 9. Projected changes in annual precipitation in the Ozarks 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	43.8 in	44.7 in (2%)	38.6 in (-12%)	46.9 in (7%)	37.4 in (-15%)
Mean of models	44.4 in	46.0 in (4%)	45.9 in (3%)	45.7 in (3%)	46.2 in (4%)
Maximum model	43.8 in	46.9 in (5%)	48.0 in (8%)	49.6 in (12%)	50.1 in (13%)

Key Climate Change Stressors

Drought and fire. Across the Ozarks ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 86% above historical conditions by the end of the century, equating to more than 1 kPa of additional VPD (RCP 8.5; Table 10). Values are expected to be fairly uniform throughout southern Missouri (Figure 7). Higher VPD means more rapid evaporation and water loss from plants and soils, which can intensify drought conditions. Annual extreme fire danger is also projected to increase by approximately 197% by mid-century under the high emissions scenario, adding more than 4 days of elevated risk (Table 11), with the highest values in the southeastern part of the state (Figure 8). Rising fire risk could increase the frequency of wildfire and push this habitat toward more historically frequent fire intervals, but it also raises the likelihood of extreme events, including crown fires, that exceed historical norms and constrain opportunities to safely apply prescribed fire.

Table 10. Projected changes in summer vapor pressure deficit in the Ozarks 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 summer vapor pressure deficit (VPD; kilopascals) 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 VPD 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	1.25 kPa	1.28 kPa (3%)	1.39 kPa (12%)	1.36 kPa (9%)	1.49 kPa (20%)
Mean of models	1.23 kPa	1.65 kPa (31%)	1.86 kPa (48%)	1.74 kPa (38%)	2.34 kPa (86%)
Maximum model	1.26 kPa	1.97 kPa (60%)	2.12 kPa (72%)	2.31 kPa (87%)	3.32 kPa (169%)

Table 11. Projected changes in annual extreme fire danger in the Ozarks 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 extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) by mid-century (2040–2069), 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 annual extreme fire danger under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069	
	Historical	RCP 4.5	RCP 8.5
Minimum model	2.4	2.3 (-2%)	1.5 (-35%)
Mean of models	2.2	4.8 (121%)	6.4 (197%)
Maximum model	3.1	5.0 (59%)	13.9 (344%)

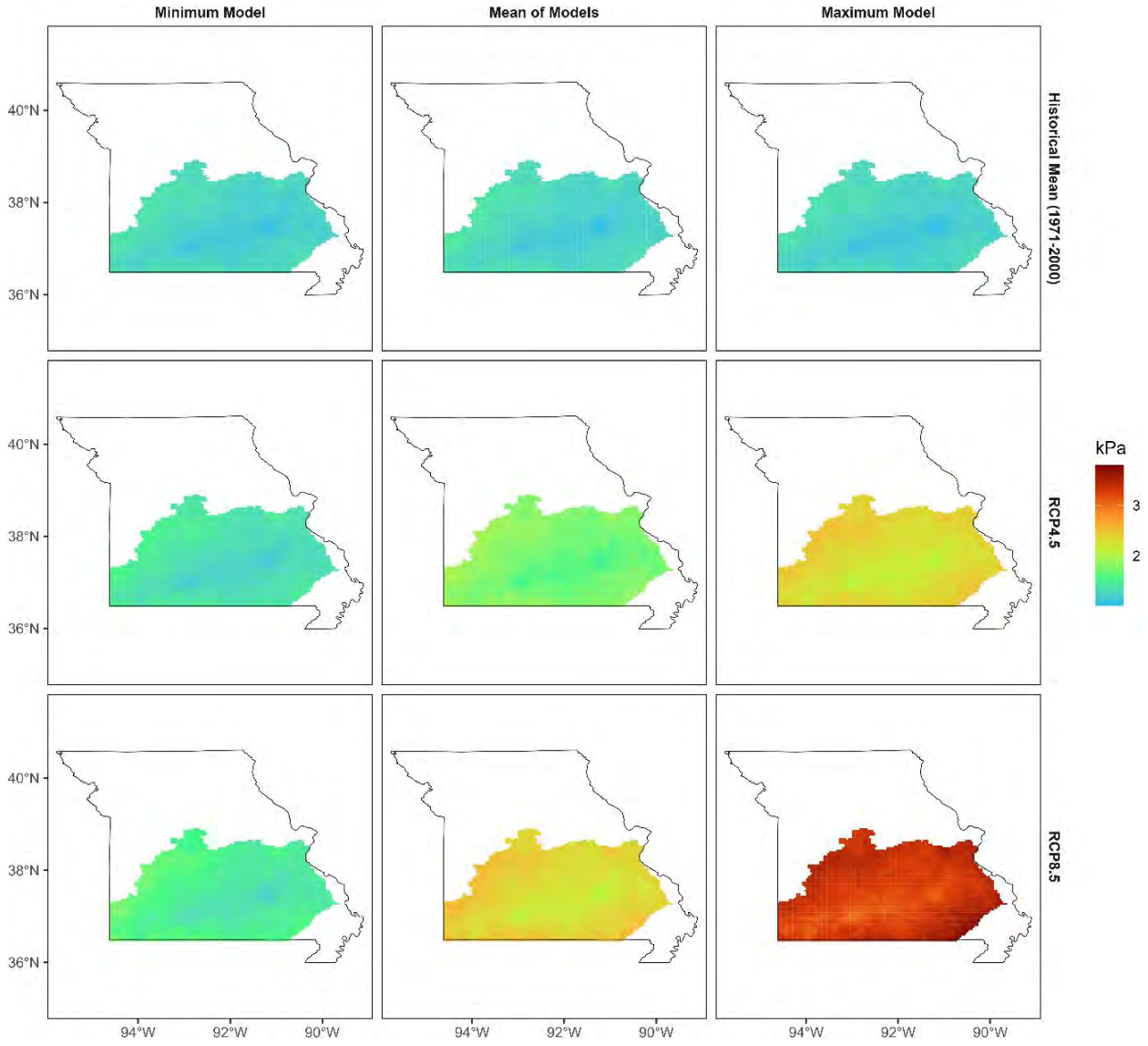

Figure 7. Projected changes in summer vapor pressure deficit in the Ozarks ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- 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 summer pressure deficit (VPD; kilopascals) 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 VPD under RCP 8.5 are presented in addition to the mean of all models (middle column).

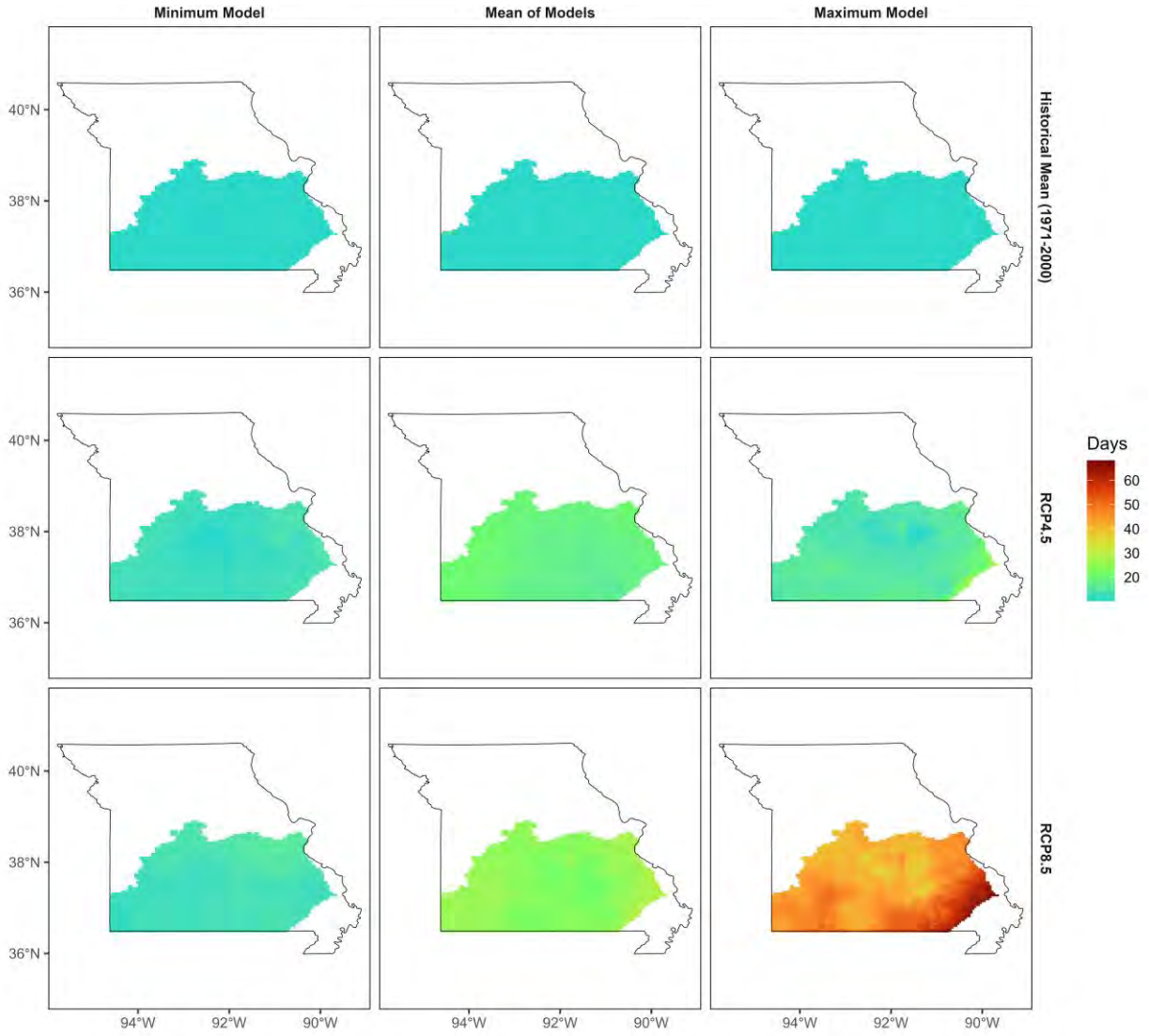

Figure 8. Projected changes in annual extreme fire danger in the Ozarks ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- 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 (2040–2069) annual extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) 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 extreme fire danger under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

Climate change is affecting the Ozark-Ouachita Oak - Shortleaf Pine Woodland and Savanna habitat through its overall influence on species distributions, adaptive capacity, and vulnerability, as well as through the specific impacts of drought and fire (Table 12). 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:

- Drought can intensify soil drying, reduce herbaceous cover, and increase erosion risk, particularly when followed by heavy rainfall on shallow, rocky uplands (Breshears and others, 2016; Ebel and others, 2022; Vose and others, 2016; Ziegler and others, 2008).
- Variability in precipitation timing and intensity can lead to large fluctuations in soil moisture, with shallow surface layers particularly sensitive to summer drying (Fay and others, 2011; O'Keefe and others, 2024; Smith and others, 2016).
- Soil warming during drought decreases moisture retention, potentially altering nutrient dynamics by reducing plant uptake and triggering post-drought nutrient flushes (Hartmann and others, 2013; Leizeaga and others, 2021; Reinsch and others, 2024; Vose and others, 2016).
- Soil microbial activity and decomposition decline under drought, reducing nutrient availability and favoring drought-tolerant microbial taxa (Baldrian and others, 2023; Lemoine and others, 2024).
- Low-intensity fire removes litter and midstory vegetation and increases light availability but can also expose mineral soil, elevating erosion risk (Arthur and others, 2021; Fernández and others, 2006; Roberts, 2004; Schlesinger and others, 2016).
- Nutrient availability often increases immediately following fire, but repeated fire may gradually deplete soil carbon and nitrogen pools in the long term, altering ecosystem function (Knicker, 2007; Pellegrini and others, 2020; Singh and others, 2025).

Community composition:

- Drought may shift communities toward xeric-adapted species, particularly drought-tolerant warm-season (C_4) grasses such as big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and Indiangrass (*Sorghastrum nutans*), and drought-tolerant forbs (Batllori and others, 2020; Havrilla and others, 2023; Randall and Walters, 2019).
- Mesophytic species like maples (*Acer* spp.), American beech (*Fagus grandifolia*), and birches (*Betula* spp.) are vulnerable to recruitment failure and canopy mortality under drought, with sugar maple (*Acer saccharum*) and yellow birch (*Betula alleghaniensis*) already showing regional decline (Coble and others, 2017; Moore and others, 2016).
- Oaks (*Quercus* spp.) and hickories (*Carya* spp.) may expand as mesic species decline, with black oak (*Q. velutina*), post oak (*Q. stellata*), and blackjack oak (*Q. marilandica*) particularly well-adapted to increasingly dry uplands (Abrams, 1990; Dey and others, 2017; Spetich and others, 2022).
- Severe or repeated drought may negate the relative drought advantage of oaks, driving recruitment failure and canopy mortality in red oak (*Q. rubra*) and black oak (*Q. velutina*) (Asbjornsen and others, 2021; Novick and others, 2022).
- Frequent, low-intensity fire promotes pyrophytic assemblages, reducing dominance of mesic trees such as sugar maple (*Acer saccharum*), increasing fire-adapted trees like white oak (*Q. alba*), black oak (*Q. velutina*), shagbark hickory (*Carya ovata*), and shortleaf pine (*Pinus echinata*), and enhancing graminoid and forb diversity (Arthur and others, 2021; Gold and others, 2023; Peterson and Reich, 2001; Schweitzer and others, 2023; Stambaugh and others, 2021).

- Elevated fire severity during drought may increase mortality of woody resprouters, reducing biodiversity and favoring only the most fire-tolerant species (Grau-Andrés and others, 2024; Meunier and others, 2021).
- In the absence of fire, drought may favor the expansion of shrubs and encroaching trees such as eastern red cedar (*Juniperus virginiana*), roughleaf dogwood (*Cornus drummondii*), and smooth sumac (*Rhus glabra*), which displace oaks and suppress future fire spread (Hanberry and others, 2012; O'Keefe and others, 2024; Ratajczak and others, 2011).
- Where fire suppression has enabled shrub expansion, increasing fire frequency alone may be insufficient to reverse encroachment, potentially locking systems into shrub-dominated states (Hanberry and others, 2012; Meunier and others, 2021).
- Drought may catalyze transitions toward either grassland, shrubland, or closed forest, depending on disturbance history and regeneration dynamics (Breshears and others, 2016; Smith and Goetz, 2021; Vose and others, 2016).

Invasive species:

- Drought can amplify invasion pressure by weakening native vegetation and favoring drought-tolerant invasives such as tree-of-heaven (*Ailanthus altissima*) and common buckthorn (*Rhamnus cathartica*) (Batllori and others, 2020; Knight and others, 2007).
- Invasive grasses including smooth brome (*Bromus inermis*), cogongrass (*Imperata cylindrica*), and Nepalese browntop (*Microstegium vimineum*) remain competitive and may expand under drought, threatening native tree regeneration (Duell and others, 2016; Kroeger and Otfinowski, 2025; Warren and others, 2011).
- Fire may suppress some invasive shrubs through topkill, but it can also promote pyrophytic invaders such as autumn olive (*Elaeagnus umbellata*) and Chinese silvergrass (*Miscanthus sinensis*) or vigorous resprouters such as common buckthorn (*Rhamnus cathartica*) and invasive honeysuckles (*Lonicera* spp.), which often increase without additional management (Culbert and others, 2025; Illinois Department of Natural Resources, 2017; Vander Yacht and others, 2017; Zouhar and others, 2008).
- Flammable invasive grasses can reinforce feedbacks that increase fire intensity and frequency, perpetuating their dominance in burned areas (Culbert and others, 2025; Fusco and others, 2019; Kerns and others, 2020).

Pests and pathogens:

- Climate change is expected to amplify pest and pathogen pressure by accelerating insect development, facilitating northward expansion, and weakening host defenses (Dukes and others, 2009; Pureswaran and others, 2018; Stanturf and others, 2024).
- Drought-stressed oaks (*Quercus* spp.) are highly susceptible to outbreak by wood-and bark-boring beetles such as red oak borer (*Enaphalodes rufulus*) and twolined chestnut borer (*Agrilus bilineatus*), contributing to widespread oak decline (Fierke and others, 2007; Haavik and others, 2015; Stump and others, 2024; USDA Forest Service, USA, 2002).
- Root- and stem-infecting pathogens including root rots (*Armillaria* spp.) and water molds (*Phytophthora* spp.) are favored under fluctuating moisture regimes, intensifying disease spread

(Haavik and others, 2015; Kolb and others, 2016; Kubiak and others, 2017; Wood and others, 2018).

- Fungal pathogens such as Hypoxylon canker (*Biscogniauxia atropunctata*) may shift to pathogenic states under drought, increasing oak mortality (Ansley and others, 2023; Wood and others, 2018).
- Fire may indirectly reduce pest and pathogen pressure by altering microclimates through vegetation removal, which can increase exposure to heat and moisture stress, and by thinning host populations; however, fire-damaged trees are also more vulnerable (Barkley, 2006; Canelles and others, 2021; Kelsey and Joseph, 2003; Progar and others, 2014).

Herbivory:

- Warmer winters may increase white-tailed deer (*Odocoileus virginianus*) populations, intensifying browsing pressure on oak (*Quercus* spp.) seedlings and saplings, which are highly preferred browse (Champagne and others, 2021; Felton and others, 2024; Frelich and others, 2015; Hanberry and Abrams, 2018; Weiskopf and others, 2019).
- Drought may reduce plant investment in chemical defenses, increasing palatability and making drought-stressed species more vulnerable to herbivory (Cole and others, 2021; Duell and others, 2024; Gutbrodt and others, 2011).
- Browsing in drought-stressed or severely burned habitats may reduce forb diversity, favor short-lived or weedy species and grasses, and limit regeneration of oaks and other native trees (Anderson and others, 2017, p. 202; Batzli and Dejaco, 2013; Urbanek and others, 2012).

Table 12. Summary of impacts from key climate change stressors on Ozark-Ouachita Oak- Shortleaf Pine Woodland and Savanna habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory impacts (e.g., showing both high and low tolerance to drought), reflecting conflicting findings in the literature.

Stressor	Impact	Affected species
Drought	When tolerance of droughts is high, competitive advantage and/or increased abundance	*Juniperus virginiana*[1–8]
		Parthenocissus quinquefolia[6]
		Pinus echinata[8]
		Quercus alba[8–17]
		Quercus coccinea[8,18,19]
		Quercus marilandica[4,6,8]
		Quercus rubra[10,13,17,18,20,21]
		Quercus stellata[6,8,22–25]
		Quercus velutina[17,18,26]
		**Rhamnus cathartica*[27]
		Schizachyrium scoparium[6,28–31]
		Vaccinium pallidum[32]
	When tolerance of droughts is low, reduced growth and/or abundance	**Berberis thunbergii*[6]
		Carya texana[4]

Stressor	Impact	Affected species
		Carya tomentosa[4]
		**Lonicera maackii*[6]
		Pinus echinata[4,33,34]
		Quercus alba[4,35,36,37]
		Quercus coccinea[4]
		Quercus rubra[4,14,38,39,40,41,42]
		Quercus stellata[14,43]
		Quercus velutina[6,14,44]
		Schizachyrium scoparium[45]
		Toxicodendron radicans[46]
Fire	When tolerance of fire is high, competitive advantage and/or increased abundance	*Carya texana*[47]
		Carya tomentosa[47]
		**Lonicera maackii*[6]
		**Lonicera tatarica*[6]
		Pinus echinata[6,47,48,49,50,51,52,53,54]
		Quercus alba[11,14,47,49,50,55,56,57]
		Quercus coccinea[6,47,49,50]
		Quercus marilandica[47,49,50]
		Quercus rubra[6,14,47,49,50,58]
		Quercus stellata[47,49,50]
		Quercus velutina[47,56]
		Schizachyrium scoparium[6,29]
		Toxicodendron radicans[46]
		Vaccinium pallidum[59]
	When tolerance of fire is low, increased mortality and/or reduced abundance	**Ailanthus altissima*[6]
		**Berberis thunbergii*[6]
		Carya texana[60]
		Carya tomentosa[61]
		Juniperus virginiana[3,6,47,56]
		Parthenocissus quinquefolia[6,62]
		Quercus coccinea[63]
		Quercus marilandica[6,64]
		Quercus rubra[21,56]
		Quercus stellata[25]
		Quercus velutina[6,66]

[1]Caterina and others (2014); [2]Ganguli and others (2016); [3]Anderson (2023); [4]Peters and others (2015); [5]Torquato and others (2020); [6]USDA, NRCS (2025); [7]Volder and others (2010); [8]Niinements and Vallardes (2006); [9]Duan and others (2022); [10]Gustafson and Sturtevant (2013); [11]Hayford and Chin (2020); [12]Maxwell (2024); [13]McEwan and others

(2011); [14]Novick and others (2022); [15]Clark and others (2016); [16]Rogers (1990);[17]Brzostek and others (2014); [18]Rauschendorfer and others (2022); [1963]Carey (1992b); [20]Walters and others (2023); [21] Sander (1990b); [22]Ni and Pallardy (1991); [23]Ni and Pallardy (1992); [24]Pallardy and Rhoads (1993); [25]Carey (1992e); [26]Carey (1992f); [27]Fisichelli and others (2014); [28]Lemoine and others (2017); [29]Steinberg (2002); [30]USDA, NRCS (2004); [31]Dekirmenjian and others (2024); [32]Tirmenstein (1991e); [33]Adhikari and others (2021); [34]Grissino-Mayer and Butler (1993); [35]Au and Maxwell (2022); [36]Kellner and Swihart (2016); [37]LeBlanc and Berland (2019); [38]Lee and Ibanez (2021); [39]Lockwood and others (2023); [40]Reich and others (2022); [41]Stangler and others (2016); [42]Wang and others (2023); [43]Prolic and Goldblum (2016); [44]Asbjornsen and others (2021); [45]Duell and others (2016); [46]Innes (2012); [47]Nowacki and Abrams (2015); [48]Bradley and others (2016); [49]Arthur and others (2021); [50]Varner (2018); [51]Carey (1992a); [52]Guldin and Black (2018); [53]Guyutte and others (2007); [54]Stambaugh and others (2021); [55]Hanberry and others (2012); [56]Hoving and others (2021); [57]Tirmenstein (1991c); [58] Peters and others (2020); [59]Apfelbaum and others (2023); [60]Burton (2010); [61]Coladonato (1992a); [62]Coladonato (1991); [63]Carey (1992b); [64]Carey (1992c); [65]Sander (1990a)

Focal Wildlife Species: Black Bear

Black bears (*Ursus americanus*) are highly adaptable generalists, a trait that has allowed them to persist, and in some areas, expand, even with changing climate conditions and expanding human footprints (Garshelis and others, 2020). In the Midwest, climate projections suggest black bears will continue to persist, with no predicted loss of suitable habitat under low emissions scenarios and only some losses in western Minnesota under high emissions scenarios (Pandey and Papeş, 2018).

Climate change may directly affect black bears by altering denning behavior and exposing them to more frequent extreme rainfall events. Warming conditions could shorten the denning period, leading to early emergence before food is available or even den abandonment (Kurth and others, 2024). Den entry is also influenced by food availability, with bears entering dens later in wet years, when resources are abundant. Heavy rainfall and flooding have mixed effects on recruitment and survival, in some cases causing little impact, but in others leading to loss of young (Kurth and others, 2024).

As generalist foragers, black bears can shift seasonally among berries, insects, nuts, and other plants (Minnesota Department of Natural Resources, n.d.), and this dietary flexibility will likely help them adapt to changing phenology (Kurth and others, 2024). However, climate-driven food shortages may test this adaptability. Drought and low mast production can reduce food availability and, coupled with extreme heat impacts on foraging efficiency, lead to nutritional deficiencies, poorer body condition, and smaller bears (Garshelis and others, 2017; Johnson and others, 2020; Kurth and others, 2024; Stump and others, 2024). Because oak acorn production is closely tied to bear abundance, declines in oak populations or acorn yields may push bears to travel farther from their home ranges in search of food (Kurth and others, 2024; Stump and others, 2024). This increases the likelihood of human-bear conflicts, particularly during droughts or other climate-induced food shortages, and can increase mortality through road crossings (Johnson and others, 2020).

Adaptive management will be important for supporting black bears in a changing climate while reducing risks of human-bear conflict. Maintaining habitat connectivity will allow bears to find resources and potentially shift their range (Krosby and others, 2016). Decreasing opportunities for human-bear conflict will also be essential, especially if bears emerge early from dens when food is not available (Kurth and others, 2024). Maintaining natural sources of food is one way to prevent bears from foraging from human food sources, but in some extreme cases, managers may want to explore diversionary feeding, though that can drive population growth (Garshelis and others, 2017).

Focal Aquatic System: Huzzah Creek

Huzzah Creek, a tributary of the Meramec River in east-central Missouri, stands out as one of the state's most resilient aquatic systems due to its intact forested watershed, minimal land use impacts, and high biodiversity (Sowa and others, 2007; The Nature Conservancy, 2014). The creek's hydrology is dominated by groundwater and springs, resulting in stable baseflows and temperatures that buffer against droughts, floods, and thermal extremes (Sowa and others, 2007; The Nature Conservancy, 2014; Westhoff and Paukert, 2014). Extensive forest cover, which comprises over 85% of the watershed, with much of it in public ownership, protects water quality, limits sediment and nutrient runoff, and supports diverse fish, mussel, and macroinvertebrate communities; conditions that have led to Huzzah Creek's status as a reference stream for ecological health in the Ozark Highlands (Missouri Department of Natural Resources, 2017; The Nature Conservancy, 2014).

Missouri's climate has undergone clear shifts over the past century, with mean annual temperature rising by nearly 1°F since 1900 and annual precipitation increasing throughout much of the state, especially in the Ozark Highlands (Marti and Heimann, 2024). There have also been widespread upward trends in annual peak streamflows statewide over the past century, especially in southern Missouri in the most recent 30- to 50-year records (Marti and Heimann, 2024). In addition, the frequency of high-flow events has increased at more than half of Missouri's stream gages over the last 75–100 years (Marti and Heimann, 2024). Climate projections suggest these trends will continue (Table 13), with future scenarios predicting more intense spring rainfall, increases in winter and spring precipitation, warmer minimum temperatures, and an elevated risk of both flooding and drought (Marti and Heimann, 2024).

The resilience of Huzzah Creek to climate-driven stressors is closely tied to its spring-fed, groundwater-dominated hydrology and intact forested riparian corridors. Groundwater inputs stabilize streamflow, buffering against drought and floods, and help regulate stream temperatures, creating cool, stable refugia for aquatic life even as air temperatures rise (Knouft and others, 2021; Sowa and others, 2007; Westhoff and Paukert, 2014). Streams with high baseflow and intact riparian canopy are among the most resistant to temperature increases in the region (Whittier and others, 2018). Because of this, Huzzah is better positioned to support native fish and mussel communities as the climate warms compared to more disturbed or runoff-dominated streams (Hinck and others, 2011; Westhoff and Paukert, 2014).

Nonetheless, as summer stream temperatures increase across the region, climate change is projected to drive community shifts in Ozark streams. Smallmouth bass (*Micropterus dolomieu*) are likely to decline in growth and abundance in warmer, runoff-dominated waters, while largemouth bass (*M. salmoides*) and other heat-tolerant species may expand, potentially altering food web dynamics (Middaugh and others, 2018; Westhoff and Paukert, 2014). These projected fish community shifts, combined with heightened risks to mussel assemblages from sedimentation, water-quality degradation, and habitat alteration, highlight the need to maintain Huzzah Creek's unique thermal and hydrologic buffers to preserve ecological function and biodiversity (Hinck and others, 2011; The Nature Conservancy, 2014).

Table 13. Projected changes in streamflow for Huzzah Creek. Projections use gridded simulations of daily runoff based on CMIP5 climate models under the high-emissions (RCP 8.5) scenario. Values show average historical (1977–2006) 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 (Authors (in review); U.S. Department of the Interior, Bureau of Reclamation, 2014; USDA Forest Service Office of Sustainability and Climate, 2022; Wenger and others, 2010).

Streamflow metric	1977–2006	2070–2099	Percent change
Mean Winter (December–February) Flow	220.8 ft³/s	189.1 ft³/s	-14%
Mean Spring (March–May) Flow	201.6 ft³/s	211.2 ft³/s	5%
Mean Summer (June–August) Flow	69.1 ft³/s	60.0 ft³/s	-13%
Mean Fall (September–November) Flow	99.2 ft³/s	82.7 ft³/s	-17%
Mean Annual Flow	147.5 ft³/s	135.6 ft³/s	-8%
1.5-year Flood (33rd percentile)	789.8 ft³/s	853.2 ft³/s	8%
10-year Flood (90th percentile)	2102.0 ft³/s	2314.3 ft³/s	10%
25-year Flood (96th percentile)	2579.9 ft³/s	2830.9 ft³/s	10%
Number of Winter Floods	10.1	9.8	-3%
Average Lowest 7-day Flow	32.7 ft³/s	30.0 ft³/s	-8%
Center of Flow Mass Timing	March 12	March 16	3%

Central Midwest Oak Openings and Barrens

The Central Midwest Oak Openings and Barrens habitat is a structurally open, fire-maintained habitat that historically occupied dry-mesic to mesic uplands throughout the upper Midwest and parts of northern Missouri and Iowa (Figure 9), forming a broad ecotone between tallgrass prairie and closed-canopied deciduous forest (USNVC, 2025). These communities typically developed on well-drained, shallow-to-bedrock soils composed of sandy substrates that slowed succession and limited canopy closure (Anderson and Bowles, 1999; Fowler and Beckage, 2019; Heikens and Robertson, 1994; Kost and others, 2007; USDA Forest Service, USA, 2005; USNVC, 2025).

Historically, this habitat covered millions of hectares, aided by Indigenous fire practices (Briere and Gajewski, 2023; Paciorek and others, 2021; Pyne, 2017; Thomas-Van Gundy and others, 2020). Canopy cover ranged from open savannas (10–30% cover) to more closed woodlands (30–80% cover), dominated by oaks (*Quercus* spp.) (Table 14) (Chapman and others, 1993; Law and others, 1994; Nelson, 1985; Taft, 1997; USDA Forest Service, USA, 2005). Repeated low-intensity fires every 2–10 years prevented canopy closure, restricted woody encroachment, and promoted a rich layer of warm-season (C_4) prairie grasses and forbs typical of mesic to dry prairies (Anderson and Bowles, 1999; Brose and others, 2014; Heikens, 1999; Heikens and Robertson, 1994; Hutchinson and others, 2008; Kost and others, 2007; USNVC, 2025; Vose and others, 2021). Shrub cover varied with fire regime, with fire-adapted species resprouting vigorously after burns (Faber-Langendoen, 1995; Grimm, 1984; USNVC, 2025).

Following Euro-American settlement, fire use declined, and wildfires were actively suppressed. Consequently, nearly all but the driest sites developed into dense shrub thickets or closed-canopy woodlands dominated by mesic species (Abrams and others, 2021; Alexander and others, 2021; Darling, 2021; Faber-Langendoen, 1995; Hutchinson and others, 2008; Nowacki and Abrams, 2008; USNVC, 2025). By the mid-1980s, approximately 98% of the original extent of the Central Midwest Oak Openings and Barrens in the Midwest had been lost (Abella and others, 2017; Dey and Kabrick, 2015; Fowler and Beckage, 2019; Fralish, 2004; Hanberry and others, 2017; Nuzzo, 1986; Pile Knapp and others, 2024). Today, this habitat is rare and highly fragmented, persisting primarily as small remnant woodlots or in areas where active management continues (USNVC, 2025).

Table 14. Plant species common to the Central Midwest Oak Openings and Barrens habitat. Daggers (†) denote some species within the genus may be introduced to the region, though most are native. Taxonomic authority, common name, growth habit, and native status are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Acer rubrum L.	Red maple	Tree
Carya ovata (Mill.) K. Koch	Shagbark hickory	Tree
Populus tremuloides Michx.	Quaking aspen	Tree
Quercus alba L.	White oak	Tree
Quercus bicolor Willd.	Swamp white oak	Tree
Quercus ellipsoidalis E.J. Hill	Northern pin oak	Tree
Quercus macrocarpa Michx.	Bur oak	Shrub, tree
Quercus palustris Münchh.	Pin oak	Tree

Species	Common name	Growth habit
Quercus spp. L.	Oak	Tree
Quercus velutina Lam.	Black oak	Tree
Ambrosia psilostachya DC.	Cuman ragweed	Forb/herb
Amphicarpaea bracteata (L.) Fernald	American hogpeanut	Forb/herb, vine
Andropogon gerardii Vitman	Big bluestem	Graminoid
Artemisia ludoviciana Nutt.	White sagebrush	Forb/herb
Calamagrostis canadensis (Michx.) P. Beauv.	Bluejoint	Graminoid
Calamovilfa longifolia (Hook.) Scribn.	Prairie sandreed	Graminoid
Carex pensylvanica Lam.	Pennsylvania sedge	Graminoid
†*Carex* spp. L.	Sedge	Graminoid
Comandra umbellata (L.) Nutt.	Bastard toadflax	Forb/herb
Hesperostipa spartea (Trin.) Barkworth	Porcupinegrass	Graminoid
Koeleria macrantha (Ledeb.) Schult.	Prairie Junegrass	Graminoid
Panicum virgatum L.	Switchgrass	Graminoid
Parthenocissus quinquefolia (L.) Planch.	Virginia creeper	Vine
Rudbeckia hirta L.	Blackeyed Susan	Forb/herb
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid
Sorghastrum nutans (L.) Nash	Indiangrass	Graminoid
Sporobolus heterolepis (A. Gray) A. Gray	Prairie dropseed	Graminoid

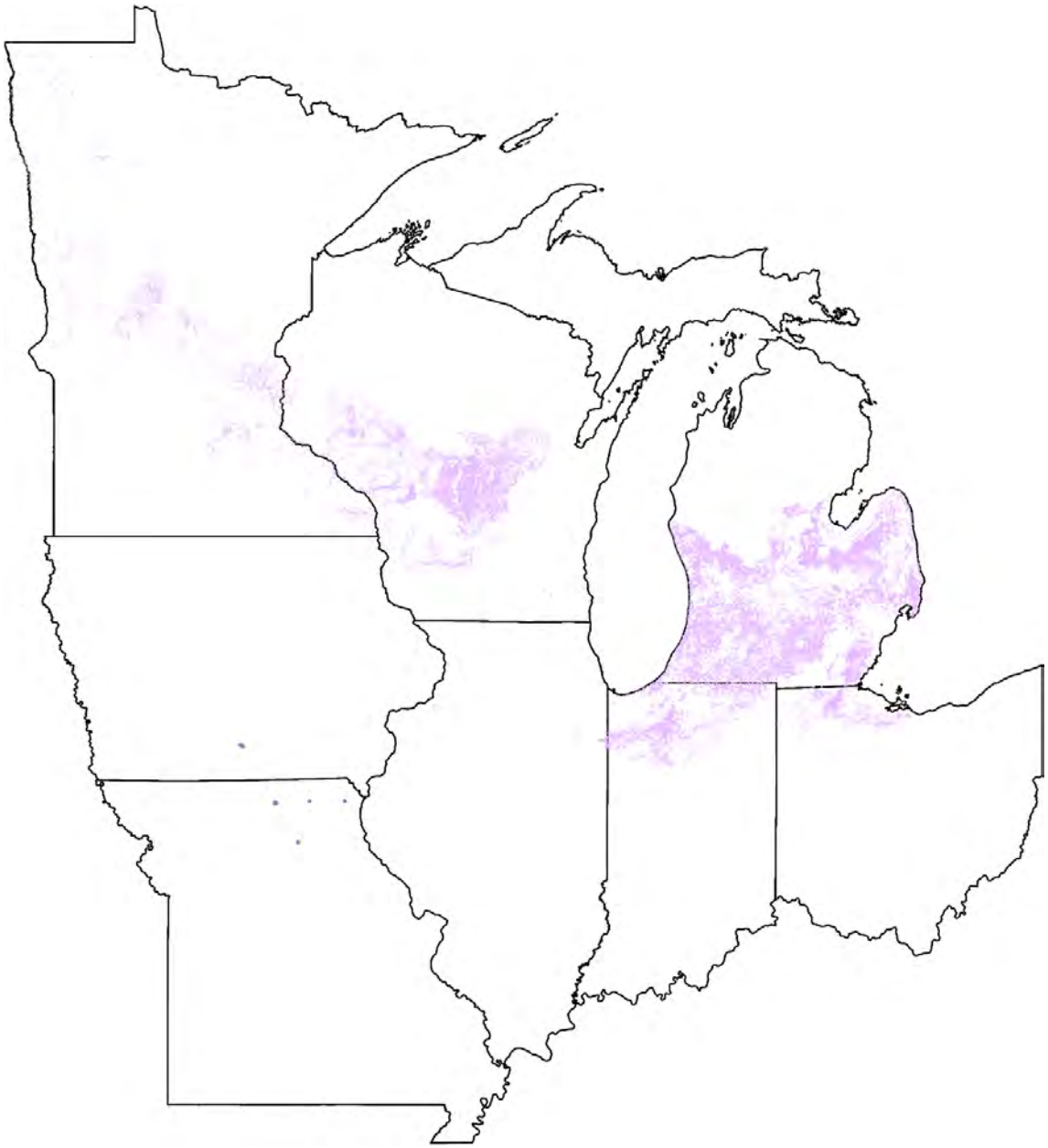

Figure 9. Central Midwest Oak Openings and Barrens habitat distribution. Geographic distribution of the Central Midwest Oak Openings and Barrens habitat (Group Code: G181) in the Midwest (USNVC, 2020). Dark purple areas indicate Central Midwest Oak Openings and Barrens habitat in the Central Irregular Plains ecoregion (small areas in northern Missouri and southern Iowa), while light purple areas mark additional locations throughout the Midwest. Habitat boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The Central Midwest Oak Openings and Barrens habitat is prioritized for the Central Irregular Plains ecoregion (Table i), and climate projections for this area largely reflect regional trends. Both average and extreme temperatures are expected to rise throughout the coming century, with model projections indicating a 9.8°F increase by the end of the century under the high emissions scenario (RCP 8.5; Table

15) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). Most models also project an increase in annual precipitation, averaging 2.1 additional inches per year, although some indicate a slight decline (Table 16) (Marvel and others, 2023). However, precipitation is expected to become more variable, with more extreme events and notably drier summers despite the overall increase (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

In the Midwest, the Central Midwest Oak Openings and Barrens habitat primarily occurs in the Great Lakes Plains ecoregion (Table i). Climate projections for this ecoregion are similar to those for the Central Irregular Plains ecoregion, with average and extreme temperatures expected to rise throughout the coming century, and model projections indicating a 10.1°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). Most models also project an increase in annual precipitation, averaging 3.4 additional inches per year, although some indicate a slight decline (Table 18) (Marvel and others, 2023).

Table 15. Projected changes in annual temperature in the Central Irregular 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	53.1°F	55.2°F (2.1)	56.2°F (3.1)	56.0°F (2.9)	58.6°F (5.5)
Mean of models	53.2°F	57.8°F (4.6)	59.3°F (6.1)	58.7°F (5.5)	63.0°F (9.8)
Maximum model	53.0°F	59.2°F (6.2)	61.1°F (8.1)	61.1°F (8.2)	66.7°F (13.7)

Table 16. Projected changes in annual precipitation in the Central Irregular 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	39.4 in	38.7 in (-2%)	35.5 in (-10%)	42.4 in (8%)	34.4 in (-13%)
Mean of models	39.2 in	40.9 in (4%)	40.9 in (4%)	40.7 in (4%)	41.3 in (5%)
Maximum model	39.1 in	41.5 in (6%)	43.9 in (12%)	45.9 in (17%)	46.7 in (19%)

Table 17. Projected changes in annual temperature in the Great Lakes 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	48.3 °F	51.0 °F (2.7)	52.1 °F (3.7)	51.7 °F (3.4)	54.7 °F (6.4)
Mean of models	48.5 °F	53.2 °F (4.6)	54.8 °F (6.4)	54.3 °F (5.8)	58.6 °F (10.2)
Maximum model	48.3 °F	54.6 °F (6.4)	56.8 °F (8.5)	57.0 °F (8.7)	63.0 °F (14.7)

Table 18. Projected changes in annual precipitation in the Great Lakes 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	35.6 in	35.1 in (-1%)	35.4 in (-1%)	38.2 in (7%)	34 in (-4%)
Mean of models	35.8 in	38.2 in (7%)	38.3 in (7%)	38.1 in (6%)	39.2 in (9%)
Maximum model	35.5 in	42 in (18%)	39.1 in (10%)	42.3 in (19%)	44.4 in (25%)

Key Climate Change Stressors

Drought and fire. Across the Central Irregular Plains ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 82% above historical conditions by the end of the century, equating to more than 1 kPa of additional VPD (RCP 8.5; Table 19). The highest values are expected in the southern part of the region, peaking in west-central Missouri (Figure 10). Higher VPD means more rapid evaporation and water loss from plants and soils, which can intensify drought conditions. Annual extreme fire danger is also projected to increase by approximately 113% by mid-century under the high emissions scenario, adding more than 12 days of elevated risk (Table 20). The greatest fire risk is projected in the northern part of the region along the Iowa-Missouri border, where the maximum model indicates as many as 60 days of extreme risk annually (Figure 11). Rising fire risk could increase the frequency of wildfire and push this habitat toward more historically frequent fire intervals, but it also raises the likelihood of extreme events, including crown fires, that exceed historical norms and constrain opportunities to safely apply prescribed fire.

In the Great Lakes Plains ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 72% above historical conditions by the end of the century, equating to more than 1 kPa of additional VPD (RCP 8.5; Table 21). Annual extreme fire danger is also projected to increase by approximately 65% by mid-century under the high emissions scenario, adding more than 4 days of elevated risk (Table 22). Increases in VPD and fire risk are slightly greater for the southern portion of the Great Lakes Plains ecoregion, where the maximum model indicates as many as 50 days of extreme risk annually (Figure 12).

Table 19. Projected changes in summer vapor pressure deficit in the Central Irregular 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 summer vapor pressure deficit (VPD; kilopascals) 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 VPD 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	1.25 kPa	1.29 kPa (3%)	1.40 kPa (13%)	1.38 kPa (10%)	1.50 kPa (20%)
Mean of models	1.26 kPa	1.64 kPa (30%)	1.81 kPa (44%)	1.75 kPa (39%)	2.29 kPa (82%)
Maximum model	1.26 kPa	1.86 kPa (48%)	2.06 kPa (64%)	2.26 kPa (79%)	3.24 kPa (158%)

Table 20. Projected changes in annual extreme fire danger in the Central Irregular 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 extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) by mid-century (2040–2069), 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 annual extreme fire danger under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069	
	Historical	RCP 4.5	RCP 8.5
Minimum model	11.0	8.7 (-21%)	10.2 (-7%)
Mean of models	11.0	19.9 (82%)	23.3 (113%)
Maximum model	11.0	55.5 (406%)	37.8 (245%)

Table 21. Projected changes in summer vapor pressure deficit in the Great Lakes 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 summer vapor pressure deficit (VPD; kilopascals) 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 VPD 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	0.96 kPa	1.03 kPa (7%)	1.08 kPa (13%)	1.07 kPa (11%)	1.17 kPa (22%)
Mean of models	0.98 kPa	1.22 kPa (24%)	1.36 kPa (39%)	1.31 kPa (33%)	1.69 kPa (72%)
Maximum model	0.99 kPa	1.47 kPa (49%)	1.66 kPa (68%)	1.83 kPa (85%)	2.8 kPa (184%)

Table 22. Projected changes in annual extreme fire danger in the Great Lakes 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 extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) by mid-century (2040–2069), 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 annual extreme fire danger under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069	
	Historical	RCP 4.5	RCP 8.5
Minimum model	11	5.7 (-48%)	4.4 (-60%)
Mean of models	11	13.6 (24%)	18.1 (65%)
Maximum model	11	19.5 (78%)	34.5 (215%)

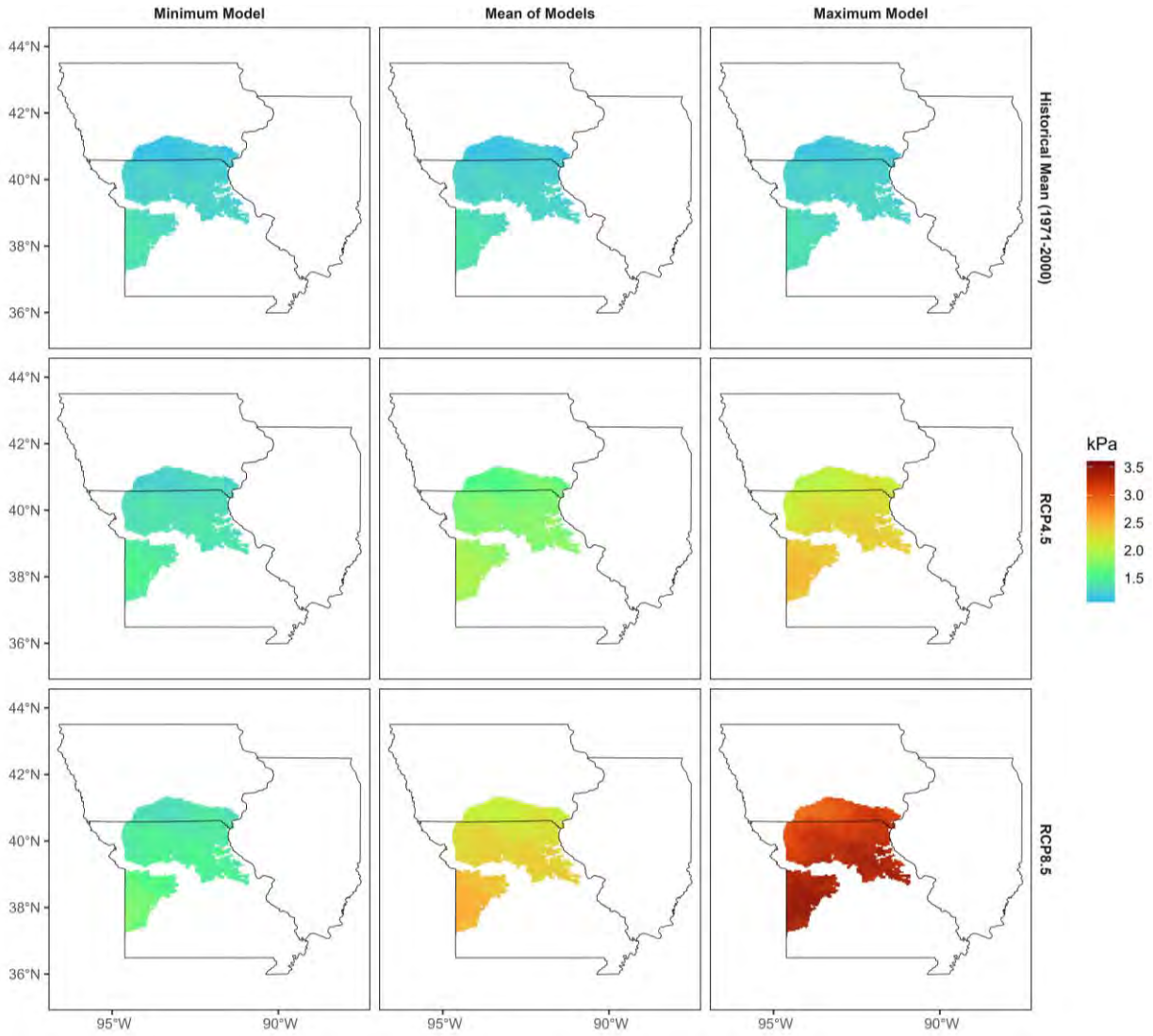

Figure 10. Projected changes in summer vapor pressure deficit in the Central Irregular Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA 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 summer pressure deficit (VPD; kilopascals) 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 VPD under RCP 8.5 are presented in addition to the mean of all models (middle column).

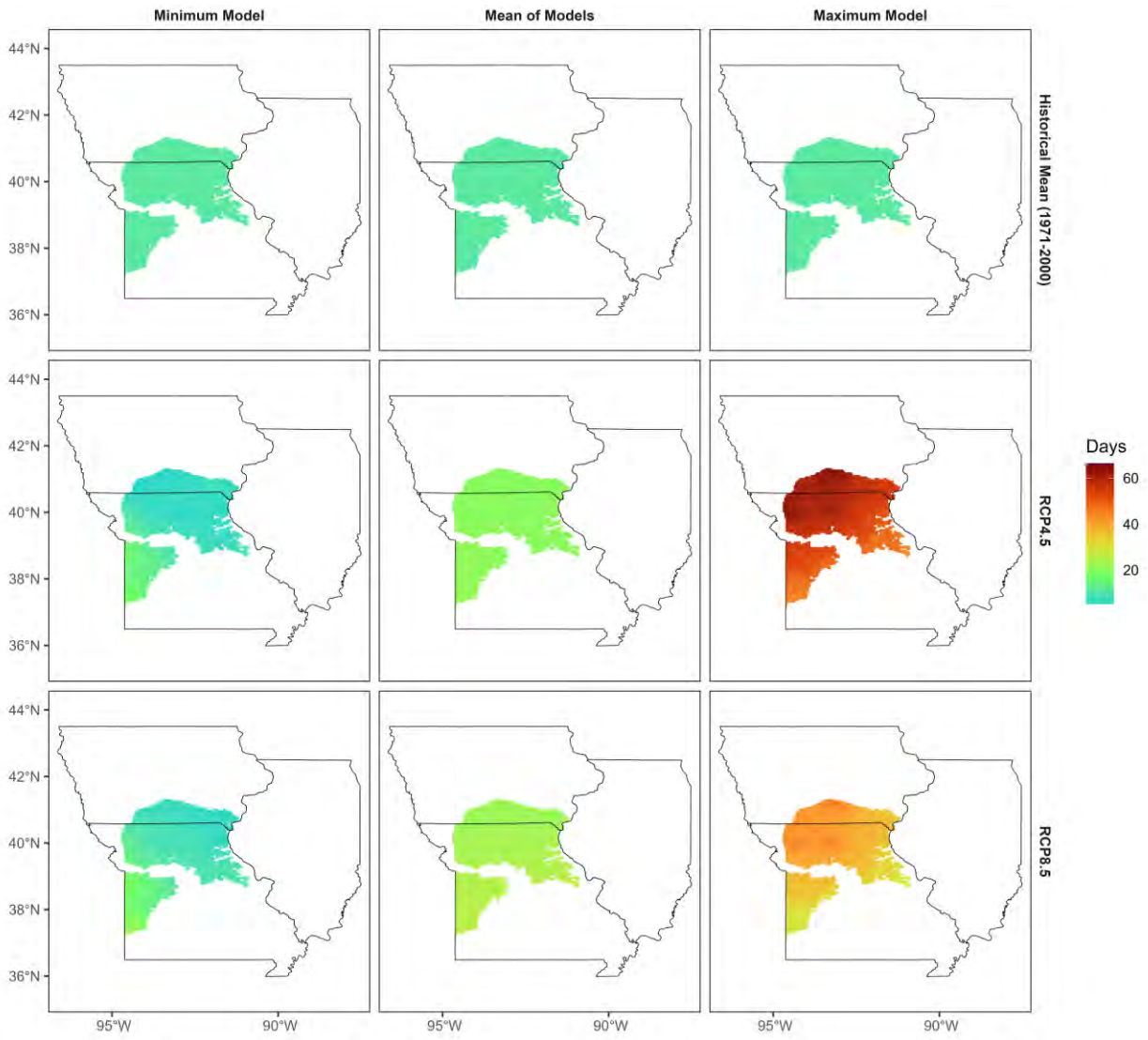

Figure 11. Projected changes in annual extreme fire danger in the Central Irregular Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA 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 (2040–2069) annual extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) 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 extreme fire danger under RCP 8.5 are presented in addition to the mean of all models (middle column).

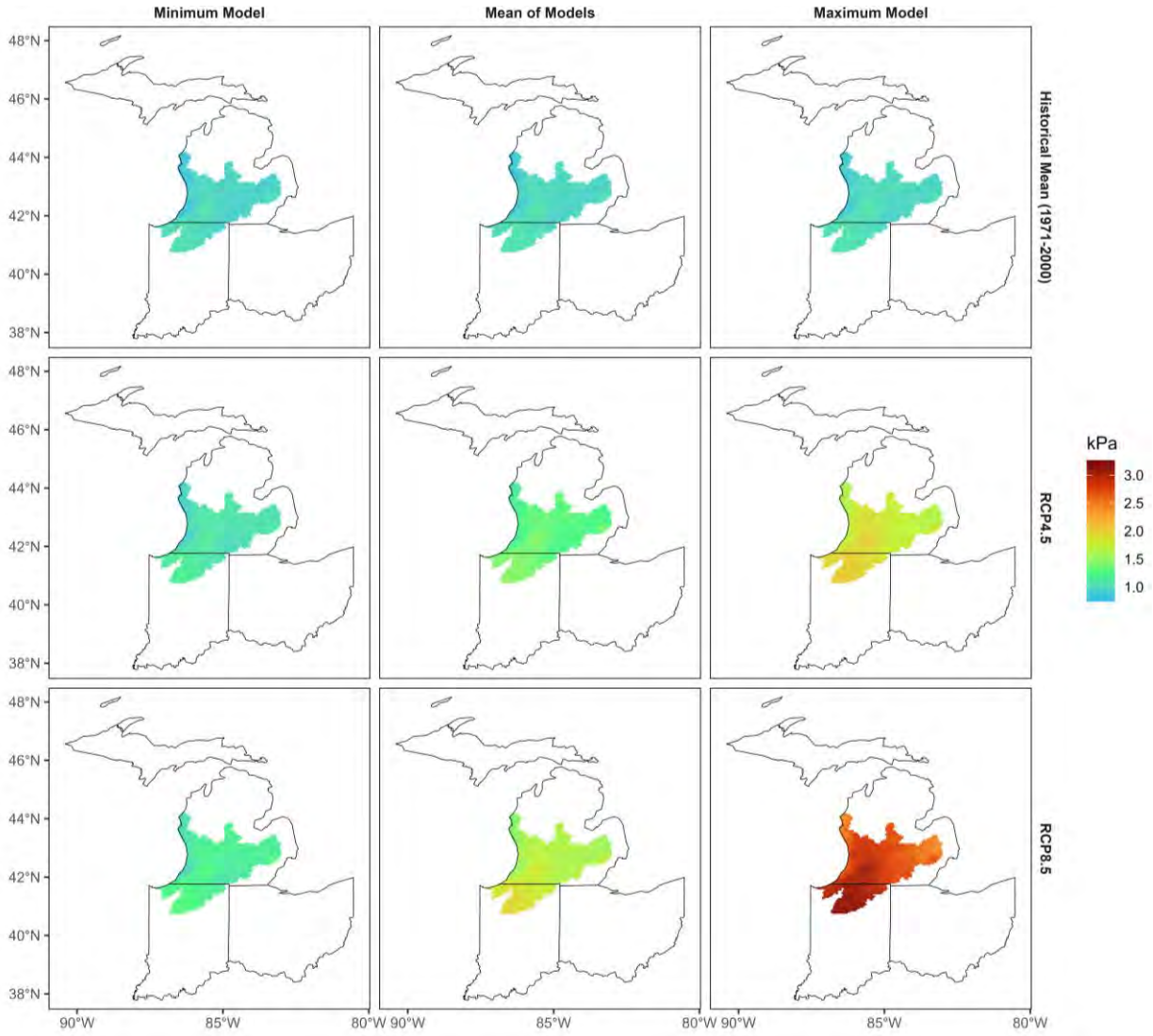

Figure 12. Projected changes in summer vapor pressure deficit in the Great Lakes Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA 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 summer pressure deficit (VPD; kilopascals) 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 VPD under RCP 8.5 are presented in addition to the mean of all models (middle column).

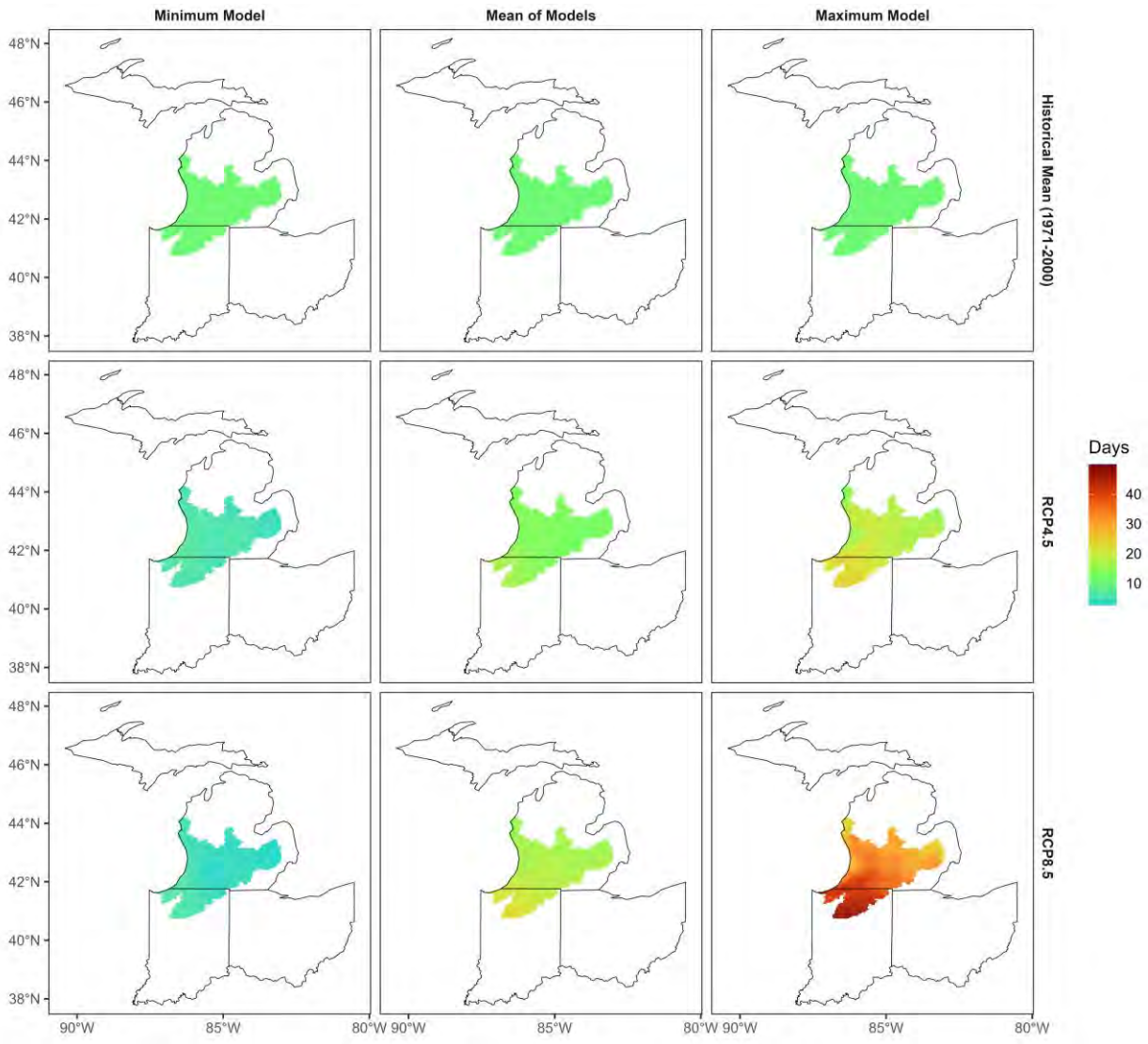

Figure 13. Projected changes in annual extreme fire danger in the Great Lakes Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for 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 (2040–2069) annual extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) 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 extreme fire danger under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

Climate change is affecting the Central Midwest Oak Openings and Barrens habitat through its overall influence on species distributions, adaptive capacity, and vulnerability, as well as through the specific impacts of drought and fire (Table 23). 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:

- Drought can reduce herbaceous cover and increase erosion risk, particularly when followed by heavy rainfall events that can flush accumulated nutrients and carbon on open, sandy, and glacially derived soils (Breshears and others, 2016; Butler and others, 2015; Homyak and others, 2017; Leizeaga and others, 2021).
- Soil microbial activity and decomposition decline under drought, reducing nutrient availability and favoring drought-tolerant microbial taxa (Baldrian and others, 2023; Lemoine and others, 2024; Reinsch and others, 2024).
- Low-intensity fire consumes litter, opens the canopy, increases understory light availability, maintains herbaceous diversity, and exposes mineral soil, while high-intensity or repeated fire can increase erosion risk, degrade soil structure, and drive long-term nutrient losses (Arthur and others, 2021; Fernández and others, 2006; Hanberry and others, 2017; Knicker, 2007; Pellegrini and others, 2020; Singh and others, 2025).

Community composition:

- Drought may shift communities toward xeric-adapted species, particularly drought-tolerant warm-season (C_4) grasses such as big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and Indiangrass (*Sorghastrum nutans*), though extreme drought may reduce their dominance in favor of opportunistic forbs or shrubs (Batllori and others, 2020; Havrilla and others, 2023; Knapp and others, 2024; Volder and others, 2010).
- Mesophytic species like sugar maple (*Acer saccharum*), red maple (*A. rubrum*), and yellow birch (*Betula alleghaniensis*) may decline under drought, while oaks (e.g., *Quercus velutina*, *Q. alba*, *Q. macrocarpa*) may maintain or expand dominance, particularly on dry sites (Boakye and others, 2023; Frelich and others, 2017; Rauschendorfer and others, 2022).
- Extreme drought may push drought-adapted oaks past tolerance thresholds, leading to recruitment failure and canopy mortality (Asbjornsen and others, 2021; Druckenbrod and others, 2019).
- Frequent, low-intensity fire promotes pyrophytic assemblages, reducing dominance of mesic trees and increasing fire-adapted species like oaks, shagbark hickory (*Carya ovata*), and grasses, while high-intensity or overly frequent fire can reduce recruitment and diversity (Arthur and others, 2012; Hutchinson and others, 2024).
- In the absence of fire, drought may favor the expansion of shrubs and encroaching trees such as eastern red cedar (*Juniperus virginiana*), roughleaf dogwood (*Cornus drummondii*), and smooth sumac (*Rhus glabra*), which outcompete graminoids and suppress future fire spread (Hanberry and others, 2012; O'Keefe and others, 2024; Ratajczak and others, 2011; Volder and others, 2010).

- Drought and fire suppression may catalyze transitions toward either grassland, shrubland, or closed-canopy forest, depending on disturbance history (Breshears and others, 2016; Frelich and Reich, 2010; Smith and Goetz, 2021; Vose and others, 2016).

Invasive species:

- Drought favors invasive species with broad tolerances and rapid recovery strategies, such as smooth brome (*Bromus inermis*), Nepalese browntop (*Microstegium vimineum*), and cogongrass (*Imperata cylindrica*), which can suppress native grasses and tree regeneration (Duell and others, 2021; Warren and others, 2011).
- Temporary nitrogen pulses following post-drought rewetting may promote invasive species with fast resource acquisition strategies (Borken and Matzner, 2009; Rembelski, 2019; Schrama and Bardgett, 2016; Zhang and others, 2023).
- Fire may suppress some invasive shrubs, but it can also promote pyrophytic invaders such as Chinese silvergrass (*Miscanthus sinensis*) and autumn olive (*Elaeagnus umbellata*) and shrubs like common buckthorn (*Rhamnus cathartica*) or invasive honeysuckles (*Lonicera* spp.) that vigorously resprout or persist through seed banks (Culbert and others, 2025; Illinois Department of Natural Resources, 2017; Vander Yacht and others, 2017; Zouhar and others, 2008).
- Flammable invasive grasses can reinforce feedbacks that increase fine fuel load and fire severity, perpetuating their dominance in burned areas (Fusco and others, 2019; Kerns and others, 2020).

Pests and pathogens:

- Warming accelerates pest and pathogen development, increasing outbreak risk from moths and bark beetles, but drought may harm moisture-reliant pathogens like root rots (*Armillaria* spp.) (Dukes and others, 2009; Harrington and McNew, 2018; Kubiak and others, 2017; Pureswaran and others, 2018; Stanturf and others, 2024).
- Drought weakens host defenses, increasing vulnerability to pests such as the twolined chestnut borer (*Agrilus bilineatus*) and red oak borer (*Enaphalodes rufulus*), particularly in red oak (*Quercus rubra*) and black oak (*Q. velutina*) (Fierke and others, 2007; Haavik and others, 2015; Stump and others, 2024; USDA Forest Service, USA, 2002).
- Root- and stem-infecting pathogens including root rots (*Armillaria* spp.) and water molds (*Phytophthora* spp.) are favored under fluctuating moisture regimes, intensifying disease spread (Haavik and others, 2015; Kolb and others, 2016; Kubiak and others, 2017; Wood and others, 2018).
- Fungal pathogens such as Hypoxylon canker (*Biscogniauxia atropunctata*) may shift to pathogenic states under drought, increasing oak decline and mortality (Ansley and others, 2023; Wood and others, 2018).
- Fire may indirectly reduce pest and pathogen pressure by thinning host populations, but fire-damaged trees can also attract wood-boring insects (Barkley, 2006; Canelles and others, 2021; Jenkins and others, 2014; Parker and others, 2006; Progar and others, 2014).

Herbivory:

- Warmer winters may increase white-tailed deer (*Odocoileus virginianus*) populations, intensifying browsing pressure on oak (*Quercus* spp.) seedlings and saplings and native forbs (Champagne and

others, 2021; Felton and others, 2024; Frelich and others, 2015; Hanberry and Abrams, 2018; Weiskopf and others, 2019).

- Drought may reduce plant investment in chemical defenses, increasing palatability and making drought-stressed species more vulnerable to herbivory, and reducing regeneration (Cole and others, 2021; Duell and others, 2024; Gutbrodt and others, 2011).
- Browsing in drought-stressed or severely burned habitats may reduce forb diversity, favor short-lived or weedy species and grasses, and limit regeneration of oaks and other native trees (Anderson and others, 2017, p. 202; Batzli and Dejaco, 2013; Urbanek and others, 2012).
- Moderate browsing pressure may reduce woody encroachment and fuel loads (Frelich and others, 2015; Lecomte and others, 2019; Urbanek and others, 2012; Velamazán and others, 2018; Zhang and others, 2019).

Table 23. Summary of impacts from key climate change stressors on Central Midwest Oak Openings and Barrens habitat species. Asterisks (*) denote species introduced to the region. Daggers (†) denote some species within the genus may be introduced to the region, though most are native. Some species appear in contradictory impacts (e.g., showing both high and low tolerance to drought), reflecting conflicting findings in the literature.

Stressor	Impact	Affected species
Drought	When tolerance of droughts is high, competitive advantage and/or increased abundance	*Andropogon gerardii*[1–4]
		Artemisia ludoviciana[4]
		Calamagrostis canadensis[5,6]
		Calamovilfa longifolia[4,7–9]
		Carex pensylvanica[10]
		Carya ovata[11]
		Koeleria macrantha[4,12,13]
		Panicum virgatum[14]
		Parthenocissus quinquefolia[4]
		Quercus alba[15–24]
		Quercus ellipsoidalis[25–27]
		Quercus macrocarpa[4,24,25,28,29]
		Quercus velutina[15,26,30]
		**Rhamnus cathartica*[31]
		Schizachyrium scoparium[4,9,32–34]
	When tolerance of droughts is low, reduced growth and/or abundance	*Acer rubrum*[4,20,24,35–41]
		Andropogon gerardii[14,32,34,42–45]
		**Berberis thunbergii*[4]
		Calamagrostis canadensis[4,9]
		Carex pensylvanica[46]
		Carya ovata[25]
		Hesperostipa spartea[47]

Stressor	Impact	Affected species
		**Lonicera maackii*[4]
		Panicum virgatum[9,48,49]
		Populus tremuloides[4,17,24,25,39,41,50–53]
		Quercus alba[11,25,54,55]
		Quercus bicolor[4,25,56]
		Quercus palustris[4,24,25]
		Quercus velutina[4,21,57]
		Schizachyrium scoparium[58]
		Sorghastrum nutans[32,48,59–61]
Fire	When tolerance of fire is high, competitive advantage and/or increased abundance	*Andropogon gerardii*[4]
		Calamagrostis canadensis[5,62]
		†*Carex* spp.[62]
		Carya ovata[4,63,64]
		Hesperostipa spartea[47]
		Koeleria macrantha[4,13]
		**Lonicera maackii*[4]
		**Lonicera tatarica*[4]
		Panicum virgatum[4,65]
		Populus tremuloides[4,50,63,64]
		Quercus alba[18,21,63,64,66–69]
		Quercus bicolor[63,64]
		Quercus ellipsoidalis[63,64,70]
		Quercus macrocarpa[4,27,63,64,66,67,71]
		Quercus palustris[64]
		Quercus velutina[63,64]
		Schizachyrium scoparium[4,33]
		Sorghastrum nutans[4,61]
		Sporobolus heterolepis[4,72]
	When tolerance of fire is low, increased mortality and/or reduced abundance	*Acer rubrum*[4,63,64,66,67,73]
		**Ailanthus altissima*[4]
		**Berberis thunbergii*[4]
		Calamagrostis canadensis[4]
		Calamovilfa longifolia[8]
		Carex pensylvanica[70]
		Carya ovata[74]
		Parthenocissus quinquefolia[4,75]
		Quercus palustris[76]
		Quercus velutina[4,78]

[1]Baty and others (2015); [2]Nippert and others (2009); [3]Slette and others (2023); [4]USDA, NRCS (2025); [5]Tesky (1992); [6]Wilcox and others (2022); [7]Duckwitz and others (2006); [8]Hauser (2005); [9]USDA, NRCS (2004); [10]Randall and Walters (2019); [11]Au and Maxwell (2022); [12]Watkins (2009); [13]Simonin (2000); [14]Watson and others (2021); [15]Brzostek and others (2014); [16]Duan and others (2022); [17]Gustafson and Sturtevant (2013); [18]Hayford and Chin (2020); [19]Maxwell (2024); [20]McEwan and others (2011); [21]Novick and others (2022); [22]Clarke and others (2016); [23]Rogers (1990); [24]Niinements and Vallardes (2006); [25]Peters and others (2015); [26]Rauschendorfer and others (2022); [27]Coladonato (1992c); [28]Johnson (1990); [29]Gucker (2011); [30]Carey (1992f); [31]Fisichelli and others (2014); [32]Lemoine and others (2017); [33]Steinberg (2002); [34]Dekirmenjian and others (2024); [35]Ibanez and others (2017); [36]Palus and others (2018); [37]Ramage and others (2023); [38]Reich and others (2022); [39]Swanston and others (2018); [40]Walters and others (2023); [41]Wang and others (2023); [42]Kooyers and others (2025); [43]Maricle and others (2017); [44]Pehim Limbu and Avolio (2025); [45]Uchytil (1988); [46]Nelson and others (2018); [47]Walkup (1991a); [48]Kazemi and others (2023); [49]Liu and others (2015); [50]Howard (1996); [51]Levesque and Hamann (2022); [52]Michaelian and others (2011); [53]Worrall and others (2013); [54]Kellner and Swihart (2016); [55]LeBlanc and Berland (2019); [56]Snyder (1992a); [57]Asbjornsen and others (2021); [58]Duell and others (2016); [59]Hoover and others (2014); [60]Swemmer and others (2006); [61]Walkup (1991b); [62]Apfelbaum and others (2023); [63]Hoving and others (2021); [64]Nowacki and Abrams (2015); [65]Uchytil (1993a); [66]Arthur and others (2021); [67]Varner (2018); [68]Hanberry and others (2012); [69]Tirmenstein (1991c); [70]Coladonato (1993); [71]Sander (1990a); [72]Snyder (1992b); [73]Walters and Yawney (1990); [74]Tirmenstein (1991a); [75]Coladonato (1991); [76]Carey (1992d); [77]Sander (1990a)

Focal Wildlife Species: Tri-colored Bat

The tricolored bat (*Perimyotis subflavus*) is one of three North American bat species that has suffered population declines exceeding 90% (U.S. Fish and Wildlife Service, 2021). Much of this loss is attributed to white-nose syndrome (*Pseudogymnoascus destructans*), which can cause 75–100% mortality in infected tri-colored bat populations. Climate change is expected to have complex interactions with white-nose syndrome in tricolored bats while also directly influencing bat physiology, behavior, and prey availability.

Bats are considered highly sensitive to climate stress due to their high surface-area-to-volume ratio (Korine and others, 2016). For tri-colored bats, yearly variation in temperature and precipitation has been correlated with birth size and growth rates, likely reflecting changes in maternal bat activity or prey availability (Hoying and Kunz, 1998). Heavy rainfall may further reduce body size and survival by limiting foraging opportunities and insect abundance, as shown in little brown bats (*Myotis lucifugus*) (Davy and others, 2022). Rain also imposes direct energetic costs through thermoregulation when bats become wet (Voigt and others, 2011).

As highly mobile species, tri-colored bats may shift their geographic range to track suitable climates (Festa and others, 2023). At the northern edge of their range, some populations already migrate south in the colder months, while others remain, moving from tree roosts to caves within the same region. With milder winters, long-distance migrations may become less necessary, and more bats may overwinter regionally (Fraser and others, 2012). Protecting migratory corridors and roosting sites will be essential as these strategies shift (Berthinussen and others, 2014).

In mild climates, tri-colored bats already forage during sporadic winter arousals when prey are available (Bernard and others, 2021). As winters in the Midwest become milder, tri-colored bats are likely to increase such intermittent winter activity. However, white-nose syndrome also elevates arousal

frequency and, when prey are not available, as is often the case in Midwestern winters, these events can deplete energy stores (Reeder and others, 2012).

Mortality from white-nose syndrome is mediated by environmental conditions, increasing with colder, drier hibernacula (Flory and others, 2012). Warming temperatures could reduce white-nose syndrome damage by shortening hibernation periods and increasing access to unfrozen water for bats that become active and dehydrated when infected, particularly in the southern part of the species' range outside the Midwest (Flory and others, 2012; McClure and others, 2022). Manipulating both hibernacula temperature and vapor pressure deficit (VPD) have been identified as possible management strategies, though the relationship between white-nose syndrome and hibernacula conditions is more complicated in tri-colored bats than other species studied (Turner and others, 2022).

Focal Aquatic System: Little Creek

Little Creek, located within Dunn Ranch Prairie in northern Missouri, is a headwater stream embedded in a landscape actively managed to retain native grassland characteristics. Draining into the Upper Grand River watershed, it exhibits intermittent flow, rapid runoff, and seasonal drying – hydrologic patterns shaped by convective storms, shallow soils, and elevated evapotranspiration (Dodds and others, 2004). These conditions support aquatic communities dominated by short-lived, drought-tolerant species such as minnows, crayfish, and invertebrates (Larson and others, 2013). Due to its intact prairie cover, low predator density, and proximity to extant populations, Little Creek is designated as a nonessential experimental population area for the federally endangered Topeka shiner (*Notropis topeka*) (U.S. Fish and Wildlife Service, 2013). Restoration efforts by The Nature Conservancy have included reconnecting the floodplain, stabilizing eroding streambanks, and installing a bioengineered ramp to facilitate native fish passage (The Nature Conservancy, 2024).

Despite these improvements, Little Creek remains vulnerable to broader climatic shifts. In tallgrass prairie watersheds, increased precipitation has paradoxically coincided with declining streamflow, largely due to woody encroachment, increased evapotranspiration, and altered subsurface flow paths (Keen and others, 2024b). Missouri's flood seasonality has shifted markedly since the mid-20th century, with peak flows now occurring 7–14 days later and lasting 20–30% longer (Barth and others, 2025), trends likely to continue under climate change (Table 24). These changes, driven by more intense rainfall and reduced snowpack storage, have transformed flood timing distributions from seasonally concentrated to prolonged and diffuse, a pattern consistent with climate-driven hydroclimatic variability. For first-order prairie streams like Little Creek, such shifts can disrupt nutrient cycling, reduce habitat availability during sensitive life stages, and weaken resilience to drought and disturbance (Barth and others, 2025; Dodds and others, 2004; Keen and others, 2024b; Larson and others, 2013).

Projected changes in fish communities further highlight this vulnerability. Small streams in the central United States, especially those supporting cool- and moderate-water species, face elevated risks of compositional turnover under warming and altered hydrology (Sievert and others, 2016). Habitat for these species is expected to contract, while warmwater generalists are likely to expand. However, streams embedded in intact prairie watersheds with minimal fragmentation may be less vulnerable to these changes, suggesting such systems may offer relative resilience and serve as important strongholds for native aquatic biodiversity in a changing climate (Larson and others, 2013; Sievert and others, 2016).

Table 24. Projected changes in streamflow for Little Creek. Projections use gridded simulations of daily runoff based on CMIP5 climate models under the high-emissions (RCP 8.5) scenario. Values show average historical (1977–2006) 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 (Authors (in review); U.S. Department of the Interior, Bureau of Reclamation, 2014; USDA Forest Service Office of Sustainability and Climate, 2022; Wenger and others, 2010).

Streamflow metric	1977–2006	2070–2099	Percent change
Mean Winter (December–February) Flow	1.4 ft^3/s	2 ft^3/s	38%
Mean Spring (March–May) Flow	3.2 ft^3/s	4.7 ft^3/s	48%
Mean Summer (June–August) Flow	2.5 ft^3/s	2.3 ft^3/s	-7%
Mean Fall (September–November) Flow	2.5 ft^3/s	2.6 ft^3/s	2%
Mean Annual Flow	2.4 ft^3/s	2.9 ft^3/s	20%
1.5-year Flood (33rd percentile)	37.2 ft^3/s	46.9 ft^3/s	26%
10-year Flood (90th percentile)	147.4 ft^3/s	213.4 ft^3/s	45%
25-year Flood (96th percentile)	192.5 ft^3/s	279.8 ft^3/s	45%
Number of Winter Floods	3.0	4.0	34%
Average Lowest 7-day Flow	0.1 ft^3/s	0.1 ft^3/s	0%
Center of Flow Mass Timing	April 13	April 9	-2%

North-Central Oak- Hickory Forest and Woodland

The North-Central Oak - Hickory Forest and Woodland habitat is a broadly distributed habitat type spanning much of the Midwest as well as parts of Pennsylvania, New York, and southeast Canada (Figure 12), commonly occurring on rolling to moderately steep uplands, slopes, and glacial till plains (Curtis, 1959; Davis, 1977; Fralish, 2003; USNVC, 2025). These communities occupy dry-mesic to mesic sites with moderately- to well-drained soils that range from shallow to deep and are often loamy in texture (Curtis, 1959; Kost and others, 2007; Leach and Givnish, 1999; Taft, 1997; USNVC, 2025; Will-Wolf and Stearns, 1999). The overstory canopy is typically dominated by drought- and fire-tolerant hardwoods, particularly oaks (*Quercus* spp.) and hickories (*Carya* spp.), with the subcanopy, where present, supporting more mesic or shade-tolerant species, and the understory supporting a mix of grasses, thicket-forming shrubs, and a diverse mix of forbs (Table 25). Sites with deeper, mesic soils tend to support moderately closed canopies with richer tree diversity, while drier, thinner soils support more open canopies with lower overstory density (Curtis, 1959; Fralish, 2004; Grimm, 1984; Kost and others, 2007; Leach and Givnish, 1999; USNVC, 2025; Will-Wolf and Stearns, 1999).

Historically, periodic low- to moderate-frequency surface fires, typically recurring every 3 to 15 years, maintained moderately open conditions by limiting encroachment of shade-tolerant, fire-sensitive species and resetting successional trajectories (Abrams, 1992; Grimm, 1984; Hanberry and others, 2020; Knapp and others, 2015; Kost and others, 2007; McClain and others, 2010; Taft, 1997). These dynamics were strongest at dry, exposed sites, whereas wetter sites shielded by natural firebreaks, such as rivers or steep ravines, burned less frequently and sometimes succeeded to closed-canopy forest (Abrams, 1992; Brose and others, 2014; Frelich and others, 2015; Grimm, 1984; Nowacki and Gundy, 2021; USNVC, 2025). Periodic drought also reinforced the dominance of oaks and hickories, particularly on xeric sites (Abrams, 1990; Anderson and Bowles, 1999; Au and Maxwell, 2022; Barros and others, 2018; Fralish, 2004; Kost and others, 2007; Will-Wolf and Stearns, 1999).

Over the past century, anthropogenic fire suppression, land conversion, timber harvesting, and invasive species have altered historical habitat structure and function. Fire-sensitive woody species have proliferated, increasing stem density and shading out fire-adapted herbs and shrubs (Brandt and others, 2014; Brose and others, 2014; Dey and Kabrick, 2015; Hanberry and others, 2020; Knapp and others, 2015; Nowacki and Abrams, 2008; Pyne, 2017; Stambaugh and others, 2021). This process of "mesophication" has shifted many stands toward closed-canopy forests dominated by shade-tolerant mesophytes, reducing biodiversity and structural heterogeneity (Alexander and others, 2021; Duan and others, 2023; Hutchinson and others, 2024; Nowacki and Abrams, 2008; Vose and others, 2021). Today, remaining habitat is further threatened by invasive species, including autumn olive (*Elaeagnus umbellata*), Canada bluegrass (*Poa compressa*), common buckthorn (*Rhamnus cathartica*), hoary alyssum (*Berteroa incana*), invasive honeysuckles (*Lonicera* spp.), Oriental bittersweet (*Celastrus orbiculatus*), multiflora rose (*Rosa multiflora*), and spotted knapweed (*Centaurea stoebe*) (Kost and others, 2007).

Table 25. Plant species common to the North-Central Oak- Hickory Forest and Woodland habitat. Asterisks (*) denote species introduced to the region. Daggers (†) denote some species within the genus may be introduced to the region. Taxonomic authority, common name, growth habit, and native status are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Acer saccharum Marshall	Sugar maple	Shrub, tree
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roem.	Saskatoon serviceberry	Shrub, tree
Betula alleghaniensis Britton	Yellow birch	Tree
Betula papyrifera Marshall	Paper birch	Tree
Carya cordiformis (Wangenh.) K. Koch	Bitternut hickory	Tree
Carya glabra (Mill.) Sweet	Pignut hickory	Tree
Carya ovalis (Wangenh.) Sarg.	Red hickory	Tree
Carya ovata (Mill.) K. Koch	Shagbark hickory	Tree
Carya spp. Nutt.	Hickory	Tree
Carya tomentosa (Lam.) Nutt.	Mockernut hickory	Tree
Cornus drummondii C.A. Mey.	Roughleaf dogwood	Shrub, tree
Corylus cornuta Marshall	Beaked hazelnut	Shrub, tree
Ostrya virginiana (Mill.) K. Koch	Eastern hophornbeam	Shrub, tree
Pinus strobus L.	Eastern white pine	Tree
†*Populus* spp. L.	Cottonwood	Tree
Populus tremuloides Michx.	Quaking aspen	Tree
Quercus alba L.	White oak	Tree
Quercus bicolor Willd.	Swamp white oak	Tree
Quercus ellipsoidalis E.J. Hill	Northern pin oak	Tree
Quercus macrocarpa Michx.	Bur oak	Shrub, tree
Quercus muehlenbergii Engelm.	Chinkapin oak	Tree
Quercus palustris Münchh.	Pin oak	Tree
Quercus rubra L.	Northern red oak	Tree
Quercus stellata Wangenh.	Post oak	Tree
Quercus velutina Lam.	Black oak	Tree
Tilia americana L.	American basswood	Tree
Tsuga canadensis (L.) Carrière	Eastern hemlock	Tree
Amphicarpaea bracteata (L.) Fernald	American hogpeanut	Forb/herb, vine
Andropogon gerardii Vitman	Big bluestem	Graminoid
Apios americana Medik.	Groundnut	Forb/herb, vine
Bromus kalmii A. Gray	Arctic brome	Graminoid
Calamagrostis canadensis (Michx.) P. Beauv.	Bluejoint	Graminoid
Carex pensylvanica Lam.	Pennsylvania sedge	Graminoid
Chasmanthium latifolium (Michx.) Yates	Indian woodoats	Graminoid
Desmodium glutinosum (Muhl. ex Willd.) Alph. Wood	Pointedleaf ticktrefoil	Forb/herb
Dichanthelium boscii (Poir.) Gould & C.A. Clark	Bosc's panicgrass	Graminoid
Dichanthelium oligosanthes (Schult.) Gould	Heller's rosette grass	Graminoid

Species	Common name	Growth habit
Elymus hystrix L.	Eastern bottlebrush grass	Graminoid
Elymus virginicus L.	Virginia wildrye	Graminoid
Festuca subverticillata (Pers.) Alexeev	Nodding fescue	Graminoid
Galium boreale L.	Northern bedstraw	Forb/herb
Geranium maculatum L.	Wild geranium	Forb/herb
**Poa pratensis* L.	Kentucky bluegrass	Graminoid
Pteridium aquilinum (L.) Kuhn	Bracken fern	Forb/herb

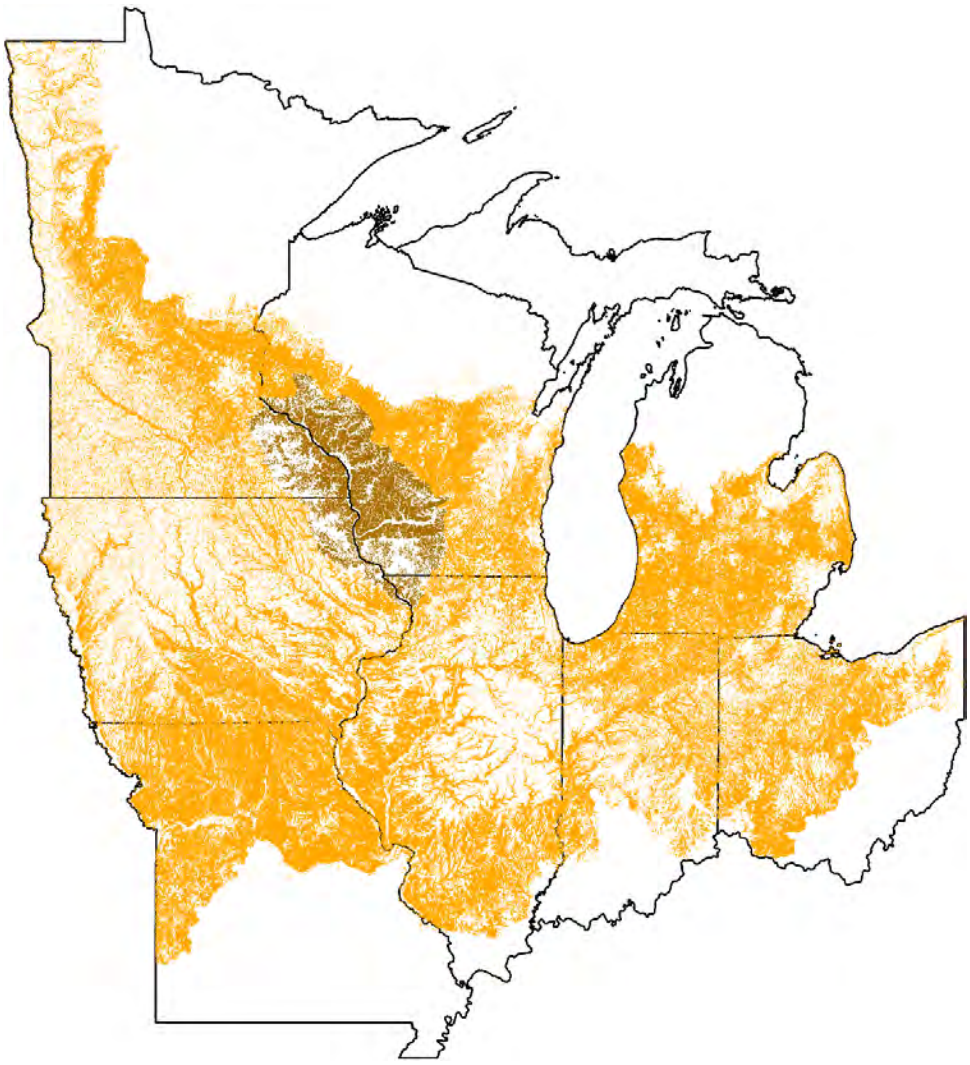

Figure 14. North-Central Oak- Hickory Forest and Woodland habitat distribution. Geographic distribution of the North-Central Oak- Hickory Forest and Woodland habitat (Group Code: G649) in the Midwest (USNVC, 2020). Dark orange areas indicate North-Central Oak- Hickory Forest and Woodland habitat in the Driftless Area ecoregion, while light orange areas mark additional locations throughout the Midwest. Habitat boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The North-Central Oak - Hickory Forest and Woodland habitat is prioritized for the Driftless Area ecoregion (Table i), and climate projections for this area largely reflect regional trends. 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 26) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). Most models also project an increase in annual precipitation, averaging 2.9 additional inches per year, although some indicate a slight decline (Table 27) (Marvel and others, 2023). However, precipitation is expected to become more variable, with more extreme events and notably drier summers despite the overall increase (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 26. 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 27. 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%)

Key Climate Change Stressors

Drought and fire. Across the Driftless Area ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 72% above historical conditions by the end of the century (RCP 8.5; Table 28), fairly uniformly throughout the region (Figure 13). Higher VPD means more rapid evaporation and

water loss from plants and soils, which can intensify drought conditions. Annual extreme fire danger is also projected to increase by approximately 81% by mid-century under the high emissions scenario, adding more than 9 days of elevated risk (Table 29). As with VPD, the fire risk is mostly uniform across the region, though may be slightly higher to the northeast, where the maximum model indicates as many as 60 days of extreme risk annually (Figure 14). Rising fire risk could increase the frequency of wildfire and push this habitat toward more historically frequent fire intervals, but it also raises the likelihood of extreme events, including crown fires, that exceed historical norms and constrain opportunities to safely apply prescribed fire.

Table 28. Projected changes in summer vapor pressure deficit 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 summer vapor pressure deficit (VPD; kilopascals) 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 VPD 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	0.91 kPa	0.88 kPa (-3%)	0.91 kPa (0%)	1.00 kPa (9%)	1.07 kPa (18%)
Mean of models	0.95 kPa	1.20 kPa (26%)	1.32 kPa (38%)	1.30 kPa (36%)	1.64 kPa (72%)
Maximum model	0.98 kPa	1.42 kPa (45%)	1.62 kPa (65%)	1.73 kPa (77%)	2.42 kPa (147%)

Table 29. Projected changes in annual extreme fire danger 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 extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) by mid-century (2040–2069), 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 annual extreme fire danger under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069	
	Historical	RCP 4.5	RCP 8.5
Minimum model	11.0	8.1 (-26%)	6.6 (-40%)
Mean of models	11.0	18.1 (65%)	19.8 (81%)
Maximum model	11.0	51.0 (365%)	40.8 (272%)

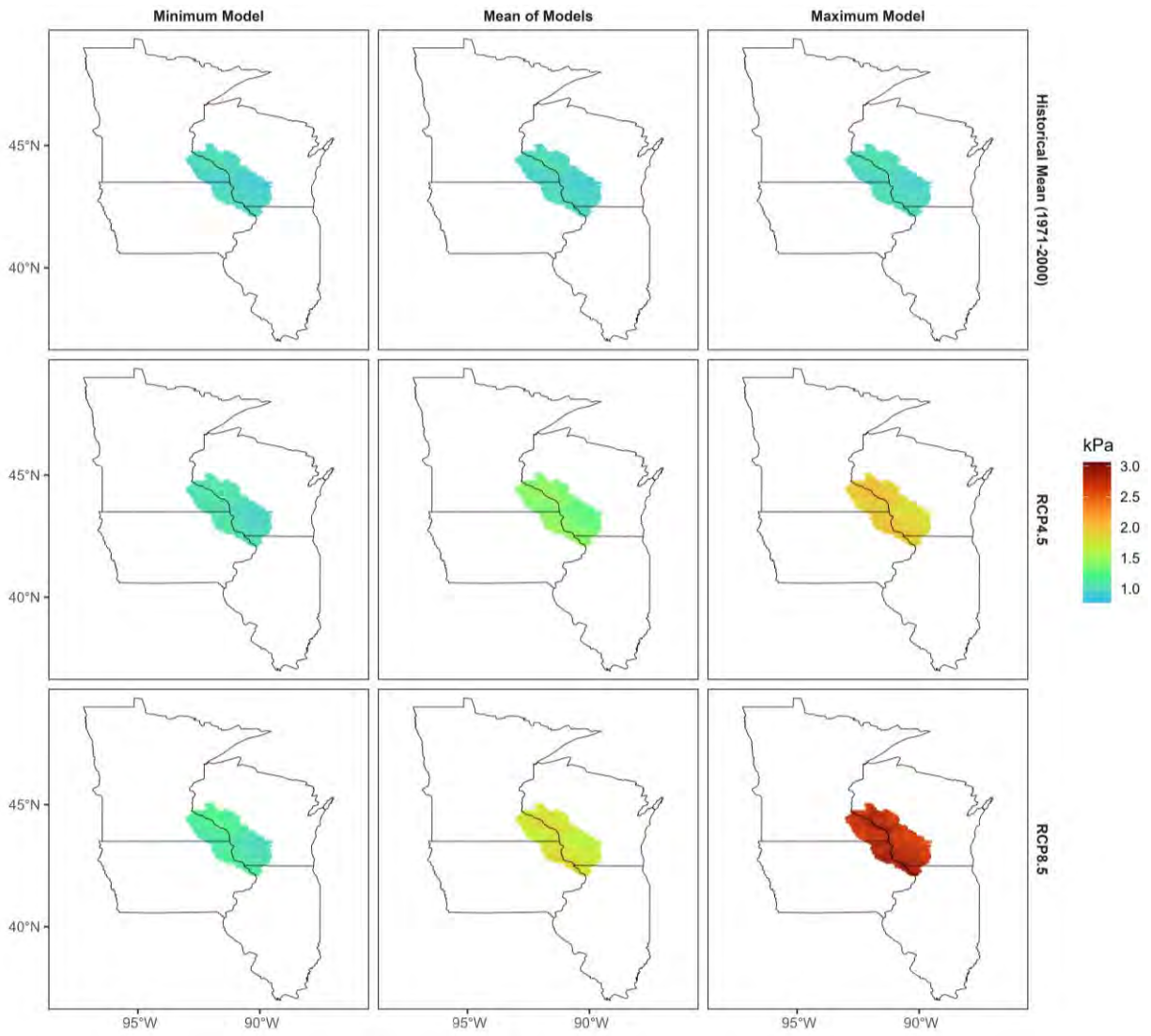

Figure 15. Projected changes in summer vapor pressure deficit in the Driftless Area ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000, top row) and projected (2070–2099) annual summer pressure deficit (VPD; kilopascals) 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 VPD under RCP 8.5 are presented in addition to the mean of all models (middle column).

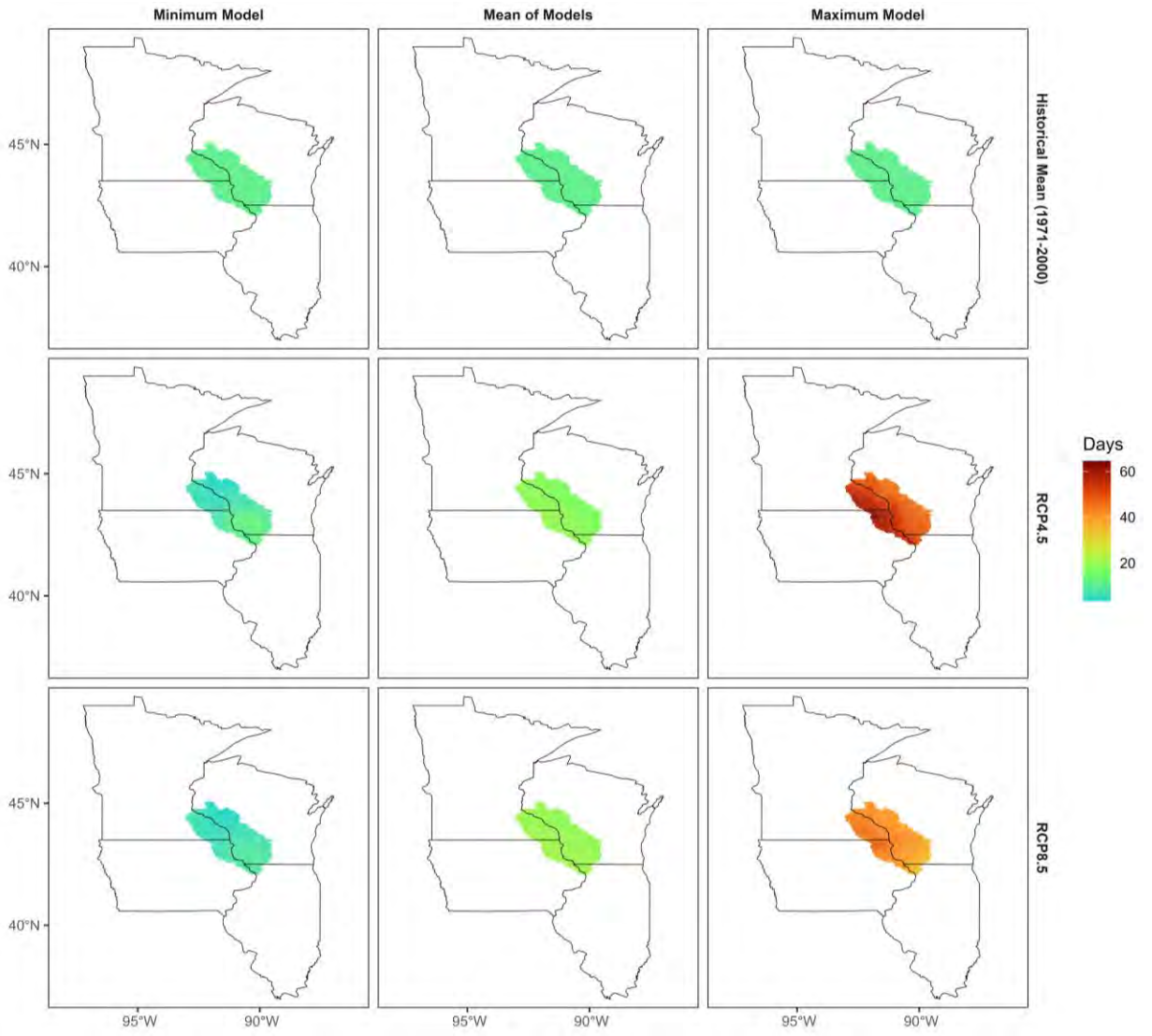

Figure 16. Projected changes in annual extreme fire danger in the Driftless Area ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show average historical (1971–2000, top row) and projected (2040–2069) annual extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) 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 extreme fire danger under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

Climate change is affecting the North-Central Oak - Hickory Forest and Woodland habitat through its overall influence on species distributions, adaptive capacity, and vulnerability, as well as through the specific impacts of drought and fire (Table 30). 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:

- Drought can intensify soil drying, reduce herbaceous cover, and increase erosion risk, particularly when followed by heavy rainfall (Breshears and others, 2016; Vose and others, 2016; Ziegler and others, 2008).
- Drought may alter nutrient dynamics by reducing plant uptake, allowing inorganic nitrogen to accumulate, and triggering post-drought nutrient flushes, destabilizing soil fertility (Hartmann and others, 2013; Homyak and others, 2017; Reinsch and others, 2024; Schlesinger and others, 2016).
- Soil microbial activity and decomposition decline under drought, reducing nutrient availability and favoring drought-tolerant microbial taxa (Baldrian and others, 2023; Lemoine and others, 2024).
- Low-intensity fire removes surface vegetation, increases understory light availability, and mobilizes nutrients through ash deposition, while high-severity or repeated fire can degrade soil structure, increase erosion risk, reduce infiltration capacity, and drive nutrient loss, especially on sloped, shallow soils typical of the Driftless Area (Arthur and others, 2021; Fernández and others, 2006; Hanberry and others, 2017; Knicker, 2007; Pellegrini and others, 2020; Singh and others, 2025).

Community composition:

- Drought may shift communities toward xeric-adapted species, particularly drought-adapted warm-season (C_4) grasses such as big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and Indiangrass (*Sorghastrum nutans*) (Batllori and others, 2020; Havrilla and others, 2023; Knapp and others, 2024).
- Mesophytic species like sugar maple (*Acer saccharum*), red maple (*A. rubrum*), and yellow birch (*Betula alleghaniensis*) are vulnerable to repeated drought, leading to recruitment failure and decline (Boakye and others, 2023; Coble and others, 2017; Moore and others, 2016).
- Oaks (e.g., *Quercus alba*, *Q. velutina*, *Q. rubra*, *Q. macrocarpa*) and hickories (*Carya* spp.) may gain a competitive advantage under moderate drought as mesic species decline, but severe or repeated drought can lead to recruitment failure and canopy mortality (Asbjornsen and others, 2021; Druckenbrod and others, 2019; Novick and others, 2022).
- Frequent, low-intensity fire promotes pyrophytic assemblages, reducing dominance of mesic trees and increasing fire-adapted species like oaks, hickories, and grasses, while severe or overly frequent fire can reduce recruitment and diversity (Arthur and others, 2012; Hutchinson and others, 2024).
- In the absence of fire, drought may favor the expansion of shrubs and encroaching trees such as eastern red cedar (*Juniperus virginiana*), roughleaf dogwood (*Cornus drummondii*), and smooth sumac (*Rhus glabra*), which suppress herbaceous diversity and future fire spread (Hanberry, 2022; O'Keefe and others, 2024; Ratajczak and others, 2011; Volder and others, 2010).
- Drought and fire suppression may catalyze transitions toward either grassland, shrubland, or closed-canopy mesic forest, depending on disturbance history and recruitment dynamics (Breshears and others, 2016; Frelich and Reich, 2010; Smith and Goetz, 2021; Vose and others, 2016).

Invasive species:

- Drought favors invasive species with broad tolerances, including smooth brome (*Bromus inermis*), Nepalese browntop (*Microstegium vimineum*), and cogongrass (*Imperata cylindrica*), which can suppress native grasses and tree recruitment (Duell and others, 2016; Warren and others, 2011).
- Woody invaders like common buckthorn (*Rhamnus cathartica*) and invasive honeysuckles (*Lonicera* spp.) persist under drought and resprout after disturbance, gaining advantage in closed-canopy, mesic sites (Knight and others, 2007; Zouhar and others, 2008).
- Fire may suppress some invasive shrubs if applied frequently or during the late growing season, but it can also favor pyrophytic invaders such as Chinese silvergrass (*Miscanthus sinensis*) and autumn olive (*Elaeagnus umbellata*) or vigorous resprouters (Culbert and others, 2025; Illinois Department of Natural Resources, 2017; Vander Yacht and others, 2017; Zouhar and others, 2008).
- Flammable invasive grasses can reinforce feedbacks that increase fine fuel load and fire severity, perpetuating their dominance in burned areas, while woody invaders can dampen fire spread by shading herbaceous fuels (Culbert and others, 2025; Dibble and others, 2008; Fusco and others, 2019; Kerns and others, 2020).

Pests and pathogens:

- Warming accelerates pest and pathogen development, increasing outbreak risk from bark beetles, moths, and pathogens such as root rots (*Armillaria* spp.) and bur oak blight (*Tubakia iowensis*) (Dukes and others, 2009; Harrington and McNew, 2018; Kubiak and others, 2017; Pureswaran and others, 2018; Stanturf and others, 2024).
- Drought weakens host defenses, increasing susceptibility to cambium- and wood-boring beetles such as emerald ash borer (*Agrilus planipennis*), twolined chestnut borer (*A. bilineatus*), and red oak borer (*Enaphalodes rufulus*), particularly in red oak (*Quercus rubra*) and black oak (*Q. velutina*) (Fierke and others, 2007; Haavik and others, 2015; Stump and others, 2024; USDA Forest Service, USA, 2002).
- Root- and stem-infecting pathogens including root rots (*Armillaria* spp.) and water molds (*Phytophthora* spp.) and foliar diseases like powdery mildew (Erysiphales spp.) are favored under fluctuating moisture regimes (Haavik and others, 2015; Kolb and others, 2016; Kubiak and others, 2017; Wood and others, 2018).
- Fungal pathogens such as Hypoxylon canker (*Biscogniauxia atropunctata*) may shift to pathogenic states under drought, increasing oak mortality (Ansley and others, 2023; Wood and others, 2018).
- Fire may indirectly reduce pest and pathogen pressure by thinning host populations, but fire-damaged trees can also attract bark beetles (Barkley, 2006; Canelles and others, 2021; Jenkins and others, 2014; Parker and others, 2006; Progar and others, 2014).

Herbivory:

- Warmer winters may increase white-tailed deer (*Odocoileus virginianus*) populations, intensifying browsing pressure on oak (*Quercus* spp.) seedlings, hickories (*Carya* spp.), and mesic hardwoods (Champagne and others, 2021; Felton and others, 2024; Frelich and others, 2015; Hanberry and Abrams, 2018; Weiskopf and others, 2019).
- Drought may reduce plant investment in chemical defenses, increasing palatability, reducing regeneration, and shifting composition (Cole and others, 2021; Duell and others, 2024; Gutbrodt and others, 2011).

- Browsing in drought-stressed or severely burned habitats may reduce forb diversity, favor short-lived or weedy species and grasses, and limit regeneration of oaks and other native trees (Anderson and others, 2017, p. 202; Batzli and Dejaco, 2013; Urbanek and others, 2012).
- Moderate browsing pressure may reduce woody encroachment and fuel loads (Frelich and others, 2015; Lecomte and others, 2019; Urbanek and others, 2012; Velamazán and others, 2018; Zhang and others, 2019).

Table 30. Summary of impacts from key climate change stressors on North-Central Oak- Hickory Forest and Woodland habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory impacts (e.g., showing both high and low tolerance to drought), reflecting conflicting findings in the literature.

Stressor	Impact	Affected species
Drought	When tolerance of droughts is high, competitive advantage and/or increased abundance	*Andropogon gerardii*[1–4]
		Calamagrostis canadensis[5,6]
		Carex pensylvanica[7]
		Carya cordiformis[4–10]
		Carya glabra[4,10]
		Carya ovata[11]
		Cornus drummondii[12–15]
		Pinus strobus[16]
		**Poa pratensis*[17]
		Quercus alba[10,16,18,–25]
		Quercus ellipsoidalis[9,26, 27]
		Quercus macrocarpa[4,9,10,28,29]
		Quercus muehlenbergii[4,10]
		Quercus rubra[16,17,21,25,26,30,31]
		Quercus stellata[4,10,32–35]
		Quercus velutina[25,26,36]
		**Rhamnus cathartica*[37]
	When tolerance of droughts is low, reduced growth and/or abundance	*Acer saccharum*[7,9,10,20,21,37–46]
		Amelanchier alnifolia[4]
		Andropogon gerardii[47–53]
		Apios americana[4]
		**Berberis thunbergii*[4]
		Betula alleghaniensis[9,16,30,54,55]
		Betula papyrifera[4,9,10,16,42,46,56]
		Calamagrostis canadensis[4,57]
		Carex pensylvanica[58]
		Carya glabra[9]
		Carya ovata[9]

		Carya tomentosa[9]
		Cornus drummondii[4]
		Corylus cornuta[4]
		Elymus virginicus[59]
		Festuca subverticillata[4]
		Galium boreale[60]
		Geranium maculatum[61]
		**Lonicera maackii*[4]
		Ostrya virginiana[9]
		Pinus strobus[4,9,10,42,62,63,64]
		**Poa pratensis*[4,65–72]
		Populus tremuloides[4,9,10,16,45,46,73–76]
		Quercus alba[9,10,77,78]
		Quercus bicolor[4,9,79]
		Quercus muehlenbergii[9]
		Quercus palustris[4,9,10]
		Quercus rubra[9,22,41,42,44,46,80]
		Quercus stellata[22,81]
		Quercus velutina[4,22,63]
		Tilia americana[4,45]
		Tsuga canadensis[4,9,10,16,82–88]
Fire	When tolerance of fire is high, competitive advantage and/or increased abundance	*Amelanchier alnifolia*[4]
		Andropogon gerardii[4]
		Betula papyrifera[62,89–92]
		Bromus kalmii[4]
		Calamagrostis canadensis[5,62]
		Carya glabra[89,90]
		Carya ovata[4,89,90]
		Carya spp.[93,94]
		Carya tomentosa[90]
		Corylus cornuta[4,95]
		Festuca subverticillata[4]
		Galium boreale[96]
		**Lonicera maackii*[4]
		**Lonicera tatarica*[4]
		Pinus strobus[82,89,90,97]
		**Poa pratensis*[4]
		Populus tremuloides[4,73,89,90]
		Pteridium aquilinum[4,98]

	Quercus alba[19,22,89,90,93,94,99,100]
	Quercus bicolor[89,90]
	Quercus ellipsoidalis[89,90,101]
	Quercus macrocarpa[4,29,89,90,93,94,102]
	Quercus muehlenbergii[90,93,94]
	Quercus palustris[90]
	Quercus rubra[4,22,82,90,93,94]
	Quercus stellata[90,93,94]
	Quercus velutina[89,90]
	Tilia americana[4]
When tolerance of fire is low, increased mortality and/or reduced abundance	*Acer saccharum*[89,90,93,94]
	**Ailanthus altissima*[4]
	Apios americana[4]
	**Berberis thunbergii*[4]
	Betula alleghaniensis[4,54,89,90,103]
	Calamagrostis canadensis[4]
	Carex pensylvanica[101]
	Carya cordiformis[89,90,104,105]
	Carya ovata[106]
	Carya tomentosa[104]
	Chasmanthium latifolium[4]
	Cornus drummondii[4,107]
	Elymus virginicus[108]
	Galium boreale[4]
	Geranium maculatum[109]
	Ostrya virginiana[89,90,110]
	Pinus strobus[4,93,94,111]
	Quercus muehlenbergii[4]
	Quercus palustris[112]
	Quercus rubra[29,89]
	Quercus stellata[31]
	Quercus velutina[4,113]
	Tilia americana[89,90,114,115]
	Tsuga canadensis[87,89,90,116]

[1]Baty and others (2023); [2]Nippert and others (2009); [3]Slette and others (2023); [4]USDA, NRCS (2025); [5]Tesky (1992); [6]Wilcox and others (2022); [7]Randall and Walters (2019); [8]Coladonato (1991); [9]Peters and others (2015); [10]Niinements and Vallardes (2006); [11]Au and Maxwell (2022); [12]Muench and others (2016); [13]O'Keefe and others (2024); [14]Ratajczak and others (2011); [15]Keen and others (2022); [16]Gustafson and Sturtevant (2013); [17]Printz and Hendrickson (2015); [18]Duan and others (2022); [19]Hayford and Chin (2020); [20]Maxwell (2024); [21]McEwan and others

(2011); [22]Novick and others (2022); [23]Clarke and others (2016); [24]Rogers (1990); [25]Brzostek and others (2014); [26]Rauschendorfer and others (2022); [27]Coladonato (1992c); [28]Johnson (1990); [29]Gucker (2011); [30]Walters and others (2023); [31]Sander (1990b); [32]Ni and Pallardy (1991); [33]Ni and Pallardy (1992); [34]Pallardy and Rhoads (1993); [35]Carey (1992e); [36]Carey (1992f); [37]Fisichelli and others (2014); [38]Au and others (2020); [39]Bapikee (2013); [40]Boakye and others (2023); [41]Lee and Ibanez (2021); [42]Reich and others (2022); [43]Rudolph and LeBlanc (2020); [44]Stangler and others (2016); [45]Swanston and others (2018); [46]Wang and others (2023); [47]Dekirmenjian and others (2024); [48]Kooyers and others (2025); [49]Lemoine and others (2017); [50]Maricle and others (2017); [51]Pehim Limbu and Avolio (2025); [52]Uchytil (1988); [53]Watson and others (2021); [54]Sullivan (1994a); [55]Erdmann (1990); [56]Perillo (2017); [57]USDA, NRCS (2004); [58]Nelson and others (2018); [59]Rudgers and Swafford (2009); [60]Schuette (2022); [61]Šajbidorová and others (2019); [62]Apfelbaum and others (2023); [63]Asbjornsen and others (2021); [64]Livingston and Kenefic (2018); [65]Ahler and others (2025); [66]Huff (2003); [67]Richardson (2008); [68]Abraham and others (2004); [69]Stevens (1950); [70]Braun and others (2022); [71]Palit and others (2021); [72]Uchytil (1993b); [73]Howard (1996); [74]Levesque and Hamann (2022); [75]Michaelian and others (2011); [76]Worrall and others (2013); [77]Kellner and Swihart (2016); [78]LeBlanc and Berland (2019); [79]Snyder (1992a); [80]Lockwood and others (2023); [81]Prolic and Goldblum (2016); [82]Peters and others (2020); [83]Goring and Williams (2017); [84]Thomas and Orwig (2025); [85]Shuman and others (2019); [86]Goerlich and Nyland (1980); [87]Godman and Lancaster (1990); [88]Smith (2008) 35; [89]Hoving and others (2021); [90]Nowacki and Abrams (2015); [91]Swanston and others (2011); [92]Uchytil (1991); [93]Arthur and others (2021); [94]Varner (2018); [95]Fryer (2007); [96]Gucker (2005); [97]Carey (1993a); [98]Crane (1990); [99]Hanberry and others (2012); [100]Tirmenstein (1991c); [101]Coladonato (1993); [102]Sander (1990a); [103]Pike (2022); [104]Coladonato (1992a); [105]Smith (1990); [106]Tirmenstein (1991a); [107]Keen and others (2024); [108]Shadow and Jensen (2020); [109]Bowles and others (2007); [110]Coladonato (1992b); [111]Wendel and Smith (1990); [112]Carey (1992d); [113]Sander (1990a); [114]Sullivan (1994b); [115]Crow (1990); [116]Carey (1993b)

Focal Wildlife Species: Timber Rattlesnake

The timber rattlesnake (*Crotalus horridus*) is a large venomous snake found in the eastern and central United States and considered vulnerable, imperiled, or critically imperiled across much of the Midwest (NatureServe, 2025). Historical declines were driven by persecution, habitat loss, road mortality, and the pet trade (Clark and others, 2010; Martin and others, 2008; New York State Department of Environmental Conservation, 2025). For remaining populations, barriers to movement and gene flow threaten long-term viability (Clark and others, 2010).

Like other ectotherms, timber rattlesnakes balance thermal regulation with other fundamental needs, including reproduction (Martin, 1993) and predator avoidance (Herr and others, 2020; Martin, 1993). Open basking sites, often on rocky outcrops, are important for maintaining body temperature (Clifford and others, 2020). While warming may benefit populations at the northern edge of the range, the species' capacity for temperature acclimation may be limited (Jesper and others, 2023). High rainfall, projected to increase under climate change in the Midwest (Marvel and others, 2023), and cloud cover are associated with lower reproductive success and, in some cases, local extirpation (Aldridge and Duvall, 2002; Martin, 1993). For example, the last population of timber rattlesnake in New Hampshire lacked genetic diversity and exhibited signs of a population bottleneck, and following a particularly wet summer, altered hibernacula conditions may have fostered fungal disease and led to their extinction (Clark and others, 2011).

Management options to support this species include tree removal to replicate thermal conditions at basking and gestation sites (Clifford and others, 2020), wildlife crossings or translocations to promote gene flow and increase adaptive capacity (Clark and others, 2011), and proactive planning to anticipate and reduce human-snake conflicts (Keener-Eck and others, 2020). Climate-informed management will need to integrate ecological requirements with social considerations to promote persistence of this vulnerable species.

Focal Aquatic System: Root River

The Root River in southeastern Minnesota drains about 4,100 km^2 of Driftless Area terrain shaped by steep relief and karstic limestone and dolomite (Minnesota Pollution Control Agency, 2012). Fractured bedrock, springs, sinkholes, and losing reaches create strong groundwater–surface water connections that sustain cool baseflows and support trout in tributaries such as Camp Creek and the South Branch, while the mainstem transitions to warmwater assemblages (Minnesota Pollution Control Agency, 2012; Minnesota Pollution Control Agency, 2024). Sediment and nutrients from intensive agriculture and legacy channel alterations remain chronic stressors. Nearly half of assessed reaches were determined to be unsuitable for aquatic life, with biological monitoring most often pointing to degraded invertebrate communities, poor habitat, and elevated bacteria (Minnesota Pollution Control Agency, 2012; Minnesota Pollution Control Agency, 2024). These conditions, combined with the watershed's karst hydrogeology and dependence on cold water inputs, make the Root River highly sensitive to shifts in precipitation and temperature.

Observed changes in the Root River, combined with regional climate trends, highlight the system's vulnerability to shifting flow and temperature regimes. The basin experienced recent damaging floods that reworked channels, accelerated bank erosion, and stressed aquatic communities (Dauwalter and Mitro, 2019; Minnesota Pollution Control Agency, 2012). More broadly across Minnesota, heavy rainfall has intensified and peak flows have risen in agricultural watersheds (Williams-Sether and Sanocki, 2025). Projections for the Upper Midwest point to wetter springs, hotter and drier late summers, and higher evaporative demand, trends that amplify flood hazards (Table 31) while increasing pollutant concentrations during low flows (Scholl and others, 2025). Although groundwater inputs buffer stream temperatures and sustain baseflow, they are expected to gradually warm and shift toward greater winter and spring recharge (Potter, 2019; Scholl and others, 2025). Regional modeling suggests cold and transition habitats will contract away from non-spring reaches (Stewart and others, 2015).

Groundwater springs are important thermal refugia, where trout remain more abundant but often in poorer condition, underscoring that springs provide only partial protection as air and groundwater temperatures rise (Varela and others, 2024). More broadly, trout recruitment in Midwestern streams has declined during warming summers and low autumn flows (Maitland and Latzka, 2022), and warmer conditions can tip species interactions, including parasite outbreaks, against brook trout (*Salvelinus fontinalis*) (Dauwalter and Mitro, 2019).

Table 31. Projected changes in streamflow for Root 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 (1977–2006) 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 (Authors (in review); U.S. Department of the Interior, Bureau of Reclamation, 2014; USDA Forest Service Office of Sustainability and Climate, 2022; Wenger and others, 2010).

Streamflow metric	1977–2006	2070–2099	Percent change
Mean Winter (December–February) Flow	116.5 ft^3/s	192.2 ft^3/s	65%
Mean Spring (March–May) Flow	428.1 ft^3/s	547.3 ft^3/s	28%
Mean Summer (June–August) Flow	361.1 ft^3/s	347.8 ft^3/s	-4%
Mean Fall (September–November) Flow	160.4 ft^3/s	170.6 ft^3/s	6%

Mean Annual Flow	267.5 ft^3/s	315.4 ft^3/s	18%
1.5-year Flood (33rd percentile)	2863.3 ft^3/s	3174.5 ft^3/s	11%
10-year Flood (90th percentile)	4959.2 ft^3/s	5843.9 ft^3/s	18%
25-year Flood (96th percentile)	6161.4 ft^3/s	7784.1 ft^3/s	26%
Number of Winter Floods	2.6	3.2	22%
Average Lowest 7-day Flow	28.4 ft^3/s	18.9 ft^3/s	-33%
Center of Flow Mass Timing	May 4	April 23	-5%

Central Appalachian-Northeast Oak Forest and Woodland

The Central Appalachian-Northeast Oak Forest and Woodland habitat occurs primarily across the western Allegheny Plateau in southeastern Ohio, extending into Pennsylvania and West Virginia (Figure 15) (USNVC, 2025). These communities are most common on upland slopes and ridge systems with shallow, well-drained, acidic soils (Braund and Zimmerman, 2022; USNVC, 2025; Zaremba and others, 2003). Vegetation reflects the drought-prone, fire-adapted nature of these sites. Oaks (*Quercus* spp.) are often dominant (Table 32), but where fire is suppressed or local conditions are wetter, dense subcanopy and midstory layers of shade-tolerant species such as red maple (*Acer rubrum*) can develop, reducing light availability to the understory (Hutchinson and others, 2014; Keyser and others, 2017; Lafon and others, 2017; USNVC, 2025). Canopy cover is variable, with xeric ridgetops remaining more open and more mesic slopes supporting denser canopies of up to 60% cover (Braund and Zimmerman, 2022; Martin and others, 2011; McCarthy and Robison, 2003).

Prior to Euro-American settlement, frequent, low- to moderate-intensity surface fires, recurring every 4–15 years (Lafon and others, 2017; Ruffner, 2006), promoted open-canopy conditions and diverse understories of grasses and dry-mesic forbs (Arthur and others, 2021; Lafon and others, 2017; Sutherland and Hutchinson, 2003b; USNVC, 2025). In the absence of fire, woody thickets of species such as blackberries (*Rubus* spp.) may increase and shade-tolerant saplings may gradually establish, leading to closed-canopy conditions (Albrecht and McCarthy, 2006; Brose, 2017; Dickinson, 2006; Hutchinson and others, 2014). As a result, many formerly open sites have transitioned into denser forests with reduced understory diversity (Goins and others, 2013; Hanberry and others, 2020; Iverson and others, 2008a; Nowacki and Abrams, 2008; Sutherland and Hutchinson, 2003b).

Today, this habitat is threatened by continued fire suppression, habitat loss and land-use conversion, white-tailed deer browsing, and invasive species such as autumn olive (*Elaeagnus umbellata*), common buckthorn (*Rhamnus cathartica*), invasive honeysuckles (*Lonicera* spp.), Japanese barberry (*Berberis thunbergii*), kudzu (*Pueraria montana*), and Oriental bittersweet (*Celastrus orbiculatus*) (Huebner, 2006; Rebbeck, 2012).

Table 32. Plant species common to the Central Appalachian-Northeast Oak Forest and Woodland habitat. Taxonomic authority, common name, growth habit, and native status are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit
Acer rubrum L.	Red maple	Tree
Acer saccharum Marshall	Sugar maple	Shrub, tree
Carya ovata (Mill.) K. Koch	Shagbark hickory	Tree
Carya spp. Nutt.	Hickory	Tree
Cornus florida L.	Flowering dogwood	Shrub, tree
Pinus rigida Mill.	Pitch pine	Tree
Quercus alba L.	White oak	Tree
Quercus coccinea Münchh.	Scarlet oak	Tree
Quercus montana Willd.	Chestnut oak	Tree
Quercus rubra L.	Northern red oak	Tree

Species	Common name	Growth habit
Quercus velutina Lam.	Black oak	Tree
Andropogon gerardii Vitman	Big bluestem	Graminoid
Danthonia spicata (L.) P. Beauv. ex Roem. & Schult.	Poverty oatgrass	Graminoid
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid
Sorghastrum nutans (L.) Nash	Indiangrass	Graminoid

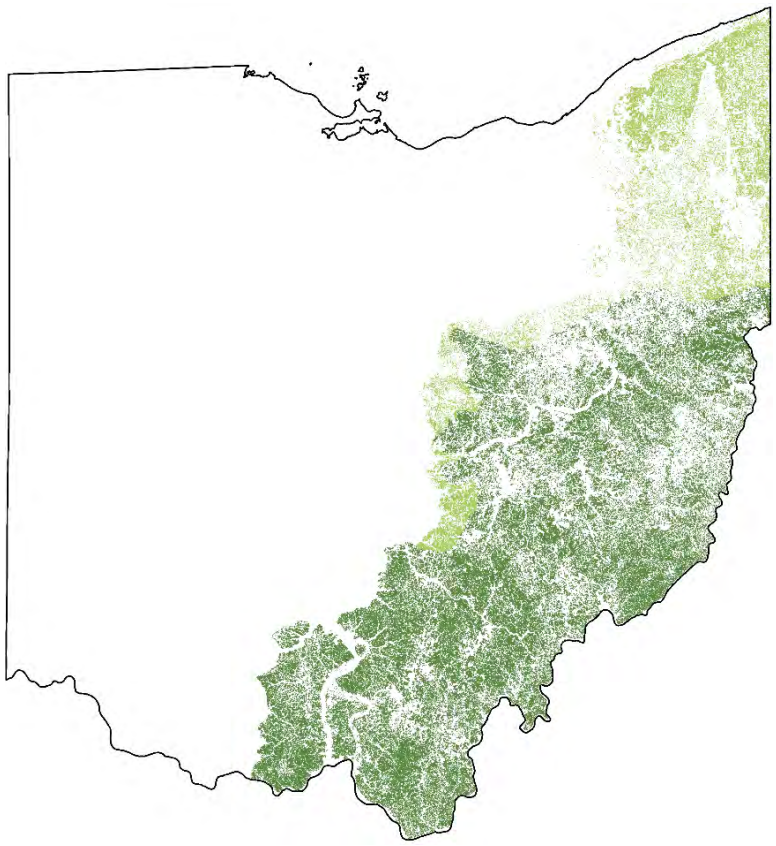

Figure 17. Central Appalachian-Northeast Oak Forest and Woodland habitat distribution. Geographic distribution of the Central Appalachian-Northeast Oak Forest and Woodland habitat (Group Code: G650) in the Midwest (USNVC, 2020). Dark green areas indicate Central Appalachian-Northeast Oak Forest and Woodland habitat in the Appalachian Forest ecoregion, while light green areas mark additional locations throughout the Midwest. Habitat boundaries have been enlarged to enhance visibility and do not reflect true proportions.

Overview of Climate Projections

The Central Appalachian-Northeast Oak Forest and Woodland habitat is prioritized for the Appalachian Forest ecoregion (Table i), and climate projections for this area largely reflect regional trends. 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 33) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). Most models also project an increase in annual precipitation, averaging 3.3 additional inches per year, although some indicate a slight decline (Table 34) (Marvel and others, 2023). However, precipitation is

expected to become more variable, with more extreme events and notably drier summers despite the overall increase (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 33. 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 34. 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 fire. Across the Appalachian Forest ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 72% above historical conditions by the end of the century (RCP 8.5; Table 35), with the highest values expected in southern Ohio (Figure 16). Higher VPD means more rapid evaporation and water loss from plants and soils, which can intensify drought conditions. Annual extreme fire danger is also projected to increase by approximately 83% by mid-century under the high emissions scenario, adding more than 9 days of elevated risk (Table 36). As with VPD, the greatest risk is projected in the southwestern part of the state, where the maximum model indicates up to 60 days of extreme fire risk annually (Figure 17). Rising fire risk could increase the frequency of wildfire and push this habitat toward more historically frequent fire intervals, but it also raises the likelihood of extreme events, including crown fires, that exceed historical norms and constrain opportunities to safely apply prescribed fire.

Table 35. Projected changes in summer vapor pressure deficit 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 summer vapor pressure deficit (VPD; kilopascals) 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 VPD 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	0.96 kPa	0.90 kPa (-6%)	0.97 kPa (1%)	0.94 kPa (-2%)	1.00 kPa (4%)
Mean of models	0.97 kPa	1.22 kPa (25%)	1.37 kPa (41%)	1.29 kPa (33%)	1.67 kPa (72%)
Maximum model	0.97 kPa	1.50 kPa (55%)	1.75 kPa (81%)	1.84 kPa (89%)	2.92 kPa (201%)

Table 36. Projected changes in annual extreme fire danger 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 extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) by mid-century (2040–2069), 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 annual extreme fire danger under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069	
	Historical	RCP 4.5	RCP 8.5
Minimum model	11.0	4.1 (-63%)	5.8 (-47%)
Mean of models	11.0	15.0 (37%)	20.1 (83%)
Maximum model	11.0	27.9 (155%)	51.5 (370%)

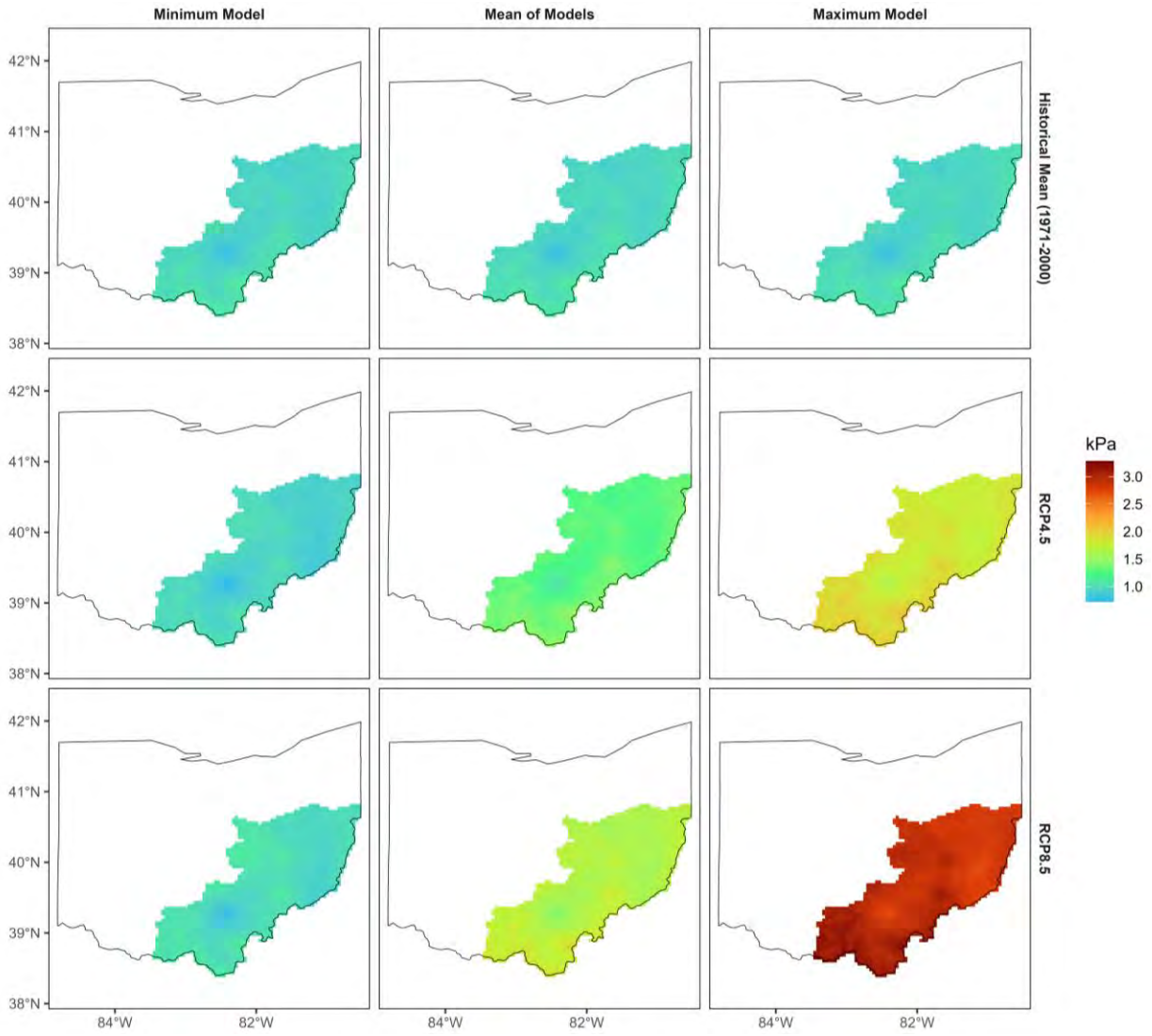

Figure 18. Projected changes in summer vapor pressure deficit in the Appalachian Forest ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA 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) summer vapor pressure deficit (VPD; kilopascals) 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 VPD under RCP 8.5 are presented in addition to the mean of all models (middle column).

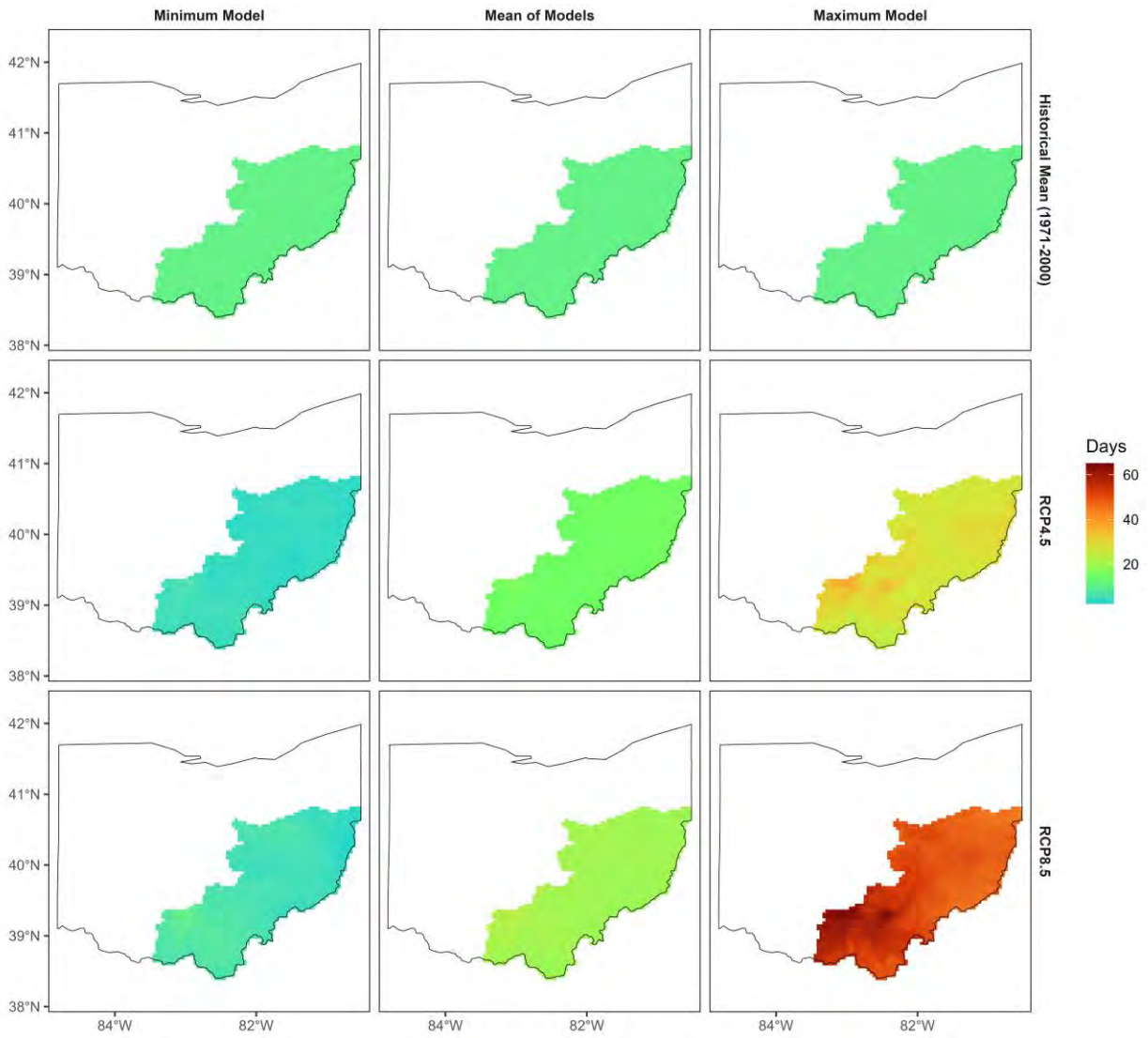

Figure 19. Projected changes in annual extreme fire danger in the Appalachian Forest ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for 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 (2040–2069) annual extreme fire danger (annual number of days when 100-hour fuel moisture is below the 3rd percentile of historical years) 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 extreme fire danger under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

Climate change is affecting the Central Appalachian-Northeast Oak Forest and Woodland habitat through its overall influence on species distributions, adaptive capacity, and vulnerability, as well as through the specific impacts of drought and fire (Table 37). 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:

- Drought can intensify soil drying, reduce herbaceous cover, and increase erosion risk, particularly when followed by heavy rainfall on xeric ridgetops and slopes (Breshears and others, 2016; Butler and others, 2015; Ebel and others, 2022; Homyak and others, 2017; Leizeaga and others, 2021; Ziegler and others, 2008).
- Shallow, acidic soils with limited water-holding capacity are prone to interannual variation in moisture, which may suppress soil microbial activity and decomposition under drought, reducing nutrient availability and favoring drought-tolerant microbial taxa (Baldrian and others, 2023; Lemoine and others, 2024; Reinsch and others, 2024).
- Low-intensity fire increases understory light and nutrient availability, while high-severity or repeated fire can degrade soil structure and deplete nutrients, particularly on thin, ridgetop soils (Hanberry and others, 2017; Knicker, 2007; Singh and others, 2025).

Community composition:

- Drought may shift communities toward xeric-adapted taxa, particularly drought-tolerant warm-season (C_4) grasses such as big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and Indiangrass (*Sorghastrum nutans*), though extreme drought may reduce their dominance in favor of opportunistic forbs and cool-season (C_3) grasses (Batllori and others, 2020; Havrilla and others, 2023; Knapp and others, 2024; Volder and others, 2010).
- Mesophytic species like sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), and birches (*Betula* spp.) are vulnerable to recruitment failure and decline under drought, while oaks (e.g., *Quercus alba*, *Q. velutina*, *Q. rubra*, *Q. coccinea*, *Q. montana*) may expand as mesic species decline (Abrams and Nowacki, 2016; Boakye and others, 2023; Dey and others, 2019; Frelich and others, 2017; Rauschendorfer and others, 2022).
- Severe or repeated drought may negate the relative drought advantage of oaks, driving recruitment failure and canopy mortality in red oak (*Q. rubra*) and black oak (*Q. velutina*) (Asbjornsen and others, 2021; Druckenbrod and others, 2019; Novick and others, 2022).
- While historical fire suppression has driven “mesophication,” increasing the dominance of shade-tolerant maples (*Acer* spp.) and American beech (*Fagus grandifolia*), altering microclimates through vegetation removal, and dampening fire spread, the return of frequent fire may reduce mesophytic species and favor fire-adapted oaks and grasses (Nowacki and Abrams, 2008; Peterson and Reich, 2001).
- Drought and intensified fire regimes may catalyze transitions toward grassland, shrubland, or savanna, depending on disturbance history and site context (Breshears and others, 2016; Frelich and Reich, 2010; Smith and Goetz, 2021; Vose and others, 2016).

Invasive species:

- Drought can amplify invasion pressure by weakening native vegetation and driving post-drought nutrient pulses, favoring species such as Nepalese browntop (*Microstegium vimineum*), smooth brome (*Bromus inermis*), and kudzu (*Pueraria montana*) (Bradley and others, 2010; Duell and others, 2021; Zhang and others, 2023).

- Fire may suppress some invasive shrubs such as common buckthorn (*Rhamnus cathartica*) and invasive honeysuckles (*Lonicera* spp.), but it can also promote shrubs that vigorously resprout or persist through seed banks, requiring repeated disturbance for effective control (Culbert and others, 2025; Illinois Department of Natural Resources, 2017; Vander Yacht and others, 2017; Zouhar and others, 2008).
- Flammable invasive grasses can reinforce feedbacks that increase fire frequency and intensity, perpetuate their dominance in burned areas, and displace native understories (Fusco and others, 2019; Kerns and others, 2020), while woody invaders such as common buckthorn (*Rhamnus cathartica*) can dampen fire spread by shading herbaceous fuels (Culbert and others, 2025; Dibble and others, 2008; Fusco and others, 2019; Kerns and others, 2020; Mandle and others, 2011).

Pests and pathogens:

- Warming and drought may increase outbreak risk from red oak borer (*Enaphalodes rufulus*) and twolined chestnut borer (*Agrilus bilineatus*), both of which disproportionately impact red oak (*Quercus rubra*) and black oak (*Q. velutina*) (Fierke and others, 2007; Haavik and others, 2015).
- Drought may increase oak vulnerability to pathogens such as Hypoxylon canker (*Biscogniauxia atropunctata*), which causes rapid mortality under warm, dry conditions, and root rots (*Armillaria* spp.) and water molds (*Phytophthora* spp.), which thrive under fluctuating soil moisture (Wood and others, 2018; Kolb and others, 2016).
- Drought may inhibit some pathogens by disrupting life cycles (Delavaux and others, 2021).
- Fire may indirectly reduce pest and pathogen pressure by thinning host populations, but fire-damaged trees can also attract bark beetles and other opportunistic pests (Barkley, 2006; Canelles and others, 2021; Jenkins and others, 2014; Parker and others, 2006; Progar and others, 2014).

Herbivory:

- Warmer winters may increase white-tailed deer (*Odocoileus virginianus*) populations, intensifying browsing pressure on oak (*Quercus* spp.) seedlings and saplings (Champagne and others, 2021; Felton and others, 2024; Frelich and others, 2015; Hanberry and Abrams, 2018; Weiskopf and others, 2019).
- Drought may reduce plant investment in chemical defenses, increasing palatability and making drought-stressed seedlings more vulnerable to herbivory (Cole and others, 2021; Duell and others, 2024; Gutbrodt and others, 2011).
- Browsing in drought-stressed or severely burned habitats may reduce forb diversity and favor weeds and grasses (Anderson and others, 2017, p. 202; Batzli and Dejaco, 2013; Urbanek and others, 2012).

Table 37. Summary of impacts from key climate change stressors on Central Appalachian-Northeast Oak Forest and Woodland habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory impacts (e.g., showing both high and low tolerance to drought), reflecting conflicting findings in the literature.

Stressor	Impact	Affected species
Drought		*Andropogon gerardii*[1–4]

Stressor	Impact	Affected species
	When tolerance of droughts is high, competitive advantage and/or increased abundance	*Carya ovata*[5]
		Danthonia spicata[6]
		Pinus rigida[7,8]
		Quercus alba[8–17]
		Quercus coccinea[8,18,19]
		Quercus rubra[10,13,17,18,20, 21]
		Quercus velutina[17,18,22]
		**Rhamnus cathartica*[23]
		Schizachyrium scoparium[4,24–27]
	When tolerance of droughts is low, reduced growth and abundance	*Acer rubrum*[4,8,13,20,28–33]
		Acer saccharum[8,12,13,23,31–41]
		Andropogon gerardii[24,27,42–46]
		**Berberis thunbergii*[4]
		Carya ovata[38]
		Cornus florida[38,47,48,49]
		**Lonicera maackii*[4]
		Pinus rigida[38]
		Quercus alba[5,38,50,51]
		Quercus coccinea[38]
		Quercus rubra[14,31,33,37,38,41,52]
		Quercus velutina[4,14,53]
		Schizachyrium scoparium[54]
		Sorghastrum nutans[24,55–58]
Fire	When tolerance of fire is high, competitive advantage and/or increased abundance	*Andropogon gerardii*[4]
		Carya ovata[4,59,60]
		Carya spp.[61,62]
		Cornus florida[48]
		Danthonia spicata[6]
		**Lonicera maackii*[4]
		**Lonicera tatarica*[4]
		Pinus rigida[7,60,61,62]
		Quercus alba[11,14,59–64]
		Quercus coccinea[4,60–62]
		Quercus rubra[4,14,47,60–62]
		Quercus velutina[59,60]
		Schizachyrium scoparium[4,25]
		Sorghastrum nutans[4,58]
	When tolerance of droughts is low, reduced growth and abundance	*Acer rubrum*[4,59–62,65]

Stressor	Impact	Affected species
	When tolerance of fire is low, increased mortality and/or reduced abundance	*Acer saccharum*[59–62]
		**Ailanthus altissima*[4]
		**Berberis thunbergii*[4]
		Carya ovata[66]
		Cornus florida[4,49,60]
		Pinus rigida[4]
		Quercus coccinea[67]
		Quercus rubra[21,59]
		Quercus velutina[4,68]

[1]Baty and others (2023); [2]Nippert and others (2009); [3]Slette and others (2023); [4]USDA, NRCS (2025); [5]Au and Maxwell (2022); [6]Covington (2000); [7]Gucker (2007); [8]Niinements and Vallardes (2006); [9]Duan and others (2022); [10]Gustafson and Sturtevant (2013); [11]Hayford and Chin (2020); [12]Maxwell (2024); [13]McEwan and others (2011); [14]Novick and others (2022); [15]Clark and others (2016); [16]Rogers (1990); [17]Brzostek and others (2014); [18]Rauschendorfer and others (2022); [19]Carey (1992b); [20]Walters and others (2023); [21]Sander (1990b); [22]Carey (1992f); [23]Fisichelli and others (2014); [24]Lemoine and others (2017); [25]Steinberg (2002); [26]USDA, NRCS (2004); [27]Dekirmenjian and others (2024); [28]Ibanez and others (2017); [29]Palus and others (2018); [30]Ramage and others (2023); [31]Reich and others (2022); [32]Swanston and others (2018); [33]Wang and others (2023); [34]Au and others (2020); [35]Bapikee (2013); [36]Boakye and others (2023); [37]Lee and Ibanez (2021); [38]Peters and others (2015); [39]Randall and Walters (2019); [40]Rudolph and LeBlanc (2020); [41]Stangler and others (2016); [42]Kooyers and others (2025); [43]Maricle and others (2017); [44]Pehim Limbu and Avolio (2025); [45]Uchytil (1988); [46]Watson and others (2021); [47](2020); [48]Tirmenstein (1991b); [49]McLemore(1990); [50]Kellner and Swihart (2016); [51]LeBlanc and Berland (2019); [52]Lockwood and others (2023); [53]Asbjornsen and others (2021); [54]Duell and others (2016); [55]Hoover and others (2014); [56]Kazemi and others (2023); [57]Swemmer and others (2006); [58]Walkup (1991b); [59]Hoving and others (2021); [60]Nowacki and Abrams (2015b); [61]Arthur and others (2021); [62]Varner (2018); [63]Hanberry and others (2012); [64]Tirmenstein (1991c); [65]Walters and Yawney (1990) (26); [66]Tirmenstein (1991a); [67]Carey (1992b); [68]Sander (1990a)

Focal Wildlife Species: Wood Thrush

The wood thrush (*Hylocichla mustelina*) is a migratory songbird that winters in Mexico and Central America, and breeds across the eastern United States and southern Canada (Audubon Society, n.d.). Populations have rapidly declined due to loss and fragmentation of their breeding habitat, moist deciduous forest (Fynn, 2022), as well as brood parasitism by brown-headed cowbirds (*Molothrus ater*) (Vernasco and others, 2018).

Climate change is expected to further affect wood thrush populations. Extreme rainfall and heat events reduce nest survival (Schöll and Hille, 2020; Taff and Shipley, 2023), while drought decreases post-fledging survival, effects that can be more detrimental than brood parasitism (Vernasco and others, 2018). These responses are likely mediated by resource availability, as drought decreases prey abundance (Vernasco and others, 2018). Protecting mature forest will be particularly important for maintaining breeding habitat, since mature forest cover can buffer drought effects (Vernasco and others, 2018). Conversely, increases in spring precipitation correspond with higher wood thrush productivity (McDermott and DeGroote, 2016), so years with wet springs may increase wood thrush numbers.

As a migratory species, the wood thrush is also sensitive to changing climate conditions across its range, with warming temperatures likely to shift the timing of migration and breeding. Currently, northern

populations migrate earlier than southern ones, which stay in breeding areas until early October (Ingenloff and Peterson, 2021). If wood thrushes respond to temperature cues, they may shift their migration timing to take advantage of longer periods of prey productivity. While warmer springs could lead to earlier breeding, hotter summers may shorten breeding season length (McDermott and DeGroote, 2016).

Overall, breeding suitability for wood thrush is likely to decline in the Midwest. By 5.4°F (3°C) of warming, much of the breeding area in the southern and central Midwest will be lost (Audubon Society, n.d.; Matthews and others, 2011; Matthews and others, 2014), reflecting both direct climate impacts and reduced persistence of moist deciduous forest. Climate change is also expected to lead to increasingly frequent droughts and decreased rainfall in overwintering sites, resulting in poorer body condition, survival, migration ability, and reproductive success in the Midwest (McKinnon and others, 2015).

Focal Aquatic System: Captina Creek

Captina Creek, a 56-kilometer stream in southeastern Ohio, flows east into the upper Ohio River at Powhatan Point (Upper Ohio Wheeling Watershed). Nearly its entire length is classified as Exceptional Warmwater Habitat and has been recognized by the U.S. Environmental Protection Agency as an Aquatic Resource of National Importance due to its exceptional water quality and biodiversity (Belmont Soil and Water Conservation District and Captina Creek Watershed Stakeholders, 2014). The creek supports one of Ohio's most significant populations of eastern hellbender (*Cryptobranchus alleganiensis alleganiensis*) and is one of only two watersheds in the state where successful reproduction has been documented (Belmont Soil and Water Conservation District and Captina Creek Watershed Stakeholders, 2014). Despite its location in one of Ohio's most heavily mined regions, Captina Creek maintains a fish community that includes pollution-sensitive species (Belmont Soil and Water Conservation District and Captina Creek Watershed Stakeholders, 2014; State of Ohio Environmental Protection Agency, 2010), a resilience attributed in part to limestone bedrock that buffers the pH impacts of acid mine drainage (Belmont Soil and Water Conservation District and Captina Creek Watershed Stakeholders, 2014). The 588-square-kilometer watershed lies within the unglaciated Western Allegheny Plateau ecoregion and is predominantly forested (64.6%), with pasture comprising the next largest land cover type (23%) (Belmont Soil and Water Conservation District and Captina Creek Watershed Stakeholders, 2014).

Climate change is projected to alter hydrologic patterns and thermal regimes across the Ohio River Basin, with direct implications for the ecological stability of streams like Captina Creek. Mid-century estimates (2045–2065) suggest a mean annual temperature increase of 1.9–3.1°C, along with greater winter and spring precipitation, more frequent high-intensity storms, and extended summer dry periods (United States Army Corps of Engineers Institute for Water Resources (U.S.), 2017). These shifts are expected to intensify flash flooding, reduce summer baseflows, and increase thermal stress (Table 38), conditions that could undermine the creek's biological integrity. Species dependent on stable flows, clean substrate, and cool, well-shaded microhabitats, such as freshwater mussels and the eastern hellbender, are particularly at risk. The Ohio River–Susquehanna River Adaptive Capacity Unit, which includes Captina Creek, is among the most vulnerable regions for hellbenders, due to a combination of high climate exposure, low landscape protection, and limited adaptive capacity (Sutton and others, 2023). Maintaining forest cover, buffering headwater streams, and improving watershed-scale connectivity will be critical for sustaining habitat suitability as environmental stressors intensify (Sutton and others, 2023).

Table 38. Projected changes in streamflow for Captina Creek. Projections use gridded simulations of daily runoff based on CMIP5 climate models under the high-emissions (RCP 8.5) scenario. Values show average historical (1977–2006) 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 (Authors (in review); U.S. Department of the Interior, Bureau of Reclamation, 2014; USDA Forest Service Office of Sustainability and Climate, 2022; Wenger and others, 2010).

Streamflow metric	1977–2006	2070–2099	Percent change
Mean Winter (December–February) Flow	170.7 ft^3/s	176.8 ft^3/s	4%
Mean Spring (March–May) Flow	193.7 ft^3/s	200.9 ft^3/s	4%
Mean Summer (June–August) Flow	67.4 ft^3/s	64.7 ft^3/s	-4%
Mean Fall (September–November) Flow	61.6 ft^3/s	60.2 ft^3/s	-2%
Mean Annual Flow	123.3 ft^3/s	125.6 ft^3/s	2%
1.5-year Flood (33rd percentile)	560.2 ft^3/s	593.1 ft^3/s	6%
10-year Flood (90th percentile)	1202.6 ft^3/s	1493.6 ft^3/s	24%
25-year Flood (96th percentile)	1674.4 ft^3/s	1852.4 ft^3/s	11%
Number of Winter Floods	12.4	11.8	-5%
Average Lowest 7-day Flow	31.6 ft^3/s	29.8 ft^3/s	-6%
Center of Flow Mass Timing	March 21	March 20	0%

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Appendix 1

Table iii. Woody plant species common to the Central Interior Oak Forest, Woodland and Savanna ecosystem. Key species were identified using USNVC floristic descriptions (USNVC, 2020), filtered by associations occurring within the Midwest that collectively comprise the Central Interior Oak Forest, Woodland and Savanna ecosystem. Daggers (†) denote some species within the genus may be introduced to the region. Taxonomic authority, common name, growth habit, and native status are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit	Interior Low Plateau Oak Forest, Woodland and Savanna	Oak-Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna	Central Midwest Oak Openings and Barrens	North-Central Oak- Hickory Forest and Woodland	Central Appalachian-Northeast Oak Forest and Woodland
Acer rubrum L.	Red maple	Tree			×		×
Acer saccharum Marshall	Sugar maple	Shrub, tree				×	×
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roem.	Saskatoon serviceberry	Shrub, tree				×	
Betula alleghaniensis Britton	Yellow birch	Tree				×	
Betula papyrifera Marshall	Paper birch	Tree				×	
Carya cordiformis (Wangenh.) K. Koch	Bitternut hickory	Tree				×	
Carya glabra (Mill.) Sweet	Pignut hickory	Tree				×	
Carya ovalis (Wangenh.) Sarg.	Red hickory	Tree				×	
Carya ovata (Mill.) K. Koch	Shagbark hickory	Tree			×	×	×
Carya spp. Nutt.	Hickory	Tree				×	×
Carya texana Buckley	Black hickory	Tree		×			
Carya tomentosa (Lam.) Nutt.	Mockernut hickory	Tree	×	×		×	
Cornus drummondii C.A. Mey.	Roughleaf dogwood	Shrub, tree				×	

Species	Common name	Growth habit	Interior Low Plateau Oak Forest, Woodland and Savanna	Oak-Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna	Central Midwest Oak Openings and Barrens	North-Central Oak- Hickory Forest and Woodland	Central Appalachian-Northeast Oak Forest and Woodland
Cornus florida L.	Flowering dogwood	Shrub, tree					×
Corylus cornuta Marshall	Beaked hazelnut	Shrub, tree				×	
†*Crataegus* spp. L.	Hawthorn	Shrub, tree		×			
Juniperus virginiana L.	Eastern redcedar	Tree		×			
Ostrya virginiana (Mill.) K. Koch	Eastern hophornbeam	Shrub, tree				×	
Pinus echinata Mill.	Shortleaf pine	Tree		×			
Pinus rigida Mill.	Pitch pine	Tree					×
Pinus strobus L.	Eastern white pine	Tree				×	
†*Populus* spp. L.	Cottonwood	Tree				×	
Populus tremuloides Michx.	Quaking aspen	Tree			×	×	
Quercus alba L.	White oak	Tree	×	×	×	×	×
Quercus bicolor Willd.	Swamp white oak	Tree			×	×	
Quercus coccinea Münchh.	Scarlet oak	Tree	×	×			×
Quercus ellipsoidalis E.J. Hill	Northern pin oak	Tree			×	×	
Quercus falcata Michx.	Southern red oak	Tree	×				
Quercus macrocarpa Michx.	Bur oak	Shrub, tree			×	×	
Quercus marilandica Münchh.	Blackjack oak	Shrub, tree		×			
Quercus montana Willd.	Chestnut oak	Tree	×				×
Quercus muehlenbergii Engelm.	Chinkapin oak	Tree				×	
Quercus palustris Münchh.	Pin oak	Tree			×	×	

Species	Common name	Growth habit	Interior Low Plateau Oak Forest, Woodland and Savanna	Oak-Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna	Central Midwest Oak Openings and Barrens	North-Central Oak- Hickory Forest and Woodland	Central Appalachian-Northeast Oak Forest and Woodland
Quercus rubra L.	Northern red oak	Tree	×	×		×	×
Quercus spp. L.	Oak	Tree			×		
Quercus stellata Wangenh.	Post oak	Tree	×	×		×	
Quercus velutina Lam.	Black oak	Tree	×	×	×	×	×
Tilia americana L.	American basswood	Tree				×	
Tsuga canadensis (L.) Carrière	Eastern hemlock	Tree				×	
Vaccinium pallidum Aiton	Blue Ridge blueberry	Shrub		×			

Table iv. Non-woody plant species common to the Central Interior Oak Forest, Woodland and Savanna ecosystem. Key species were identified using USNVC floristic descriptions (USNVC, 2020), filtered by associations occurring within the Midwest that collectively comprise the Central Interior Oak Forest, Woodland and Savanna ecosystem. Asterisks (*) denote species introduced to the region. Daggers (†) denote some species within the genus may be introduced to the region. Taxonomic authority, common name, growth habit, and native status are determined according to the PLANTS Database (USDA, NRCS, 2025).

Species	Common name	Growth habit	Interior Low Plateau Oak Forest, Woodland and Savanna	Oak-Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna	Central Midwest Oak Openings and Barrens	North-Central Oak- Hickory Forest and Woodland	Central Appalachian-Northeast Oak Forest and Woodland
Ambrosia psilostachya DC.	Cuman ragweed	Forb/herb			×		
Amphicarpaea bracteata (L.) Fernald	American hogpeanut	Forb/herb, vine		×	×	×	
Andropogon gerardii Vitman	Big bluestem	Graminoid			×	×	×
Apios americana Medik.	Groundnut	Forb/herb, vine				×	
Artemisia ludoviciana Nutt.	White sagebrush	Forb/herb			×		
Bromus kalmii A. Gray	Arctic brome	Graminoid				×	
Chasmanthium latifolium (Michx.) Yates	Indian woodoats	Graminoid				×	
Calamagrostis canadensis (Michx.) P. Beauv.	Bluejoint	Graminoid			×	×	
Calamovilfa longifolia (Hook.) Scribn.	Prairie sandreed	Graminoid			×		
Carex pensylvanica Lam.	Pennsylvania sedge	Graminoid			×	×	
Carex picta Steud.	Boott's sedge	Graminoid	×				
†*Carex* spp. L.	Sedge	Graminoid			×		
Comandra umbellata (L.) Nutt.	Bastard toadflax	Forb/herb			×		

Species	Common name	Growth habit	Interior Low Plateau Oak Forest, Woodland and Savanna	Oak-Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna	Central Midwest Oak Openings and Barrens	North-Central Oak- Hickory Forest and Woodland	Central Appalachian-Northeast Oak Forest and Woodland
Danthonia spicata (L.) P. Beauv. ex Roem. & Schult.	Poverty oatgrass	Graminoid					×
Desmodium glutinosum (Muhl. ex Willd.) Alph. Wood	Pointedleaf ticktrefoil	Forb/herb		×		×	
Desmodium nudiflorum (L.) DC.	Nakedflower tick-trefoil	Forb/herb		×			
Dichanthelium boscii (Poir.) Gould & C.A. Clark	Bosc's panicgrass	Graminoid		×		×	
Dichanthelium oligosanthes (Schult.) Gould	Heller's rosette grass	Graminoid				×	
Elymus hystrix L.	Eastern bottlebrush grass	Graminoid				×	
Elymus virginicus L.	Virginia wildrye	Graminoid				×	
Festuca subverticillata (Pers.) Alexeev	Nodding fescue	Graminoid				×	
Galium boreale L.	Northern bedstraw	Forb/herb				×	
Geranium maculatum L.	Wild geranium	Forb/herb				×	
Hesperostipa spartea (Trin.) Barkworth	Porcupinegrass	Graminoid			×		
Koeleria macrantha (Ledeb.) Schult.	Prairie Junegrass	Graminoid			×		
**Lonicera japonica* Thunb.	Japanese honeysuckle	Vine	×				
Monarda bradburiana Beck	Eastern beebalm	Forb/herb		×			
Panicum virgatum L.	Switchgrass	Graminoid			×		

Species	Common name	Growth habit	Interior Low Plateau Oak Forest, Woodland and Savanna	Oak-Shortleaf Pine Ozark-Ouachita Forest, Woodland and Savanna	Central Midwest Oak Openings and Barrens	North-Central Oak- Hickory Forest and Woodland	Central Appalachian-Northeast Oak Forest and Woodland
Parthenocissus quinquefolia (L.) Planch.	Virginia creeper	Vine		×	×		
**Poa pratensis* L.	Kentucky bluegrass	Graminoid				×	
Polystichum acrostichoides (Michx.) Schott	Christmas fern	Forb/herb	×				
Pteridium aquilinum (L.) Kuhn	Bracken fern	Forb/herb				×	
Rudbeckia hirta L.	Blackeyed Susan	Forb/herb			×		
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid		×	×		×
Sorghastrum nutans (L.) Nash	Indiangrass	Graminoid			×		×
Sporobolus heterolepis (A. Gray) A. Gray	Prairie dropseed	Graminoid			×		
Toxicodendron radicans (L.) Kuntze	Eastern poison ivy	Forb/herb, vine		×			

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 the RCP8.5 end of the century (2070–2099) scenario across each region.

Ecoregion	Climate Variable	Season	Minimum Model	Maximum Model
Midwest Region	Extreme Dry Months	Annual	MRI-CGCM3_r1i1p1	HadGEM2-CC365_r1i1p1
Interior Plateau	Extreme Fire Danger Days	Annual	GFDL-ESM2M	HadGEM2-ES365
Appalachian Forest	Extreme Fire Danger Days	Annual	MRI-CGCM3	HadGEM2-ES365
Great Lakes Plains	Extreme Fire Danger Days	Annual	MRI-CGCM3	HadGEM2-ES365
Midwest Region	Extreme Fire Danger Days	Annual	MRI-CGCM3	IPSL-CM5A-MR
Central Irregular Plains	Extreme Fire Danger Days	Annual	MRI-CGCM3	MIROC5
Driftless Area	Extreme Fire Danger Days	Annual	MRI-CGCM3	MIROC5
Ozarks	Extreme Fire Danger Days	Annual	bcc-csm1-1-m	HadGEM2-ES365
Appalachian Forest	Precipitation	Annual	IPSL-CM5A-MR	CSIRO-Mk3-6-0
Driftless Area	Precipitation	Annual	IPSL-CM5A-MR	CSIRO-Mk3-6-0
Great Lakes Plains	Precipitation	Annual	IPSL-CM5A-MR	CSIRO-Mk3-6-0
Central Irregular Plains	Precipitation	Annual	IPSL-CM5A-MR	IPSL-CM5B-LR
Interior Plateau	Precipitation	Annual	IPSL-CM5A-MR	IPSL-CM5B-LR
Ozarks	Precipitation	Annual	IPSL-CM5A-MR	IPSL-CM5B-LR
Appalachian Forest	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Central Irregular Plains	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Driftless Area	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Great Lakes Plains	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Interior Plateau	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Ozarks	Mean Temperature	Annual	MRI-CGCM3	MIROC-ESM-CHEM
Appalachian Forest	Vapor Pressure Deficit	Summer	MRI-CGCM3	HadGEM2-CC365
Central Irregular Plains	Vapor Pressure Deficit	Summer	MRI-CGCM3	HadGEM2-CC365
Driftless Area	Vapor Pressure Deficit	Summer	MRI-CGCM3	HadGEM2-CC365
Great Lakes Plains	Vapor Pressure Deficit	Summer	MRI-CGCM3	HadGEM2-CC365
Interior Plateau	Vapor Pressure Deficit	Summer	MRI-CGCM3	HadGEM2-CC365
Ozarks	Vapor Pressure Deficit	Summer	MRI-CGCM3	HadGEM2-CC365