Effects of Climate Change on Midwestern Ecosystems: Central and Eastern North American Grassland and Shrubland

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Foreword

Series Overview

The following is the fifth 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 and Eastern North American Grassland and Shrubland ecosystem may be increasingly shaped by intensifying drought and shifting seasonality. Rising temperatures and more variable precipitation, marked by longer dry spells, are projected to increase evapotranspiration and soil moisture deficits, and yield more frequent drought. At the same time, warming temperatures are projected to advance spring onset and extend the growing season. Drought may alter habitat structure by accelerating soil erosion, disrupting nutrient cycling, increasing physiological stress on plants, and reducing productivity. These changes are expected to shift community composition toward species adapted to water limitation and fluctuating resources, reducing much of the herbaceous cover that characterizes this ecosystem. Seasonal shifts may restructure habitat by altering phenology and flowering dynamics, potentially increasing productivity but also heightening the risk of late-season frost damage. Community composition is expected to shift toward early-emerging species and species with phenological flexibility. Altered phenology may also lead to mismatches between plants and pollinators and increase pollinator competition at the beginning and end of the growing season, with potential consequences for reproduction.

Although these overarching stressors affect the entire ecosystem, their specific impacts likely vary with local habitat conditions. In the Central and Northern Tallgrass Prairie habitat, which are historically fire-maintained habitats dominated by a mix of warm-season (C_4) and cool-season (C_3) grasses and forbs, climate change may shift community composition by favoring deep-rooted forbs and established shrubs while displacing shallow-rooted species, including many native grasses. These changes, especially in the absence of fire, may promote woody encroachment and drive long-term community reassembly. In the Central Interior Acidic Open Glade and Barrens habitat, characterized by shallow, drought-prone soils, climate change may reinforce xeric assemblages and reduce the abundance of mesic species. In the absence of fire, shrubs rather than larger woody species, are more likely to increase, as water limitations constrain the establishment of trees. In the Eastern North American Ruderal Meadow and Shrubland habitat, which lack native species richness and structural stability, disturbance-tolerant invaders may increasingly dominate. Drought and earlier springs are expected to reinforce early successional dynamics and further constrain the restoration potential of these already degraded habitats.

Across the region, invasive species, herbivory, and microbial and fungal communities are also expected to respond to climate change. Invasive plants with ruderal traits and flexible phenologies are likely to benefit from drought-driven disturbance, post-drought resource pulses, and longer, earlier growing seasons. These species often germinate and flower earlier than natives, gaining priority access to resources as seasonal timing shifts. Browse pressure from increasing white-tailed deer (*Odocoileus virginianus*) populations is expected to intensify, particularly during drought, when plant defenses are weakened, and during extended growing seasons, which prolong forage availability. This selective browsing may contribute to declines in native forbs while indirectly promoting non-native grasses. Microbial and fungal communities, like plant communities, are likely vulnerable to both drought and shifting seasonality. Reduced soil moisture may suppress microbial activity and decomposition, while shifts in fungal community composition, particularly declines in arbuscular mycorrhizal fungi, may impair plant drought tolerance.

Adaptation strategies for the Central and Eastern North American Grassland and Shrubland may require managers to anticipate and respond to these changes through both resistance-based approaches, such as restoring fire regimes and reinforcing native species dominance, and acceptance of some potential

transitions, such as facilitating drought-tolerant and phenologically flexible species establishment and adjusting fire regimes to align with altered phenology.

Central and Eastern North American Grassland and Shrubland

Key Climate Change Stressors

Climate change is reshaping the regimes that sustain the Central and Eastern North American Grassland and Shrubland ecosystem that spans much of the Midwest. These temperate systems are fundamentally structured by moisture availability and seasonal timing, factors that may now be shifting beyond historical variability due to increased temperature, altered precipitation patterns, and earlier spring onset.

Drought

Drought is an influential and potentially intensifying stressor for the Central and Eastern North American Grassland and Shrubland ecosystem. Over the past millennium, episodic and locally confined droughts, particularly those of prolonged duration or extreme severity, have shaped grassland and shrubland ecology and function (Anderson, 2006; Anderson and Bowles, 1999; Belesky and Malinowski, 2016; Briggs and others, 2005; Changnon and others, 2002; Finch and others, 2016; Jones and others, 2016). Severe droughts occurred up to twice as often in the prairie region as in adjacent forested ecosystems, influencing species composition, ecosystem function, and ecosystem boundaries (Anderson, 2006; Changnon and others, 2002; Williams and others, 2009). Since the early 20th century, the Midwest has experienced increased precipitation and a decrease in the frequency of drought. More recently, however, a pattern of highly variable precipitation has emerged, characterized by rapid transitions between flood and drought conditions, and longer periods between precipitation events (Ford and others, 2021; Hoell and others, 2021; Marvel and others, 2023).

Climate projections indicate that drought frequency will likely increase in the Midwest (Figure 1), due to warming-induced evapotranspiration and variability in seasonal precipitation (Chen and Ford, 2023; Ford and others, 2021; Marvel and others, 2023; Zhao and others, 2020; Zhao and others, 2023). While some models suggest increases in annual precipitation, higher evaporative water losses are likely to offset these increases and contribute to more frequent and severe soil moisture deficits and drought (Gamelin and others, 2022; Iglesias and others, 2022; Zhao and others, 2020; Zhuang and others, 2024). Both chronic seasonal drought and flash drought (i.e., rapid onset and high evapotranspiration demand) are projected to become more common (Christian and others, 2021; Koster and others, 2019; Lesinger and Tian, 2022; Pendergrass and others, 2020). Grassland and shrubland ecosystems are highly sensitive to rainfall variability, as annual changes in growing season precipitation can lead to large shifts in soil moisture and productivity (Fay and others, 2011; Griffin-Nolan and others, 2021; Hsu and Adler, 2014; Liu and others, 2024b; Ross and others, 2012). Consequently, increasing drought frequency and intensity will likely be a dominant driver of future changes in grassland and shrubland extent, structure, and function (Breshears and others, 2016a; Felton and others, 2020; Knapp and others, 2024; Smith and others, 2024).

Seasonal Shifts

Changing seasonality, including earlier spring onset, more frost-free days, and longer growing seasons, is a key and accelerating feature of climate change across the Central and Eastern North American Grassland and Shrubland ecosystem. Since the early 20th century, winter warming in the Midwest has lengthened the frost-free season by approximately two weeks (Belesky and Malinowski, 2016; Walsh and others, 2014; Zhong and others, 2017). Climate projections indicate these warming trends will likely continue throughout the 21st century, further extending growing seasons and accelerating the onset of spring conditions (Figure 2) (Abendroth and others, 2019; Monahan and others, 2016; Rawlins and others, 2016; Wang and others, 2021). These changes initiate a cascade of responses related to plant and animal phenology, community productivity, and ecological function (Inouye, 2022; Petrie and others, 2012; Post and others, 2022; Schuerings and others, 2014). Coincident with seasonal shifts, the risk of false springs, when plants break dormancy early and are subsequently damaged by late-season frosts, will continue to challenge early-season plant performance (Allstadt and others, 2015; Lamichhane, 2021; Martinuzzi and others, 2019; Richardson and others, 2018). For temperate ecosystems, which are shaped by their temporal relationships to distinct seasonal periods, these shifting seasonal parameters may exert strong influences on community composition and function.

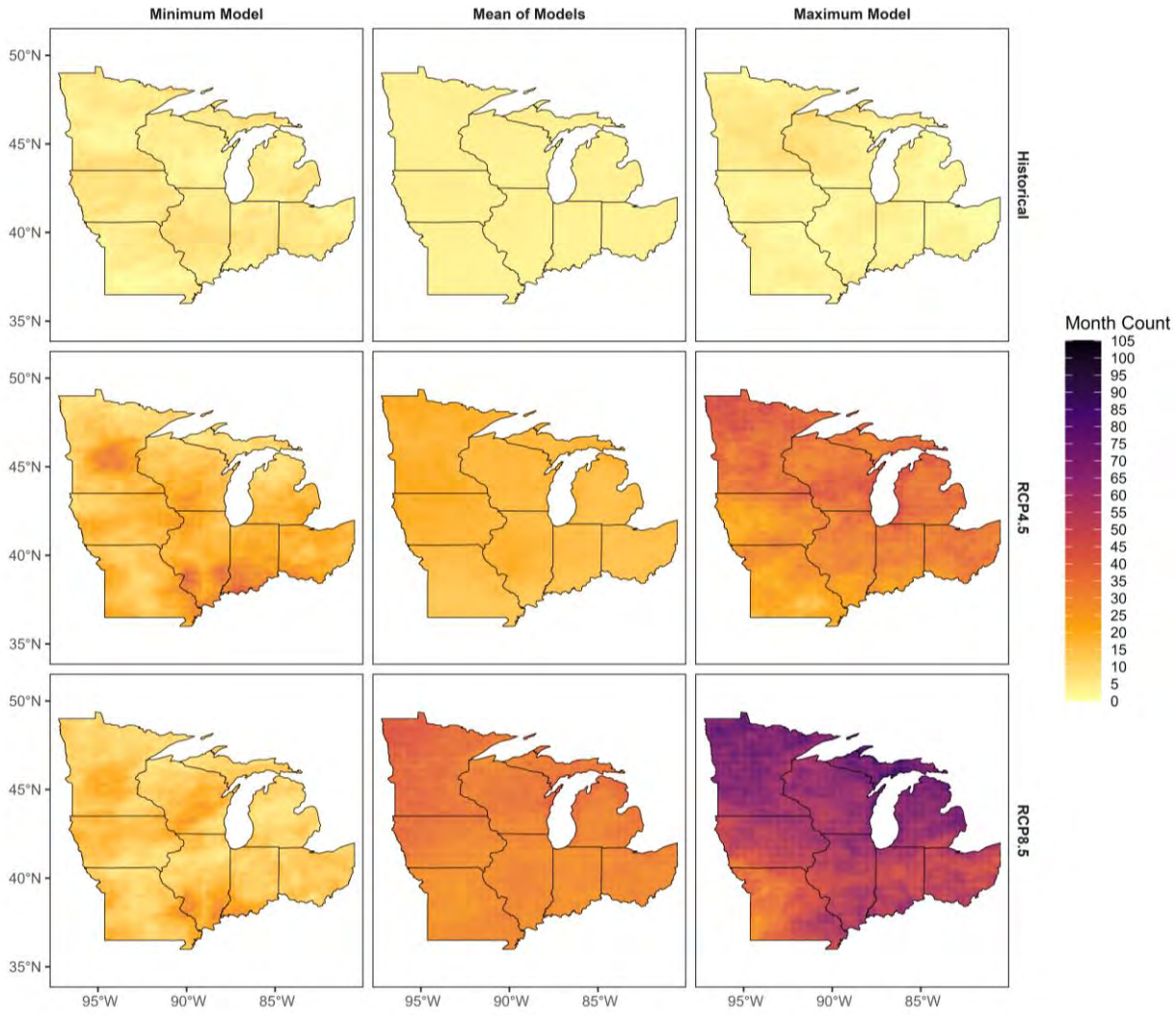

Figure 1. Projected changes in 30-year extreme drought incidence in the Midwest region. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions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

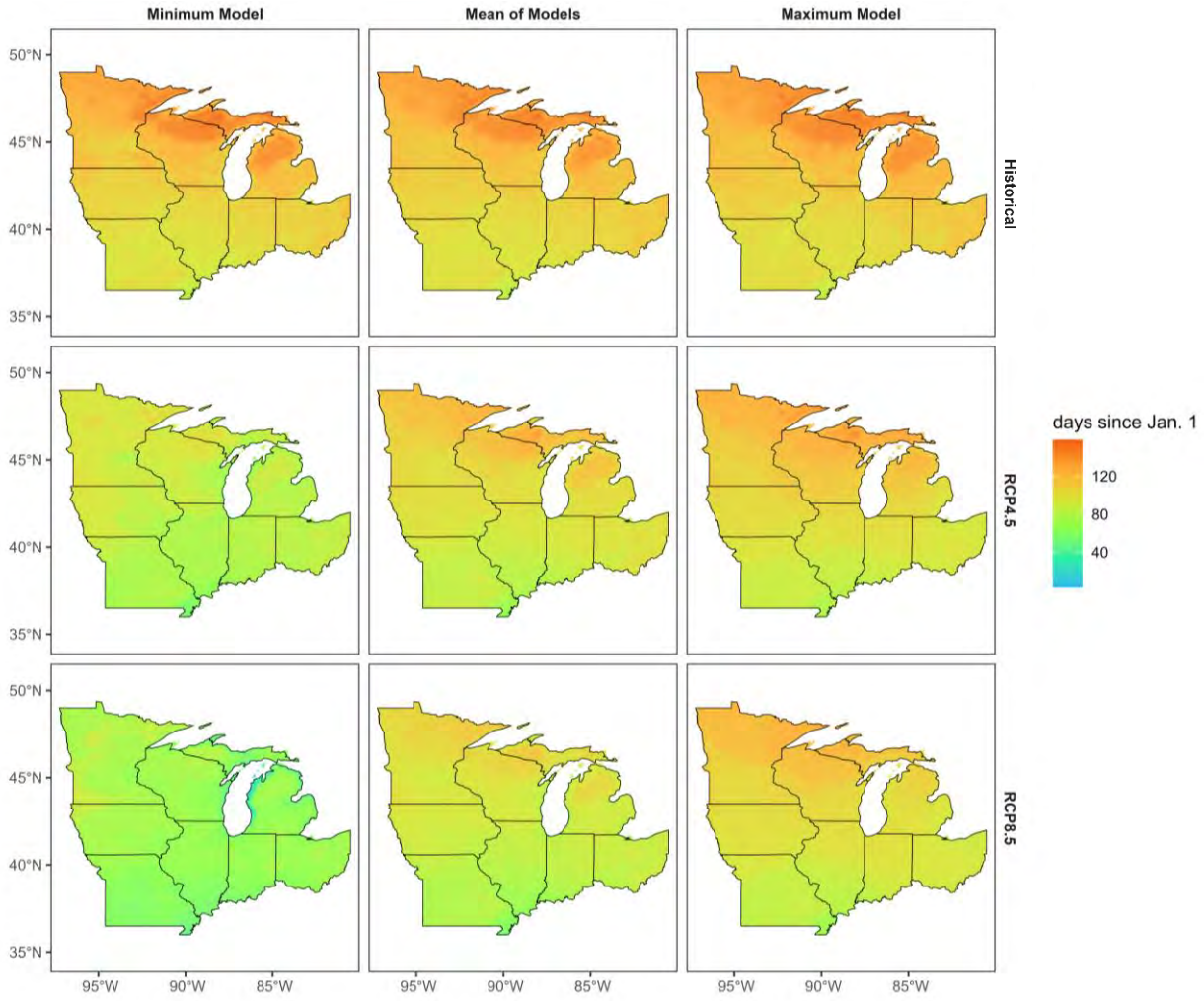

Figure 2. Projected last spring freeze day in the Midwest region. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean historical (1971–2000, top row) and projected (2070–2099) last spring freeze day (the last day after January 1 with a minimum temperature below freezing) 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 last spring freeze day 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 environmental conditions that underpin the structure and function of the Central and Eastern North American Grassland and Shrubland ecosystem. Increasing drought frequency and shifting seasonal cues are expected to drive major changes in habitat structure, community composition, and species interactions. These ecosystems are responsive to patterns of moisture availability and seasonal timing, and even modest disruptions to these drivers can have cascading effects.

Habitat Structure

Abiotic responses to drought. Increasingly intermittent precipitation and novel drought patterns are expected to intensify soil moisture variability between and within years in the Central and Eastern North American Grassland and Shrubland. Generally, warmer and drier conditions are expected to reduce forest extent and facilitate a northeastern expansion of prairie and savanna (Galatowitsch and others, 2009; Klemm and others, 2020; Lyon and others, 2019; Reich and others, 2022; Toot and others, 2020). During the growing season, these changes will limit moisture availability and water storage capacity, particularly in upper layers of the soil (Fay and others, 2011; Felton and others, 2020; Jonas and others, 2015; Petrie and others, 2012; Vose and others, 2016). Biotic responses to drought, such as declining herbaceous cover, can amplify its physical effects – enhancing wind erosion, increasing runoff, and limiting infiltration (Felton and others, 2020; Hanberry and others, 2019; Knapp and others, 2024; Vose and others, 2016; Yin and others, 2018). Due to reduced plant uptake, drought can also lead to nutrient accumulation in the soil, and subsequent onset of rainfall can rapidly release these accumulated nutrients (Austin and others, 2004; Burt and others, 2015; Castillioni and others, 2022b; Hoover and Rogers, 2016; Rosenblatt, 2018; Vose and others, 2016). Drought may also alter root structures and hydrologic flow paths, accelerating chemical weathering, especially in systems with woody encroachment (Sadayappan and others, 2023; Wen and others, 2021). Even short droughts can have prolonged impacts on soil carbon and nutrient dynamics (Hoover and Rogers, 2016).

Biotic responses to drought. By lowering soil moisture and elevating vapor pressure deficit (VPD), drought places physiological stress on Central and Eastern North American Grassland and Shrubland plants. High VPD can induce stomatal closure, limiting both water loss and carbon uptake, and ultimately constraining photosynthesis and growth (Pehim Limbu and Avolio, 2025; Vose and others, 2016; Wang and others, 2008). The severity of these impacts depends on both environmental context, such as historical levels of aridity and rainfall variability, and plant traits like photosynthetic pathway and rooting depth. While drier regions often support more drought-adapted species, water is a limiting resource and drought sensitivity tends to increase with aridity, although this trend can vary considerably between sites and species (Knapp and others, 2015; Knapp and others, 2024). This trend broadly suggests grasslands in the Midwest may be less vulnerable to drought than those in the Great Plains. Moreover, eastern areas of the Midwest may be less vulnerable than those closer to the western edge of the region.

Grassland plants employ various physiological strategies to survive droughts, including tolerance strategies, where traits allow plants to withstand drought stress (e.g., dehydration stress tolerance), and avoidance strategies, where traits prevent plants from experiencing drought stress (e.g., responsive root morphology) (Barkaoui and Volaire, 2023; Kazemi and others, 2023; Norton and others, 2016; Vose and others, 2016). For example, warm-season (C_4) grasses exhibit high water-use efficiency, allowing them to maintain photosynthesis and carbon fixation under low moisture conditions (drought tolerance). While

generally more drought-tolerant than cool-season (C_3) grasses, warm-season (C_4) grass advantages can diminish under severe drought (Hoover and others, 2019; Ripley and others, 2007; Taylor and others, 2011; Taylor and others, 2014; Volder and others, 2010). Rooting strategies also influence drought tolerance. During extended dry periods, deep-rooted forbs, some native grasses, and most shrubs can access subsurface moisture (drought avoidance) (Keen and others, 2024; Nippert and Knapp, 2007b; Padilla and others, 2019; Vose and others, 2016; Wilcox and others, 2023). In contrast, plants with predominantly shallow root structures, including many grasses, rely on rapid uptake from rainfall and are more vulnerable to drying (Asbjornsen and others, 2008; Nippert and Knapp, 2007a; Padilla and others, 2019). However, even drought-adapted species can deteriorate in prolonged or repeated drought, which can exceed their physiological tolerance and impair their future recovery (Slette and others, 2023; Smith and others, 2024; Zhou and others, 2022). As drought intensifies, the prevalence of different drought strategies in the landscape may shift in response to drought characteristics (Knapp and others, 2024). Further, plants may reduce leaf and root growth, enter dormancy, shift carbon allocation, or face mortality, particularly when droughts are coupled with extreme heat (Dreesen and others, 2015; Poirier and others, 2012; Rosenblatt, 2018; Vose and others, 2016; Wang and others, 2008; Weaver and Albertson, 1936; Will and others, 2013).

Drought negatively affects productivity and ecosystem function in the Central and Eastern North American Grassland and Shrubland ecosystem. As drought severity intensifies, aboveground net primary productivity (ANPP) decreases (Knapp and others, 2024; Smith and others, 2024). Under extreme drought, productivity losses can exceed 35–60%, especially in drier or species-poor sites (Isbell and others, 2015; Jones, 2019; Slette and others, 2023; Smith and others, 2024). Reduced productivity is often accompanied by declines in other ecosystem functions, including rainfall-use efficiency, herbaceous cover, and nutrient cycling (Fay and others, 2011; Felton and others, 2020; Knapp and others, 2024; Rosenblatt, 2018; Smith and others, 2024). Legacy effects from prior droughts, such as lower root biomass, slower growth, and reduced reproductive capacity, may further compound these impacts (Dudney and others, 2017; Hoover and others, 2014; Knapp and others, 2024; Luo and others, 2023; Sherry and others, 2008). However, interactions with other climate change stressors will influence and potentially dampen the negative impact of drought on ecosystem functioning. Elevated CO_2 and climate warming, for example, may offset some drought-driven productivity losses by enhancing water-use efficiency and maintaining biomass (Ahler and others, 2025; Hamerlynck and others, 1997; Havrilla and others, 2023; Jung and others, 2019; Lei and others, 2016; Pehim Limbu and Avolio, 2025; Reich and others, 2020; Xu and others, 2014).

Biotic responses to seasonal shifts. Across the Central and Eastern North American Grassland and Shrubland, early spring warming often accelerates green-up, extends flowering periods, and shifts the seasonal distribution of flowering times, particularly at the beginning and end of the growing season (Austin and others, 2024; Dunnell and Travers, 2011; Ramirez-Parada and others, 2022; Whittington and others, 2015; Zettlemoyer and others, 2021). Flowering in temperate prairies has advanced by 2–6 days per 1.8°F (1°C) of warming, with recent evidence of some species now flowering 8–16 days earlier (Geissler and others, 2023; Memmott and others, 2007; Prevéy and others, 2020). These advances are not uniform across taxa, as spring-flowering species typically respond more to warming than summer bloomers (Geissler and others, 2023; Jiang and others, 2024; Pearson, 2019; Sherry and others, 2007).

Phenological responses to seasonal shifts can have broad implications for ecosystem function. An extended growing season may advance reproductive activity and increase productivity, particularly when coupled with elevated CO_2 and wetter early-season conditions (Briske, 2017; Fridley and others, 2016; Hufkens and others, 2016; Li and others, 2025). However, these advances in growth and flowering often

occur during periods still prone to frosts (i.e., false springs), increasing the risk of freeze damage (Allstadt and others, 2015; Augspurger, 2009; Augspurger, 2013; Chamberlain and others, 2019; Martinuzzi and others, 2019; Zettlemoyer and others, 2021). This can impair growth, reproduction, and survival, particularly for species that respond quickly to seasonal shifts but lack frost tolerance during critical developmental phases (Allstadt and others, 2015; Chamberlain and others, 2019; Kral-O'Brien and others, 2019; Zettlemoyer and others, 2021).

Community Composition

Community responses to drought. Climate change is expected to alter community composition within the Central and Eastern North American Grassland and Shrubland ecosystem by reshaping the relative abundance and distribution of species. While this ecosystem is typically considered adapted to a broad range of climate conditions, drought can trigger not only short-term shifts in species dominance but also lasting reconfiguration of community structure (Craine and others, 2011; Eckhoff and others, 2023; Henry, 2019; Hoover and others, 2014; Ratajczak and Ladwig, 2019; Tucker and others, 2011). Species most capable of tolerating drought stress may become locally dominant (Eckhoff and others, 2023; Galatowitsch and others, 2009; Tucker and others, 2011). At the same time, species that rely on drought avoidance strategies, such as rapid life cycles, opportunistic growth, or heightened responsiveness to brief periods of moisture availability, may also increase in abundance (Castillioni and others, 2022a; Hoover and Rogers, 2016; Volder and others, 2010).

Fluctuating resource availability may undermine the stability of long-standing dominance patterns by creating new establishment opportunities and disrupting priority effects that typically reinforce dominance by early establishing species (Castillioni and others, 2020b; Goodale and Wilsey, 2018; Mount and others, 2024; Pehim Limbu and Avolio, 2025; Ratajczak and Ladwig, 2019). For example, mid-summer droughts can dramatically reduce flowering biomass for dominant grasses such as big bluestem (*Andropogon gerardii*), by up to 94% in some cases (Dietrich and Smith, 2016). As dominant species lose their competitive edge, less common or more opportunistic species may expand, leading to community reassembly and reordering of functional groups (Castillioni and others, 2020a; Mount and others, 2024; Pehim Limbu and Avolio, 2025). For example, Indiangrass (*Sorghastrum nutans*) may capitalize on heavy rainfall events but tends to lose its competitive advantage under prolonged drought (Smith and others, 2016; Swemmer and others, 2006). Big bluestem may maintain an advantage in moderately dry conditions, but sustained drought may ultimately favor more drought-hardy species such as little bluestem (*Schizachyrium scoparium*) or gramas (*Bouteloua* spp.) (Baty and others, 2015; Dekirmenjian and others, 2024; Griffin-Nolan and others, 2021; Kooyers and others, 2025; Smith and others, 2017; Watson and others, 2021). Compositional shifts may also occur among regional ecotypes of the same species. For example, drought tolerance varies between ecotypes of big bluestem, with western populations exhibiting greater resilience to water stress (Johnson and others, 2015; Smith and others, 2017). As drought intensifies, these more drought-tolerant ecotypes may be favored and expand eastward (Johnson and others, 2015; Smith and others, 2017).

Photosynthetic pathway and rooting depth are key traits influencing species' responses to drought. The higher water-use efficiency and temperature optima of warm-season (C_4) grasses suggest their dominance may be reinforced under rising temperatures and increasing aridity (Bardgett and Semchenko, 2019; Belesky and Malinowski, 2016; Catford and Jones, 2019; Forrestel and Edwards, 2019; Havrilla and others, 2023; Keen and others, 2024). However, in some cases, extreme drought may negatively impact C_4 species more than cool-season (C_3) species and reduce their abundance, particularly under elevated atmospheric CO_2 concentrations. C_3 species, including many introduced grasses, are

better adapted to exploit variable and intermittent rainfall, whereas C_4 species tend to rely on growing season precipitation (Ahler and others, 2025; DeKeyser and others, 2015; Griffin-Nolan and others, 2021; Havrilla and others, 2023; Knapp and others, 2024; O'Keefe and others, 2024; Polley and others, 2013; Smith and others, 2016; Taylor and others, 2011). Collectively, these advantages may result in greater abundances of C_3 grasses, particularly when severe droughts occur during the growing season. Where C_4 grasses decline, other functional groups may establish and dominate. Because they are capable of accessing subsurface water, deep-rooted shrubs and forbs are also often more resilient during prolonged drought and, as shallow-rooted grasses decline, may increase in relative abundance (Nippert and Knapp, 2007b; Smith and others, 2024; Volder and others, 2010; Vose and others, 2016; Wilcox and others, 2023).

Over the past century, many grasslands have experienced increasing woody plant cover, a process known as woody encroachment, driven by the loss of historical disturbance regimes (e.g., fire, grazing), rising atmospheric CO_2 and nitrogen deposition, and, in the Midwest, increased precipitation (Archer and others, 2017; Keen and others, 2024; Kulmatiski and Beard, 2013; Muench and others, 2016; Ratajczak and others, 2011; Twidwell and others, 2016; Vose and others, 2016). These changes facilitated woody plant establishment from seed. Now that many woody species are established, ongoing shifts in climate, particularly increased drought frequency and precipitation variability, may paradoxically reinforce their persistence (Archer and others, 2017; Breshears and others, 2016b; Ratajczak and others, 2011; Schuster and Dukes, 2017; Smith and Goetz, 2021; Volder and others, 2010). Deeply rooted shrubs can access subsurface moisture when surface soils dry, allowing them to maintain function during drought (Dodds and others, 2023; Sadayappan and others, 2023; Schuster and Dukes, 2017; Smith and Goetz, 2021; Volder and others, 2010). Established woody plants can also spread vegetatively and reinforce their dominance through feedbacks such as competitive shading, increased evapotranspiration, and altered soil infiltration, all of which suppress grass establishment (Ansley and others, 2023; Archer and others, 2017; Dodds and others, 2023; Grant and others, 2017; Keen and others, 2022; Keen and others, 2024; Riddle and others, 2014; Sadayappan and others, 2023; Vose and others, 2016). Drought can further tip the balance in favor of woody expansion by weakening herbaceous competitors and creating recruitment opportunities for woody plants during periods of reduced ground cover (Archer and others, 2017; Bond, 2008; Keen and others, 2024; Polley and others, 2013; Slette and others, 2023; Smith and Goetz, 2021; Volder and others, 2010; Zhou and others, 2022). These effects are especially pronounced under repeated or severe drought, which can impair the recovery of native grassland species and accelerate transitions to shrubland (Slette and others, 2022; Slette and others, 2023; Smith and others, 2024; Zhou and others, 2022), particularly in species-poor systems with limited adaptive capacity (Isbell and others, 2015; Vose and others, 2016). Only exceptionally extreme droughts are likely to cause woody mortality or disrupt the self-reinforcing feedbacks that maintain this encroached state (Keen and others, 2024; Muench and others, 2016; Ratajczak and others, 2011; Twidwell and others, 2016; Vose and others, 2016).

Community responses to seasonal shifts. By favoring species that can capitalize on earlier or extended growing seasons, seasonal shifts may contribute to community reassembly in the Central and Eastern North American Grassland and Shrubland ecosystem, potentially altering relative species abundance and dominant growth forms. Species with life histories or physiological traits that enable early growth or reproduction, particularly those adapted to cool-season moisture and temperature cues, are poised to gain a competitive advantage (Briske, 2017; Jones, 2019). Shifting phenology, particularly earlier spring green-up, may also alter competitive hierarchies, favoring species most capable of adjusting phenology (Cleland and Wolkovich, 2024; Dawson-Glass and others, 2025a; Ratajczak and Ladwig, 2019). Over time, this may shift community composition toward early-season growers and species whose phenology is

highly responsive to climatic signals, potentially at the expense of those unable to adjust their phenology to changing seasonal conditions (Craine and others, 2012; Lee and others, 2024; Palit and others, 2021; Wolkovich and Donahue, 2021). For instance, annuals tend to exhibit stronger phenological shifts than perennials, likely due to their shorter life cycles and greater responsiveness to environmental cues (Cleland and others, 2006; Dawson-Glass and others, 2025b; Stuble and others, 2021). Conversely, species that cannot reliably align development with new seasonal patterns may decline in abundance (Miller-Rushing and others, 2010; Thackeray and others, 2016; Willis and others, 2008; Zettlemoyer and others, 2021). For example, while many native prairie species consistently advance flowering in response to warmer spring and winter temperatures, species with limited phenological plasticity, genetic migration potential, or those that inappropriately respond to shifting phenological cues may be susceptible to local extinction (Zettlemoyer and others, 2021).

As species shift their phenology in response to climate change, communities are reassembling, with some species becoming temporally isolated and others experiencing greater overlap in flowering times with co-occurring species (Austin and others, 2024; Ramirez-Parada and others, 2022; Whittington and others, 2015). For example, warming may advance reproduction in mid-season panicgrasses (*Panicum* spp.) while delaying it in late-season dominants like bluestems (*Andropogon* spp.), reducing mid-season flowering (Sherry and others, 2007). However, in many cases, longer flowering periods lead to greater temporal overlap as species either advance or delay flowering, intensifying competition for pollinators at the beginning and end of the growing season (Austin and others, 2024; Kehrberger and Holzschuh, 2019; Mesgaran and others, 2017; Reyes-Fox and others, 2014). Collectively, these shifts can reduce reproductive synchrony and diminish floral resources available to pollinators with specialized dietary needs (Memmott and others, 2007; Sherry and others, 2007). As a result, the reproductive fitness of plants may be affected, particularly for species vulnerable to trophic mismatch with key pollinators (Memmott and others, 2007; Ramirez-Parada and others, 2022).

Woody species may be particularly well positioned to exploit emerging seasonal niches under climate change. For some woody species like the evergreen eastern redcedar (*Juniperus virginiana*), warmer winter temperatures can increase physiological activity, enabling greater carbon gain and root growth during the winter months when most herbaceous and deciduous woody species remain dormant (Ansley and others, 2023; Caterina and others, 2014; Zou and others, 2014). This extended activity window may reduce spring soil moisture available to neighboring plants and further suppress herbaceous competitors. In addition, some woody species may exhibit stronger phenological responses to warming than their herbaceous counterparts (Geissler and others, 2023), allowing them to shift growth timing more effectively than neighboring forbs and grasses.

Invasive Species

General invasive species responses to climate change. The Central and Eastern North American Grassland and Shrubland ecosystem is already among the most heavily invaded ecosystems in the world, and climate change is expected to exacerbate this pattern (Catford and Jones, 2019; Fan and others, 2013; Poland and others, 2021). Several broad drivers and their effects are likely to promote further establishment and spread of invasive species under climate change: shifts in temperature and moisture, altered competitive dynamics, increased disturbance from extreme events, land use change, and enhanced dispersal (Catford and Jones, 2019; Clements and Jones, 2021; Poland and others, 2021).

Many invasive species possess physiological, reproductive, and phenological traits that confer consistent advantages over native species, enabling them to rapidly exploit changing environmental conditions.

Ruderal traits, such as rapid growth, early maturation, short generation times, high fecundity, and low root biomass, are common among invasive grasses (Catford and Jones, 2019; Schrama and Bardgett, 2016). These characteristics may confer a competitive advantage following extreme events, particularly in disturbance-prone or low-diversity communities (Catford and Jones, 2019; Clements and Jones, 2021; Fynn and others, 2009; Smith and Knapp, 2001). As exposure to climate extremes can reduce grassland community stability and resistance to invasion, particularly in the absence of canopy thermal buffering, grasslands with limited herbaceous cover and low topographic complexity may be especially vulnerable (Bernath-Plaisted and others, 2023; Bernath-Plaisted and others, 2024; Liu and others, 2024a; Ratcliffe and others, 2023; White and others, 2001). At finer scales, certain weather conditions likely to increase in the future (e.g., drought, heatwaves) may reduce the effectiveness of herbicides and other control efforts, further complicating management in already degraded areas (Hanberry and others, 2019; Vose and others, 2016; Wagle and Gowda, 2018).

Invasive species responses to drought. Projected increases in drought frequency and severity may shift competitive dynamics in ways that disproportionately favor invasive grasses and forbs across the Central and Eastern North American Grassland and Shrubland ecosystem, suggesting that invasive species may be equally or even more problematic in the future (Duell and others, 2016; Duell and others, 2021). Generally, drought may have more negative impacts on invasive species than natives (Braasch and others, 2025; Ruppert and others, 2024; Valliere and others, 2019). However, drought does not always disproportionately harm invasive species (Liu and others, 2017; Ratcliffe and others, 2023; Zhang and others, 2023) and may even promote invasion, negatively impacting native species, weakening community resistance, and increasing competition for resources (Jiménez and others, 2011; LaForgia and others, 2020; Mojzes and others, 2020; Sanders and others, 2024). For instance, invasive species may outperform natives in biomass production, reproductive output, and survival under drought conditions (Ali and Bucher, 2022; Dong and others, 2014; Sobuj and others, 2024). Moreover, the combination of extreme drought and heatwaves has been shown to increase overall community invasibility (Dreesen and others, 2015).

Common ruderal characteristics may allow many invasive species to tolerate or avoid drought stress and then capitalize on post-drought resource pulses and newly available niche space (Schrama and Bardgett, 2016; Wolkovich and others, 2013). For example, under drought conditions, smooth brome (*Bromus inermis*) has been shown to outperform the native western wheatgrass (*Pascopyrum smithii*) (Duell and others, 2021) and recover more rapidly from drought stress (Catford and Jones, 2019). Smooth brome also exhibits drought avoidance strategies, including morphological adjustments in the root system during dry periods (Kroeger and Otfinowski, 2025). Similarly, despite being generally classified as hydrophilic and drought-intolerant, Kentucky bluegrass (*Poa pratensis*) can maintain a competitive advantage over native grasses following drought (Kirwan and others, 2021).

Invasive species responses to seasonal shifts. In the Central and Eastern North American Grassland and Shrubland ecosystem, many invasive grasses and forbs exhibit earlier phenologies than native species, as they germinate, leaf out, flower, and fruit earlier in the season (Calinger and others, 2013; Dawson-Glass and others, 2025a; Wolkovich and others, 2013; Zettlemoyer and others, 2019). In the context of earlier onset of growing seasons, early invasive phenologies permit priority access to light and other resources, suppressing the establishment and growth of later-emerging natives (Catford and Jones, 2019; Dickson and others, 2012a; Wilsey and others, 2015). Invasive species often demonstrate greater responsiveness to environmental cues than natives, particularly to warming in mesic environments (Dawson-Glass and others, 2025a; Dickson and others, 2012b; Sorte and others, 2013; Willis and others, 2010; Wolkovich and others, 2013; Zettlemoyer and others, 2022). For example, some invasive annual herbs can advance

flowering more rapidly than native species in response to spring warming – by as much as ~4 days per 1.8°F (1°C) (Calinger and others, 2013; Keller and Shea, 2021). This plasticity may allow invasive species to maintain or expand their dominance across a broader range of climate conditions (Dawson-Glass and others, 2025a; Garretson and others, 2022; Wolkovich and Cleland, 2014). These advantages are especially evident in cool-season (C_3) grasses like smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*), which are photosynthetically active early in spring and benefit from extended growing seasons (Ahler and others, 2025; Grant and others, 2020; Palit and others, 2021; Stotz and others, 2017). Their ability to initiate growth earlier in spring and remain active later into fall may allow them to accumulate more biomass and outcompete native species with shorter periods of activity (Catford and Jones, 2019; Palit and others, 2021).

Furthermore, for invasive species, extended growing seasons may enhance reproductive output and establishment success. Warming tends to advance early-season life stages such as germination and flowering, while delaying later stages like seed maturation, effectively accelerating or lengthening the reproductive period for some invasives (Dawson-Glass and others, 2025a; Gioria and others, 2018; Keller and Shea, 2021; Peng and others, 2018).

Herbivory

Herbivory responses to drought and seasonal shifts. Changes in drought regimes and seasonality may alter herbivore-plant interactions in ways that intensify pressure on native forbs and shift community composition in the Central and Eastern North American Grassland and Shrubland ecosystem. Drought can weaken plant defenses, increasing palatability and susceptibility to herbivores (Duell and others, 2024). Shifts in the timing and duration of the growing season may result in earlier forage availability, thereby increasing foraging pressure, particularly if plant and herbivore phenologies shift in synchrony (Hamann and others, 2021; Smith and Hall, 2016). White-tailed deer preferentially browse forbs over warm-season (C_4) grasses, especially during peak summer months when drought stress is often highest (Anderson and others, 2001; Anderson and others, 2017). This selective pressure reduces reproductive output in many prairie forbs, shifting competitive dynamics in favor of grasses and contributing to declines in forb diversity (Anderson and others, 2001; Nisi and others, 2015). Species such as Culver's root (*Veronicastrum virginicum*), leadplant (*Amorpha canescens*), and purple prairie clover (*Dalea purpurea*) are particularly vulnerable (Anderson and others, 2001). In addition, higher deer densities can limit recruitment of native woody species and indirectly facilitate the expansion of non-native grasses by suppressing competitors (Batzli and Dejaco, 2013; DiTommaso and others, 2014; Pruszenski and Hernández, 2020).

Microbial and Fungal Communities

Microbial responses to drought. Soil microbial communities in the Central and Eastern North American Grassland and Shrubland ecosystem are particularly sensitive to moisture loss (Lemoine and others, 2024; Ochoa-Hueso and others, 2018). Drought can drastically reduce overall microbial biomass while increasing richness and evenness (i.e., balance of population sizes) among surviving taxa (Sheik and others, 2011). These changes in microbial abundance and composition can persist and have lasting impacts on soil function (Barnard and others, 2013; Ochoa-Hueso and others, 2018; Preece and others, 2019; Sheik and others, 2011). Under severe moisture stress, microbial activity declines, suppressing decomposition and respiration (Lemoine and others, 2024). Even moderate, sublethal droughts can reduce decomposition by half, with potential long-term implications for carbon cycling, including greater carbon release (Lemoine and others, 2024).

Fungal responses to drought and seasonal shifts. Drought and seasonal shifts also influence diversity and compositional turnover of root-associated fungi (Barber and others, 2017; Hetrick and others, 1988; Lagueux and others, 2021). Arbuscular mycorrhizal fungi (AMF), which are key to plant nutrient and water uptake, exhibit stress-dependent responses to drought; in current CO_2 conditions, colonization increases under drought, but when drought is combined with elevated CO_2, colonization decreases (Pehim Limbu and Avolio, 2025). Extreme drought can reorder fungal community composition, with outcomes varying by host grass species and local soil conditions (Fu and others, 2022; Lagueux and others, 2021; Ochoa-Hueso and others, 2018). In particular, reduced AMF function may impair root conductivity and soil structure, potentially intensifying drought stress on host plants (Chandrasekaran, 2022; Duell and others, 2021; Fu and others, 2022; Li and others, 2019; Wilson and others, 2009). Responses among pathogenic fungi are more variable, with the impact of soil-borne pathogens increasing under warming in some cases (Delgado-Baquerizo and others, 2020; Yan and others, 2024), while decreasing when precipitation is limited (Elad and Pertot, 2014; Hu and others, 2025; Jeger, 2022). Warmer temperatures can simultaneously enhance fungal growth and stimulate plant immune defenses (Hu and others, 2025), potentially inhibiting pathogens such as rusts under warmer, drier conditions (Agrios, 2005; Helfer, 2014; Pugnaire and others, 2019).

Climate Change Impacts by Habitat

Central Tallgrass Prairie

The Central Tallgrass Prairie habitat encompasses a range of open grassland community types distributed across the central Midwest (Figure 3) (Taft and others, 2009; USNVC, 2025). These prairies span diverse geomorphic settings, including sandy outwash plains, glacial till and loessal uplands, and gravelly terraces, supporting a spectrum of dry to mesic prairie types (Faber-Langendoen and others, 2020). Soil texture and depth are key drivers of community differentiation (Wernerehl and Givnish, 2015). Dry sand prairies occur on excessively drained, coarse soils, where drought stress restricts woody establishment and favors graminoid dominance (Faber-Langendoen and others, 2020; Kost and others, 2007; Taft and others, 2009). Conversely, mesic prairies typically develop in areas with deep and relatively nutrient-rich soils that retain more moisture (Faber-Langendoen and others, 2020; Kost and others, 2007). Community structure and composition vary predictably across moisture gradients both locally and regionally, with more xeric-adapted species prevalent in drier prairie types and western prairie systems, and mesic species occurring in wetter, eastern systems (Faber-Langendoen and others, 2020; Markham, 2019; Nelson and Anderson, 1983; USNVC, 2025). Central Tallgrass Prairie habitat vegetation is generally dominated by a diverse mix of warm- (C_4) and cool-season (C_3) grasses, perennial forbs, and the occasional woody species (consisting of 60 dominant USNVC species) (Table 1) (Grman and others, 2021; USNVC, 2025).

Historically, the persistence of tallgrass prairie was maintained by frequent natural and anthropogenic fire, drought, and herbivory (Briggs and others, 2005; Harty, 2023; McClain and others, 2021). Fire regimes, with return intervals ranging from one to five years, suppressed woody encroachment and maintained open canopy structure, favoring fire-adapted grasses and forbs (Faber-Langendoen and others, 2020; Gaetani and others, 2010; Leis and Morrison, 2022; Taft and others, 2009). Large herbivores such as bison and elk historically contributed to disturbance by grazing, trampling, and nutrient deposition (Changnon and others, 2002; Harty, 2023). These combined stressors produced a dynamic mosaic of plant communities across the landscape. Today, habitat loss and fragmentation, fire suppression, and invasive species are among the most pressing threats to prairie integrity. Due to their suitability for agricultural production, currently existing tallgrass prairie habitat represents a small fraction of its historical extent, lacks connectivity between remnant areas, and ranks among the most threatened ecosystems globally (Hoekstra and others, 2005; Robertson and others, 1997; Samson and others, 2004). This habitat may be at increasing risk of habitat conversion as agricultural areas shift under climate change (Cunningham, 2022; Wright and Wimberly, 2013). Woody encroachment by shrubs and trees has transformed many remnant tallgrass prairie systems, especially where fire return intervals have lengthened (Briggs and others, 2005; Gaetani and others, 2010; Leis and Morrison, 2022). Numerous invasive shrubs, graminoids, and forbs have further reduced native plant cover and diversity, including autumn olive (*Elaeagnus umbellata*), buckthorns (*Rhamnus* spp.), honeysuckles (*Lonicera* spp.), Japanese knotweed (*Polygonum cuspidatum*), Kentucky bluegrass (*Poa pratensis*), multiflora rose (*Rosa multiflora*), quackgrass (*Elymus repens*), smooth brome (*Bromus inermis*), timothy (*Phleum pratense*), and yellow sweet-clover (*Melilotus officinalis*) (Kost and others, 2007; Taft and others, 2009; USNVC, 2025).

Table 1. Plant species common to the Central Tallgrass Prairie 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
Agrostis hyemalis (Walter) Britton, Sterns & Poggenb.	Winter bentgrass	Graminoid
Andropogon gerardii Vitman	Big bluestem	Graminoid
Aristida basiramea Engelm. ex Vasey	Forked threeawn	Graminoid
Aristida tuberculosa Nutt.	Seaside threeawn	Graminoid
Blephilia ciliata (L.) Benth.	Downy pagoda-plant	Forb/herb
Bouteloua curtipendula (Michx.) Torr.	Sideoats grama	Graminoid
Brickellia eupatorioides (L.) Shinners	False boneset	Forb/herb
Calamagrostis canadensis (Michx.) P. Beauv.	Bluejoint	Graminoid
Camassia scilloides (Raf.) Cory	Atlantic camas	Forb/herb
Carex muehlenbergii Schkuhr ex Willd.	Muhlenberg's sedge	Graminoid
Carex pensylvanica Lam.	Pennsylvania sedge	Graminoid
Carex siccata Dewey	Dryspike sedge	Graminoid
Carex tonsa (Fernald) E.P. Bicknell	Shaved sedge	Graminoid
Cicuta maculata L.	Spotted water hemlock	Forb/herb
Clinopodium arkansanum (Nutt.) House	Limestone calamint	Forb/herb
Cyperus lupulinus (Spreng.) Marcks	Great Plains flatsedge	Graminoid
Cyperus schweinitzii Torr.	Schweinitz's flatsedge	Graminoid
Dalea candida Michx. ex Willd.	White prairie clover	Forb/herb
Dalea purpurea Vent.	Purple prairie clover	Forb/herb
Danthonia spicata (L.) P. Beauv. ex Roem. & Schult.	Poverty oatgrass	Graminoid
Desmanthus illinoensis (Michx.) MacMill. ex B.L. Rob. & Fernald	Illinois bundleflower	Forb/herb
Dichanthelium linearifolium (Scribn. ex Nash) Gould	Slimleaf panicgrass	Graminoid
Dichanthelium ovale (Elliott) Gould & C.A. Clark	Eggleaf rosette grass	Graminoid
Eleocharis compressa Sull.	Flatstem spikerush	Graminoid
Eryngium yuccifolium Michx.	Button eryngo	Forb/herb
Euphorbia corollata L.	Flowering spurge	Forb/herb
Helianthus pauciflorus Nutt.	Stiff sunflower	Forb/herb
Hesperostipa spartea (Trin.) Barkworth	Porcupinegrass	Graminoid
Koeleria macrantha (Ledeb.) Schult.	Prairie Junegrass	Graminoid
Lithospermum canescens (Michx.) Lehm.	Hoary puccoon	Forb/herb
Muhlenbergia cuspidata (Torr. ex Hook.) Rydb.	Plains muhly	Graminoid
Panicum virgatum L.	Switchgrass	Graminoid
Penstemon hirsutus (L.) Willd.	Hairy beardtongue	Forb/herb

Species	Common name	Growth habit
**Poa compressa* L.	Canada bluegrass	Graminoid
**Poa pratensis* L.	Kentucky bluegrass	Graminoid
Polygala verticillata L.	Whorled milkwort	Forb/herb
Pycnanthemum virginianum (L.) T. Dur. & B.D. Jacks. ex B.L. Rob. & Fernald	Virginia mountainmint	Forb/herb
Rudbeckia hirta L.	Blackeyed Susan	Forb/herb
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid
Scutellaria parvula Michx.	Small skullcap	Forb/herb
Selaginella rupestris (L.) Spring	Northern selaginella	Forb/herb
Solidago canadensis L.	Canada goldenrod	Forb/herb
Solidago nemoralis Aiton	Gray goldenrod	Forb/herb
Sorghastrum nutans (L.) Nash	Indiangrass	Graminoid
Sporobolus cryptandrus (Torr.) A. Gray	Sand dropseed	Graminoid
Sporobolus heterolepis (A. Gray) A. Gray	Prairie dropseed	Graminoid
Symphyotrichum ericoides (L.) G.L. Nesom	White heath aster	Forb/herb
Symphyotrichum sericeum (Vent.) G.L. Nesom	Western silver aster	Forb/herb
Amorpha canescens Pursh	Leadplant	Shrub
Ceanothus americanus L.	New Jersey tea	Shrub
Cornus drummondii C.A. Mey.	Roughleaf dogwood	Shrub
Corylus americana Walter	American hazelnut	Shrub
†*Crataegus* spp. L.	Hawthorn	Shrub, tree
Malus ioensis (Alph. Wood) Britton	Prairie crab apple	Shrub, tree
Prunus americana Marshall	American plum	Shrub, tree
Rosa carolina L.	Carolina rose	Subshrub
Quercus macrocarpa Michx.	Bur oak	Shrub, tree
Rhus glabra L.	Smooth sumac	Shrub, tree
Salix humilis Marshall	Prairie willow	Shrub

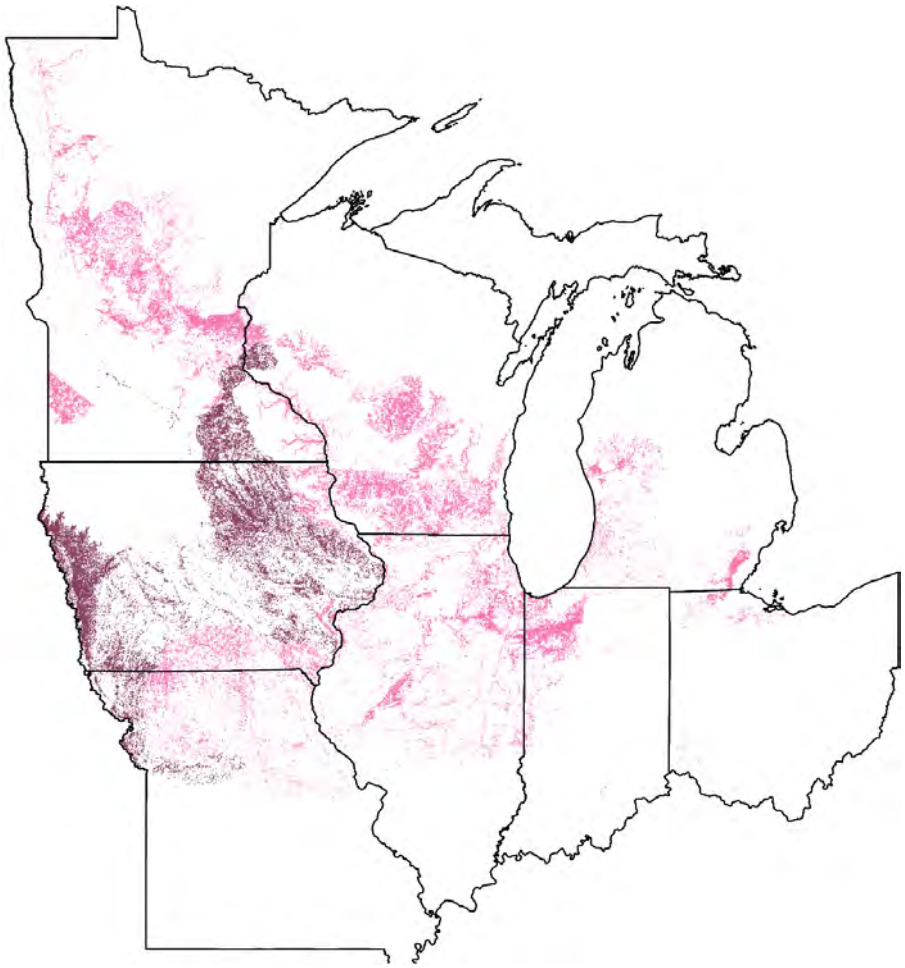

Figure 3. Central Tallgrass Prairie habitat distribution. Current geographic distribution of the Central Tallgrass Prairie habitat (Group Code: G333) in the Midwest (Faber-Langendoen and others, 2020; USNVC, 2025). Dark pink areas indicate Central Tallgrass Prairie habitat in the Temperate Prairie ecoregion, while light pink 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 Tallgrass Prairie habitat is prioritized for the Temperate Prairie 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 10.4°F increase in mean annual temperature 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 2.4 additional inches per year, although some indicate a slight decline (RCP 8.5; 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 in total precipitation (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 2. Projected changes in mean annual temperature in the Temperate Prairie 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 mean 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	47.9°F	50.2°F (2.4)	51.2°F (3.3)	51.1°F (3.2)	53.7°F (5.8)
Mean of models	47.9°F	52.9°F (4.9)	54.4°F (6.5)	53.9°F (6.0)	58.3°F (10.3)
Maximum model	47.8°F	54.3°F (6.4)	56.4°F (8.6)	56.3°F (8.4)	62.4°F (14.6)

Table 3. Projected changes in annual precipitation in the Temperate Prairie 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 mean 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	33.5 in	33.7 in (1%)	31.8 in (-5%)	36.2 in (8%)	30.3 in (-9%)
Mean of models	32.8 in	34.5 in (5%)	34.6 in (6%)	34.3 in (5%)	35.2 in (7%)
Maximum model	34.5 in	36.4 in (6%)	36.6 in (6%)	35.4 in (3%)	38.4 in (11%)

Key Climate Change Stressors

Drought and seasonal shifts. Across the Temperate Prairie ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 81% compared to historical conditions by the end of the century (RCP 8.5; Table 4), with the highest projected VPD in Missouri (Figure 4). Higher VPD means more rapid evaporation and water loss from plants and soils, which can intensify drought conditions. The length of the growing season is also projected to increase by approximately 20% by the end of the century, translating to more than 33 additional consecutive days with minimum temperatures above freezing (RCP 8.5; Table 5). These increases are expected to be most pronounced in Missouri and parts of southern Iowa (Figure 5).

Table 4. Projected changes in summer vapor pressure deficit in the Temperate Prairie 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 mean 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	1.08 kPa	1.12 kPa (4%)	1.22 kPa (13%)	1.20 kPa (11%)	1.29 kPa (19%)
Mean of models	1.09 kPa	1.43 kPa (31%)	1.56 kPa (43%)	1.55 kPa (42%)	1.97 kPa (81%)
Maximum model	1.11 kPa	1.58 kPa (42%)	1.82 kPa (64%)	1.96 kPa (77%)	2.79 kPa (151%)

Table 5. Projected changes in growing season length in the Temperate Prairie 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 mean historical (1971–2000) and projected growing season length (consecutive days with minimum temperatures above freezing) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected growing season length under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	168.2	181.9 (8%)	186.1 (11%)	180.2 (7%)	183.7 (9%)
Mean of models	171.1	191 (12%)	199.2 (16%)	186.9 (9%)	204.5 (20%)
Maximum model	174.4	213 (22%)	217.4 (25%)	213 (22%)	229.5 (32%)

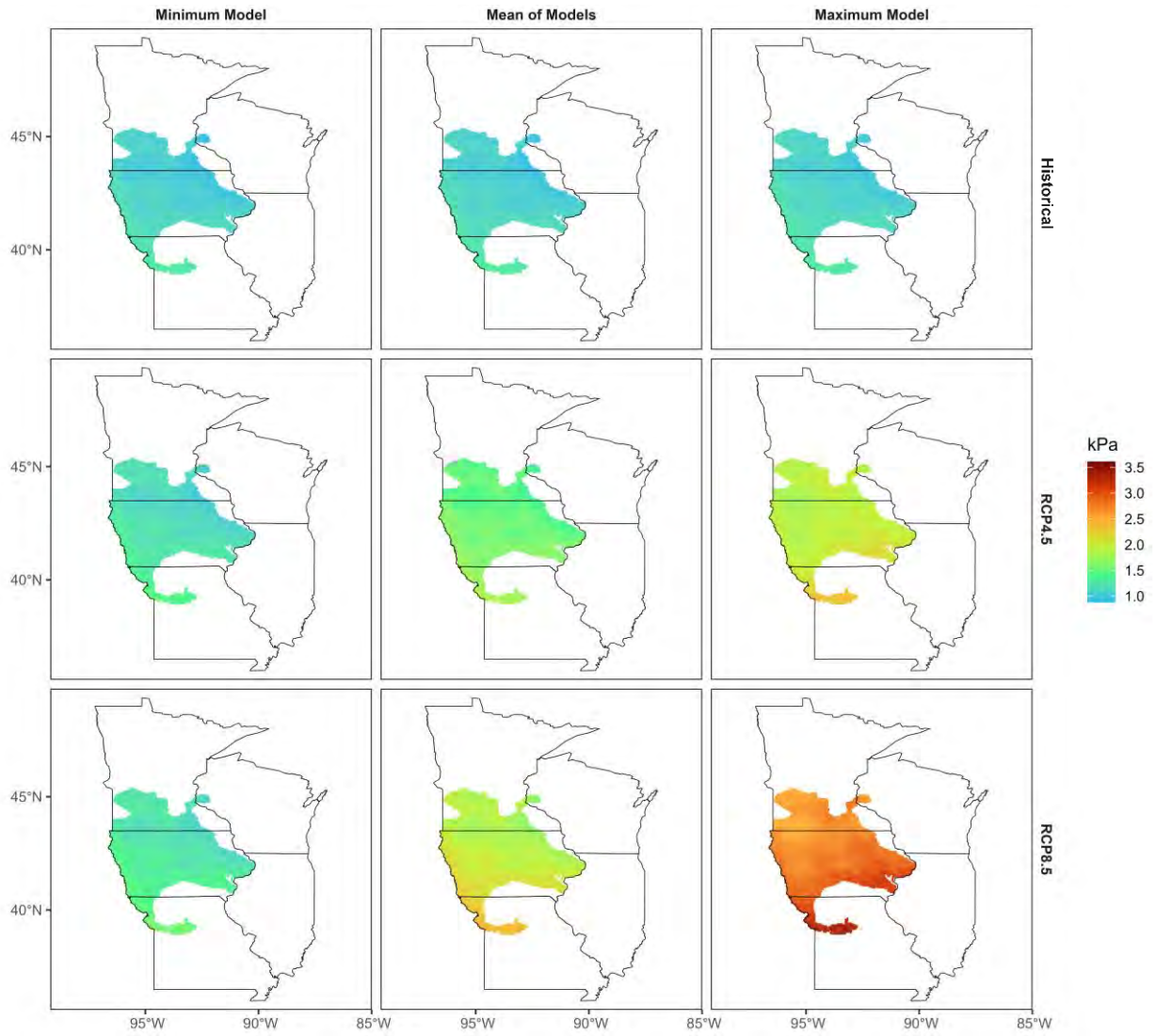

Figure 4. Projected summer vapor pressure deficit in the Temperate Prairie ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean 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).

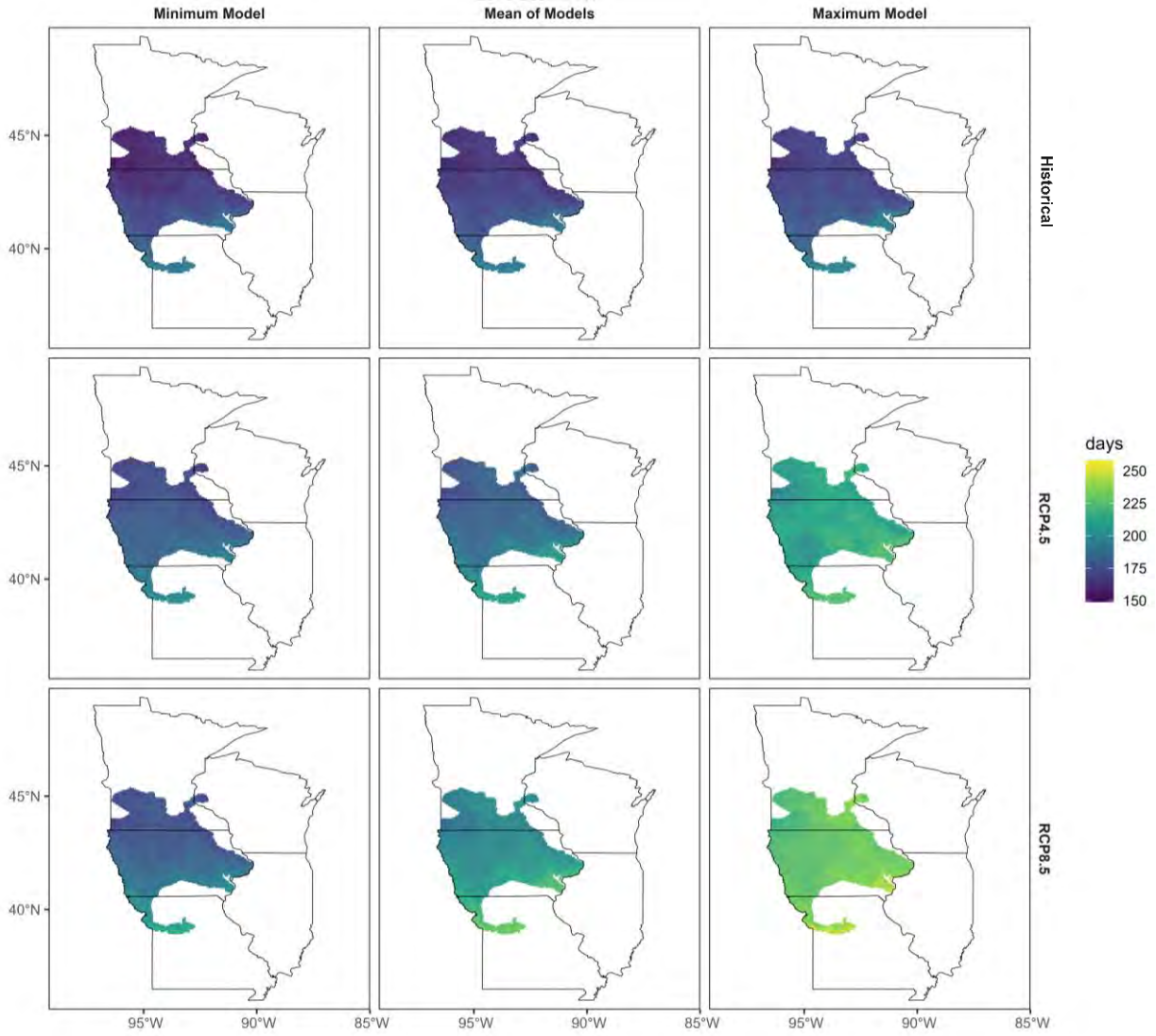

Figure 5. Projected annual growing season length in the Temperate Prairie ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean historical (1971–2000, top row) and projected (2070–2099) annual growing season length (consecutive days with minimum temperatures above freezing) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected annual growing season length under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

This section summarizes how climate change is impacting the Central Tallgrass Prairie habitat. It covers both the overall influence on species distributions, adaptive capacity, and vulnerability (Table 6) and the species-specific effects of drought and seasonal shifts summarized from available literature (Table 7). The bullet points below provide a concise overview of these impacts on habitat structure, community composition, invasive species, herbivory, and microbial and fungal communities.

Habitat structure:

- Drought and increasingly variable precipitation will likely reduce soil moisture and plant cover, intensifying erosion, accelerating nutrient loss, and impairing long-term soil function (Castillioni and others, 2022b; Fay and others, 2011; Felton and others, 2020; Hanberry and others, 2019; Hoover and Rogers, 2016; Lemoine and others, 2024; Vose and others, 2016).
- Physiological stress from drought may suppress plant growth and increase the risk of mortality and reproductive failure, particularly during high-temperature events (Dreesen and others, 2015; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Smith and others, 2024; Vose and others, 2016; Wang and others, 2008; Will and others, 2013).
- Species vulnerability to drought varies by traits such as rooting depth and photosynthetic pathway (Barkaoui and Volaire, 2023; Hoover and others, 2019; Kazemi and others, 2023; Keen and others, 2024a; Nippert and Knapp, 2007; Padilla and others, 2019; Slette and others, 2023; Smith and others, 2024; Taylor and others, 2011; Wilcox and others, 2023; Zhou and others, 2022).
- Drought may reduce productivity and impair ecosystem functioning, especially in areas with high aridity or low species diversity (Briske, 2017; Fay and others, 2011; Jones, 2019; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Slette and others, 2023; Smith and others, 2024).
- Warming-driven shifts in seasonal timing are expected to alter flowering phenology and growing season dynamics (Austin and others, 2024; Dunnell and Travers, 2011; Geissler and others, 2023; Memmott and others, 2007; Ramirez-Parada and others, 2022; Sherry and others, 2007; Zettlemoyer and others, 2021).
- Phenological shifts associated with seasonal warming may boost productivity by extending the growing season but also increase vulnerability to late frosts (Allstadt and others, 2015; Briske, 2017; Cleland and Wolkovich, 2024; Fridley and others, 2016; Li and others, 2025; Zettlemoyer and others, 2021).

Community composition:

- Changing mean annual temperature and precipitation patterns may promote prairie-savanna expansion into areas currently dominated by forest at the western grassland-forest ecotype, while wetter conditions in the eastern Midwest may continue to facilitate conversion of grasslands into forest and shrubland (Ahler and others, 2025; Galatowitsch and others, 2009; Havrilla and others, 2023; Reich and others, 2022; Toot and others, 2020).
- Droughts may lead to short- and long-term shifts in dominant species by intensifying aboveground competition and weakening priority effects, favoring drought-tolerant species such as little bluestem (*Schizachyrium scoparium*) or gramas (*Bouteloua* spp.) over drought-sensitive species like Indiangrass (*Sorghastrum nutans*) (Table 7) (Baty and others, 2015; Dekirmenjian and others, 2024; Griffin-Nolan and others, 2021; Kooyers and others, 2025; Smith and others, 2017; Watson and others, 2021).
- Whether drought stress reinforces or weakens warm season (C_4) grass dominance over cool season (C_3) species may depend on drought severity and interacting factors, such as atmospheric CO_2 concentrations and precipitation variability (Ahler and others, 2025; Catford and Jones, 2019; Griffin-Nolan and others, 2021; Havrilla and others, 2023; O'Keefe and others, 2024; Polley and others, 2013).

- Functional groups such as deep-rooted shrubs and forbs, as well as species with drought avoidance or escape strategies, may increase in abundance as graminoids decline under drought (Table 7) (Dodds and others, 2023; Sadayappan and others, 2023; Smith and Goetz, 2021; Vose and others, 2016).
- Drought may facilitate woody encroachment by reducing herbaceous competition and favoring deep-rooted woody species, such as roughleaf dogwood (*Cornus drummondii*), that can access subsurface moisture, particularly in the absence of effective fire regimes (Archer and others, 2017; Breshears and others, 2016b; Dodds and others, 2023; Keen and others, 2024; Sadayappan and others, 2023; Smith and Goetz, 2021; Volder and others, 2010; Vose and others, 2016).
- Seasonal shifts may favor early-growing or phenologically responsive species, leading to gradual changes in abundance as less flexible taxa decline (Cleland and Wolkovich, 2024; Dawson-Glass and others, 2025a; Lee and others, 2024; Ramirez-Parada and others, 2022; Wolkovich and Donahue, 2021; Zettlemoyer and others, 2021).
- Earlier and extended growing seasons may result in novel co-flowering patterns and disrupt synchrony between plant flowering and pollinator activity, increasing the risk of trophic mismatches that reduce reproductive success for species unable to align phenology with shifting seasonal conditions (Austin and others, 2024; Memmott and others, 2007; Ramirez-Parada and others, 2022; Whittington and others, 2015; Zettlemoyer and others, 2021).

Invasive species:

- Climate change is expected to intensify invasive species pressure by favoring species with ruderal traits and high plasticity that can rapidly exploit warmer temperatures, variable precipitation, and resource availability following extreme events (Catford and Jones, 2019; Clements and Jones, 2021; Fan and others, 2013; Poland and others, 2021; Vose and others, 2016).
- Drought may promote invasion by reducing community resistance, weakening native competitors, and generating post-drought resource pulses that favor traits common to invasive grasses and forbs (Ali and Bucher, 2022; Catford and Jones, 2019; Duell and others, 2016; Duell and others, 2021; Jiménez and others, 2011; LaForgia and others, 2020; Schrama and Bardgett, 2016).
- Invasive grasses with ruderal traits, such as rapid growth, high fecundity, and drought avoidance, may tolerate or recover from drought stress more effectively than many native species, leading to increased abundance (Table 7) (Braasch and others, 2025; Catford and Jones, 2019; Duell and others, 2021; Kirwan and others, 2021; Kroeger and Otfinowski, 2025; Schrama and Bardgett, 2016; Wolkovich and others, 2013).
- Warming-driven shifts in seasonality and longer growing seasons may advantage invasive species with early phenology and prolonged activity, such as smooth brome (*Bromus inermus*) and Kentucky bluegrass (*Poa pratensis*), allowing them to gain priority access to resources and suppress later-emerging natives (Table 7) (Ahler and others, 2025; Calinger and others, 2013; Catford and Jones, 2019; Dawson-Glass and others, 2025a; Dickson and others, 2012a; Keller and Shea, 2021a; Sorte and others, 2013; Willis and others, 2010; Wilsey and others, 2015; Wolkovich and others, 2013; Zettlemoyer and others, 2019).

Herbivory:

- Drought and seasonal shifts may intensify herbivory pressure, particularly from increasing white-tailed deer (*Odocoileus virginianus*) populations, by weakening plant defenses, extending forage

windows, and contributing to declines in native forb diversity (Anderson and others, 2001; Duell and others, 2024; Felton and others, 2024; Hamann and others, 2021; Nisi and others, 2015; Smith and Hall, 2016).

Microbial and fungal communities:

- Drought and seasonal shifts may drive fungal community reassembly and reduce the functional role of arbuscular mycorrhizal fungi in supporting plant drought tolerance (Fu and others, 2022; Hu and others, 2025; Lagueux and others, 2021; Ochoa-Hueso and others, 2021; Pehim Limbu and Avolio, 2025).
- Drought may reduce overall microbial activity and biomass, negatively impacting soil function and carbon cycling (Barnard and others, 2013; Ochoa-Hueso and others, 2018; Preece and others, 2019; Sheik and others, 2011).

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

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and/or vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity/ high vulnerability (–)	High adaptive capacity/ low vulnerability (+)
Amorpha canescens[1]	*Amorpha canescens*[1]	*Bouteloua curtipendula*[8]	*Quercus macrocarpa*[9,10,11]
Panicum virgatum[2]	*Andropogon gerardii*[3]	*Quercus ellipsoidalis*[9]	
**Poa pratensis*[2]	*Cornus drummondii*[4]		
Rudbeckia hirta[1]	*Dalea purpurea*[1]		
	Koeleria macrantha[1]		
	Quercus ellipsoidalis[5,6]		
	Quercus macrocarpa[4,6,7]		
	Schizachyrium scoparium[1]		

[1]Rana and others ; [2]Kane and others (2017); [3]Smith and others (2017); [4]Hanberry (2024); [5]Rauschendorfer and others (2022); [6]Peters and others (2020); [7]Prasad and others (2020); [8]Brinker and others (2018); [9]Potter and others (2017); [10]Etterson and others (2020); [11]Handler and others (2014)

Table 7. Summary of impacts from key climate change stressors on Central Tallgrass Prairie 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		*Agrostis hyemalis*[1]
		Andropogon gerardii[2–8]

Stressor	Impact	Affected species
	When tolerance of droughts is low, potential for reduced growth rates and abundance	*Amorpha canescens*[9]
		Calamagrostis canadensis[1,10,11]
		Carex pensylvanica[12]
		Cornus drummondii[1]
		Hesperostipa spartea[13]
		Panicum virgatum[11,14,15]
		**Poa pratensis*[1,16–19]
		Prunus americana[1,20,21]
		Quercus macrocarpa[22]
		Salix humilis[1]
		Schizachyrium scoparium[23,24]
		Sorghastrum nutans[4,15,25–27]
	When tolerance of droughts is high, potential for competitive advantage and increased abundance	*Amorpha canescens*[28]
		Andropogon gerardii[1,29–31]
		Bouteloua curtipendula[32,33]
		Calamagrostis canadensis[10]
		Ceanothus americanus[1,34]
		Cornus drummondii[35–37]
		Koeleria macrantha[1,38,39]
		Malus ioensis[1]
		Muhlenbergia cuspidata[40]
		Panicum virgatum[8]
		**Poa pratensis*[41]
		Quercus ellipsoidalis[42,43]
		Quercus macrocarpa[1,42,44,45]
		Rhus glabra[46]
		Schizachyrium scoparium[1,2,4,11,47]
		Sporobolus cryptandrus[1,48]
Seasonal shifts	When phenological shifts are well-matched to changing seasonality, potential for competitive advantage	*Polygala verticillata*[49]
		Solidago canadensis[50]
		Solidago nemoralis[49]
	When phenological shifts are poorly matched to changing seasonality, potential for reduced regeneration success and competitive capacity	*Ceanothus americanus*[49]
		Cicuta maculata[49]
		Cornus drummondii[51]
		Dalea purpurea[52]
		Rudbeckia hirta[49]

[1]USDA , NRCS 2025; [2]Dekirmenjian and others (2024); [3]Kooyers and others (2025); [4]Lemoine and others (2017); [5]Maricle and others (2017); [6]Pehim Limbu and Avolio (2025); [7]Uchytil (1988); [8]Watson and others (2021); [9]Fryer (2011); [10]Tesky (1992); [11]USDA, NRCS (2004); [12]Nelson and others (2018); [13]Walkup (1991a); [14]Liu and others (2015); [15]Kazemi and others (2023); [16]Ahler and others (2025); [17] Huff (2003); [18]Palit and others (2021); [19]Uchytil (1993); [20]Fryer (2010); [21]Wedel and others (2025); [22]Dannenberg and others (2019); [23]Duell and others (2016); [24]Volder and others (2010); [25]Hoover and others (2014); [26]Swemmer and others (2006); [27]Walkup (1991b); [28]Casey (2011); [29]Baty and others (2015); [30]Nippert and others (2009); [31]Slette and others (2023); [32]Chadwick (2003); [33]Engstrom (2004); [34]Wynia (2010); [35]Keen and others (2022); [36]Muench and others (2016); [37]O'Keefe and others (2024); [38]Goklany (2012); [39]Simonin (2000a); [40]Fryer (2009b); [41]Printz and Hendrickson (2015); [42]Peters and others (2015); [43]Rauschendorfer and others (2022); [44]Niinemets and Vallardares (2006); [45]Abrams(1990); [46]Johnson (1990); [47]Steinberg (2002); [48]Simomin (2000b); [49]Bertin (2017); [50]Zettlemoyer and others (2019); [51]Prevéy and others (2020); [52]Whittington and others (2015)

Focal Wildlife Species: Prairie Vole

The prairie vole (*Microtus ochrogaster*) is a small rodent found in grasslands across the central United States and south-central Canada. It occupies non-agricultural fields, where it builds shallow burrows and runways through dense vegetation (Sexton, 2024). Prairie voles are short-lived, typically surviving only about a year in the wild, but can breed multiple times each year. Compared to meadow voles, prairie voles inhabit drier environments, though they remain sensitive to drought and local rainfall variability (Meserve and Klatt, 1985). Population size appears to correlate with precipitation patterns (Hazard, 1982; Minnesota Department of Natural Resources, 2025a). In Illinois, both increased temperatures and higher rainfall were associated with reduced life expectancy, but the combination of high temperatures and low precipitation had the most pronounced negative effect (Getz and others, 1997). These responses are seasonally dependent, with summer rainfall appearing to buffer the impacts of heat, while precipitation during other parts of the year may exacerbate thermal stress (Getz and others, 1997). Male reproductive output also declines under drought conditions (Nelson and others, 1989).

However, during droughts, prairie voles are generally not constrained by access to water, but rather by reduced food availability resulting from lower plant productivity (Abdellatif and others, 1982). As generalist herbivores, prairie voles consume a wide range of plants and plant parts depending on availability (Meserve and Klatt, 1985; Sexton, 2024). While their short generation times allow prairie vole populations to recover quickly following resource pulses, prolonged or widespread droughts may outpace this capacity, potentially leading to local population declines or extirpation. If drought conditions are localized, immigration from adjacent areas could help offset population declines (Meserve and Klatt, 1985).

Habitat quality is also critical to prairie vole persistence, with mixed vegetation of forbs and grasses supporting their habitat needs, while areas with lower plant diversity tend to lack vole populations (Sexton, 2024). Prescribed fire may help maintain plant diversity, but it can also lead to mortality as prairie voles are surface-active throughout the year.

Focal Aquatic System: Chanarambie Creek

Chanarambie Creek, a tributary of the Rock River in southwestern Minnesota, drains a roughly 75-square-mile agricultural watershed dominated by cropland (approximately two-thirds) and rangeland (approximately one-third) (Koschak, 2015). It lies within the Missouri River Basin, a major hydrologic system spanning much of the central United States. Intensive land use and widespread subsurface tile

drainage have transformed this former tallgrass prairie landscape, accelerating surface runoff, reducing groundwater infiltration, and producing flashier flows with elevated sediment and nutrient loads (Adelsperger and others, 2024; Koschak, 2015). Several stream segments within the watershed fail to meet water quality standards for turbidity and aquatic life, with biological assessments linking these impairments to habitat degradation, nutrient enrichment, and altered flow regimes (Boettcher and others, 2018; Minnesota Pollution Control Agency, 2014). Despite these pressures, the Chanarambie Creek valley contains some of the largest remaining native prairie tracts in southwestern Minnesota, supporting rare species and high-quality plant communities that elevate its conservation value (Minnesota Prairie Plan Working Group, 2018). This juxtaposition of hydrologic vulnerability and conservation value highlights the urgency of understanding how climate variability and long-term change may further alter water dynamics in the Chanarambie Creek watershed (Boettcher and others, 2018; Williams-Sether and Sanocki, 2025; Wilson and others, 2023).

Both recent observations and future projections indicate that climate change is reshaping hydrologic patterns in southwestern Minnesota, with implications for streamflow timing, runoff intensity, and seasonal water availability (Table 8) (Anurag and Ng, 2022; Barth and others, 2025; Williams-Sether and Sanocki, 2025; Wilson and others, 2023). In the southern half of Minnesota, peak stream flows are occurring later in the water year, likely due to shifts in precipitation timing (Williams-Sether and Sanocki, 2025). However, warming is advancing snowmelt timing across the Upper Midwest, and in years when snowmelt drives peaks, earlier melt could offset the current tendency toward later flows (Barth and others, 2025; Uzun and others, 2021). Across the Midwest, agricultural watersheds are projected to experience greater seasonal variability in streamflow, including increased runoff in winter and spring and reduced baseflows during summer (Barth and others, 2025; Chien and others, 2013). In the Missouri River Basin, cumulative runoff has already increased and is expected to continue rising due to more intense spring precipitation and a heightened risk of flooding, while increasingly variable summer rainfall may lead to prolonged low-flow conditions (Wilson and others, 2023). Despite projected increases in total precipitation, groundwater recharge may decline or remain stable, as higher evapotranspiration rates and runoff-prone soils limit water infiltration (Anurag and Ng, 2022). Collectively, these shifts point to a future in which Chanarambie Creek's hydrology is increasingly shaped by extreme seasonal contrasts.

Table 8. Projected changes in streamflow for Chanarambie Creek. Projections use gridded simulations of daily runoff based on CMIP5 climate models under the high-emissions (RCP 8.5) scenario. Values show mean 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 (Luce and others, 2025; 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	4.9 ft^3/s	255%
Mean Spring (March–May) Flow	9.8 ft^3/s	23.5 ft^3/s	139%
Mean Summer (June–August) Flow	9.3 ft^3/s	8.9 ft^3/s	-5%
Mean Fall (September–November) Flow	4.9 ft^3/s	7.0 ft^3/s	43%
Mean Annual Flow	6.4 ft^3/s	11.1 ft^3/s	74%
1.5-year Flood (33rd percentile)	104.2 ft^3/s	144.5 ft^3/s	39%
10-year Flood (90th percentile)	304.8 ft^3/s	444.3 ft^3/s	46%

25-year Flood (96th percentile)	378.4 ft^3/s	605.4 ft^3/s	60%
Number of Winter Floods	2.0	2.8	45%
Average Lowest 7-day Flow	0.4 ft^3/s	0.4 ft^3/s	6%
Center of Flow Mass Timing	May 11	April 21	-9%

Northern Tallgrass Prairie

The Northern Tallgrass Prairie habitat occupies the cooler, northern extent of the tallgrass prairie region, spanning portions of Minnesota, Iowa, and the eastern Dakotas (Figure 6) (Faber and Markham, 2011; USNVC, 2025). While compositionally similar to more southern prairie types, this habitat is distinguished by its unique climatic gradients, glacial history, and soil characteristics (Danz and others, 2011; McAndrews, 1966). Mesic prairies typically occur on nutrient-rich, moderately to poorly drained soils that retain moisture, whereas dry prairies are found on well-drained uplands, sandy outwashes, or calcareous tills (Faber and Markham, 2011; Faber-Langendoen and others, 2020; McAndrews, 1966). Local variation in plant community composition reflects underlying gradients in soil depth, texture, and moisture availability (Briggs and Knapp, 1995; Faber and Markham, 2011; Gibson and Hulbert, 1987), with even subtle changes in elevation, hydrology, or climate often producing sharp, nonlinear shifts in species composition (Danz and others, 2011; Gibson and Hulbert, 1987). Northern Tallgrass Prairie habitat vegetation is generally dominated by a diverse mix of warm- (C_4) and cool-season (C_3) grasses, perennial forbs, and the occasional woody species (consisting of 34 common USNVC species) (Table 9) (USNVC, 2025).

Historically, natural disturbance regimes, particularly fire, drought, and herbivory, were essential for maintaining open grassland structure and limiting woody encroachment (Allen and Palmer, 2011; Collins, 2000; Robertson and others, 1997; Samson and others, 2004). Fire return intervals of three to five years promoted community stability and prevented succession to shrubland or forest, especially in mesic sites where tree establishment would otherwise be favored (Archer and others, 2017; Collins, 2000; Faber and Markham, 2011; Knapp and others, 1998). Large herbivores such as bison and elk also historically contributed to spatial heterogeneity and nutrient cycling, but white-tailed deer are now the primary herbivore influence on woody vegetation dynamics in remnant systems (Changnon and others, 2002; Harty, 2023).

Today, the Northern Tallgrass Prairie habitat is among the most imperiled habitats in North America, with over 97% of its historical extent lost to agriculture, fragmentation, and woody encroachment (Gibson, 2009; Samson and others, 2004). These habitats may face increasing risk of land-use conversion as agricultural zones shift in response to climate change (Cunningham, 2022; Wright and Wimberly, 2013). Active management is often required to preserve fragmented habitats (Briske, 2017; Kaul and Wilsey, 2021; Koper and others, 2010). Numerous introduced species, including autumn olive (*Elaeagnus umbellata*), buckthorns (*Rhamnus* spp.), honeysuckles (*Lonicera* spp.), Japanese knotweed (*Polygonum cuspidatum*), Kentucky bluegrass (*Poa pratensis*), multiflora rose (*Rosa multiflora*), quackgrass (*Elymus repens*), smooth brome (*Bromus inermis*), timothy (*Phleum pratense*), and yellow sweet clover (*Melilotus officinalis*) threaten native plant cover and diversity (Kost and others, 2007; Taft and others, 2009; USNVC, 2025). In the coming years, the combined influence of environmental gradients, disturbance regimes, and ongoing anthropogenic change will continue to shape the future of this habitat.

Table 9. Plant species common to the Northern Tallgrass Prairie habitat. Asterisks (*) denote species introduced to the region. 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
Ambrosia artemisiifolia L.	Annual ragweed	Forb/herb
Andropogon gerardii Vitman	Big bluestem	Graminoid
**Berteroa incana* (L.) DC.	Hoary alyssum	Forb/herb
Blephilia ciliata (L.) Benth.	Downy pagoda-plant	Forb/herb
Bouteloua curtipendula (Michx.) Torr.	Sideoats grama	Graminoid
Bouteloua gracilis (Willd. ex Kunth) Lag. ex Griffiths	Blue grama	Graminoid
Bouteloua hirsute Lag.	Hairy grama	Graminoid
Brickellia eupatorioides (L.) Shinners	False boneset	Forb/herb
Calamovilfa longifolia (Hook.) Scribn.	Prairie sandreed	Graminoid
Carex inops L.H. Bailey	Long-stolon sedge	Graminoid
Carex siccata Dewey	Dryspike sedge	Graminoid
**Centaurea stoebe* L.	Spotted knapweed	Forb/herb
Cicuta maculata L.	Spotted water hemlock	Forb/herb
**Cirsium arvense* (L.) Scop.	Canada thistle	Forb/herb
Cyperus schweinitzii Torr.	Schweinitz's flatsedge	Graminoid
Hesperostipa spartea (Trin.) Barkworth	Porcupinegrass	Graminoid
Koeleria macrantha (Ledeb.) Schult.	Prairie Junegrass	Graminoid
**Melilotus officinalis* (L.) Lam.	Sweetclover	Forb/herb
Muhlenbergia richardsonis (Trin.) Rydb.	Mat muhly	Graminoid
Pascopyrum smithii (Rydb.) Á. Löve	Western wheatgrass	Graminoid
Rudbeckia hirta L.	Blackeyed Susan	Forb/herb
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid
**Securigera varia* (L.) Lassen	Crownvetch	Forb/herb, vine
Solidago canadensis L.	Canada goldenrod	Forb/herb
Sporobolus cryptandrus (Torr.) A. Gray	Sand dropseed	Graminoid
Sporobolus heterolepis (A. Gray) A. Gray	Prairie dropseed	Graminoid
Symphyotrichum ericoides (L.) G.L. Nesom	White heath aster	Forb/herb, vine
**Trifolium pratense* L.	Red clover	Forb/herb
**Trifolium* spp. L.	Clover	Forb/herb
**Verbascum Thapsus* L.	Common mullein	Forb/herb
Amorpha canescens Pursh	Leadplant	Shrub
Populus tremuloides Michx.	Quaking aspen	Tree
Quercus macrocarpa Michx.	Bur oak	Shrub, tree
†*Salix* spp. L.	Willow	Shrub, tree

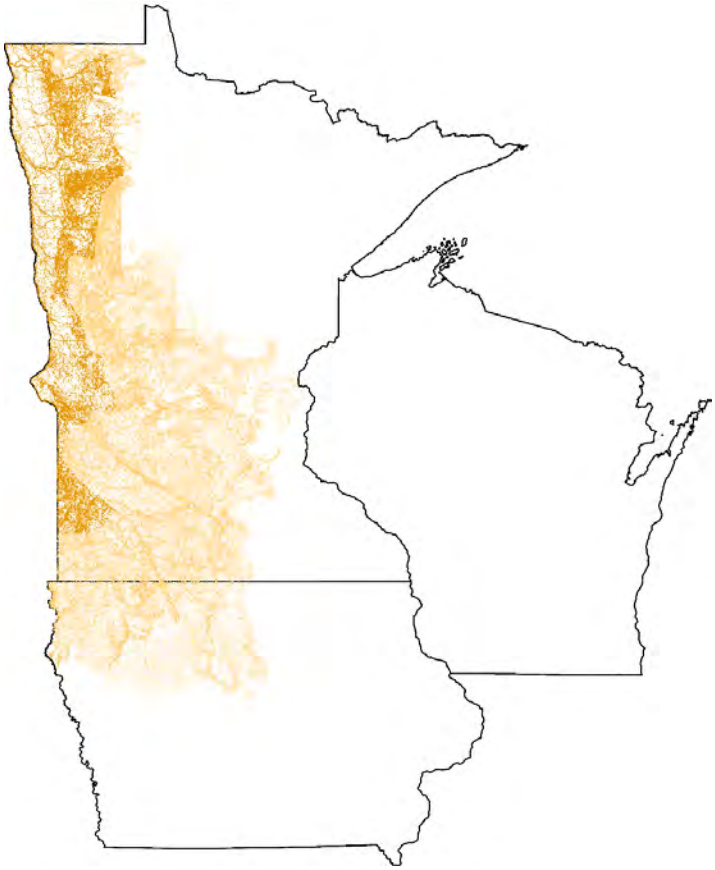

Figure 6. Northern Tallgrass Prairie habitat distribution. Geographic distribution of the Northern Tallgrass Prairie habitat (Group Code: G075) in the Midwest (Faber-Langendoen and others, 2020; USNVC, 2025). Dark orange areas indicate Northern Tallgrass Prairie habitat in the Northern Plains ecoregion, 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 Northern Tallgrass Prairie habitat is prioritized for the Northern 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 11.1°F increase by the end of the century under the high emissions scenario (RCP 8.5; Table 10) (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 no change (RCP 8.5; Table 11) (Marvel and others, 2023). However, precipitation is expected to become more variable, with more extreme events and notably drier summers despite the projected increase in total precipitation (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 10. Projected changes in mean annual temperature in the Northern 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 mean 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	41.6°F	43.6°F (2.0)	45.4°F (3.9)	44.3°F (2.8)	47.6°F (6.0)
Mean of models	41.4°F	46.7°F (5.3)	48.3°F (6.9)	47.8°F (6.4)	52.5°F (11.1)
Maximum model	41.5°F	48.3°F (6.8)	50.6°F (9.2)	50.1°F (8.6)	57.0°F (15.6)

Table 11. Projected changes in annual precipitation in the Northern 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 mean 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	22.6 in	22.8 in (1%)	24.2 in (7%)	23.5 in (4%)	22.7 in (0%)
Mean of models	23.1 in	24.5 in (6%)	24.5 in (6%)	24.6 in (7%)	25.2 in (9%)
Maximum model	23.4 in	26.6 in (14%)	26.1 in (11%)	25.8 in (10%)	28.3 in (21%)

Key Climate Change Stressors

Drought and seasonal shifts. Across the Northern Plains ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 69% compared to historical conditions by the end of the century (Table 12), with slightly smaller increases at only the northernmost latitudes (Figure 7). Higher VPD means more rapid evaporation and water loss from plants and soils, which can intensify drought conditions. The length of the growing season is also projected to increase by approximately 25% by the end of the century, translating to more than 37 additional days consecutive days with minimum temperatures above freezing (Table 13). These increases are expected to be generally consistent across the region, though some models project especially large increases at higher northern latitudes (Figure 8).

Table 12. Projected changes in summer vapor pressure deficit in the Northern 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 mean 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	1.04 kPa	0.90 kPa (-14%)	1.05 kPa (1%)	1.01 kPa (-3%)	1.03 kPa (0%)
Mean of models	1.04 kPa	1.31 kPa (27%)	1.43 kPa (38%)	1.40 kPa (35%)	1.75 kPa (69%)
Maximum model	1.07 kPa	1.51 kPa (41%)	1.68 kPa (56%)	1.75 kPa (63%)	2.35 kPa (119%)

Table 13. Projected changes in growing season length in the Northern 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 mean historical (1971–2000) and projected growing season length (consecutive days with minimum temperatures above freezing) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected growing season length under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	147.6	154.8 (5%)	160.6 (9%)	146.8 (-1%)	160.1 (8%)
Mean of models	149.2	168.9 (13%)	178.3 (20%)	165.7 (11%)	187.0 (25%)
Maximum model	150.6	196.5 (30%)	207.9 (38%)	192.0 (27%)	217.8 (45%)

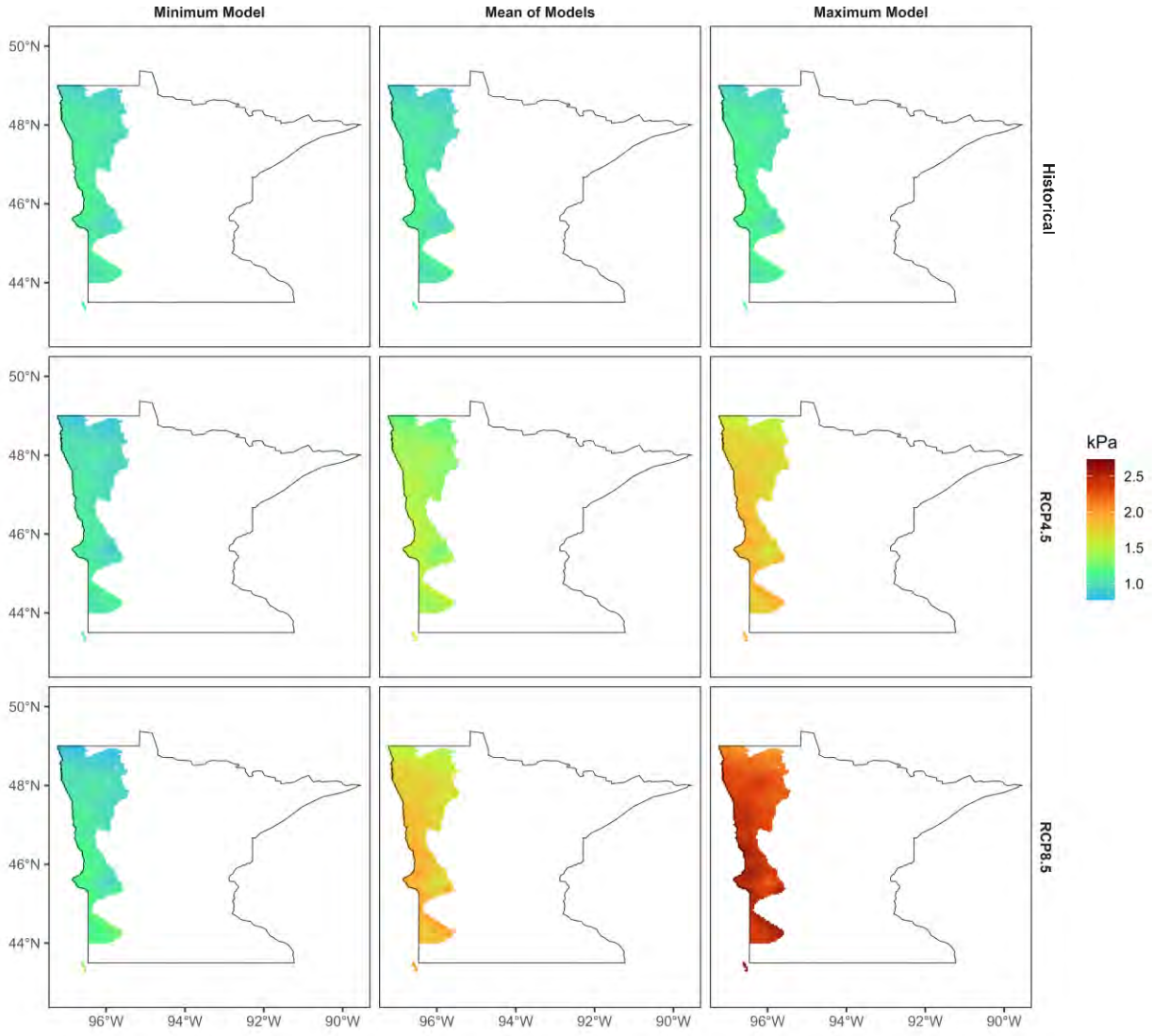

Figure 7. Projected changes in summer vapor pressure deficit in the Northern Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean 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).

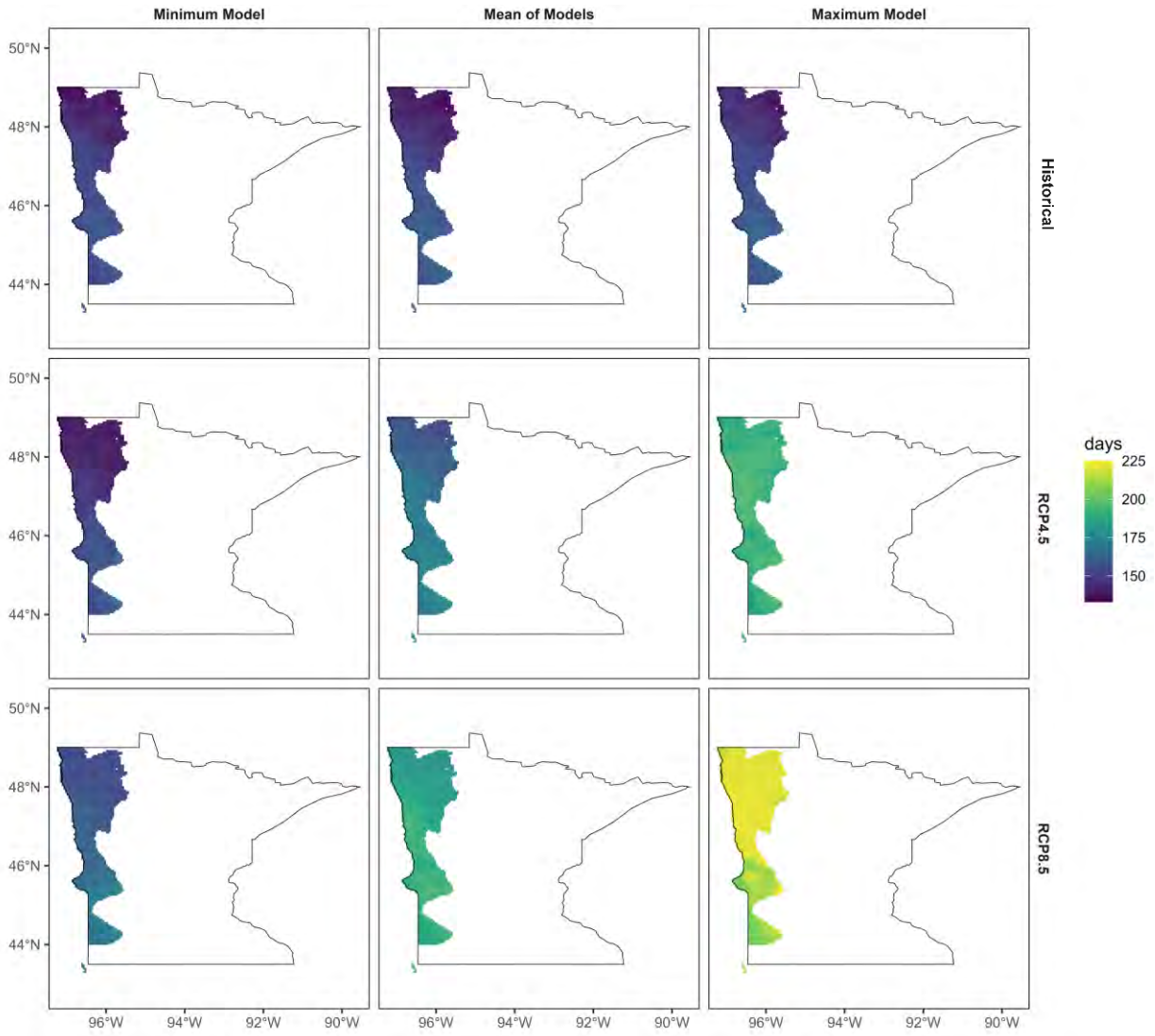

Figure 8. Projected changes in annual growing season length in the Northern Plains ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean historical (1971–2000, top row) and projected (2070–2099) annual growing season length (consecutive days with minimum temperatures above freezing) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected annual growing season length under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

This section summarizes how climate change is impacting the Northern Tallgrass Prairie habitat. It covers both the overall influence on species distributions, adaptive capacity, and vulnerability (Table 14) and the specific effects of drought and seasonal shifts (Table 15). The bullet points below provide a concise overview of these impacts on habitat structure, community composition, invasive species, herbivory, and microbial and fungal communities.

Habitat structure:

- Drought and increasingly variable precipitation will likely reduce soil moisture and plant cover, intensifying erosion, accelerating nutrient loss, and impairing long-term soil function (Castillioni and others, 2022b; Fay and others, 2011; Felton and others, 2020; Hanberry and others, 2019; Hoover and Rogers, 2016; Lemoine and others, 2024; Vose and others, 2016).
- Physiological stress from drought may suppress plant growth and increase the risk of mortality and reproductive failure, particularly during high-temperature events (Dreesen and others, 2015; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Smith and others, 2024; Vose and others, 2016; Wang and others, 2008; Will and others, 2013).
- Species vulnerability to drought varies by traits such as rooting depth and photosynthetic pathway (Barkaoui and Volaire, 2023; Hoover and others, 2019; Kazemi and others, 2023; Keen and others, 2024a; Nippert and Knapp, 2007; Padilla and others, 2019; Slette and others, 2023; Smith and others, 2024; Taylor and others, 2011; Wilcox and others, 2023; Zhou and others, 2022).
- Drought may reduce productivity and impair ecosystem functioning, especially in areas with high aridity or low species diversity (Briske, 2017; Fay and others, 2011; Jones, 2019; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Slette and others, 2023; Smith and others, 2024).
- Warming-driven shifts in seasonal timing are expected to alter flowering phenology and growing season dynamics (Austin and others, 2024; Dunnell and Travers, 2011; Geissler and others, 2023; Memmott and others, 2007; Ramirez-Parada and others, 2022; Sherry and others, 2007; Zettlemoyer and others, 2021).
- Phenological shifts associated with seasonal warming may boost productivity by extending the growing season but also increase vulnerability to late frosts (Allstadt and others, 2015; Briske, 2017; Cleland and Wolkovich, 2024; Fridley and others, 2016; Li and others, 2025; Zettlemoyer and others, 2021).

Community composition:

- Changing mean annual temperature and precipitation patterns may promote prairie-savanna expansion into areas currently dominated by the Northeastern Great Plains Aspen Woodland habitat (Group Code: G146) by reducing the sparse tree cover that characterizes this transitional system (Ahler and others, 2025; Galatowitsch and others, 2009; Havrilla and others, 2023; Reich and others, 2022; Toot and others, 2020).
- Droughts may lead to short- and long-term changes in dominant species by intensifying aboveground competition and weakening priority effects, favoring drought-tolerant species such as little bluestem (*Schizachyrium scoparium*) or gramas (*Bouteloua* spp.) over drought-sensitive species like quaking aspen (*Populus tremuloides*) (Table 15) (Baty and others, 2015; Dekirmenjian and others, 2024; Griffin-Nolan and others, 2021; Kooyers and others, 2025; Smith and others, 2017; Watson and others, 2021).
- Whether drought stress reinforces or weakens warm season (C_4) grass dominance over cool season (C_3) species may depend on drought severity and interacting factors, such as atmospheric CO_2 concentrations and precipitation variability (Ahler and others, 2025; Catford and Jones, 2019; Griffin-Nolan and others, 2021; Havrilla and others, 2023; O'Keefe and others, 2024; Polley and others, 2013).

- Functional groups such as deep-rooted shrubs and forbs, as well as species with drought avoidance or escape strategies, may increase in abundance as graminoids decline under drought (Table 15) (Dodds and others, 2023; Sadayappan and others, 2023; Smith and Goetz, 2021; Vose and others, 2016).
- Drought may facilitate woody encroachment and shrubland conversion by reducing herbaceous competition and drought-sensitive quaking aspen (*Populus tremuloides*) and favoring deep-rooted woody species, such as buckthorns (*Rhamnus* spp.) and oleasters (*Elaeagnus* spp.), that can access subsurface moisture, particularly in the absence effective fire regimes (Archer and others, 2017; Breshears and others, 2016b; Dodds and others, 2023; Keen and others, 2024; Sadayappan and others, 2023; Smith and Goetz, 2021; Volder and others, 2010; Vose and others, 2016).
- Seasonal shifts may favor early growing or phenologically responsive species, leading to gradual shifts in abundance as less flexible taxa decline (Table 15) (Cleland and Wolkovich, 2024; Dawson-Glass and others, 2025a; Lee and others, 2024; Ramirez-Parada and others, 2022; Wolkovich and Donahue, 2021; Zettlemoyer and others, 2021).
- Earlier and extended growing seasons may result in novel co-flowering patterns and disrupt synchrony between plant flowering and pollinator activity, increasing the risk of trophic mismatches that reduce reproductive success for species unable to align phenology with shifting seasonal conditions (Austin and others, 2024; Memmott and others, 2007; Ramirez-Parada and others, 2022; Whittington and others, 2015; Zettlemoyer and others, 2021).
- Woody plants may gain a competitive advantage under warming through extended winter activity and stronger phenological responsiveness, enhancing their establishment and long-term persistence (Ansley and others, 2023; Caterina and others, 2014; Geissler and others, 2023; Zou and others, 2014).

Invasive species:

- Climate change is expected to intensify invasive species pressure by favoring species with ruderal traits and high plasticity that can rapidly exploit warmer temperatures, variable precipitation, and resource availability following extreme events (Catford and Jones, 2019; Clements and Jones, 2021; Fan and others, 2013; Poland and others, 2021; Vose and others, 2016).
- Drought may promote invasion by reducing community resistance, weakening native competitors, and generating post-drought resource pulses that favor traits common to invasive grasses and forbs (Ali and Bucher, 2022; Catford and Jones, 2019; Duell and others, 2016; Duell and others, 2021; Jiménez and others, 2011LaForgia and others, 2020; Schrama and Bardgett, 2016).
- Invasive grasses with ruderal traits, such as rapid growth, high fecundity, and drought avoidance, may tolerate or recover from drought stress more effectively than many native species, leading to increased abundance (Table 15) (Braasch and others, 2025; Catford and Jones, 2019; Duell and others, 2021; Kirwan and others, 2021; Kroeger and Otfinowski, 2025; Schrama and Bardgett, 2016; Wolkovich and others, 2013).
- Warming-driven shifts in seasonality and longer growing seasons may advantage invasive species with early phenology and prolonged activity, such as smooth brome (*Bromus inermus*) and Kentucky bluegrass (*Poa pratensis*), allowing them to gain priority access to resources and suppress later-emerging natives (Table 15) (Ahler and others, 2025; Calinger and others, 2013; Catford and Jones, 2019; Dawson-Glass and others, 2025a; Dickson and others, 2012a; Keller and

Shea, 2021a; Sorte and others, 2013; Willis and others, 2010; Wilsey and others, 2015; Wolkovich and others, 2013; Zettlemoyer and others, 2019).

Herbivory:

- Drought and seasonal shifts may intensify herbivory pressure, particularly from expanding white-tailed deer (*Odocoileus virginianus*) populations, by weakening plant defenses, extending forage windows, and contributing to declines in native forb diversity (Anderson and others, 2001; Duell and others, 2024; Felton and others, 2024; Hamann and others, 2021; Nisi and others, 2015; Smith and Hall, 2016).

Microbial and fungal communities:

- Drought and seasonal shifts may drive fungal community reassembly and reduce the functional role of arbuscular mycorrhizal fungi in supporting plant drought tolerance (Fu and others, 2022; Hu and others, 2025; Lagueux and others, 2021; Ochoa-Hueso and others, 2021; Pehim Limbu and Avolio, 2025).
- Drought may reduce overall microbial activity and biomass, negatively impacting soil function and carbon cycling (Barnard and others, 2013; Ochoa-Hueso and others, 2018; Preece and others, 2019; Sheik and others, 2011).

Table 14. Summary of the expected general influence of climate change on Northern Tallgrass Prairie habitat species.
Asterisks (*) denote species introduced to the region. Some species appear in contradictory columns (e.g., showing both negative and positive influences on distribution), reflecting conflicting findings in the literature.

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and/or vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity/ high vulnerability (–)	High adaptive capacity/ low vulnerability (+)
Amorpha canescens[1]	*Amorpha canescens*[1]	*Bouteloua curtipendula*[7]	*Bouteloua gracilis*[9]
**Cirsium arvense*[2]	*Andropogon gerardii*[3]	*Populus tremuloides*[8]	*Quercus macrocarpa*[10–12]
Rudbeckia hirta[1]	*Bouteloua gracilis*[1]		
	Koeleria macrantha[1]		
	Quercus macrocarpa[4–6]		
	Schizachyrium scoparium[1]		

[1]Rana and others (2024); [2]Liu and others (2014); [3]Smith and others (2017); [4]Peters and others (2020) [5]Hanberry (2024); [6]Prasad and others (2020); [7]Brinker and others (2018); [8]Rogers and others (2016); [9]Molano-Flores and others (2019); [10]Etterson and others (2020); [11]Handler and others (2014); [12]Potter and others (2017)

Table 15. Summary of impacts from key climate change stressors on Northern Tallgrass Prairie 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, potential for competitive advantage and increased abundance	*Andropogon gerardii*[1–4]
		Amorpha canescens[5]
		**Berteroa incana*[6]
		Bouteloua curtipendula[7,8]
		Bouteloua gracilis[4,9]
		Bouteloua hirsuta[4,10]
		Calamovilfa longifolia[4,11–13]
		**Cirsium arvense*[14]
		Koeleria macrantha[4,15,16]
		Pascopyrum smithii[17]
		Quercus macrocarpa[4,18–20]
		Schizachyrium scoparium[4,13,21–23]
		**Securigera varia*[24]
		Sporobolus cryptandrus[4,26]
	When tolerance of droughts is low, potential for reduced growth rates and abundance	*Amorpha canescens*[27]
		Andropogon gerardii[21,22,28–32]
		Carex inops[33]
		Hesperostipa spartea[34]
		Muhlenbergia richardsonis[4,35]
		Pascopyrum smithii[36]
		Populus tremuloides[4,37–42]
		Quercus macrocarpa[43]
		Schizachyrium scoparium[44,45]
		**Trifolium pratense*[4]
Seasonal shifts	When phenological shifts are well-matched to changing seasonality, potential for competitive advantage	**Berteroa incana* [46]
		**Centaurea stoebe*[47]
		Cicuta maculata[46]
		Rudbeckia hirta[46,48]
		**Trifolium pratense*[49]
	When phenological shifts are poorly matched to changing seasonality, potential for reduced regeneration success and competitive capacity	*Solidago canadensis*[47]

[1]Baty and others (2015); [2]Nippert and others (2009); [3]Slette and others (2023); [4] USDA, NRCS (2025); [5]Casey (2011); [6]Noack and others (2020); [7]Chadwick (2003); [8]Engstrom (2004); [9]Anderson (2003); [10]Zlatnik (1999); [11]Duckwitz and Wynia (2006); [12]Hauser (2005); [13]USDA, NRCS (2004); [14]Hogenbirk and Wein (1991); [15]Simomin (2000a) ; [16]Goklany (2012); [17]Tirmenstein (1999); [18]Niinemets and Vallardares (2006); [19]Peters and others (2015); [20]USDA; [21]Dekirmenjian and others (2024); [22]Lemoine and others (2017); [23]Steinberg (2002); [24]Gucker (2009);; [26]Simomin (2000b); [27]Fryer (2011); [28]Kooyers and others (2025); [29]Maricle and others (2017); [30]Pehim Limbu and Avolio (2025);

[31]Uchytil (1988); [32]Watson and others (2021); [33]Fryer (2009a); [34]Walkup (1991a); [35]Sedivec and Barker (2002); [36]Duell and others (2021); [37]Gustafson and Sturtevant (2013); [38]Howard (1996); [39]Levesque and Hamann (2022); [40]Michaelian and others (2011); [41]Wang and others (2023); [42]Worrall and others (2013); [43]Dannenberg and others (2019); [44]Volder and others (2010); [45]Duell and others (2016); [46]Bertin (2017); [47]Zettlemoyer and others (2019); [48]Bertin (2017); [49]Abu-Asab and others (2001)

Focal Wildlife Species: Regal Fritillary

The regal fritillary (*Speyeria idalia*) is a butterfly found in grasslands of the east-central United States. This species may be especially vulnerable to climate change during the larval and pupal stages, when extreme weather events and temperature fluctuations can increase mortality (U.S. Fish and Wildlife Service, 2023). As the climate warms, the range of the regal fritillary may shift northward (Post van der Burg and others, 2023).

Larvae of the regal fritillary are specialists on native violet (*Viola* spp.), and their life cycle is closely synchronized with the growth and availability of these host plants (U.S. Fish and Wildlife Service, 2023). Habitat suitability depends on the presence and abundance of violets, though the specific species used varies across the fritillary's range (Illinois Department of Natural Resources, 2024). In addition to direct effects on the species, climate change may indirectly impact the regal fritillary through changes to its host plants (Henderson and others, 2018). Violets are sensitive to changes in moisture and temperature, and heavy rainfall can drown both violets and developing larvae (U.S. Fish and Wildlife Service, 2023). Moreover, if violets and fritillaries respond to different environmental cues (e.g., photoperiod vs. temperature), climate change could lead to phenological mismatches, where larvae emerge when violets are not at their most suitable stage for consumption (Reed and others, 2019; Singer and Parmesan, 2010).

Adult regal fritillaries are generalist nectar feeders and rely on a diversity of prairie wildflowers. Drought may reduce nectar availability for adults by suppressing flowering (U.S. Fish and Wildlife Service, 2023), while shifts in both temperature and rainfall may alter the timing of floral resources (Martén-Rodríguez and others, 2025). Additionally, climate change may promote the spread of non-native plants, which can outcompete both native nectar sources and violets (Barnes and others, 2024; Grant and others, 2009; Zettlemoyer and others, 2019). Maintaining a diverse floral community is therefore critical for adult foraging success. Prescribed fire can be a useful management tool to promote both violet abundance and flowering plant diversity (Henderson and others, 2018). However, regal fritillaries are sensitive to fire, so burns may be applied strategically at a scale, frequency, and timing that allow for post-fire recovery (McCullough and others, 2021).

Focal Aquatic System: Otter Tail River

The Otter Tail River, a major tributary of the Red River of the North, flows approximately 192 miles from Elbow Lake near Detroit Lakes to its confluence with the Bois de Sioux River in Breckenridge (Minnesota Pollution Control Agency, 2019). The river, which fosters a warmwater aquatic community, crosses a diverse landscape that transitions from intact forests and wetlands in the north to intensive agriculture and urban development in the south. This land-use gradient has led to increased habitat fragmentation, particularly in downstream reaches where channel straightening, agricultural drainage, and impoundments have altered channel form, reduced habitat heterogeneity, and impaired ecological function (Minnesota Pollution Control Agency, 2019).

Since 1993, the Otter Tail River watershed has experienced pronounced climate-driven hydrologic changes, including a 12% increase in annual precipitation, a 122% increase in stream discharge, and a 75% increase in bankfull flows (Minnesota Department of Natural Resources, 2023). Flow variability has also increased, with high-pulse events, measured as the number of annual flows that exceed the 75th percentile of historical daily discharge, now 150% more frequent, and minimum flows substantially higher than pre-1993 levels (Minnesota Department of Natural Resources, 2023). These patterns are consistent with basin-wide trends across the Red River watershed, where rising annual flows and reduced low-flow frequency point to increasing hydrologic instability (Novotny and Stefan, 2007).

Future projections suggest these trends will continue (Table 16). The U.S. Army Corps of Engineers' Climate Hydrology Assessment Tool (U.S. Army Corps of Engineers, 2023) indicates that streamflow magnitudes in the Otter Tail River are expected to rise through the end of the century (Minnesota Department of Natural Resources, 2023). Moreover, broader climate assessments project earlier runoff, more frequent and intense rainfall events, and reduced snowpack in northern watersheds (Scholl and others, 2025; Williams-Sether and Sanocki, 2025). Although the region has experienced wetter conditions since the 1990s, future drought risk remains high. National assessments indicate that rising temperatures will increase evaporative demand, contributing to seasonal shifts across the Upper Midwest, namely, wetter springs followed by drier summers (Scholl and others, 2025). These changes may exacerbate soil moisture deficits, particularly in agricultural areas where legacy drainage accelerates groundwater depletion (Adelsperger and others, 2024).

Table 16. Projected changes in streamflow for Otter Tail River. Projections use gridded simulations of daily runoff based on CMIP5 climate models under the high-emissions (RCP 8.5) scenario. Values show mean 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	244.3 ft^3/s	310.1 ft^3/s	27%
Mean Spring (March–May) Flow	351.3 ft^3/s	485.2 ft^3/s	38%
Mean Summer (June–August) Flow	376.4 ft^3/s	392.8 ft^3/s	4%
Mean Fall (September–November) Flow	322.4 ft^3/s	346.8 ft^3/s	8%
Mean Annual Flow	324.0 ft^3/s	384.2 ft^3/s	19%
1.5-year Flood (33rd percentile)	1821.0 ft^3/s	1975.4 ft^3/s	8%
10-year Flood (90th percentile)	4404.9 ft^3/s	5541.6 ft^3/s	26%
25-year Flood (96th percentile)	5374.9 ft^3/s	7917.8 ft^3/s	47%
Number of Winter Floods	1.6	2.7	70%
Average Lowest 7-day Flow	172.5 ft^3/s	171.6 ft^3/s	-1%
Center of Flow Mass Timing	April 14	April 8	-3%

Central Interior Acidic Open Glade and Barrens

The Central Interior Acidic Open Glade and Barrens habitat encompasses a suite of sparsely vegetated plant communities that occur on thin, drought-prone soils overlying acidic bedrock such as sandstone, shale, chert, or igneous substrates (Faber-Langendoen and others, 2020; Heikens, 1999; USNVC, 2025). These systems are distributed throughout the Interior Highlands of the Ozarks and adjacent regions (Figure 9), often on south- or west-facing slopes where sun exposure intensifies moisture stress (Erickson and others, 1942; Heikens, 1999; USNVC, 2025; Ware, 2002). Soils are shallow, ranging from exposed bedrock to pockets less than 40 cm deep, and highly permeable, limiting water retention and plant rooting depth (Heikens, 1999; Kucera and Martin, 1957; Ware, 2002). These habitats support a diverse array of xeric and prairie-associated species not found in surrounding forests, including several regional endemics, such as limestone glade bladderpod (*Physaria filiformis*) and Bush's skullcap (*Scutellaria bushii*) (Heikens, 1999; Ware, 2002). Vegetation is characterized by the dominance of 28 species, comprising warm-season (C_4) grasses, low-stature forbs, lichens, mosses, and sparse, stunted shrubs (Table 17) (Heikens, 1999; Heikens and Robertson, 1995; USNVC, 2025). Plant distribution tends to be patchy, reflecting microtopographic variability in soil depth and exposure, forming a mosaic of exposed rock, bare soil, and vegetated pockets (Ware, 2002).

These habitats are primarily maintained by a combination of fire, drought, and other disturbances such as browsing, which collectively limit forest succession and maintain open structure (Anderson and Bowles, 1999; Erickson and others, 1942; Faber-Langendoen and others, 2020; Heikens, 1999; Heikens and Robertson, 1994; USNVC, 2025). Historically, frequent low-intensity fires, both natural and anthropogenic, suppressed woody encroachment and reinforced herbaceous dominance. However, active fire suppression and lengthened fire return intervals over recent decades have contributed to canopy closure and a decline in herbaceous plant diversity in many glades and barrens (Cutter and Guyette, 1994; Heikens, 1999; Ware, 2002).

Today, these communities persist mostly as fragmented remnants on harsh sites resistant to intensive land use (Heikens, 1999; Heikens and Robertson, 1995). Their persistence is increasingly threatened by woody encroachment, invasive species, and legacy impacts from overgrazing and habitat conversion (Baskin and Baskin, 2000; Heikens, 1999; USNVC, 2025).

Table 17. Plant species common to the Central Interior Acidic Open Glade and Barrens 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
Ambrosia bidentata Michx.	Lanceleaf ragweed	Forb/herb
Andropogon virginicus L.	Broomsedge bluestem	Graminoid
Aristida dichotoma Michx.	Churchmouse threeawn	Graminoid
Carex pensylvanica Lam.	Pennsylvania sedge	Graminoid
Clitoria mariana L.	Atlantic pigeonwings	Forb/herb, vine
Croton michauxii G.L. Webster	Michaux's croton	Forb/herb
Danthonia spicata (L.) P. Beauv. ex Roem. & Schult.	Poverty oatgrass	Graminoid
Dichanthelium spp. (Hitchc. & Chase) Gould	Rosette grass	Graminoid

Species	Common name	Growth habit
Diodia teres Walter	Poorjoe	Forb/herb
Koeleria macrantha (Ledeb.) Schult.	Prairie Junegrass	Graminoid
Liatris squarrosa (L.) Michx.	Scaly blazing star	Forb/herb
Rudbeckia hirta L.	Blackeyed Susan	Forb/herb
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid
**Securigera varia* (L.) Lassen	Crownvetch	Forb/herb, vine
Selaginella rupestris (L.) Spring	Northern selaginella	Forb/herb
Solidago juncea Aiton	Early goldenrod	Forb/herb
Sorghastrum nutans (L.) Nash	Indiangrass	Graminoid
Sporobolus clandestinus (Biehler) Hitchc.	Rough dropseed	Graminoid
Sporobolus neglectus Nash	Puffsheath dropseed	Graminoid
Symphyotrichum ericoides (L.) G.L. Nesom	White heath aster	Forb/herb
**Elaeagnus umbellata* Thunb.	Autumn olive	Shrub
Juniperus virginiana L.	Eastern redcedar	Tree
**Populus alba* L.	White poplar	Tree
*Quercus marilandi*ca Münchh.	Blackjack oak	Shrub, tree
Quercus montana Willd.	Chestnut oak	Tree
*Quercus stella*ta Wangenh.	Post oak	Tree
**Rosa multiflora* Thunb.	Multiflora rose	Shrub, vine
Smilax bona-nox L.	Saw greenbrier	Shrub, vine

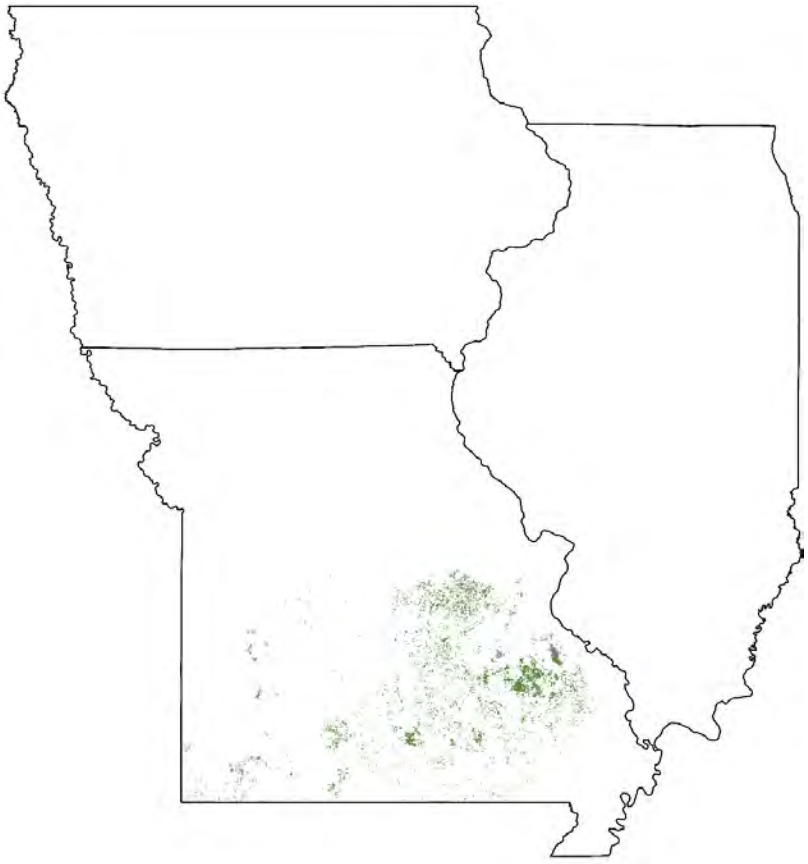

Figure 9. Central Interior Acidic Open Glade and Barrens habitat distribution. Current geographic distribution of the Central Interior Acidic Open Glade and Barrens habitat (Group Code: G178) in the Midwest (Faber-Langendoen and others, 2020; USNVC, 2025). Dark green areas indicate Central Interior Acidic Open Glade and Barrens 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 Central Interior Acidic Open Glade and Barrens 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 18) (Clarke and others, 2022; Marvel and others, 2023; Masson-Delmotte and others, 2021). Most models also project a moderate increase in annual precipitation, averaging 1.8 additional inches per year, although some indicate a slight decline in the high emission scenario (Table 19) (Marvel and others, 2023). However, precipitation is expected to become more variable, with more extreme events and notably drier summers despite the overall increase in total precipitation (Dollan and others, 2022; Marvel and others, 2023; Polasky and others, 2021).

Table 18. Projected changes in mean 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 mean 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 19. 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 mean 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	44.5 in	46.9 in (5%)	48.0 in (8%)	49.6 in (12%)	50.1 in (13%)

Key Climate Change Stressors

Drought and seasonal shifts. Across the Ozarks ecoregion, summer vapor pressure deficit (VPD) is projected to increase by approximately 86% compared to historical conditions by the end of the century (RCP 8.5; Table 20), with increases generally consistent across the region (Figure 10). Higher VPD means more rapid evaporation and water loss from plants and soils, which can intensify drought conditions. The length of the growing season is also projected to increase by approximately 21% by the end of the century under the high emission scenario, translating to more than 40 additional days consecutive days with minimum temperatures above freezing (Table 21). These increases are expected to be generally consistent across the region (Figure 11).

Table 20. 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 mean 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	1.26 kPa	1.65 kPa (31%)	1.86 kPa (48%)	1.74 kPa (38%)	2.34 kPa (86%)
Maximum model	1.23 kPa	1.97 kPa (60%)	2.12 kPa (72%)	2.31 kPa (87%)	3.32 kPa (169%)

Table 21. Projected changes in growing season length 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 mean historical (1971–2000) and projected growing season length (consecutive days with minimum temperatures above freezing) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected growing season length under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	189.6	201.9 (6%)	208.5 (10%)	194.4 (2%)	209.7 (11%)
Mean of models	191.4	214.6 (12%)	222.6 (16%)	210.7 (10%)	231.6 (21%)
Maximum model	192.4	231.2 (20%)	244.3 (27%)	225.7 (17%)	257.4 (34%)

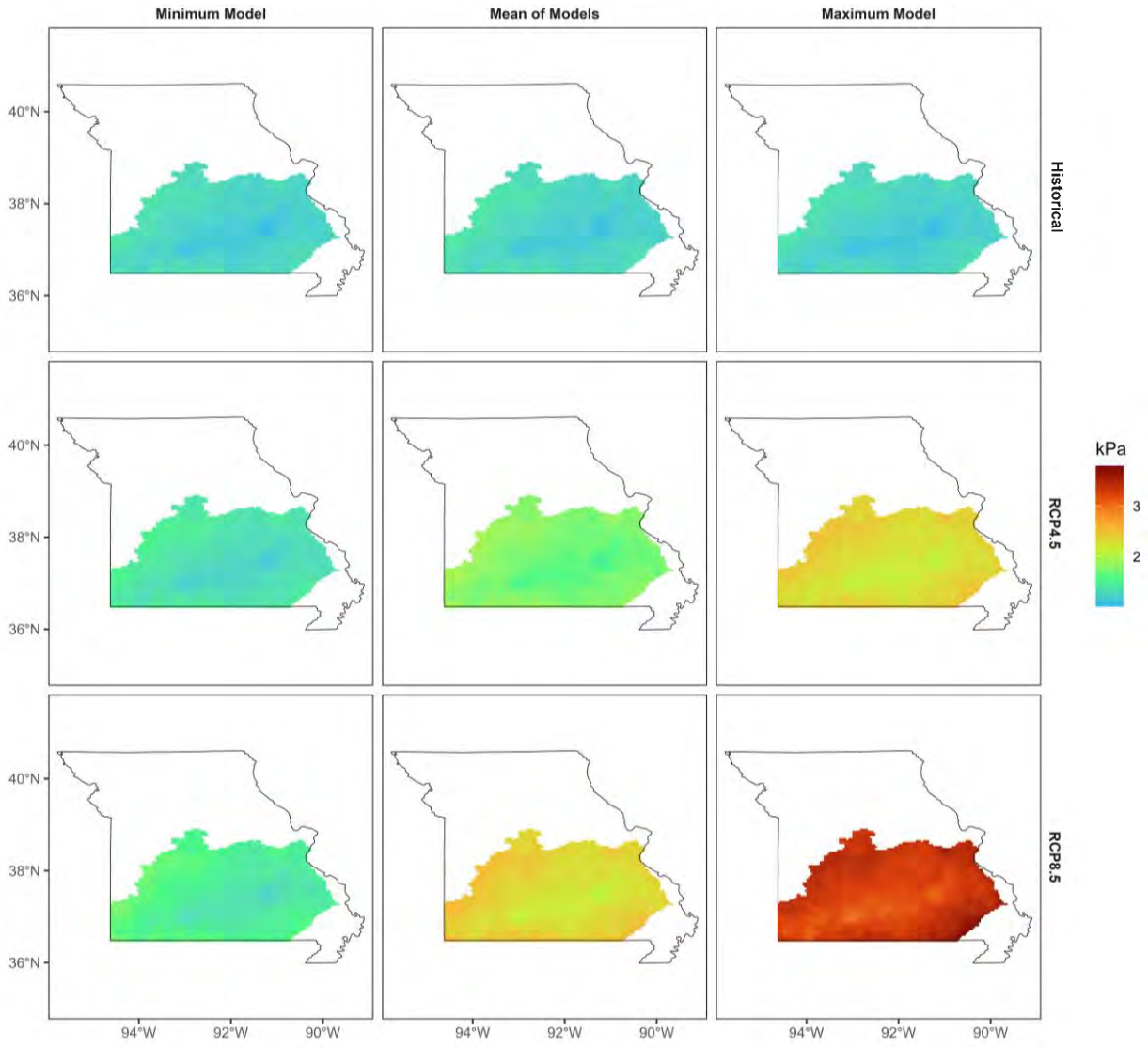

Figure 10. Projected changes in summer vapor pressure deficit in the Ozarks ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean 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).

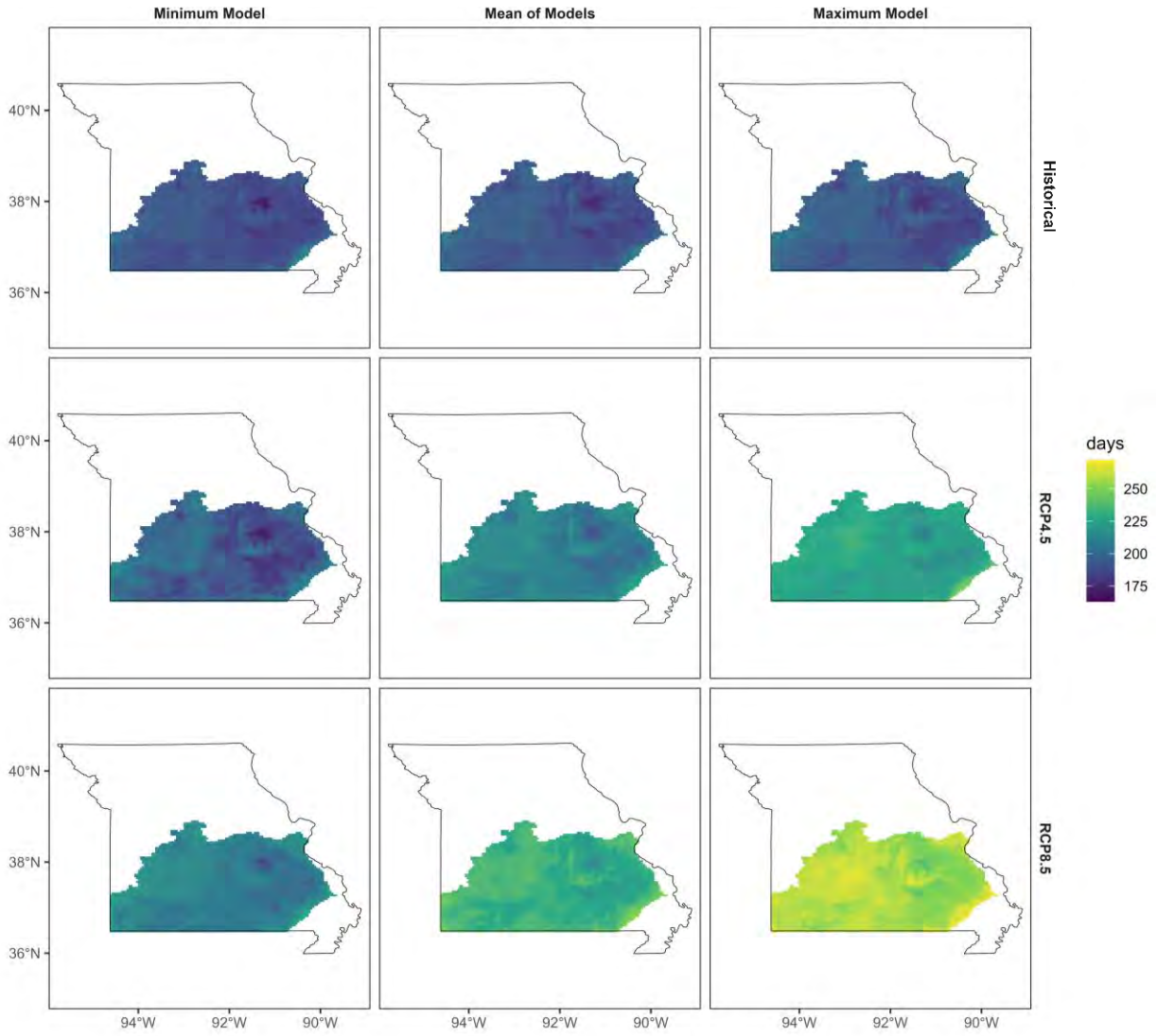

Figure 11. Projected changes in annual growing season length in the Ozarks ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean historical (1971–2000, top row) and projected (2070–2099) annual growing season length (consecutive days with minimum temperatures above freezing) for each emissions scenario (RCP 4.5, middle row; RCP 8.5, bottom row). To capture variation across climate models, models with the lowest (minimum model, left column) and highest (maximum model, right column) projected annual growing season length under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

This section summarizes how climate change is impacting the Central Interior Acidic Open Glade and Barrens. It covers both the overall influence on species distributions, adaptive capacity, and vulnerability (Table 22) and the specific effects of drought and seasonal shifts (Table 23). The bullet points below provide a concise overview of these impacts on habitat structure, community composition, invasive species, herbivory, and microbial and fungal communities.

Habitat structure:

- Drought and increasingly variable precipitation will likely reduce soil moisture and plant cover, intensifying erosion, accelerating nutrient loss, and impairing long-term soil function (Castillioni and others, 2022b; Fay and others, 2011; Felton and others, 2020; Hanberry and others, 2019; Hoover and Rogers, 2016; Lemoine and others, 2024; Vose and others, 2016).
- Physiological stress from drought may suppress plant growth and increase the risk of mortality and reproductive failure, particularly during high-temperature events (Dreesen and others, 2015; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Smith and others, 2024; Vose and others, 2016; Wang and others, 2008; Will and others, 2013).
- Drought may reduce productivity and impair ecosystem functioning, especially in areas with high aridity or low species diversity (Briske, 2017; Fay and others, 2011; Jones, 2019; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Slette and others, 2023; Smith and others, 2024).
- Warming-driven shifts in seasonal timing are expected to alter flowering phenology and growing season dynamics (Austin and others, 2024; Dunnell and Travers, 2011; Geissler and others, 2023; Memmott and others, 2007; Ramirez-Parada and others, 2022; Sherry and others, 2007; Zettlemoyer and others, 2021).
- Phenological shifts associated with seasonal warming may boost productivity by extending the growing season but also increase plant vulnerability to late frosts (Allstadt and others, 2015; Briske, 2017; Cleland and Wolkovich, 2024; Fridley and others, 2016; Li and others, 2025 Zettlemoyer and others, 2021).

Community composition:

- Droughts may reinforce xeric species composition by intensifying seasonal soil moisture depletion, leading to declines in more mesic species such as Indiangrass (*Sorghastrum nutans*) (Table 23) (Baty and others, 2015; Dekirmenjian and others, 2024; Griffin-Nolan and others, 2021; Kooyers and others, 2025; Smith and others, 2017; Watson and others, 2021).
- Whether drought stress reinforces or weakens warm season (C_4) grass dominance over cool season (C_3) species may depend on drought severity and interacting factors, such as atmospheric CO_2 concentrations and precipitation variability (Ahler and others, 2025; Catford and Jones, 2019; Griffin-Nolan and others, 2021; Havrilla and others, 2023; O'Keefe and others, 2024; Polley and others, 2013).
- Low water retention capacity, combined with drought, may facilitate woody encroachment by reducing herbaceous competition while simultaneously hindering encroachment of mesic forest species, promoting shrubby expansion and savannafication while inhibiting transition to closed-canopy forest (Archer and others, 2017; Breshears and others, 2016b; Dodds and others, 2023; Keen and others, 2024; Sadayappan and others, 2023; Smith and Goetz, 2021; Volder and others, 2010; Vose and others, 2016).
- Seasonal shifts may favor early growing or phenologically responsive species, leading to gradual shifts in abundance as less flexible taxa decline (Table 23) (Cleland and Wolkovich, 2024; Dawson-Glass and others, 2025a; Lee and others, 2024; Ramirez-Parada and others, 2022; Wolkovich and Donahue, 2021; Zettlemoyer and others, 2021).
- Earlier and extended growing seasons may result in novel co-flowering patterns and disrupt synchrony between plant flowering and pollinator activity, increasing the risk of trophic

mismatches that reduce reproductive success for species unable to align phenology with seasonal shiftings (Austin and others, 2024; Memmott and others, 2007; Ramirez-Parada and others, 2022; Whittington and others, 2015; Zettlemoyer and others, 2021).

Invasive species:

- Climate change is expected to intensify invasive species pressure by favoring species with ruderal traits and high plasticity that can rapidly exploit warmer temperatures, variable precipitation, and resource availability following extreme events (Catford and Jones, 2019; Clements and Jones, 2021; Fan and others, 2013; Poland and others, 2021; Vose and others, 2016).
- Invasive grasses with ruderal traits, such as rapid growth, high fecundity, and drought avoidance, may tolerate or recover from drought stress more effectively than many native species, leading to increased abundance (Table 23) (Braasch and others, 2025; Catford and Jones, 2019; Duell and others, 2021; Kirwan and others, 2021; Kroeger and Otfinowski, 2025; Schrama and Bardgett, 2016; Wolkovich and others, 2013).

Herbivory:

- Climate-driven drought and seasonal shifts may intensify herbivory pressure, particularly from increasing white-tailed deer (*Odocoileus virginianus*) populations, potentially contributing to declines in native forb diversity while simultaneously limiting woody encroachment (Anderson and others, 2001; Duell and others, 2024; Felton and others, 2024; Hamann and others, 2021; Nisi and others, 2015; Smith and Hall, 2016).

Microbial and fungal communities:

- Drought and seasonal shifts may drive fungal community reassembly and reduce the functional role of arbuscular mycorrhizal fungi in supporting plant drought tolerance (Fu and others, 2022; Hu and others, 2025; Lagueux and others, 2021; Ochoa-Hueso and others, 2021; Pehim Limbu and Avolio, 2025).
- Drought may reduce overall microbial activity and biomass, negatively impacting soil function and carbon cycling (Barnard and others, 2013; Ochoa-Hueso and others, 2018; Preece and others, 2019; Sheik and others, 2011).

Table 22. Summary of the expected general influence of climate change on Central Interior Acidic Open Glade and Barrens species. Asterisks (*) denote species introduced to the region.

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and/or vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity/ high vulnerability (–)	High adaptive capacity/ low vulnerability (+)
Rudbeckia hirta[1]	*Koeleria macrantha*[1]		**Rosa multiflora*[4]
	Quercus marilandica[2]		
	Rosa multiflora[3]		
	Schizachyrium scoparium[1]		

[1]Rana and others (2024); [2]Rauschendorfer and others (2022); [3]Reinhardt and others (2020); [4]Mortsch and others (2006)

Table 23. Summary of impacts from key climate change stressors on Central Interior Acidic Open Glade and Barrens 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, potential for competitive advantage and increased abundance	*Andropogon virginicus*[1]
		Juniperus virginiana[1–4]
		Koeleria macrantha[1,5,6]
		Quercus marilandica[1]
		Quercus stellata[1,7–9]
		Schizachyrium scoparium[1,10–13]
		**Securigera varia*[14]
	When tolerance of droughts is low, potential for reduced growth rates and abundance	*Carex pensylvanica* [15]
		Juniperus virginiana[14]
		Quercus marilandica[16]
		Quercus stellata[4,17–19]
		Schizachyrium scoparium[4,20]
		Sorghastrum nutans[11,21–24]
Seasonal shifts	When phenological shifts are well-matched to changing seasonality, potential for competitive advantage	**Populus alba*[25]
		Rudbeckia hirta[26]
		Solidago juncea[26]

[1]USDA, NRCS (2025); [2]Caterina and others (2014); [3]Torquato and others (2020); [4]Volder and others (2010); [5]Simomin (2000a); [6] Goklany (2012); [7]Carey (1992); [8]Niinemets and Vallardares (2006); [9]Pallardy and Rhoads (1993); [10]Dekirmenjian and others (2024); [11]Lemoine and others (2017); [12]Steinberg (2002); [13]USDA, NRCS (2004); [14]Gucker (2009); [15]Nelson and others (2018); [16]Matheus (2021); [17]Dannenberg and others (2019); [18]Novick and others (2022); [19]Prolic and Goldblum (2016); [20]Duell and others (2016); [21]Hoover and others (2014); [22]Kazemi and others (2023); [23]Swemmer and others (2006); [24]Walkup (1991b); [25]Miller and others (Miller and others, 2023); [26]Bertin (2017)

Focal Wildlife Species: Eastern Collared Lizard

The eastern collared lizard (*Crotaphytus collaris*) is distributed across the south-central United States in hot, arid environments (NatureServe Explorer, 2025). In the Midwest, it reaches its northern range limit in the glade habitats of the Ozark Highlands. These open, rocky clearings provide the sun exposure and thermal conditions necessary for activity and reproduction.

Eastern collared lizards are ectothermic and sensitive to rising temperatures, which can have both beneficial and detrimental effects. In the cooler parts of their range, such as the Midwest, warming may initially increase energy assimilation and feeding rates (Brewster and others, 2020). However, higher body temperatures also carry metabolic costs and can lead to mortality if cool refuges are not available (Adams and Gifford, 2025). To avoid overheating, lizards retreat to shaded microhabitats or pant to dissipate heat, both behaviors that incur energetic costs. Panting, while effective for cooling, elevates metabolic rates and increases water loss (Loughran and Wolf, 2023). Additionally, eastern collared lizards

exhibit temperature-dependent sex determination, with warmer incubation conditions producing disproportionately more female offspring, potentially altering population sex ratios (Santoyo-Brito and others, 2017). As generalist predators, eastern collared lizards consume a variety of invertebrates and small vertebrates (Ruppert, 1980), and are preyed upon by snakes, birds, and mammals (Myers and others, 2025). Shifts in prey and predator abundance or availability, mediated by climate change, may influence the behavioral trade-offs eastern collared lizards face between maintaining optimal body temperatures, securing food resources, and avoiding predation.

The impacts of climate change on habitat structure represent an additional threat to eastern collared lizards. Fire suppression has facilitated woody plant encroachment into glade habitats and has been linked to reduced growth and reproduction in eastern collared lizards, contributing to observed population declines (Brewster and others, 2020). Maintaining open glade structure and preventing woody encroachment will be key to ensuring that eastern collared lizards retain access to the thermal environments necessary for healthy physiological function (Brewster and others, 2021).

Eastern North American Ruderal Meadow and Shrubland

The Eastern North American Ruderal Meadow and Shrubland habitat encompasses a range of early-successional communities that emerge on lands cleared by human activity and later abandoned or subject to intermittent management. These habitats are widespread throughout the Midwest (Figure 12), commonly occupying former agricultural fields, development sites, utility corridors, and other heavily disturbed areas (USNVC, 2025). Vegetation is typically dominated by a mix of 36 species, including introduced grasses, forbs, and shrubs, as well as weedy native species tolerant of fluctuating soil moisture, elevated nutrient availability, and frequent disturbance (Table 25). These communities lack the floristic integrity of native grasslands and shrublands, instead exhibiting low species diversity, high dominance by generalists, and compositional instability (Faber-Langendoen and others, 2020; USNVC, 2025).

These habitats are highly dynamic and shaped by ongoing interactions among disturbance, colonization pressure, and management. Initial site history, including prior land use, herbicide application, and proximity to seed sources, can strongly influence community composition and successional trajectories (Gross and Emery, 2007). Some sites may persist for decades in an open, herbaceous-dominated state, while others transition rapidly to woody dominance, depending on disturbance frequency, propagule pressure, and soil characteristics (Gross and Emery, 2007). Efforts to restore these habitats to native grassland or savanna conditions often face substantial barriers, including entrenched dominance of introduced species, altered nutrient cycling, and the loss of conservative native flora (Gross and Emery, 2007; USNVC, 2025).

Table 25. Plant species common to the Eastern North American Ruderal Meadow and Shrubland 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
Ambrosia artemisiifolia L.	Annual ragweed	Forb/herb
**Anthoxanthum odoratum* Boiss.	Annual vernalgrass	Graminoid
**Arctium minus* Bernh.	Lesser burdock	Forb/herb
**Berteroa incana* (L.) DC.	Hoary alyssum	Forb/herb
**Bromus inermis* Leyss.	Smooth brome	Graminoid
**Centaurea stoebe* L.	Spotted knapweed	Forb/herb
**Cirsium arvense* (L.) Scop.	Canada thistle	Forb/herb
Conyza canadensis (L.) Cronquist	Canadian horseweed	Forb/herb
**Daucus carota* L.	Queen Anne's lace	Forb/herb
†*Digitaria* spp. Haller	Crabgrass	Graminoid
**Elymus repens* (L.) Gould	Quackgrass	Graminoid
**Leucanthemum vulgare* Lam.	Oxeye daisy	Forb/herb
**Melilotus officinalis* (L.) Lam.	Sweetclover	Forb/herb
**Phleum pratense* L.	Timothy	Graminoid
**Poa pratensis* L.	Kentucky bluegrass	Graminoid

Species	Common name	Growth habit
Rudbeckia hirta L.	Blackeyed Susan	Forb/herb
**Securigera varia* (L.) Lassen	Crownvetch	Forb/herb, vine
†*Setaria* spp. P. Beauv.	Bristlegrass	Graminoid
**Setaria viridis* (L.) P. Beauv.	Green bristlegrass	Graminoid
Solidago canadensis L.	Canada goldenrod	Forb/herb
Cornus drummondii C.A. Mey.	Roughleaf dogwood	Shrub
Corylus americana Walter	American hazelnut	Shrub
Corylus cornuta Marshall	Beaked hazelnut	Shrub, tree
†*Crataegus* spp. L.	Hawthorn	Shrub, tree
**Elaeagnus umbellata* Thunb.	Autumn olive	Shrub
**Lonicera japonica* Thunb.	Japanese honeysuckle	Vine
†*Lonicera* spp. L.	Honeysuckle	Shrub, vine
**Lonicera tatarica* L.	Tatarian honeysuckle	Shrub
**Lonicera* × *bella* Zabel [*morrowii* × *tatarica*]	Showy fly honeysuckle	Shrub
Populus tremuloides Michx.	Quaking aspen	Tree
**Rhamnus cathartica* L.	Common buckthorn	Shrub, tree
Rhus glabra L.	Smooth sumac	Shrub, tree
Rhus typhina L.	Staghorn sumac	Shrub, tree
**Rosa multiflora* Thunb.	Multiflora rose	Shrub, vine
†*Rubus* spp. L.	Blackberry	Shrub, vine
Zanthoxylum americanum Mill.	Common pricklyash	Shrub, tree

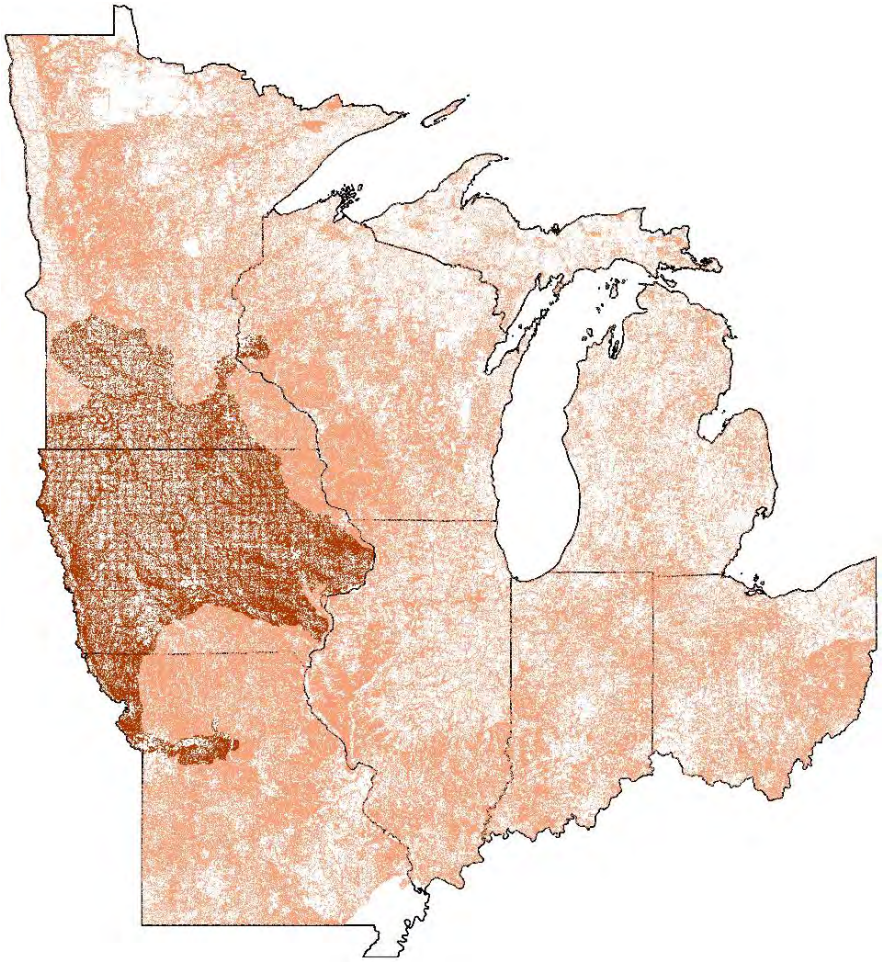

Figure 12. Eastern North American Ruderal Meadow and Shrubland habitat distribution. Current geographic distribution of the Eastern North American Ruderal Meadow and Shrubland habitat (Group Code: G609) in the Midwest (Faber-Langendoen and others, 2020; USNVC, 2025). Dark orange areas indicate Eastern North American Ruderal Meadow and Shrubland habitat in the Temperate Prairie 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 Eastern North American Ruderal Meadow and Shrubland habitat is prioritized for the Temperate Prairie 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 10.3°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 2.4 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 mean annual temperature in the Temperate Prairie 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 mean 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	47.9°F	50.2°F (2.4)	51.2°F (3.3)	51.1°F (3.2)	53.7°F (5.8)
Mean of models	47.9°F	52.9°F (4.9)	54.4°F (6.5)	53.9°F (6.0)	58.3°F (10.3)
Maximum model	47.8°F	54.3°F (6.4)	56.4°F (8.6)	56.3°F (8.4)	62.4°F (14.6)

Table 27. Projected changes in annual precipitation in the Temperate Prairie 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 mean 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	33.5 in	33.7 in (1%)	31.8 in (-5%)	36.2 in (8%)	30.3 in (-9%)
Mean of models	32.8 in	34.5 in (5%)	34.6 in (6%)	34.3 in (5%)	35.2 in (7%)
Maximum model	34.5 in	36.4 in (6%)	36.6 in (6%)	35.4 in (3%)	38.4 in (11%)

Key Climate Change Stressors

Drought and seasonal shifts. Across the Temperate Prairie ecoregion, 30-year extreme drought incidence is projected to increase by approximately 777% compared to historical conditions by the end of the century, with 30.0 months of extreme drought expected within a 30-year period (RCP 8.5; Table 28). These increases are expected to be generally consistent across the region, though some models project especially large increases in the northwest (Figure 13). This dramatic increase in extreme drought frequency reflects a severe moisture deficit. The last spring freeze day is projected to advance by approximately 16% by the end of the century under the high emissions scenario, meaning the last day after January 1 with a minimum temperature below freezing would occur about 18 days earlier (Table 29). These advances are expected to be most pronounced in Missouri (Figure 14).

Table 28. Projected changes in 30-year extreme drought incidence in the Temperate Prairie ecoregion. Projections use Standardized Precipitation-Evapotranspiration Index (SPEI) data 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 mean historical (1971–2000) and projected 30-year extreme drought incidence (months) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected 30-year extreme drought incidence under RCP 8.5 are presented in addition to the mean of all models.

	1971–2000	2040–2069		2070–2099	
	Historical	RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
Minimum model	6.4	8.3 (30%)	6.9 (8%)	13.4 (109%)	13.3 (108%)
Mean of models	3.8	14.8 (294%)	17.2 (356%)	19.0 (405%)	33.0 (777%)
Maximum model	1.0	33.5 (3132%)	28.5 (2647%)	30.1 (2800%)	74.2 (7052%)

Table 29. Projected changes in last spring freeze day in the Temperate Prairie 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 mean historical (1971–2000) and projected last spring freeze day (the last day after January 1 with a minimum temperature below freezing) by mid- (2040–2069) and end of the century (2070–2099), as well as the percent change relative to the historical value (in parentheses), for each emissions scenario. To capture variation across climate models, models with the lowest (minimum model) and highest (maximum model) projected last spring freeze day 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	106.5	84.7 (-20%)	76.6 (-28%)	76.8 (-28%)	70.0 (-34%)
Mean of models	109.8	101.7 (-7%)	98.4 (-10%)	99.1 (-10%)	91.8 (-16%)
Maximum model	111.9	101.7 (-9%)	100.4 (-10%)	100.0 (-11%)	101.3 (-9%)

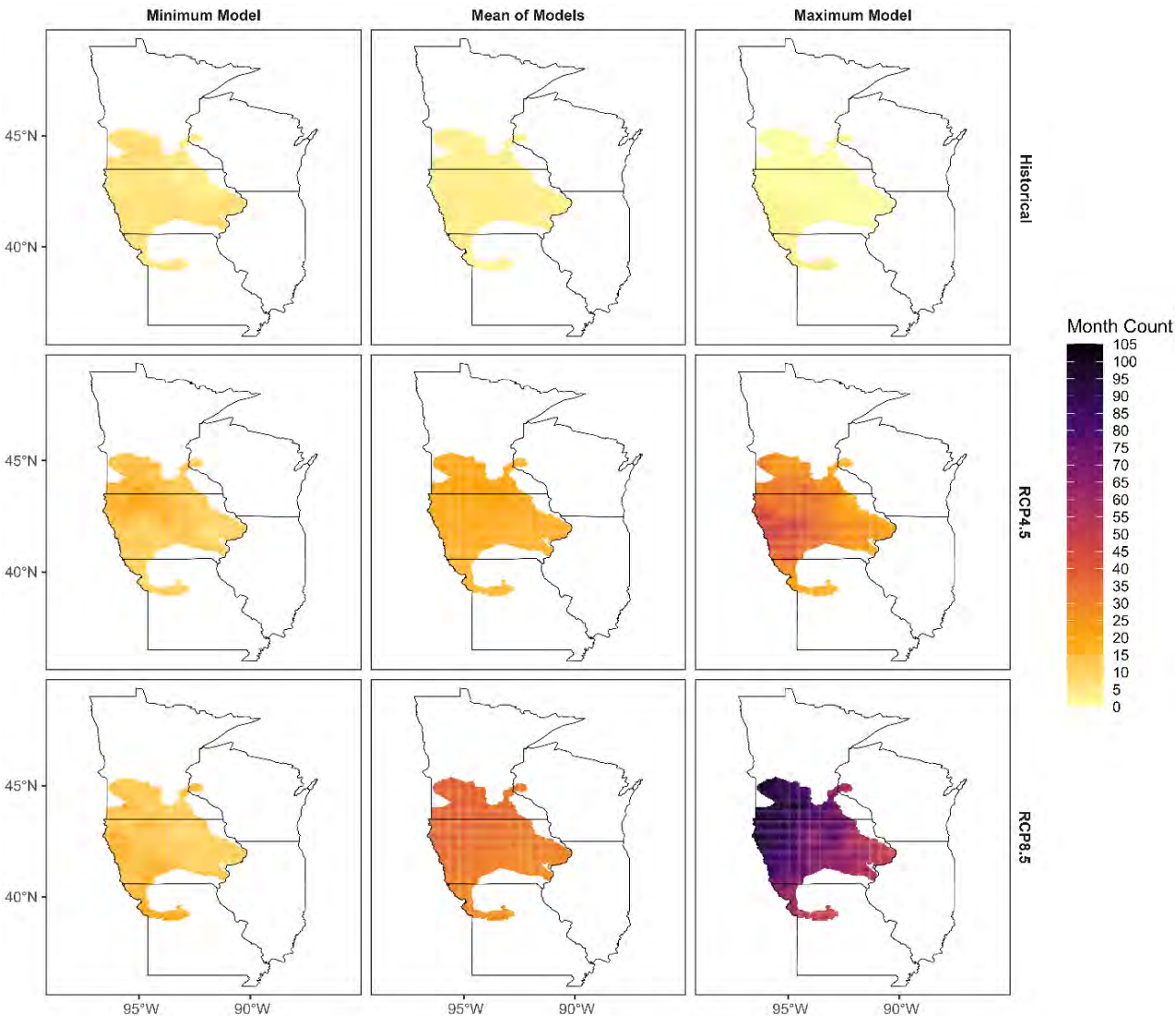

Figure 13. Projected changes in 30-year extreme drought incidence in the Temperate Prairie ecoregion. Projections use gridded Standardized Precipitation-Evapotranspiration Index (SPEI) values obtained via Thota and others (2025) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate-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

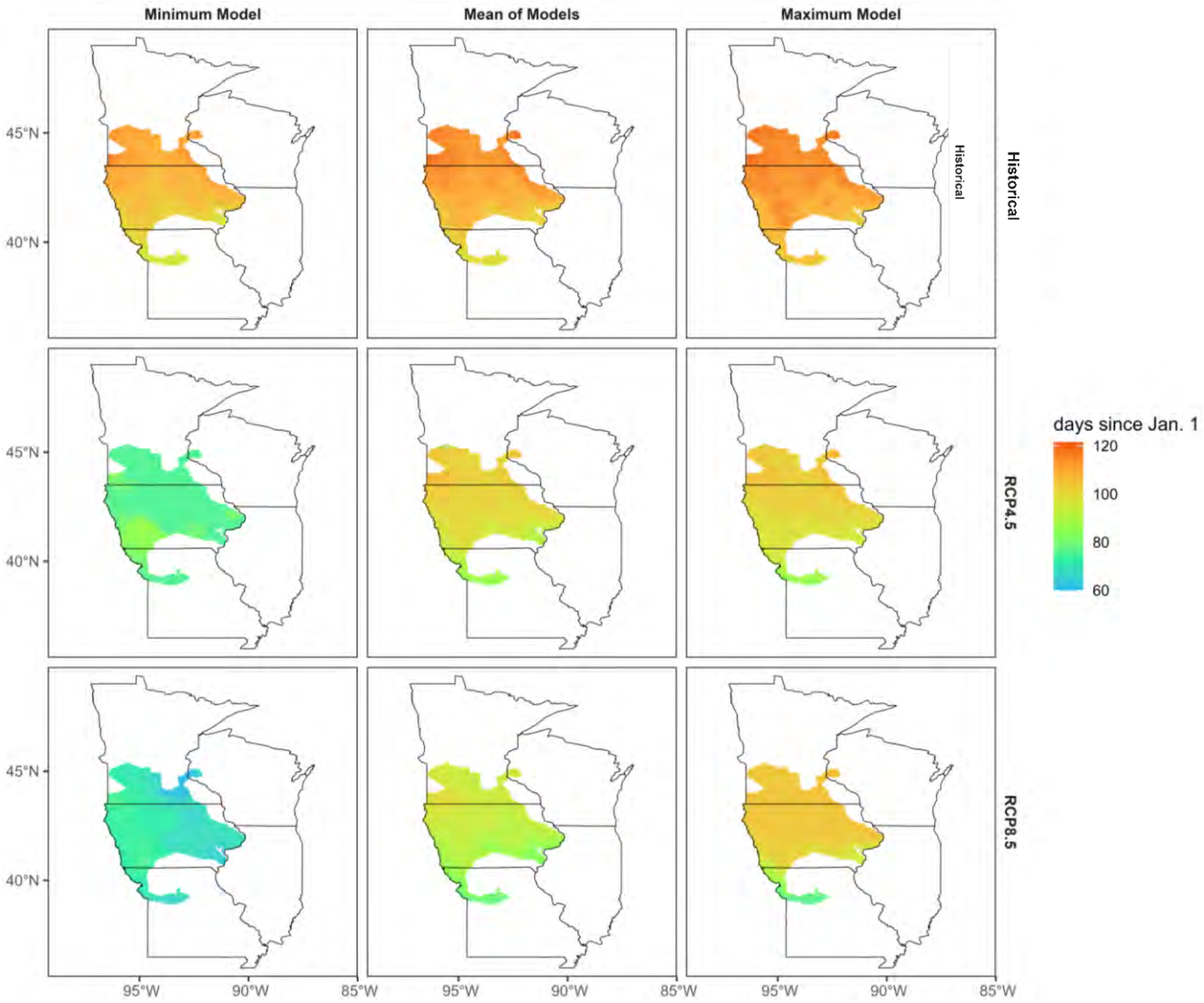

Figure 14. Projected changes in last spring freeze day in the Temperate Prairie ecoregion. Projections use gridded climate data obtained via the Climate Toolbox Climate Mapper at a 4-km resolution (Hegewisch and Abatzoglou, 2024) based on 20 CMIP5 climate models downscaled using MACAv2-METDATA for moderate- and high-emissions (RCP 4.5 and 8.5) scenarios (Abatzoglou, 2013; Abatzoglou and Brown, 2012). Values show mean historical (1971–2000, top row) and projected (2070–2099) last spring freeze day (the last day after January 1 with a minimum temperature below freezing) 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 last spring freeze day under RCP 8.5 are presented in addition to the mean of all models (middle column).

Key Climate Change Impacts

This section summarizes how climate change is impacting the Eastern North American Ruderal Meadow and Shrubland. It covers both the overall influence on species distributions, adaptive capacity, and vulnerability (Table 30) and the specific effects of drought and seasonal shifts (Table 31). The bullet points below provide a concise overview of these impacts on habitat structure, community composition, invasive species, herbivory, and microbial and fungal communities.

Habitat structure:

- Drought and increasingly variable precipitation will likely reduce soil moisture and plant cover, intensifying erosion, accelerating nutrient loss, and impairing long-term soil function (Castillioni and others, 2022b; Fay and others, 2011; Felton and others, 2020; Hanberry and others, 2019; Hoover and Rogers, 2016; Lemoine and others, 2024; Vose and others, 2016).
- Physiological stress from drought may suppress plant growth and increase the risk of mortality and reproductive failure, particularly during high-temperature events (Dreesen and others, 2015; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Smith and others, 2024; Vose and others, 2016; Wang and others, 2008; Will and others, 2013).
- Drought may reduce productivity and impair ecosystem functioning, especially in areas with high aridity or low species diversity (Briske, 2017; Fay and others, 2011; Jones, 2019; Knapp and others, 2024; Pehim Limbu and Avolio, 2025; Slette and others, 2023; Smith and others, 2024).
- Warming-driven shifts in seasonal timing are expected to alter flowering phenology and growing season dynamics (Austin and others, 2024; Dunnell and Travers, 2011; Geissler and others, 2023; Memmott and others, 2007; Ramirez-Parada and others, 2022; Sherry and others, 2007; Zettlemoyer and others, 2021).
- Phenological shifts associated with seasonal warming may boost productivity by extending the growing season but also increase plant vulnerability to late frosts (Allstadt and others, 2015; Briske, 2017; Cleland and Wolkovich, 2024; Fridley and others, 2016; Li and others, 2025; Zettlemoyer and others, 2021).

Community composition:

- Droughts may drive community turnover by favoring drought-tolerant species and reinforcing ruderal dominance and early-successional states (Table 31) (Baty and others, 2015; Dekirmenjian and others, 2024; Griffin-Nolan and others, 2021; Kooyers and others, 2025; Smith and others, 2017; Watson and others, 2021).
- Seasonal shifts may favor early growing or phenologically responsive species, leading to gradual shifts in abundance as less flexible taxa decline (Table 31) (Cleland and Wolkovich, 2024; Dawson-Glass and others, 2025a; Lee and others, 2024; Ramirez-Parada and others, 2022; Wolkovich and Donahue, 2021; Zettlemoyer and others, 2021).
- Earlier and extended growing seasons may result in novel co-flowering patterns and disrupt synchrony between plant flowering and pollinator activity, increasing the risk of trophic mismatches that reduce reproductive success for species unable to align phenology with shifting seasonal conditions (Austin and others, 2024; Memmott and others, 2007; Ramirez-Parada and others, 2022; Whittington and others, 2015; Zettlemoyer and others, 2021).

Invasive species:

- Climate change may reduce the extent of suitable habitat for several invasive species, such as smooth brome (*Bromus inermis*) and common buckthorn (*Rhamnus cathartica)*, while expanding it for others like multiflora rose (*Rosa multiflora*), potentially facilitating introductions and range shifts (Table 30) (Evans and others, 2024; Kane and others, 2017; Reinhardt and others, 2020).
- Climate change is expected to maintain invasive species dominance by favoring species with ruderal traits and high plasticity that can rapidly exploit warmer temperatures, variable

precipitation, and resource availability following extreme events (Catford and Jones, 2019; Clements and Jones, 2021; Fan and others, 2013; Poland and others, 2021; Vose et al., 2016).

- Drought is unlikely to reduce invasive species dominance in ruderal systems dominated by drought-tolerant invaders and may hinder restoration efforts aimed at increasing native species diversity (Orbán and others, 2021; Sobuj and others, 2024; Wilsey and others, 2014).
- Warming-driven shifts in seasonality and longer growing seasons may advantage invasive species with early phenology and prolonged activity, such as smooth brome (*Bromus inermus*), allowing them to gain priority access to resources and suppress later-emerging natives (Table 31) (Ahler and others, 2025; Calinger and others, 2013; Catford and Jones, 2019; Dawson-Glass and others, 2025a; Dickson and others, 2012a; Keller and Shea, 2021a; Sorte and others, 2013; Willis and others, 2010; Wilsey and others, 2015; Wolkovich and others, 2013; Zettlemoyer and others, 2019).

Herbivory:

- Climate-driven drought and seasonal shifts may intensify herbivory pressure, particularly from increasing white-tailed deer (*Odocoileus virginianus*) populations, potentially contributing to declines in native forb diversity while simultaneously limiting woody encroachment (Anderson and others, 2001; Duell and others, 2024; Felton and others, 2024; Hamann and others, 2021; Nisi and others, 2015; Smith and Hall, 2016).

Microbial and fungal communities:

- Drought and seasonal shifts may drive fungal community reassembly and reduce the functional role of arbuscular mycorrhizal fungi in supporting plant drought tolerance (Fu and others, 2022; Hu and others, 2025; Lagueux and others, 2021; Ochoa-Hueso and others, 2021; Pehim Limbu and Avolio, 2025).
- Drought may reduce overall microbial activity and biomass, negatively impacting soil function and carbon cycling (Barnard and others, 2013; Ochoa-Hueso and others, 2018; Preece and others, 2019; Sheik and others, 2011).

Table 30. Summary of the expected general influence of climate change on Eastern North American Ruderal Meadow and Shrubland habitat species. Asterisks (*) denote species introduced to the region. Some species appear in contradictory columns (e.g., potential for both high and low vulnerability), reflecting conflicting findings in the literature.

Expected influence of climate change on species distributions		Hypothesized adaptive capacity and/or vulnerability to climate change	
Reduced extent or amount (–)	Increased extent or amount (+)	Low adaptive capacity/ high vulnerability (–)	High adaptive capacity/ low vulnerability (+)
**Bromus inermis*[1]	*Cornus drummondii*[6]	*Corylus cornuta*[7]	*Corylus cornuta*[10]
**Cirsium arvense*[2]	**Rosa multiflora*[4]	*Populus tremuloides*[8]	**Rhamnus cathartica*[9]
**Phleum pratense*[1]		**Rhamnus cathartica*[9]	*Rosa multiflora*[11]
**Poa pratensis*[1]			
**Rhamnus cathartica*[3,4]			
Rudbeckia hirta[5]			

[1]Kane and others (2017); [2]Liu and others (2014); [3]Evans (2024); [4]Reinhardt and others (2020); [5]Rana and others (2024); [6]Hanberry (2024);; [7]Molano-Flores and others (2019); [8]Rogers and others (2016); [9]Sansom (2020); [10]Inter-Tribal Council of Michigan, Inc (2016); [11]Mortsch and others (2006)

Table 31. Summary of impacts from key climate change stressors on Eastern North American Ruderal Meadow and Shrubland 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, potential for competitive advantage and increased abundance	**Berteroa incana*[1]
		**Bromus inermis*[2–6]
		**Cirsium arvense*[7]
		Cornus drummondii[8–10]
		**Poa pratensis*[11]
		**Rhamnus cathartica*[12]
		Rhus glabra[13]
		**Securigera varia*[14]
	When tolerance of droughts is low, potential for reduced growth rates and abundance	**Bromus inermis*[16]
		Conyza canadensis[18]
		Cornus drummondii[18]
		Corylus cornuta[18]
		**Elymus repens*[18]
		**Lonicera japonica*[19]
		**Phleum pratense*[18,20]
		**Poa pratensis*[16,18,21–23]
		Populus tremuloides[18,24–29]
		Zanthoxylum americanum[30]
Seasonal shifts	When phenological shifts are well-matched to changing seasonality, potential for competitive advantage	**Anthoxanthum odoratum*[31]
		**Berteroa incana*[32]
		**Bromus inermis*[33]
		**Centaurea stoebe*[33]
		Cornus drummondii[34]
		**Leucanthemum vulgare*[32]
		Rudbeckia hirta[32]
	When phenological shifts are poorly matched to changing seasonality, potential for reduced regeneration success and competitive capacity	**Lonicera japonica*[31]
		Solidago canadensis[33]

[1]Noack and others (2020); [2]Duell and others (2021); [3]Duell and others (2016); [4]Kroeger and Otfinowski (2025); [5]Palit and DeKeyser (2022); [6]Raave(2004); [7]Hogenbirk and Wein (1991); [8]Keen and others (2022); [9]O'Keefe and others (2024); [10]Muench and others (2016); [11]Printz and Hendrickson (2015); [12]Fisichelli and others (2014); [13]Johnson (1990); [14]Gucker (2009);; [16]Ahler and others (2025); [18]USDA, NRCS (2025); [19]Munger (2002); [20]Esser (1993); [21]Huff (2003); [22]Palit and others (2021); [23]Uchytil (1993); [24]Gustafson and Sturtevant (2013); [25]Howard (1996); [26]Levesque and Hamann (2022); [27]Michaelian and others (2011); [28]Wang and others (2023); [29]Worrall and others (2013); [30]Wedel and others (2025); [31]Abu-Asab and others (2001); [32]Bertin (2017); [33]Zettlemoyer and others (2019); [34]Prevéy and others (2020)

Focal Wildlife Species: Dakota Skipper

The Dakota skipper (*Hesperia dacotae*) is a butterfly endemic to northern prairies and was listed as federally threatened in 2014 due to widespread habitat loss from agricultural conversion (USDA, NRCS, 2024). The species depends on intact native mixed- and tallgrass prairie, where larvae feed on native bunchgrasses that provide both food and shelter as they develop slowly (Seidle and others, 2018). Because a significant portion of the historical extent of prairie systems have been lost to land-use change or have been converted to invasive-dominated ruderal habitat types, conservation of this species will in part depend on restoration or maintenance of native species within these ruderal habitats. For example, adult skippers depend on nectar from flowering forbs, showing a particular preference for the native narrow-leafed purple coneflower (*Echinacea angustifolia*) (USDA, NRCS, 2024).

Climate change is expected to impact Dakota skipper populations by altering the structure and availability of the prairie habitats they require. Species distribution models project that suitable habitat will contract in Minnesota and the Dakotas and shift northward into southern Canada as temperatures rise (Barnes and others, 2024). In Saskatchewan, Dakota skippers are currently restricted to warmer microclimates within otherwise suitable prairies, suggesting the species is temperature-limited and that northern areas may become increasingly suitable under climate change (Seidle and others, 2018). Adult emergence is also closely tied to growing degree days and is therefore likely to shift earlier in the season (Dearborn and Westwood, 2014). This phenological shift could lead to mismatches with nectar availability if floral cues do not advance similarly (Davies, 2019; Renner and Zohner, 2018). Seasonal shifts may also increase the dominance of invasive cool-season grasses, which impact Dakota skipper survival as larvae that feed on these grass species have lower body condition compared to larvae feeding on native grasses (Barnes and others, 2024). At the same time, increased drought frequency may accelerate woody encroachment, altering the open prairie conditions essential to the survival of the Dakota skipper (Minnesota Department of Natural Resources, 2025b).

Preventing woody encroachment and maintaining, establishing, and restoring prairies will be important strategies for supporting Dakota skipper populations under climate change. Because Dakota skippers are vulnerable to fire throughout their life, the timing of prescribed burns relative to their emergence may be less important than ensuring sufficient area is burned to maintain open habitat (U.S. Fish and Wildlife Service, 2016; USDA, NRCS, 2024). In addition to fire, mechanical thinning and targeted herbicide application may be necessary in some areas to reduce woody cover and promote the persistence of native bunchgrasses and nectar-producing forbs (Minnesota Department of Natural Resources, 2025b).

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Appendix 1

Table iii. Woody plant species common to the Central and Eastern North American Grassland and Shrubland. Key species were identified using USNVC floristic descriptions (USNVC, 2020), filtered by associations occurring within the Midwest that collectively comprise the Central and Eastern North American Grassland and Shrubland 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	Central Tallgrass Prairie	Northern Tallgrass Prairie	Central Interior Acidic Open Glade and Barrens	Eastern North American Ruderal Meadow and Shrubland
Amorpha canescens Pursh	Leadplant	Shrub	×	×		
Ceanothus americanus L.	New Jersey tea	Shrub	×			
Cornus drummondii C.A. Mey.	Roughleaf dogwood	Shrub	×			×
Corylus americana Walter	American hazelnut	Shrub	×			×
Corylus cornuta Marshall	Beaked hazelnut	Shrub, tree				×
†*Crataegus* spp. L.	Hawthorn	Shrub, tree	×			×
**Elaeagnus umbellata* Thunb.	Autumn olive	Shrub			×	×
Juniperus virginiana L.	Eastern redcedar	Tree			×	
**Lonicera japonica* Thunb.	Japanese honeysuckle	Vine				×
†*Lonicera* spp. L.	Honeysuckle	Shrub, vine				×
**Lonicera tatarica* L.	Tatarian honeysuckle	Shrub				×
**Lonicera* × *bella* Zabel [*morrowii* × *tatarica*]	Showy fly honeysuckle	Shrub				×
Malus ioensis (Alph. Wood) Britton	Prairie crab apple	Shrub, tree	×			
**Populus alba* L.	White poplar	Tree			×	
Populus tremuloides Michx.	Quaking aspen	Tree		×		×
Prunus americana Marshall	American plum	Shrub, tree	×			
Quercus ellipsoidalis E.J. Hill	Northern pin oak	Tree	×			

Species	Common name	Growth habit	Central Tallgrass Prairie	Northern Tallgrass Prairie	Central Interior Acidic Open Glade and Barrens	Eastern North American Ruderal Meadow and Shrubland
Quercus macrocarpa Michx.	Bur oak	Shrub, tree	×	×		
*Quercus marilandi*ca Münchh.	Blackjack oak	Shrub, tree			×	
Quercus montana Willd.	Chestnut oak	Tree			×	
Quercus stellata Wangenh.	Post oak	Tree			×	
**Rhamnus cathartica* L.	Common buckthorn	Shrub, tree				×
Rhus glabra L.	Smooth sumac	Shrub, tree	×			×
Rhus typhina L.	Staghorn sumac	Shrub, tree				×
**Rosa multiflora* Thunb.	Multiflora rose	Shrub, vine			×	×
†*Rubus* spp. L.	Blackberry	Shrub, vine				×
†*Salix* spp. L.	Willow	Shrub, tree		×		
Salix humilis Marshall	Prairie willow	Shrub	×			
Smilax bona-nox L.	Saw greenbrier	Shrub, vine			×	
Zanthoxylum americanum Mill.	Common pricklyash	Shrub, tree				×

Table iv. Non-woody plant species common to the Central and Eastern North American Grassland and Shrubland. Key species were identified using USNVC floristic descriptions (USNVC, 2020), filtered by associations occurring within the Midwest that collectively comprise the Central and Eastern North American Grassland and Shrubland 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	Central Tallgrass Prairie	Northern Tallgrass Prairie	Central Interior Acidic Open Glade and Barrens	Eastern North American Ruderal Meadow and Shrubland
Agrostis hyemalis (Walter) Britton, Sterns & Poggenb.	Winter bentgrass	Graminoid	×			
Ambrosia artemisiifolia L.	Annual ragweed	Forb/herb		×		×
Ambrosia bidentata Michx.	Lanceleaf ragweed	Forb/herb			×	
Andropogon gerardii Vitman	Big bluestem	Graminoid	×	×		
Andropogon virginicus L.	Broomsedge bluestem	Graminoid			×	
**Anthoxanthum odoratum* Boiss.	Annual vernalgrass	Graminoid				×
**Arctium minus* Bernh.	Lesser burdock	Forb/herb				×
Aristida basiramea Engelm. ex Vasey	Forked threeawn	Graminoid	×			
Aristida dichotoma Michx.	Churchmouse threeawn	Graminoid			×	
Aristida tuberculosa Nutt.	Seaside threeawn	Graminoid	×			
**Berteroa incana* (L.) DC.	Hoary alyssum	Forb/herb		×		×
Blephilia ciliata (L.) Benth.	Downy pagoda-plant	Forb/herb	×	×		
Bouteloua curtipendula (Michx.) Torr.	Sideoats grama	Graminoid	×	×		
Bouteloua gracilis (Willd. ex Kunth) Lag. ex Griffiths	Blue grama	Graminoid		×		
Bouteloua hirsute Lag.	Hairy grama	Graminoid		×		
Brickellia eupatorioides (L.) Shinners	False boneset	Forb/herb	×	×		
**Bromus inermis* Leyss.	Smooth brome	Graminoid				×
Calamagrostis canadensis (Michx.) P. Beauv.	Bluejoint	Graminoid	×			
Calamovilfa longifolia (Hook.) Scribn.	Prairie sandreed	Graminoid		×		

Species	Common name	Growth habit	Central Tallgrass Prairie	Northern Tallgrass Prairie	Central Interior Acidic Open Glade and Barrens	Eastern North American Ruderal Meadow and Shrubland
Camassia scilloides (Raf.) Cory	Atlantic camas	Forb/herb	×			
Carex inops L.H. Bailey	Long-stolon sedge	Graminoid		×		
Carex muehlenbergii Schkuhr ex Willd.	Muhlenberg's sedge	Graminoid	×			
Carex pensylvanica Lam.	Pennsylvania sedge	Graminoid	×		×	
Carex siccata Dewey	Dryspike sedge	Graminoid	×	×		
Carex tonsa (Fernald) E.P. Bicknell	Shaved sedge	Graminoid	×			
**Centaurea stoebe* L.	Spotted knapweed	Forb/herb		×		×
Cicuta maculata L.	Spotted water hemlock	Forb/herb	×	×		
**Cirsium arvense* (L.) Scop.	Canada thistle	Forb/herb		×		×
Clinopodium arkansanum (Nutt.) House	Limestone calamint	Forb/herb	×			
Clitoria mariana L.	Atlantic pigeonwings	Forb/herb, vine			×	
Conyza canadensis (L.) Cronquist	Canadian horseweed	Forb/herb				×
Croton michauxii G.L. Webster	Michaux's croton	Forb/herb			×	
Cyperus lupulinus (Spreng.) Marcks	Great Plains flatsedge	Graminoid	×			
Cyperus schweinitzii Torr.	Schweinitz's flatsedge	Graminoid	×	×		
Dalea candida Michx. ex Willd.	White prairie clover	Forb/herb	×			
Dalea purpurea Vent.	Purple prairie clover	Forb/herb	×			
Danthonia spicata (L.) P. Beauv. ex Roem. & Schult.	Poverty oatgrass	Graminoid	×		×	
**Daucus carota* L.	Queen Anne’s lace	Forb/herb				×
Desmanthus illinoensis (Michx.) MacMill. ex B.L. Rob. & Fernald	Illinois bundleflower	Forb/herb	×			
Dichanthelium spp. (Hitchc. & Chase) Gould	Rosette grass	Graminoid			×	

Species	Common name	Growth habit	Central Tallgrass Prairie	Northern Tallgrass Prairie	Central Interior Acidic Open Glade and Barrens	Eastern North American Ruderal Meadow and Shrubland
Dichanthelium linearifolium (Scribn. ex Nash) Gould	Slimleaf panicgrass	Graminoid	×			
Dichanthelium ovale (Elliott) Gould & C.A. Clark	Eggleaf rosette grass	Graminoid	×			
†*Digitaria* spp. Haller	Crabgrass	Graminoid				×
Diodia teres Walter	Poorjoe	Forb/herb			×	
Eleocharis compressa Sull.	Flatstem spikerush	Graminoid	×			
**Elymus repens* (L.) Gould	Quackgrass	Graminoid				×
Eryngium yuccifolium Michx.	Button eryngo	Forb/herb	×			
Euphorbia corollate L.	Flowering spurge	Forb/herb	×			
Helianthus pauciflorus Nutt.	Stiff sunflower	Forb/herb	×			
Hesperostipa spartea (Trin.) Barkworth	Porcupinegrass	Graminoid	×	×		
Koeleria macrantha (Ledeb.) Schult.	Prairie Junegrass	Graminoid	×	×	×	
**Leucanthemum vulgare* Lam.	Oxeye daisy	Forb/herb				×
Liatris squarrosa (L.) Michx.	Scaly blazing star	Forb/herb			×	
Lithospermum canescens (Michx.) Lehm.	Hoary puccoon	Forb/herb	×			
**Melilotus officinalis* (L.) Lam.	Sweetclover	Forb/herb		×		×
Muhlenbergia cuspidate (Torr. ex Hook.) Rydb.	Plains muhly	Graminoid	×			
Muhlenbergia richardsonis (Trin.) Rydb.	Mat muhly	Graminoid		×		
Panicum virgatum L.	Switchgrass	Graminoid	×			
Pascopyrum smithii (Rydb.) Á. Löve	Western wheatgrass	Graminoid		×		
Penstemon hirsutus (L.) Willd.	Hairy beardtongue	Forb/herb	×			
**Phleum pratense* L.	Timothy	Graminoid				×

Species	Common name	Growth habit	Central Tallgrass Prairie	Northern Tallgrass Prairie	Central Interior Acidic Open Glade and Barrens	Eastern North American Ruderal Meadow and Shrubland
**Poa compressa* L.	Canada bluegrass	Graminoid	×			
**Poa pratensis* L.	Kentucky bluegrass	Graminoid	×			×
Polygala verticillate L.	Whorled milkwort	Forb/herb	×			
Pycnanthemum virginianum (L.) T. Dur. & B.D. Jacks. ex B.L. Rob. & Fernald	Virginia mountainmint	Forb/herb	×			
Rosa Carolina L.	Carolina rose	Subshrub	×			
Rudbeckia hirta L.	Blackeyed Susan	Forb/herb	×	×	×	×
Schizachyrium scoparium (Michx.) Nash	Little bluestem	Graminoid	×	×	×	
Scutellaria parvula Michx.	Small skullcap	Forb/herb	×			
**Securigera varia* (L.) Lassen	Crownvetch	Forb/herb, vine		×	×	×
Selaginella rupestris (L.) Spring	Northern selaginella	Forb/herb	×		×	
†*Setaria* spp. P. Beauv.	Bristlegrass	Graminoid				×
**Setaria viridis* (L.) P. Beauv.	Green bristlegrass	Graminoid				×
Solidago canadensis L.	Canada goldenrod	Forb/herb	×	×		×
Solidago juncea Aiton	Early goldenrod	Forb/herb			×	
Solidago nemoralis Aiton	Gray goldenrod	Forb/herb	×			
Sorghastrum nutans (L.) Nash	Indiangrass	Graminoid	×		×	
Sporobolus clandestinus (Biehler) Hitchc.	Rough dropseed	Graminoid			×	
Sporobolus cryptandrus (Torr.) A. Gray	Sand dropseed	Graminoid	×	×		
Sporobolus heterolepis (A. Gray) A. Gray	Prairie dropseed	Graminoid	×	×		
Sporobolus neglectus Nash	Puffsheath dropseed	Graminoid			×	

Species	Common name	Growth habit	Central Tallgrass Prairie	Northern Tallgrass Prairie	Central Interior Acidic Open Glade and Barrens	Eastern North American Ruderal Meadow and Shrubland
Symphyotrichum ericoides (L.) G.L. Nesom	White heath aster	Forb/herb	×	×	×	
Symphyotrichum sericeum (Vent.) G.L. Nesom	Western silver aster	Forb/herb	×			
**Trifolium pratense* L.	Red clover	Forb/herb		×		
**Trifolium* spp. L.	Clover	Forb/herb		×		
**Verbascum Thapsus* L.	Common mullein	Forb/herb		×		

Appendix 2

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

CMIP5 Models:

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

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

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

Ecoregion	Climate Variable	Season	Minimum Model	Maximum Model
Ozarks	Extreme Dry Months	Annual	CNRM-CM5_r1i1p1	IPSL-CM5A-MR_r1i1p1
Midwest Region	Extreme Dry Months	Annual	MRI-CGCM3_r1i1p1	HadGEM2-CC365_r1i1p1
Temperate Prairie	Growing Season Length	Annual	CSIRO-Mk3-6-0	MIROC5
Northern Plains	Growing Season Length	Annual	MRI-CGCM3	MIROC-ESM
Midwest Region	Last Spring Freeze	Annual	MIROC-ESM	CSIRO-Mk3-6-0
Ozarks	Last Spring Freeze	Annual	MIROC-ESM	GFDL-ESM2M
Northern Plains	Precipitation	Annual	HadGEM2-CC365	GFDL-ESM2M
Temperate Prairie	Precipitation	Annual	IPSL-CM5A-MR	CanESM2
Ozarks	Precipitation	Annual	IPSL-CM5A-MR	IPSL-CM5B-LR
Northern Plains	Mean Temperature	Annual	GFDL-ESM2M	HadGEM2-CC365
Temperate Prairie	Mean Temperature	Annual	MRI-CGCM3	HadGEM2-CC365
Ozarks	Mean Temperature	Annual	MRI-CGCM3	MIROC-ESM-CHEM
Northern Plains	Vapor Pressure Deficit	Annual	GFDL-ESM2M	HadGEM2-CC365
Temperate Prairie	Vapor Pressure Deficit	Annual	MRI-CGCM3	HadGEM2-CC365