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Plant Community Trends at Tallgrass Prairie National Preserve

2002–2023

Green antelopehorn (*Asclepias viridis*), a common milkweed at Tallgrass Prairie National Preserve, Kansas.
NPS / HEARTLAND INVENTORY AND MONITORING NETWORK

Plant community trends at Tallgrass Prairie National Preserve: 2002–2023

Science Report NPS/SR—2025/266

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Abstract

The Heartland Inventory and Monitoring Network has monitored vegetation at Tallgrass Prairie National Preserve for more than 20 years. Monitoring assists the preserve in managing the nearly 10,000-acre (4,047 ha) site by providing status and trends of vegetation and disturbance metrics. The site protects critical remnant tallgrass prairie as well as the ranching legacy of the Flint Hills region. Monitoring was conducted in 30 permanent sites across the western preserve. We assessed current and future climate trends, fire and grazing rates, and vegetation metrics, including ground cover, guild, plant diversity, and species composition. We also assessed observer error using a double sampling technique.

Historical climate data indicate a trend toward increasing moisture and days with heavy precipitation. Future climate scenarios include the possibility of a warm-wet future with a slight increase in excess moisture (runoff) in the spring and the possibility of a hot-dry future where water deficit spans most of the year. Disturbance intensity continued to be lower since 2006 when management shifted from more of a ranching-based paradigm towards an ecological paradigm with heterogeneity as a central theme (Leis and Morrison 2018). Stocking rates were similar or lighter than the prior monitoring period (2015–2018), and fire return intervals were longer since 2006. Of ground cover metrics, bare areas meet greater prairie chicken habitat thresholds preserve-wide but fall below expectations in two pastures. The native woody plant guild continues to increase in cover in Big and Windmill pastures, but other plant guilds are stable. There was a small increase in tree seedlings, however. Measures of species richness were within the range of variation of prior monitoring events, but Shannon Diversity had a declining trend. Combined with Sørensen index analysis we noted that species composition shifts were likely a key factor in the decline of Shannon Diversity. Pseudoturnover was less than the previous monitoring event—likely the result of having an expert botanist and improved double sampling techniques and communication. Indications of a changing flora coincide with reduced disturbance intensity and a changing climate. An increase in disturbance intensity and/or timing shifts may be needed to avoid detrimental woody plant expansion. Future management actions that take climate scenarios into consideration may also be beneficial. For example, planning fires around changing plant phenology and grazing plans that reduce animal stress and provide adequate water in the event of drought would be beneficial in the short and long term.

Acknowledgments

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Introduction

Tallgrass Prairie National Preserve in the heart of the Flint Hills geographic province (Kindscher et al. 2011) protects 10,894 acres (4,409 ha) of remnant tallgrass prairie. The tallgrass prairie ecosystem is imperiled and the public-private partnership that protects this ecosystem and rich cultural history is unique (NPS 2000). Natural resource managers apply ecological disturbances, such as fire and grazing, to maintain habitat for the many species of plants and animals that live there. Because these organisms have disparate needs, the principle of heterogeneity guides the application of management techniques, providing a wide array of conditions, vegetation structures, and microhabitats to maximize biodiversity at the landscape scale (Hase 2016; Leis et al. 2013).

The greater prairie-chicken (*Tympanuchus cupido*) is an iconic species present at the preserve that requires heterogeneous habitat for its life cycle. Bare areas are used for lekking, warming, and finding insects to eat. More densely vegetated areas are often used for nesting and escape cover, while areas with a grass canopy and open ground layer are favored for chicks to navigate in safety while feeding (McKee et al. 1998; Svedarsky et al. 2003). Ideally, these distinct structures are juxtaposed in a matrix of grasses and forbs. Because such heterogeneity also provides habitat for a variety of other organisms, managing for greater prairie-chickens can help meet goals for the whole community (Fuhlendorf et al. 2006).

To achieve heterogeneity across the preserve, prescribed fire intervals ranging from 1 to 3 years are planned for burn units (Hase 2016). Grazing is also deployed using a variety of grazing systems, including moderate stocking rates. Fire frequency on the land that eventually became the Tallgrass Prairie National Preserve varied historically, with the period from the late 1970s to the preserve's establishment in 1996 having very frequent fires (intervals of 1 to 3 years; Allen and Palmer 2011; Earls 2006). Since 2006, fire frequency changed from nearly annual burns to more variable fire frequency, and grazing intensity declined (Leis and Morrison 2018). Key indicators of desired vegetation conditions identified by preserve staff include thresholds for cover of bare areas and litter and low abundance of woody and nonnative plants.

The Heartland Inventory and Monitoring Network (Heartland Network) began monitoring plant community trends in 1998. Herein, we report on trends in management actions (fire and grazing) and key plant community indicators for 2002–2023. Vegetation sampling methods were under development from 1998 to 2001, so botanical data from this time period are not included in this report (HTLN 2021). We include discussion of climate trends to frame long-term vegetation trends and to consider future effects on the prairie.

Methods

Study Site

The preserve is nearly 10,000 acres, although our monitoring and analyses have focused on the western preserve where long-term monitoring sites were installed to observe tallgrass prairie plant communities (Figure 1). The western portion of the preserve is remnant tallgrass prairie. The preserve is embedded in a matrix of remnant rangelands within the Flint Hills of Kansas. Monitoring sites were located primarily in the tallgrass prairie vegetation type with some sites in the rocky mixed prairie type (Figure 1; Kindscher et al. 2011).

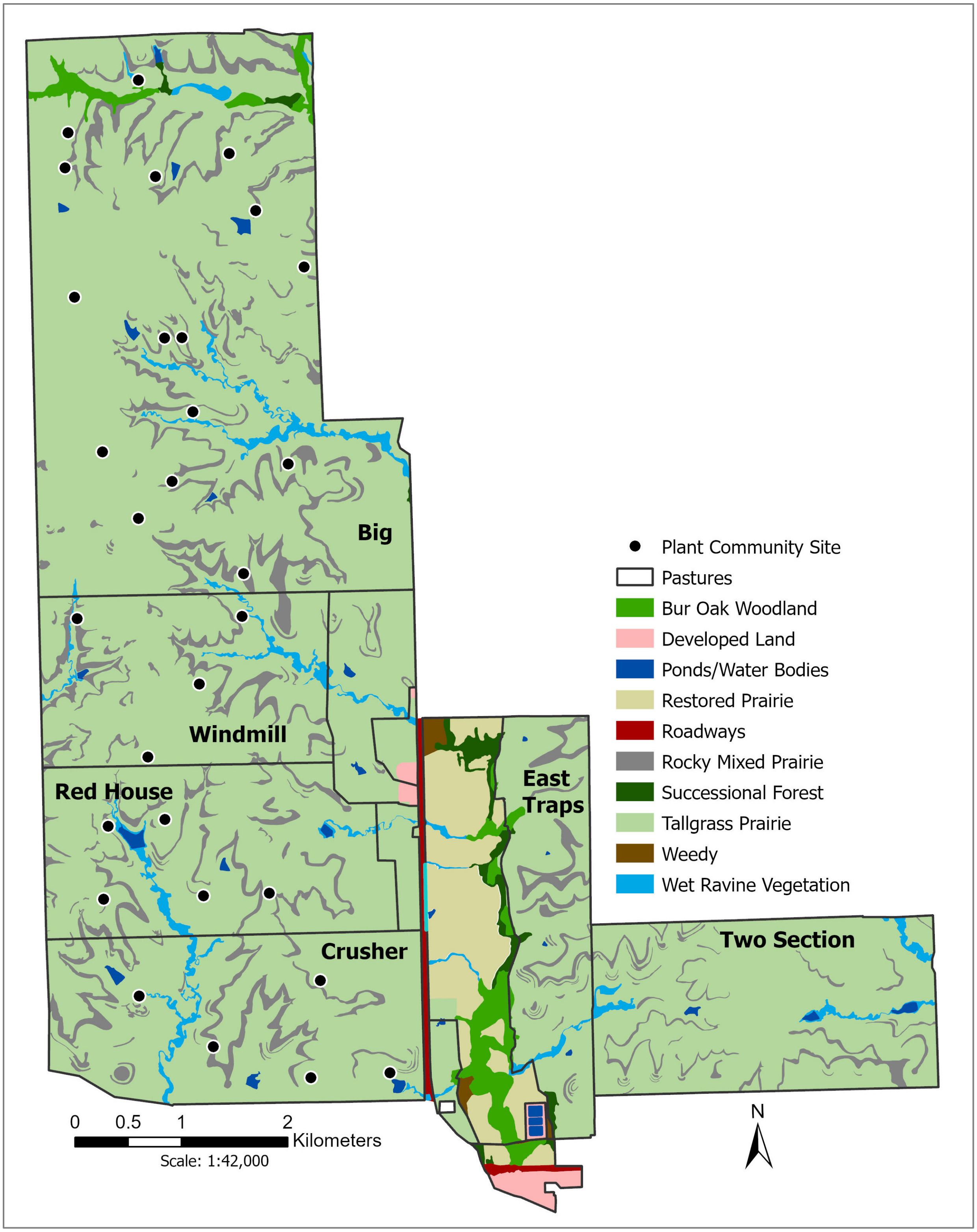

Figure 1. Vegetation types with respect to long-term monitoring sites at Tallgrass Prairie National Preserve, Kansas. This map is based on the vegetation map by Kindscher et al. (2011) but modified to reflect recent prairie reconstruction in the bottomland area of the preserve.

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Sampling Design

We established vegetation monitoring sites in the prairie beginning in 1997, and the design was finalized in 2002 (Figure 1; HTLN 2021). See Leis et al. (2022) for details on the sampling design. Monitoring methods followed the prairie standard operating procedures outlined in the vegetation community monitoring protocol (Leis et al. 2022). Monitoring sites were 50 m × 20 m (0.1 ha) in size with two focal transects bounding the site on the 50 m sides (Figure 2). Ten subplots were established in each site along the 50 m transects. Each subplot consisted of a series of nested frames (0.01 m^2, 0.1 m^2, 1 m^2, and 10 m^2), but only observations at the 10 m^2 scale were summarized to the site scale (0.1 ha) for this study.

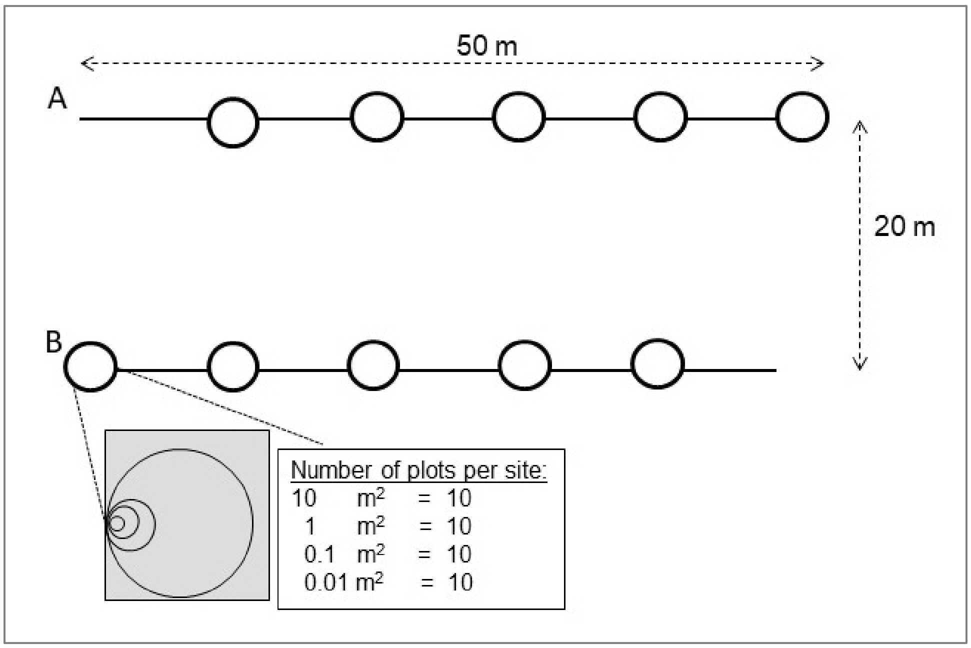

Figure 2. The Heartland Inventory and Monitoring Network basic long-term vegetation monitoring sampling site design. Each site consists of 10 nested plots.
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Data Summary

We included vegetation data from 2002 to 2023 in our analyses. Vegetation data were collected annually from 2002 to 2008, and then every four years (2010, 2014, 2018, and 2023), except for the last interval, which was 5 years (N = 11 sampled years). The revisit design changed through time (DeBacker et al. 2004; James et al. 2009) from a two-season (mid-May, early October), annual visit design with a core of 18 sites and a rotating panel of additional sites, to a one-season (mid-June), one-visit-every-four-years design with 30 sites (Table 1). Conversion of the two-season sampling to a single dataset was done by using the maximum cover of the two seasons for each species observed.

Table 1. Sample sizes (N) for monitored years at Tallgrass Prairie National Preserve, Kansas, 2002–2023.

Year	N
2002	19
2003	19
2004	19
2005	20
2006	25
2007	24
2008	23
2010	30
2014	30
2018	30
2023	30

Sample inclusion was guided by analyses conducted by Leis and Morrison (2018) who determined that the original core 18 and final core 30 sites came from the same population. The original 18 core sites were included in the final 30 core sites (Figure 2) and some of the 30 core sites were sometimes included as panel sites in prior years. The 30 core sites were included in any year that they were sampled. This resulted in the preserve-scale sample sizes given in Table 1.

Site values were used to calculate pasture-scale and preserve-scale statistics. Vegetation monitoring at Tallgrass Prairie National Preserve consists of ground cover, ground flora species, and tree regeneration observations. Vegetation and ground cover data were collected using a modified Daubemire cover class system in the 10 m^2 subplots (Table 2; Leis et al. 2022).

Table 2. Modified Daubenmire cover value scale used to determine ground cover for the Heartland Network Parks.

Cover Class Codes	Range of Cover (%)	Class Midpoints (%)
7	95–100	97.5
6	75–95	85.0
5	50–75	62.5
4	25–50	37.5
3	5–25	15.0
2	1–5	2.5
1	0–0.99	0.5

Climate

We included a brief description of past and future climate conditions to provide context for ecological and vegetation metrics. Annual precipitation (1905–2023) was calculated using [The Climate Analyzer tool](climateanalyzer.org) (climateanalyzer.org; Tercek 2024) for Heartland Network parks as a function of drought and days with heavy precipitation. National Weather Service data from the station in Cottonwood Falls, Kansas (Cottonwood Falls station; ID 7182200), were used to calculate days with excessive rainfall and the normalized reconnaissance drought index (RDI) metrics via The Climate Analyzer. Of the local weather stations, Cottonwood Falls station (6.3 mi [10.1 km] from TAPR) provided the longest and most consistent precipitation record.

The RDI calculates drought using precipitation and potential evapotranspiration (PET) and thus represents the deficit between precipitation and evaporative demand (Tsakiris et al. 2007). RDI represents a better fit for the conditions relative to vegetation than the standardized precipitation index (SPI), referred to in our previous report because of the inclusion of PET (Leis and Morrison 2022). Water balance metrics (deficit, PET, and actual evapotranspiration [AET]) were used to analyze current and future climate trends for Tallgrass Prairie National Preserve (Runyon et al. 2024). Climatic water deficit describes water availability as the difference of potential evapotranspiration minus actual evapotranspiration. Deficit is the additional amount of water plants would use if available (Stephenson 1998; Thoma et al. 2020). The historical period for baseline comparison was 1979–2012 because it relates to standardized 30-year normal calculations (Runyon et al. 2024). The climate futures selected included the warm-wet and hot-dry scenarios based on current climate trends and relevance to park ecology.

Climate change data related to changes in fire risk for the park were accessed using the [Climate Tool Box online platform](www.ClimateToolBox.org) (www.ClimateToolBox.org; Hegewisch and Abatzoglou 2024).

Recent Fire History

Fire history records (primarily prescribed fire) for 1998–2023 are included. Recent fire history for Tallgrass Prairie National Preserve was determined using a fire history geodatabase (Leis 2022). Although preserve management included fire for much of the last century, earlier years (pre-1998) must be evaluated qualitatively. A 30 m buffer was constructed from the center point of each monitoring site. If the buffered area was greater than 30% burned (i.e., $\geq$ 848 m^2), the whole site was considered burned. On-the-ground observations since 2009 validated the spatial data. Time since last fire (TSF) was then calculated from this dataset for each monitoring site-year combination. We did not account for seasonality in the time since fire calculations; rather we assigned burned status on an annual basis. We used a histogram approach to visualize and understand trends at the preserve and pasture scales.

Grazing

Annual grazing data were collected from the preserve lessee, stored in a Microsoft Access database, and used for stocking rate calculations. Rates were based on a 1000 lb (453.6 kg) animal; note this is a change in stocking calculation from our previous work that used 750 lbs (340.2 kg) as the basis for stocking calculations (Leis et al. 2013; Leis and Morrison 2022). Months were calculated from days on pasture using 30 days for a month. We averaged the head in and out counts to mitigate any losses or births during the season. In and out weights were also averaged to better reflect stocking over the grazing events. When there was more than one grazing event in a single pasture over the year, we summed the total Animal Unit Months (AUM) and the AUM/acre for that pasture.

Grazing systems at Tallgrass Prairie National Preserve have included pasture-based early intensive stocking (Smith and Owensby 1978), modified early intensive stocking (less than double stocking rates or extended season, based on preserve records), season long stocking (six months; Holechek et al. 2001; Owensby et al. 1988) and patch burn grazing (implemented in Big Pasture in 2006; Fuhlendorf and Engle 2001; Leis et al. 2013). Pastures that have not been grazed within the period of the study were not included in the analysis (i.e., Cemetery and Brome pastures).

In 2022–2023, the preserve also implemented a virtual fencing study in two of the pastures. Although yearling stocker cattle were deployed across the preserve in most years, a mixture of cattle demographics were deployed in the virtual fence study (Crusher Hill and Red House pastures). As a result of the irregularity in the grazing regime, we used a total weight-based calculation. Bison replaced cattle in Windmill Pasture beginning in 2009, but weight-based calculations were not available to us for bison. Therefore, we did not include stocking data beginning in 2010 for Windmill Pasture where the bison reside. Data for Crusher Hill Pasture in 2006 were not used because stocking rates could not be adequately determined. Some pastures were jointly grazed in some years (Crusher Hill and Red House pastures, Two Section and southern East Traps pastures), and the combined and individual grazing events were presented. Grazing has been deferred for a number of years in Cemetery Pasture and portions of West Traps and East Traps pastures. We assessed stocking rate at both the preserve and pasture scales to better understand trends on a spatial basis.

Ground Cover

Ground cover was assessed using cover classes (Table 2). A site mean was calculated by averaging the cover class midpoints for ten subplots in each site. We observed foliar cover of grass litter, leaf litter (deciduous plant leaves), exposed rock, bare soil, and the cover of woody debris (e.g., branches and sticks). Total unvegetated area reflects space unoccupied by stem basal area in the subplots (Leis et al. 2022). Exposed rock and bare soil were combined into a "bare" class during analysis. We also measured litter depth in each plot in 2018–2023. Plot litter measurements were averaged, and those site means were used for further analysis (3 measurements/plot in 2018, 1 measurement/plot in 2023).

Community Diversity

Native plant community richness metrics evaluate how species richness differs across monitoring sites and the park. We limited these calculations to native ground flora species (not including tree regeneration).

Alpha diversity is synonymous with species richness (S) at the site scale (i.e., mean number of species per monitoring site; Shannon and Weaver 1949). *Gamma* diversity is the community-level richness (i.e., total number of species in the community) observed across all monitoring sites. *Beta* diversity is a measure of heterogeneity in species distributions across monitoring sites such that small values (near 0) indicate a high degree of similarity in species occurrence across monitoring sites and greater values (> 5) indicate a higher degree of variation in species between sites (more differentiated communities; McCune and Grace 2002).

$$\textit{Beta}\ \text{Diversity} = \frac{gamma}{alpha} - 1$$

Plant diversity for each site was calculated using the Shannon diversity index (H′):

$$H' = -\sum_{i=1}^{n} p_i \ln p_i$$

where p_i is the relative cover of species i (Shannon and Weaver 1949).

Species distribution evenness (J′) was calculated by site according to Pielou (1969):

$$J' = \frac{H'}{\ln (S)}$$

where H′ is the Shannon diversity index and ln(S) is the maximum possible Shannon diversity for a given number of species if all species were present in equal abundance. Evenness is a measure of distribution of species within a community as compared to equal distribution and maximum diversity (Pielou 1969). A value of 1 indicates even communities, whereas 0 indicates heterogeneous communities.

PC-ORD (version 7.02) was used to calculate these diversity indices (McCune and Mefford 2016). A grand mean was then calculated for all sites at the preserve. A linear regression was conducted to determine the directional relationship of H′ over time.

Guild and Species Summary

Species guilds were assigned as designated in the USDA Plants database (USDA NRCS 2017). Woody plants included non-tree shrub and subshrub species. Species-level observations were averaged at the site and summed by guild (Appendix A).

Tree regeneration stems were tallied by species in the 10 m^2 subplots and reported in three size classes: (1) seedlings = stems < 0.5 m tall; (2) small saplings = stems ≥ 0.5 m tall and < 2.5 cm DBH; and (3) large saplings = stems ≥ 0.5 m tall and DBH ≥ 2.5 cm and < 5.0 cm. The small number of resulting observations were simply tallied for reporting.

Other guilds summarized include grasses, forbs, and grasslike plants. We include discussion of nonnative plants as another plant guild. Site mean cover of nonnative species was summed across sites. Number of nonnative species was also counted for each sampling year.

Native species richness was calculated by averaging the mean site richness of the sites sampled that year (see Table 1). The Sørensen Index was calculated to compare changes in species composition between the initial year of sampling, 2002, and the most recent year, 2023 (Sørensen 1948). Only the initial core 18 sites, which were sampled in both years, were used for this analysis. The Sørensen Index was calculated for each site, and a park-wide mean was determined. The Sørensen Index (SI) was calculated as

$$SI = \frac{2a}{(2a + b + c)} \times 100$$

where *a* is the number of species present in both years, *b* is the number of species present in the first year only, and *c* is the number of species present in the second year only. The Sørensen Index is expressed as a percentage, where 100% represents complete similarity and 0% is complete dissimilarity.

Observer Error

We assessed our error in observing the ground flora with a double sampling method. At 15 of the 30 monitoring sites, after the transects were read, one plot from each of the two transects was randomly chosen for resampling. Sampling teams switched transects and sampled the ground flora in the randomly chosen plot. In 2023, we also resampled ground cover. Pseudoturnover was calculated by comparing the two species lists from each plot and identifying species that were overlooked (a species was recorded by one observer but not the other), misidentified (the same specimen was apparently recorded as different species by the two observers), or recorded at different specificity levels (cautious errors, in which one observer identified a plant to species and the other identified it only to genus).

Pseudoturnover was calculated as in Nilsson and Nilsson (1985):

$$\frac{A + B}{S_A + S_B} \times 100$$

where *A* is the number of species recorded exclusively by observer 1, B is the number of species recorded exclusively by observer 2, S_A is the total number of species recorded by observer 1, and S_B is the total number of species recorded by observer 2. Pseudoturnover is expressed as a percentage.

We also calculated percent cover agreement for ground cover class assignments and litter depth measurements.

Statistical Analysis

Climate analyses included linear regression of long-term drought and precipitation trends. Descriptive (graphical) analyses of ground cover and nonnative plants included all sampled sites (209 total observations) to obtain more precise parameter estimates. We evaluated trends by calculating statistics at both the preserve and pasture scales. Vegetation statistics were calculated in Minitab (Minitab Statistical Software 2020). Fire and grazing trends were analyzed using R tidyverse package (Wickam et al. 2019; R Core Team 2021). Statistical significance was assessed at the alpha = 0.05 level.

Results and Discussion

Climate

Climate in the Great Plains is characterized by variability. Periodic drought, variable weather, and plant adaptations contribute to the composition and productivity of plant communities in the region (Thornthwaite 1941; Rosenberg 1987). Linear regression of normalized reconnaissance drought index from 1910 to 2023 demonstrated a significant trend toward increasing moisture (RDI = 0.0025 × year − 4.95, $p < 0.001$; Figure 3). Although the trend is toward increased moisture, the extended droughts of the 1930s (the Dust Bowl) and 1950s exhibit strong influences over the trend.

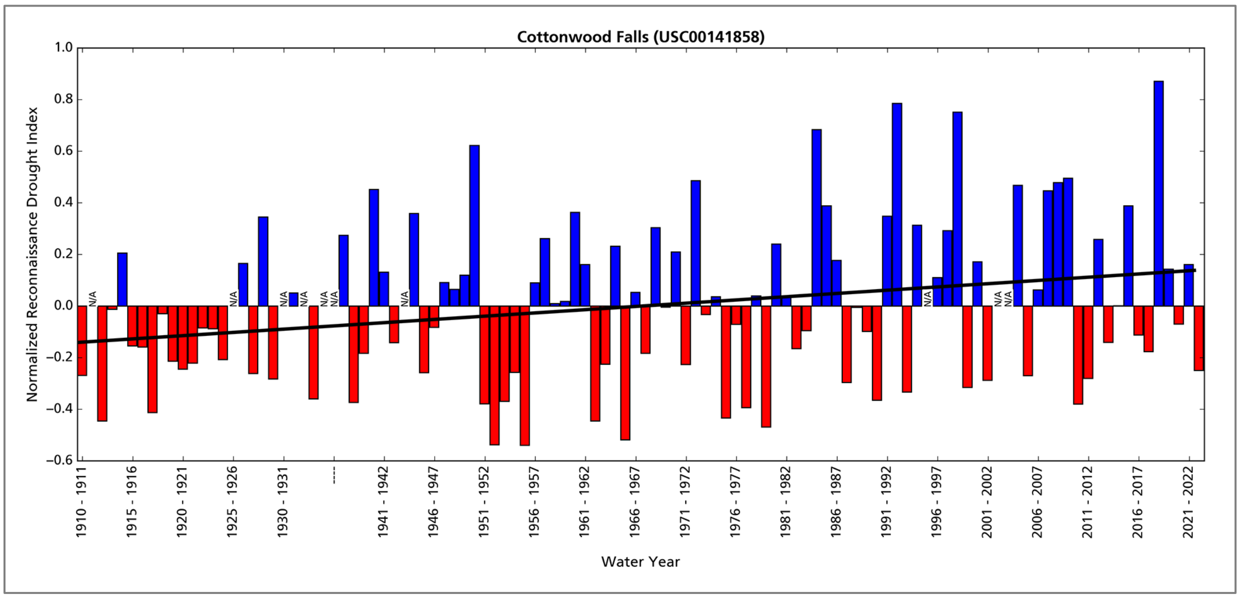

Figure 3. Normalized reconnaissance drought index for Cottonwood Falls, Kansas, 1910–2023. Blue upward bars indicate wet conditions while red downward bars indicate dry conditions.
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Linear regressions of the number of days with moderate to heavy precipitation events through time demonstrated a significant but weak relationship (days/yr = 0.0272 × year − 49.19, $r^2 = 0.12$, $p = 0.00012$; Figure 4). The Climate Change Response Program (2024) reported a 21% increase in the amount of rain falling during rain events since 1958. Although the effect of changing precipitation frequency as opposed to rainfall magnitude is not well understood, heavy rainfall events are more likely to result in surface runoff than infiltration. Increased intensity with more intermittent drought may also decrease biomass and root mass (Padilla et al. 2019). The amount of precipitation may be more critical than frequency, however (Gibson-Forty et al. 2016). Vegetation and/or surface litter can intercept the excess water better than bare ground and absorb the heavy rains, lessening runoff into streams and ponds.

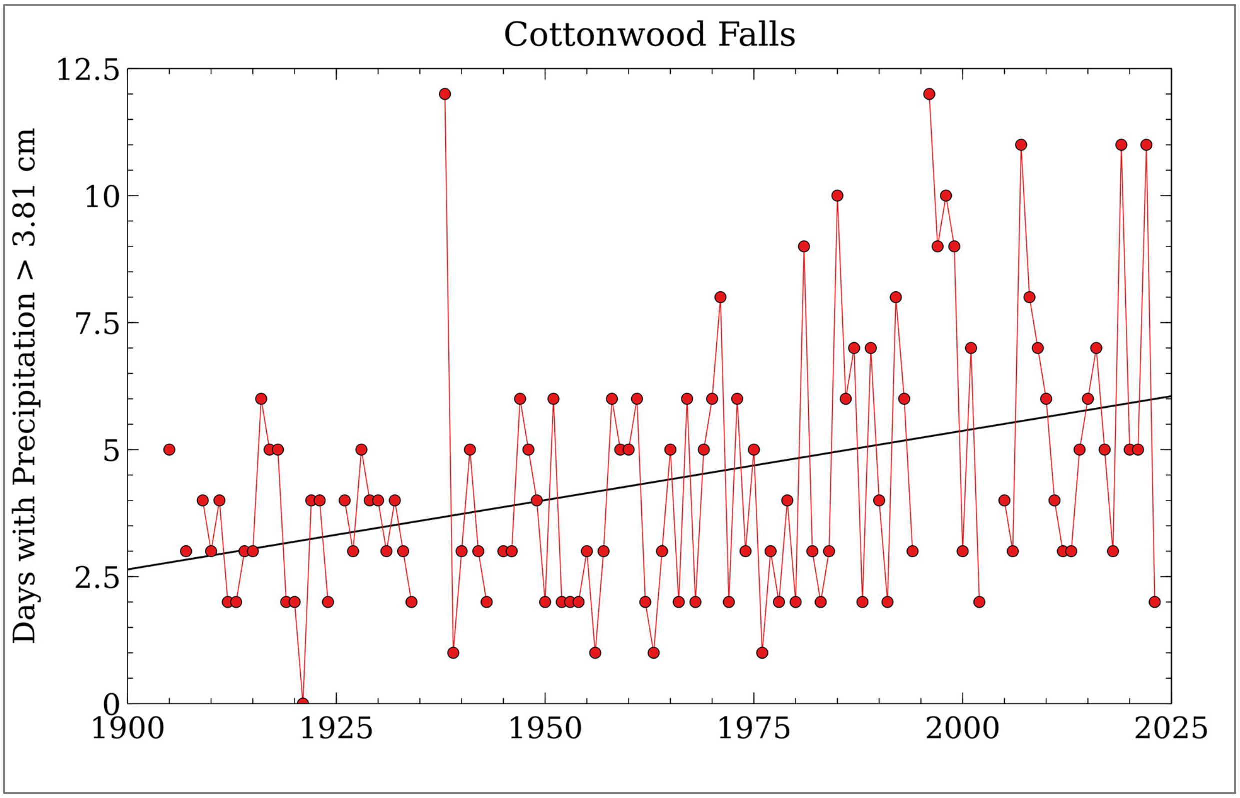

Figure 4. Days with heavy precipitation (> 3.81 cm [1.5 in]) for Tallgrass Prairie National Preserve, Kansas, 1905–2023. Weather station: Cottonwood Falls (USC00141858). Available from www.ClimateAnalyzer.org (Tercek 2024).

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Water balance metrics, including precipitation and potential and actual evapotranspiration, are predicted to deviate from the historical water balance on a seasonal basis at midcentury (2050; Figure 5). Divergent climate futures were selected to include the range of variability that may occur so that parks can plan for the broadest range of possibilities (Runyon et al. 2024). Historically, surplus precipitation occurred from October to May with deficits occurring from mid-May through mid-September. The warm-wet future is very similar to the historical model while a hot-dry future indicates greater deficit and monthly components model deficits over most of the year (February–October; Runyon et al. 2024). Although it is not clear which future we will encounter, the hot-dry model would result in prolonged water stress for plants. Potential vegetation effects of a hot-dry future are reduced vegetation cover, changes in species composition, or shifts in certain species guilds over time. Planning for these divergent scenarios can help the park to avoid unanticipated consequences of the changing climate.

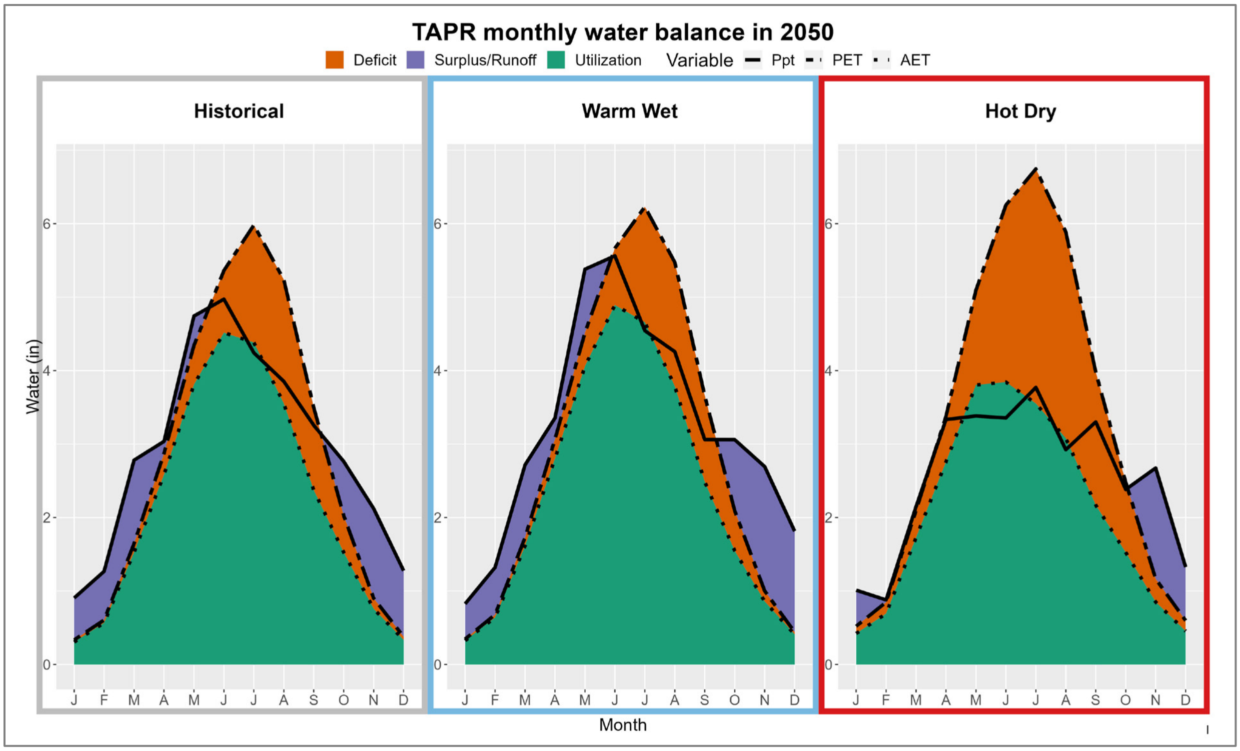

Figure 5. Water balance in the historical period (1979–2012) as compared to midcentury projections for two climate scenarios (warm-wet and hot-dry) for Tallgrass Prairie National Preserve, Kansas. Ppt = precipitation, PET = potential evapotranspiration, and AET = actual evapotranspiration. Water is measured in inches.

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Changes in climate may affect wildfire risk at the park. Models of low emission increases (RCP 4.5) for the 2055 era predict increases of 5.1 (summer) and 3.0 (fall) extreme fire risk days/year over the number of extreme fire days in the 1990s. Winter and spring models predict increases of 0.2 (winter) and 1.5 (spring) extreme fire risk days/year by midcentury (Abatzoglou and Brown 2012). Higher emissions scenarios (RCP 8.5) predict increases of 6.5 and 3.8 extreme fire risk days for summer and fall, respectively (Hegewisch and Abatzoglou 2024)

Recent Fire History

Frequent fire is critical to maintaining ecosystem processes and biotic community needs in the tallgrass prairie (Collins and Steinauer 1998; Bragg 1995; Pyne 2016). As noted in prior reports (Leis and Morrison 2018, 2022), fire frequency shifted from an annual all burn type management paradigm to a more heterogeneity-based strategy. For example, in Figure 6, the top panel shows that most monitoring sites were burned annually (0 years since fire). The most recent panel indicates a trend towards fewer fires and longer time since fire intervals for monitoring sites. In the recent period, nine sites remained unburned for seven years and one site for 10 years.

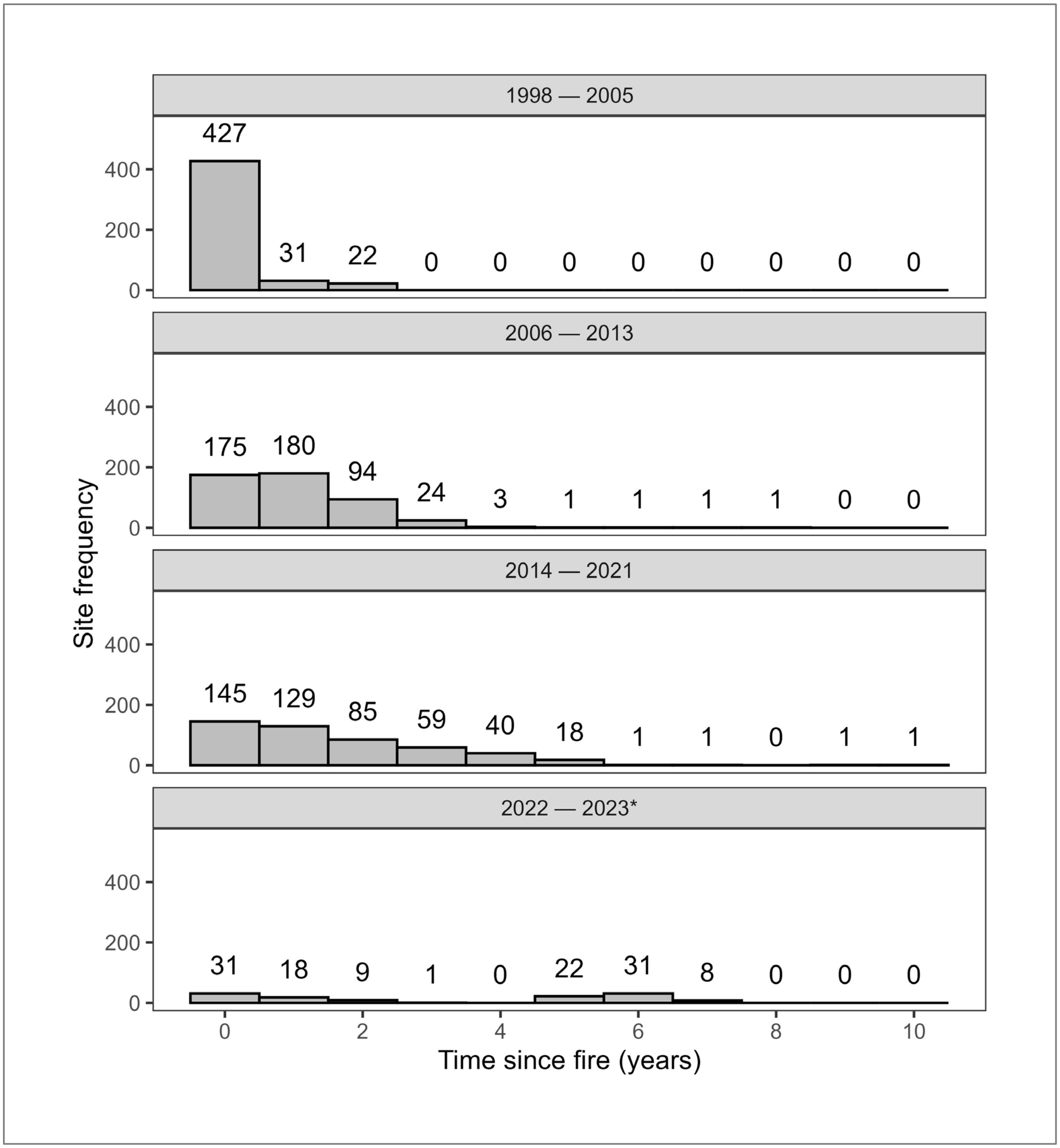

Figure 6. Cumulative frequency of time since fire (years) assignments for each monitoring site for periods between 1998 and 2023 at Tallgrass Prairie National Preserve, Kansas. *The fire history is paneled by eight-year increments except for the 2022–2023 panel where N = 2 years.
NPS

Fire is applied to pastures or more recently to portions of pastures. Mean time since fire intervals were longer for monitoring sites in Crusher Hill Pasture than other pastures (Figure 7). Considering the climate future models discussed above, the hot-dry future may present additional risk for wildfire occurrence and could present constraints for prescribed fire operations (Jonko et al. 2024). While historical fire intervals are a helpful source of information for fire planning, current ecosystem pressures, landscape-level vegetation change (including woody plant encroachment) and climate changes may require modifications to future fire regimes (Miller Hesed 2023).

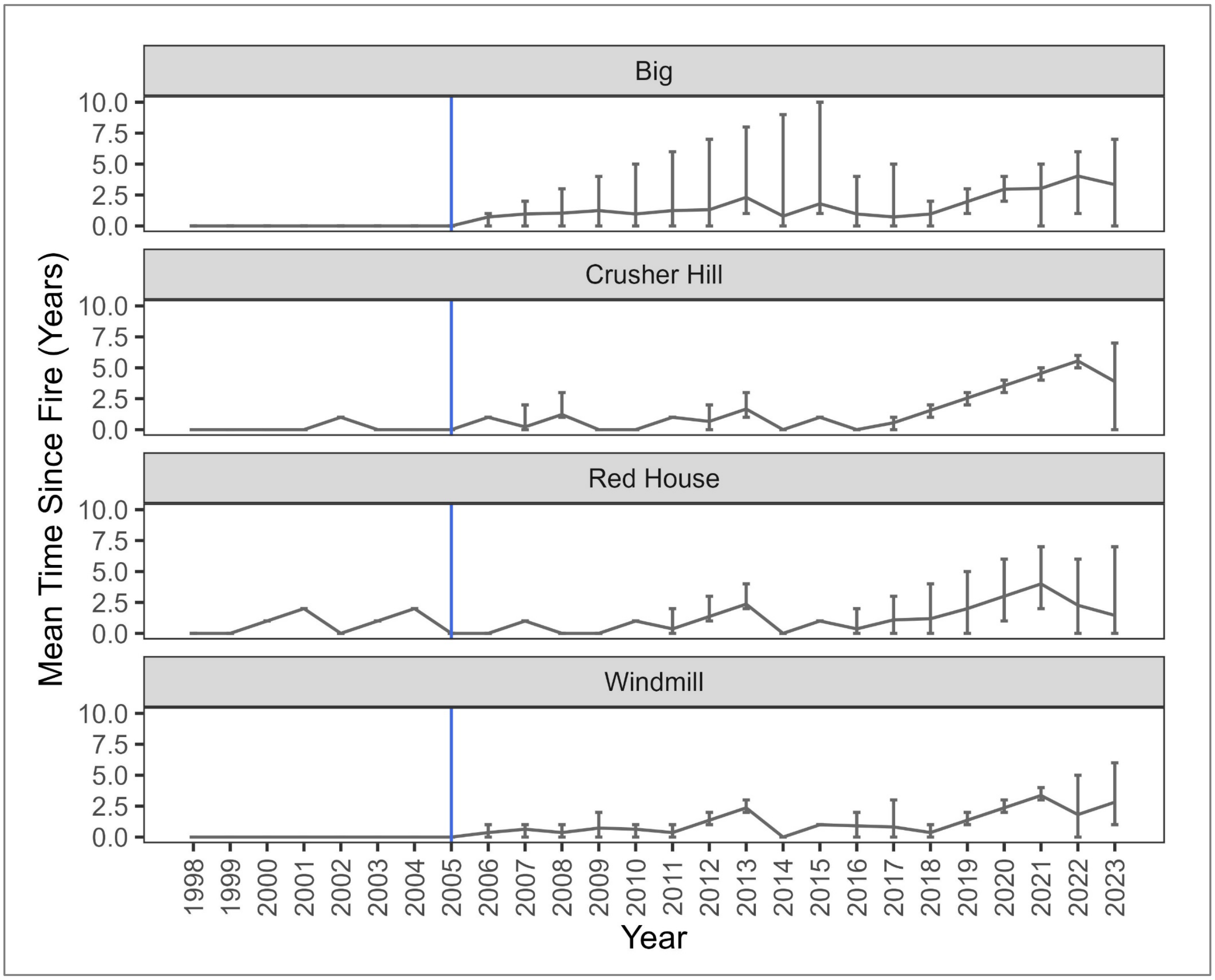

Figure 7. Mean time since fire status for monitoring sites in the western Tallgrass Prairie National Preserve, Kansas, 1995–2023. Fire history is paneled by management unit (pasture). Error bars denote the maximum and minimum time since fire values for that year-pasture combination. The vertical blue bar denotes a shift in management paradigm from livestock production to ecological heterogeneity-based management.

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Challenges to fire application include balancing ecological fire effects with operational needs and availability. For example, woody control might be most effective in the late summer through early fall, but burn crews are often unavailable at that time (during the lengthening wildfire season). Furthermore, previous studies have detected minimal effects of prescribed fire on wind erosion in grasslands (Soto and Diaz-Fierros 1997; Vermeire et al. 2005) but early spring burning may be more likely to affect wind erosion and soil water balance than fall or late spring burning (Anderson 1965; Vermeire et al. 2005).

Grazing

Grazing is an important component of the preserve's mission. It supports the interpretation of the ranching legacy and supports the ecological processes needed for a functioning tallgrass prairie ecosystem. The approach to grazing has changed over time (Leis and Morrison 2018). The density of ungulates was reduced in 2006, and animals were deployed with the synergy of grazing and fire in mind. This was especially true for Big Pasture where a patch-burn grazing system has been in use since 2006. Drought periods have also prompted managers to review stocking rates annually. At the preserve scale, stocking was reduced from 2006 to 2020, and there were small increases in stocking from 2021 to 2023 (Figure 8). In 2022 and 2023, a more flexible grazing strategy was applied to Red House and Crusher Hill pastures to accommodate a research study.

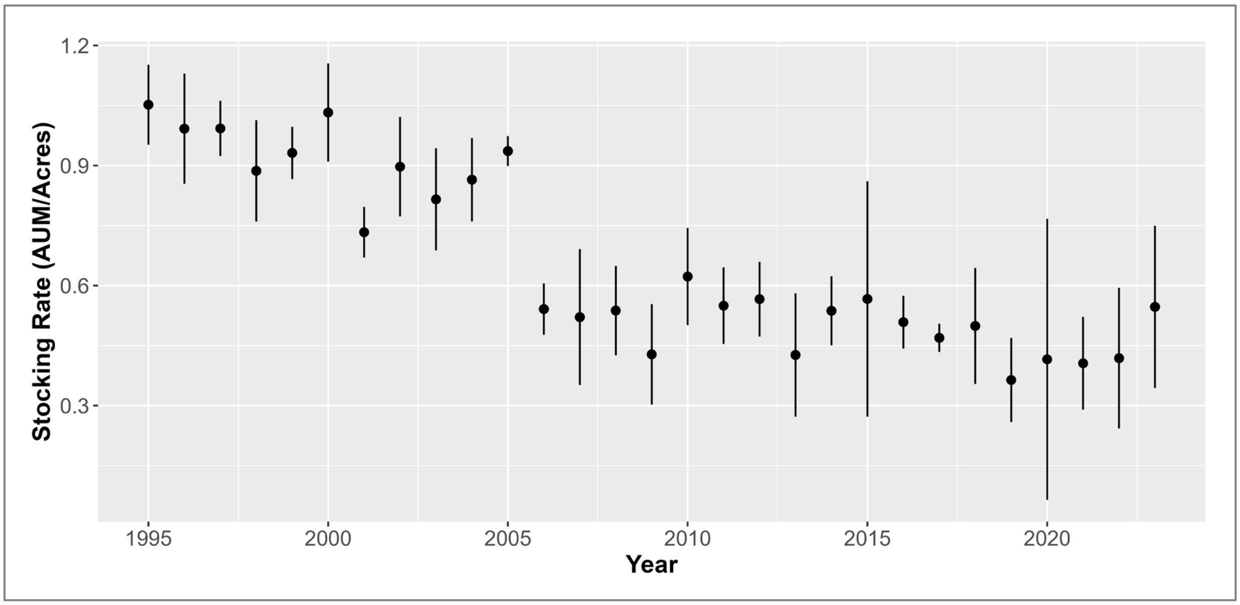

Figure 8. Mean stocking rate (animal unit months [AUM] per acre) at the preserve scale at Tallgrass Prairie National Preserve, Kansas, 1995–2023. Error bars are standard deviations.
NPS

Pasture level stocking trends vary by pasture, but generally represent lighter stocking than pre-2006 values (Figure 9). Big Pasture in particular has continued to receive lighter grazing. The record is incomplete for Windmill Pasture because we did not have weight data once the bison were deployed. We hope to be able to complete the stocking calculations for Windmill Pasture in the future as weight data for the bison become available.

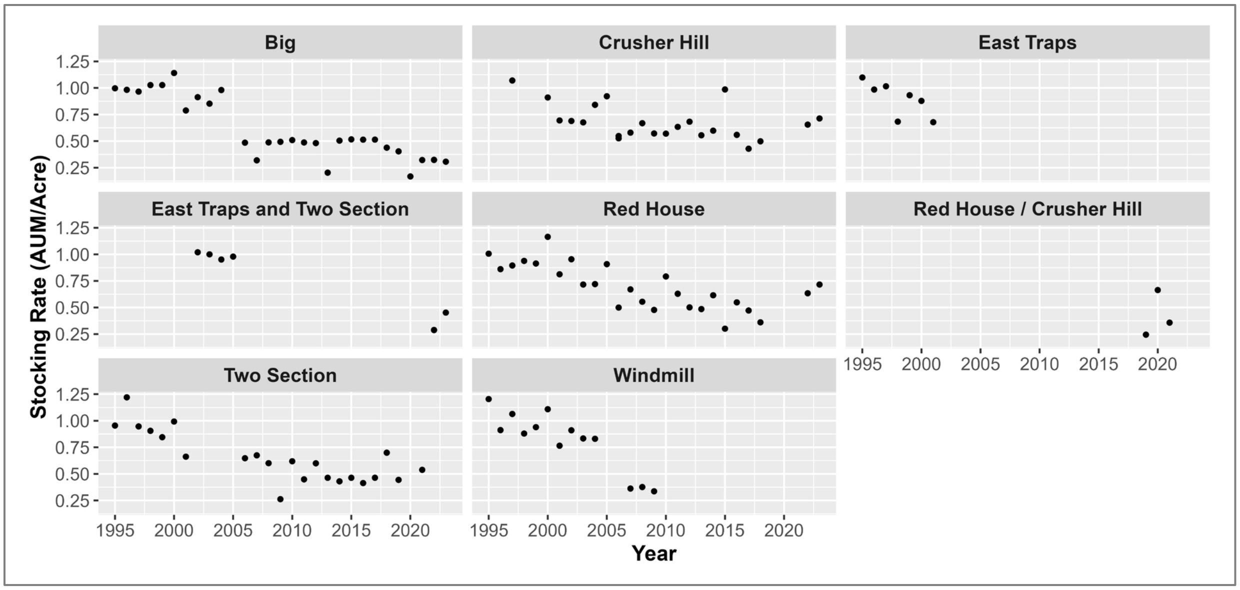

Figure 9. Stocking rate (animal unit months [AUM] per acre) at the pasture scale at Tallgrass Prairie National Preserve, Kansas, 1995–2023. Note that all pastures were stocked with cattle except for Windmill Pasture, which was stocked with bison (*Bison bison*). We were not able to calculate stocking rates once bison were released in 2009.

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Ground Cover

Deciduous leaf litter and woody debris remained minimal (less than 1%) through time (Figure 10). Bare ground exhibited a strong decline since 2007 that coincides with a one- to two-year lag since the shift in management paradigms. Maintaining some bare ground is critical to some wildlife species; however, excessive bare ground can raise concerns for erosion and increased soil evapotranspiration. A management threshold of 30–60% bare ground was established with respect to ideal greater prairie chicken habitat. Mean bare ground for the preserve was within the desired range in 2023 (Figure 11). At the pasture scale, we observed that bare ground fell below desired levels in Crusher Hill and Windmill pastures (Figure 12).

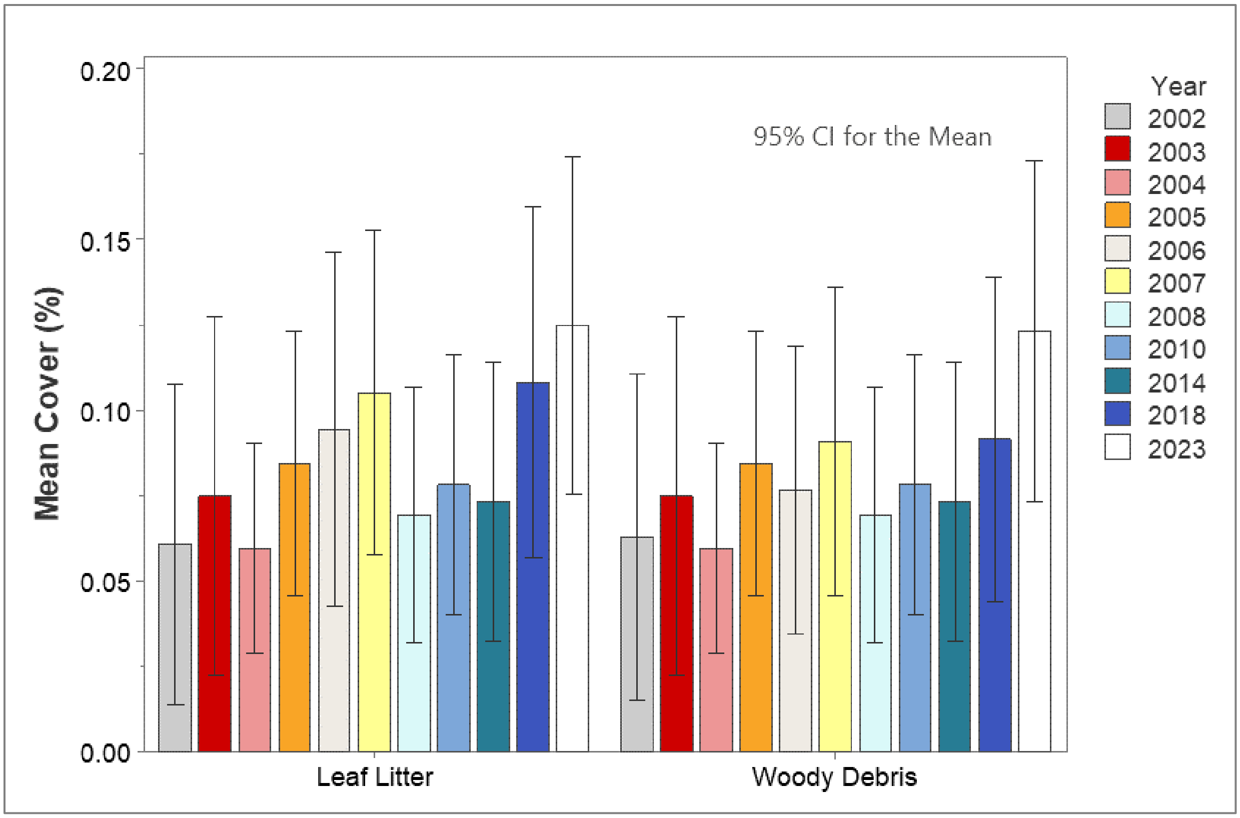

Figure 10. Mean cover for two ground cover types in monitoring sites at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.

NPS

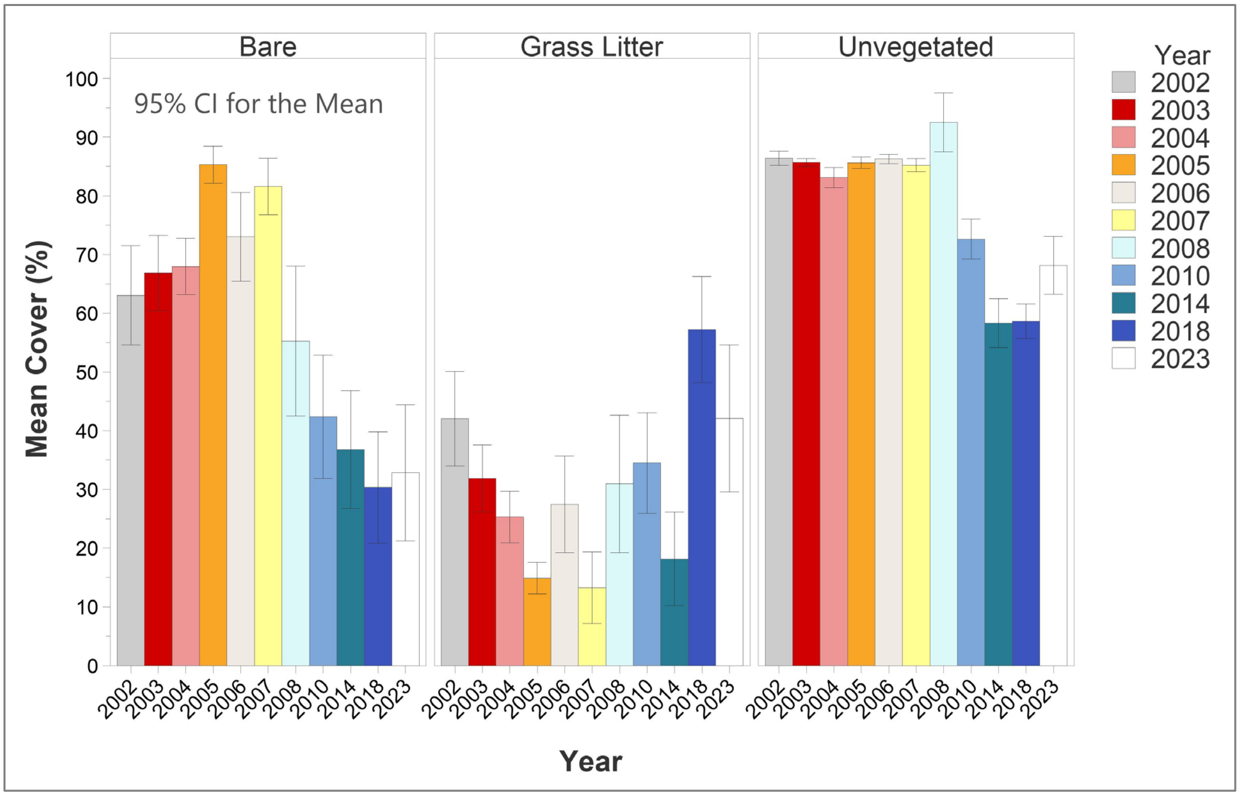

Figure 11. Mean cover for three ground cover types in monitoring sites at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.
NPS

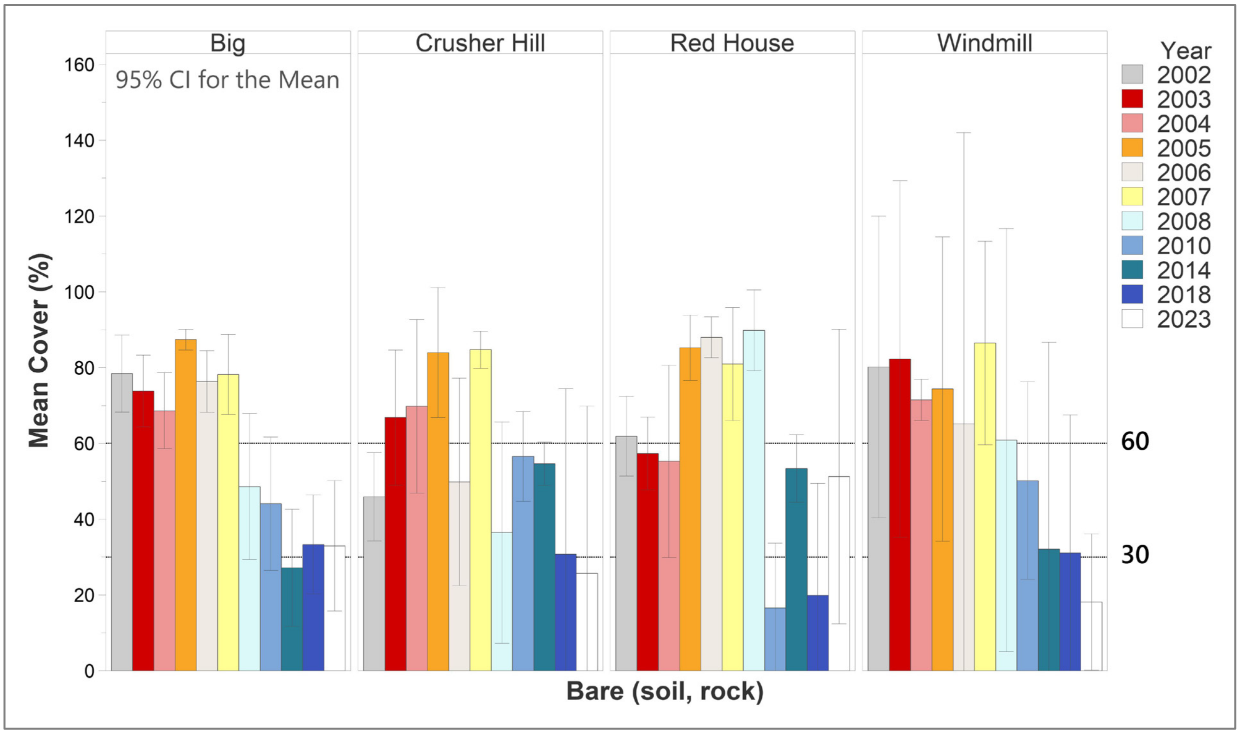

Figure 12. Mean cover of bare soil and rocks in monitoring sites at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Dashed lines indicate desired levels for bare ground cover. Error bars are 95% confidence intervals.
NPS

Grass litter cover varied through time and is somewhat reflective of fire return intervals where increasing litter occurs with increasing time since burn (Figure 11). The confidence interval patterns also reflect how widespread fire might have been across the preserve as well as vegetation productivity via precipitation. Mean litter depth was similar between the 2018 (mean = 5 cm, 95% CI = 3.5–6.8) and 2023 (mean = 4 cm, 95% CI = 2.7–5.2) monitoring events. Management goals for litter cover reflect greater prairie chicken needs for nesting cover where nest success is optimal with < 25% litter cover (McKee et al. 1998). Mean grass litter cover in 2018–2023 exceeded this threshold by a factor of 1.5–2 times, not including deciduous leaf litter.

Deep litter in the tallgrass prairie suppresses productivity (Hulbert 1988). Removal of litter via prescribed fire may allow for soil to warm, increasing recovery and germination rates. Litter removal may also result in increased loss of soil moisture, whereas litter may help facilitate water infiltration and reduce soil respiration (Bragg 1995; Wagle and Gowda 2018)

The preserve's goal for bare ground, 30–60% cover, also reflects habitat needs for greater prairie chickens. At the preserve scale, mean bare areas were similar in the last two monitoring events and were within the desired range in 2023 (Figure 11). At the pasture scale however, mean bare cover was below the threshold for Crusher Hill and Windmill pastures (Figure 12). Crusher Hill Pasture and Red House Pasture exhibited more heterogeneity in bare areas than the other monitored pastures, likely because prescribed fires didn't burn all the sites in the pasture. Windmill pasture, where half the pasture is typically burned at a time, may have additional bare areas resulting from bison behavior (e.g., wallowing) or other inherent soil or geologic variation.

Community Diversity

Guilds

<u>Woody Plants</u>

Preserve mean native woody plant cover continued a slow increase over the last four monitoring events (Figure 13). Woody cover has increased more than 16-fold since 2022 but remains < 1% on average. Although no nonnative species observed were in the woody guild, some native woody plant species can be problematic. Mean preserve native woody cover remains below the 5% management threshold, but it has increased the most in Big and Windmill pastures (Figure 14). The most common and abundant native woody species were Jersey tea (*Ceanothus herbaceous*; most common and abundant in 2023), smooth sumac (*Rhus glabra)*, prairie rose (*Rosa arkansana*), and coralberry (*Symphoricarpos orbiculatus*).

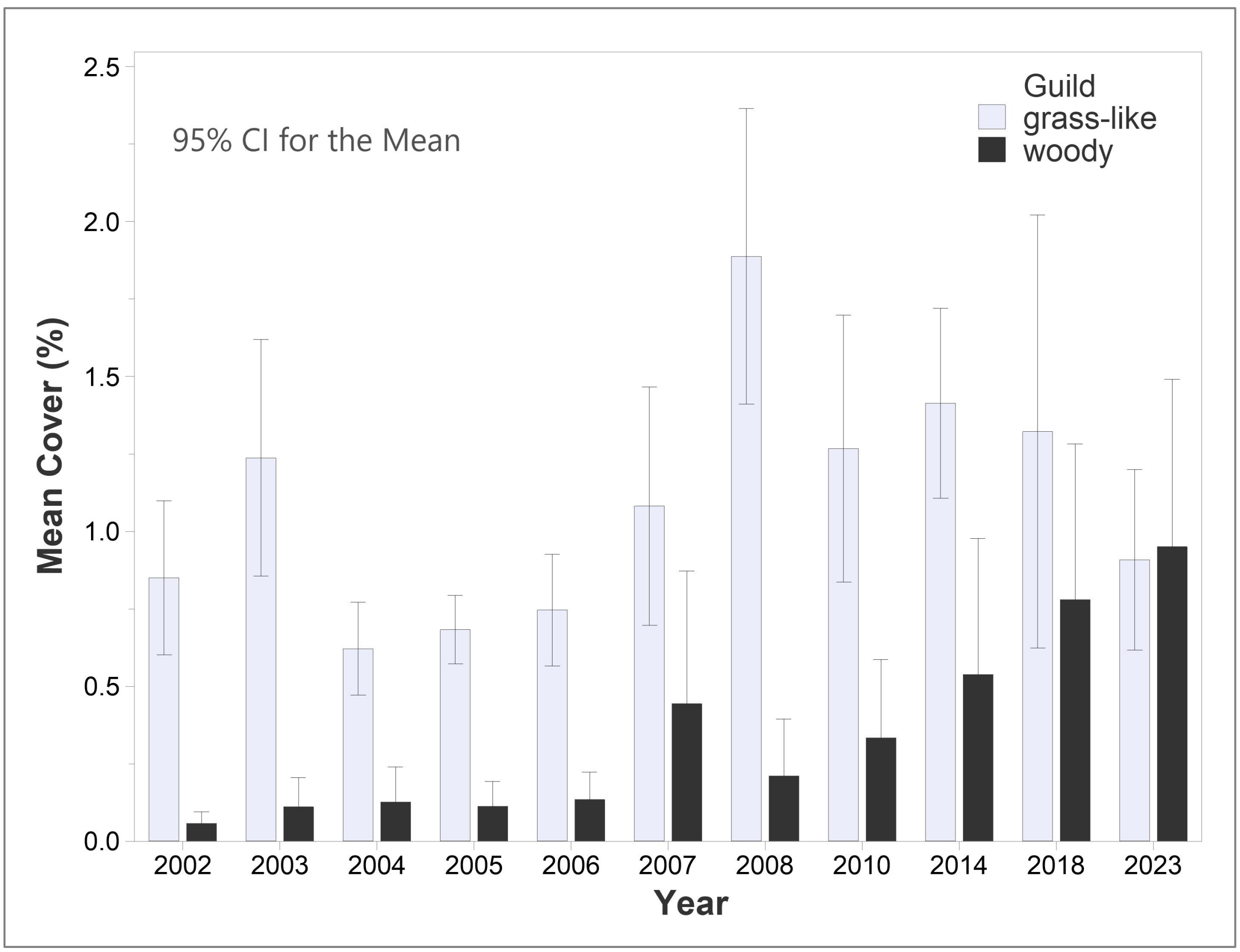

Figure 13. Mean percent cover of native woody plants (non-tree species) and grass-like plants at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals. *NPS*

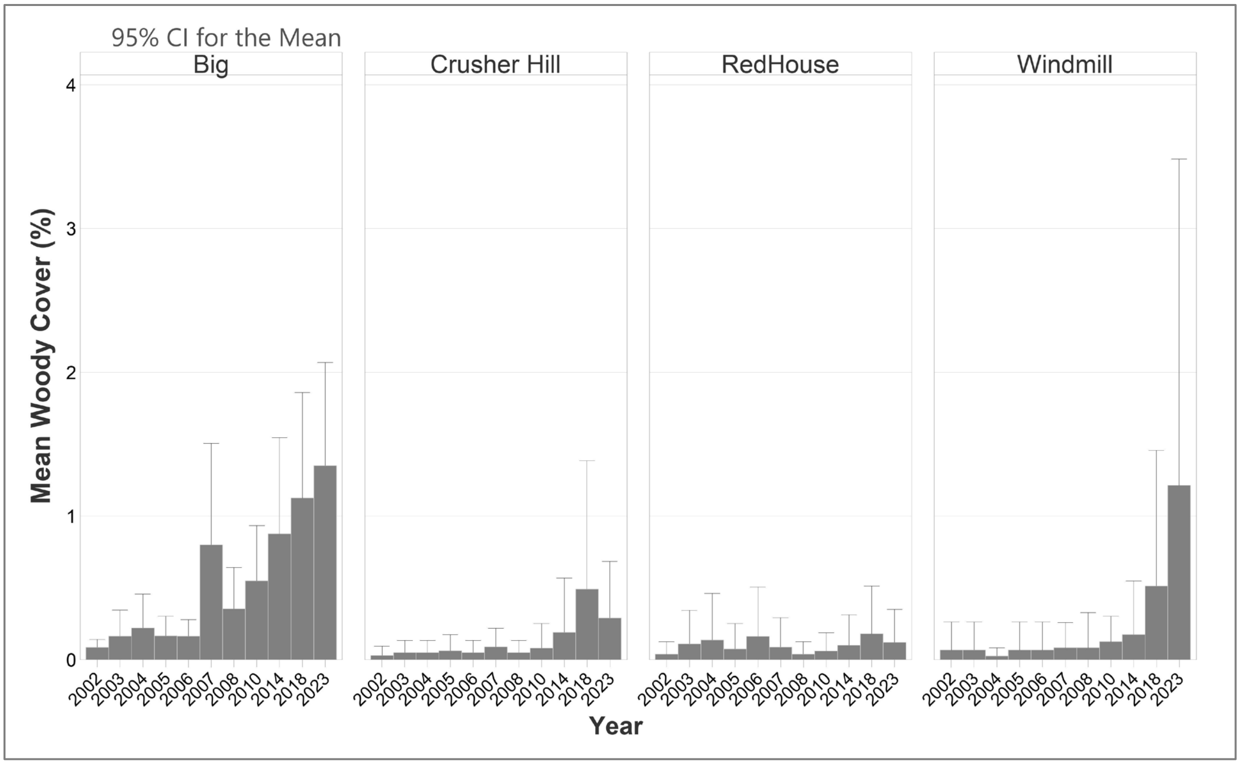

Figure 14. Mean percent cover of native woody plants (non-tree species) by pasture at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.
NPS

We have only rarely observed tree regeneration (seedlings or saplings) in the plant community monitoring sites at Tallgrass Prairie National Preserve. Single elm (*Ulmus* spp.) seedlings were found in site 65 in 2010 and 2014, but seven seedlings of elm and hackberry (*Celtis* spp.) were found across four sites in 2023. While this is a very small number of stems (7 stems), it may be an indicator that tree propagules are increasing. Increased treatments to reduce or prevent tree growth now may slow the spread and maintain the prairie over the coming decades.

A study in nearby Konza Prairie Biological Station noted that although increases in woody plants led to increased carbon sequestration, rates of evapotranspiration also increased. The shrubs were also found to use deep soil water sources (Logan and Brunsell 2015). Increased woody growth in riparian areas led to reductions in stream flows with implications for stream biota and upland species composition (Briggs et al. 2005; Ratajczak et al. 2014; Dodds et al. 2023). Importantly, fire return intervals > 3 years exacerbated woody plant growth, especially after establishment (Briggs et al. 2005). Dogwood species (*Cornus* spp.) were found to be ecosystem transforming at the Konza Prairie Biological Station but were not recorded in our monitoring sites at Tallgrass Prairie National Preserve. However, dogwood has established in other areas of the preserve (personal observation). We did not observe eastern redcedar (*Juniperus virginiana*), another transformative species in the Great Plains, in our monitoring sites (Hoch et al. 2002).

Native Grass-like, Grasses, and Forb Guilds

Grass-like plants peaked in 2008 but remain a small component of the flora in these monitoring sites (Figure 13). Grass cover has ranged from 21.5–84.4% across the monitored years (50.6% ± 5.7 CI in 2023; Figure 15). The least grass cover was observed in 2006, a year when forb cover was similarly low because of drought conditions. Grass cover averaged 51.2% ± 5.5 CI across the whole monitoring record. Forb cover was less variable overall, ranging from 11.7–37.9% cover over time (23.6 ± 5.7 CI in 2023; Figure 15). Two grass species compose the majority of grass cover: big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*; Appendix B).

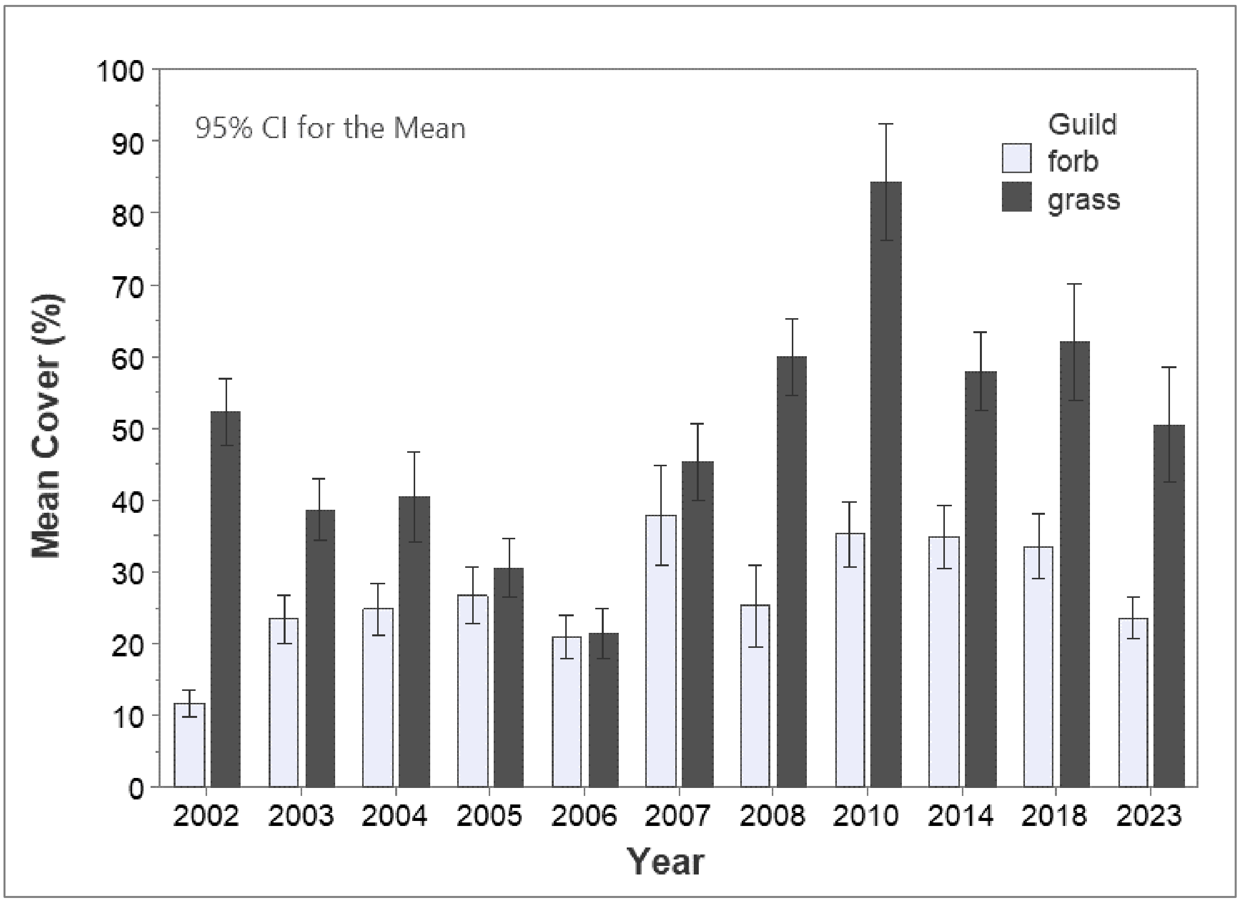

Figure 15. Mean percent cover for native forb and grass guilds at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.
NPS

Of the species observed in 2023, 77% of them were forbs. We noted an increase in the occurrence of goldenrod cover (*Solidago* sp.) across the monitoring sites in the last two monitoring events (Appendix B). We were interested in whether Baldwin's ironweed (*Vernonia baldwinii*) changed over time as it can be an indicator of change in disturbance intensity but saw that the trend varied through time (greatest 2010–2018), and current levels were within the range of variation (Appendix B). It is not clear what was driving the temporary increase.

Nonnative Plants
Monitoring the introduction and expansion of nonnative, especially problematic species, is critical for developing treatments to protect native vegetation. We have observed a limited abundance of nonnative plants since 2002 (Figure 16). The lowest nonnative species cover was from 2005 to 2007, and cover remains within the variation that we have measured through time. However, cover reached its second highest level (13%) in 2023. We observed seven nonnative species in 2018 and 2023, but plants expanded from about half the sites in 2018 to 67% of the sites in 2023. Annual brome (*Bromus arvensis*) was the most abundant of this guild with mean preserve cover of 0.27% but a summed mean site cover of 8%. The annual brome species were lumped in some years, but as a group were previously below 2.5% preserve-wide cover (2014 level), if present. Kentucky bluegrass (*Poa pratensis*) was the second most abundant of the nonnative species with mean preserve cover of < 1% but summed mean site cover was 4.4%. This meets the preserve's goal of < 5% for this species but is approaching the threshold (Hase 2016).

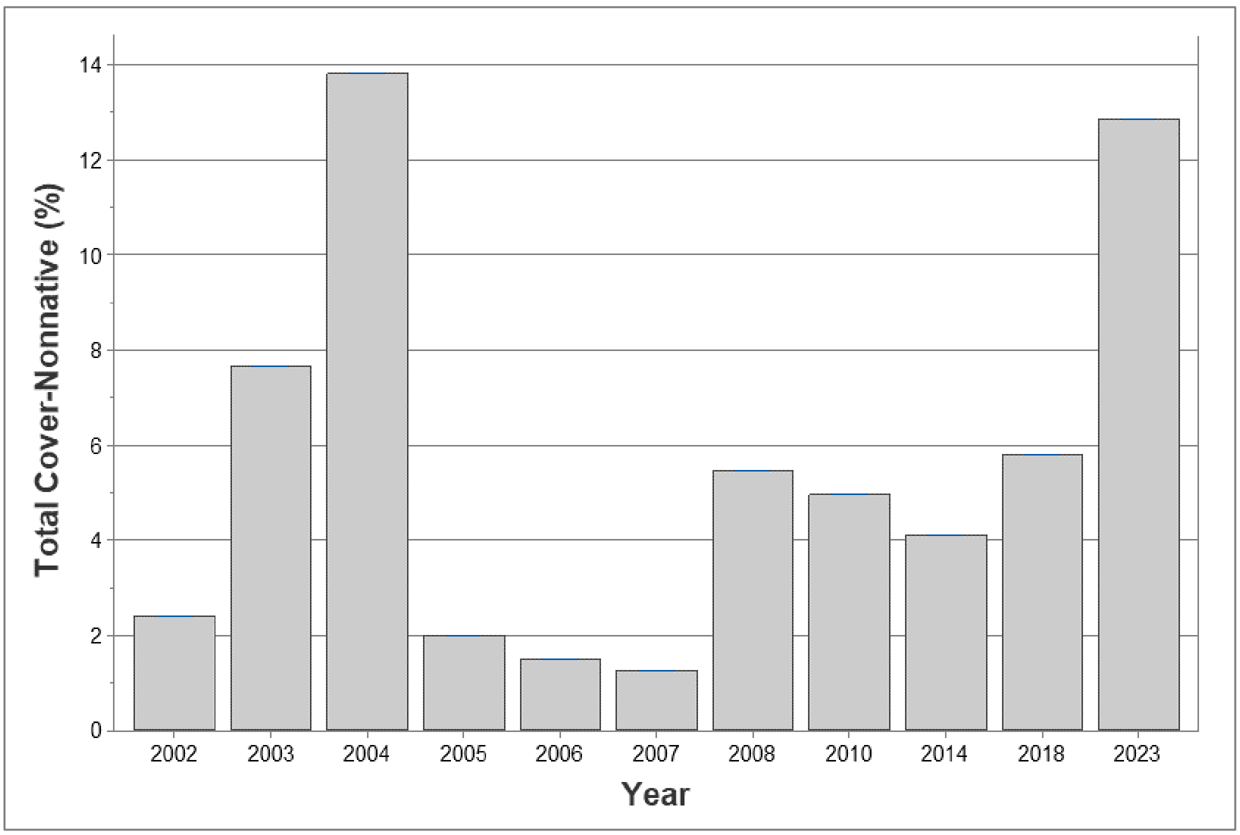

Figure 16. Total percent cover of nonnative species in monitoring sites across Tallgrass Prairie National Preserve, Kansas, 2002–2023.
NPS

Targeted invasive species monitoring will provide more spatially comprehensive results, but the plant community monitoring results reported here provide an indication of nonnative species trends (Kull et al. 2022). Maintaining cool season invasive grasses below target thresholds may become more challenging as climate change progresses. However, as nonnative plants like Kentucky bluegrass increase, management may be more difficult (Toledo et al. 2014). Changes in phenology and precipitation patterns may support cool season plants, like nonnative grasses and woody plants, more so than the climate of the recent past (Knapp et al. 2020). The earlier onset of spring, which is characteristically wet in the Great Plains, may provide an extended period of productivity for C3 plants during which native C4 plants remain dormant. This potential shift could affect the abundance of C3 and C4 functional groups. Prescribed fire during the dormant season may also provide early growth opportunities for C3 plants when C4 plants are still dormant. Vigilant monitoring and prioritization of problematic plant treatment strategies, along with supporting dominance of native species through ecological processes like grazing and fire, may contribute to resilience as climate change progresses (Hobbs and Huenneke 1992; Vujnovic et al. 2002).

Species Richness and Composition

We calculated native species richness (*alpha* diversity) and preserve-wide richness (*gamma* diversity) for all sites (Figure 17). Although we noticed an increasing pattern of *gamma* diversity when all 30 sites were included, the pattern was likely the result of an increasing number of sites sampled over time (Table 1). For this reason, we also plotted *gamma* diversity for the original 18 sites.

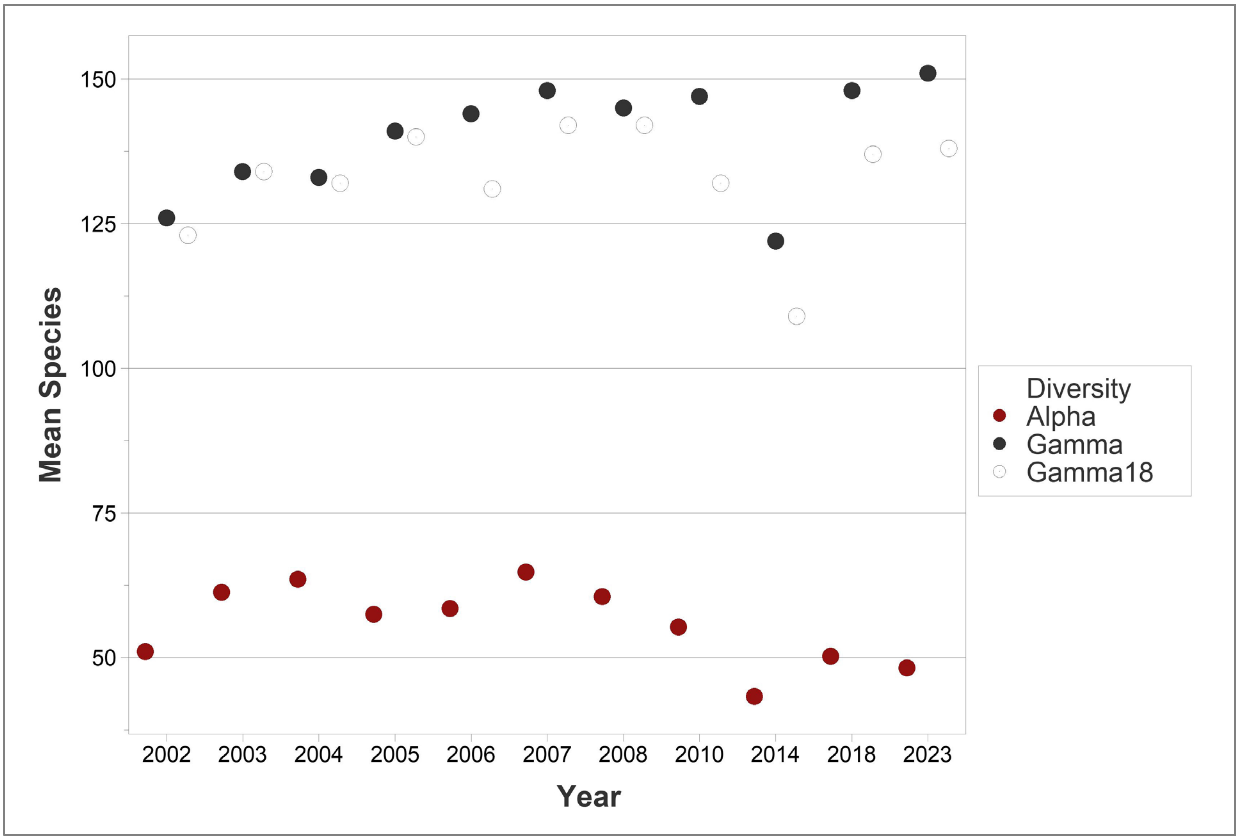

Figure 17. Native plant community diversity metrics (*gamma* diversity [preserve-wide richness], and *alpha* diversity [site-level richness]) for Tallgrass Prairie National Preserve, Kansas, 2002–2023. Gamma 18 represents the original core 18 monitoring sites while the other metrics include the 30 core sites.
NPS

Species stability at the preserve is greater at the landscape scale than at the site scale. Comparison of native species composition in the core 18 sites between 2002 and 2023 revealed a 62.9% (± 5.7 CI) similarity based on the Sørensen Index. Composition based on the Sørensen Index was predictably more similar between 2018 and 2023 (74.1% ± 5.4 CI; Meng et al. 2023). Our assessment was at the plot scale, but Meng et al. (2023) found that compositional stability over time differed at local versus landscape scales. Similar to the findings in Meng et al. (2023), we found that a parkwide comparison of species for 2002–2023 and 2018–2023 revealed more similar composition (Sørensen Index was 73.6% and 84.4%, respectively) than the site-wise comparisons. Heterogeneity resulting from varied disturbance (fire and grazing) and edaphic features (e.g., soil type, slope, and aspect) allows species to be maintained across the preserve even when local communities are in flux because of these disturbances. Further investigation of the relationships of compositional change over time at different scales may be informative (Xu et al. 2021).

The Shannon diversity index (H′) includes both species richness and percent cover in the calculation, with larger values indicating greater diversity (Figure 18). Mean H′ declined over time, especially 2006–2014 ($H' = 57.99 - 0.03 \times year$, $F = 55.40$, $P < 0.001$, adjusted $R^2 = 21.6\%$). H′ peaked in 2003 (2.90 ± 0.06 SE). The declining diversity lagged about a year behind the decrease in disturbance intensity that began in 2006. With time since fire estimates being more similar from 2018 to the present than in the earlier years of monitoring, diversity may reflect a reduction in fire and grazing. Although we did not test this, reduced disturbance may shift the abundance of annual versus perennial plants. Moreover, a reduction in shortgrass graminoid species noted in 2018 may also reflect the reduced disturbance (Leis and Morrison 2022). Fire and grazing reduce competition for light, favoring common species in the shortgrass prairie region of the Great Plains that require more open canopy and drier conditions.

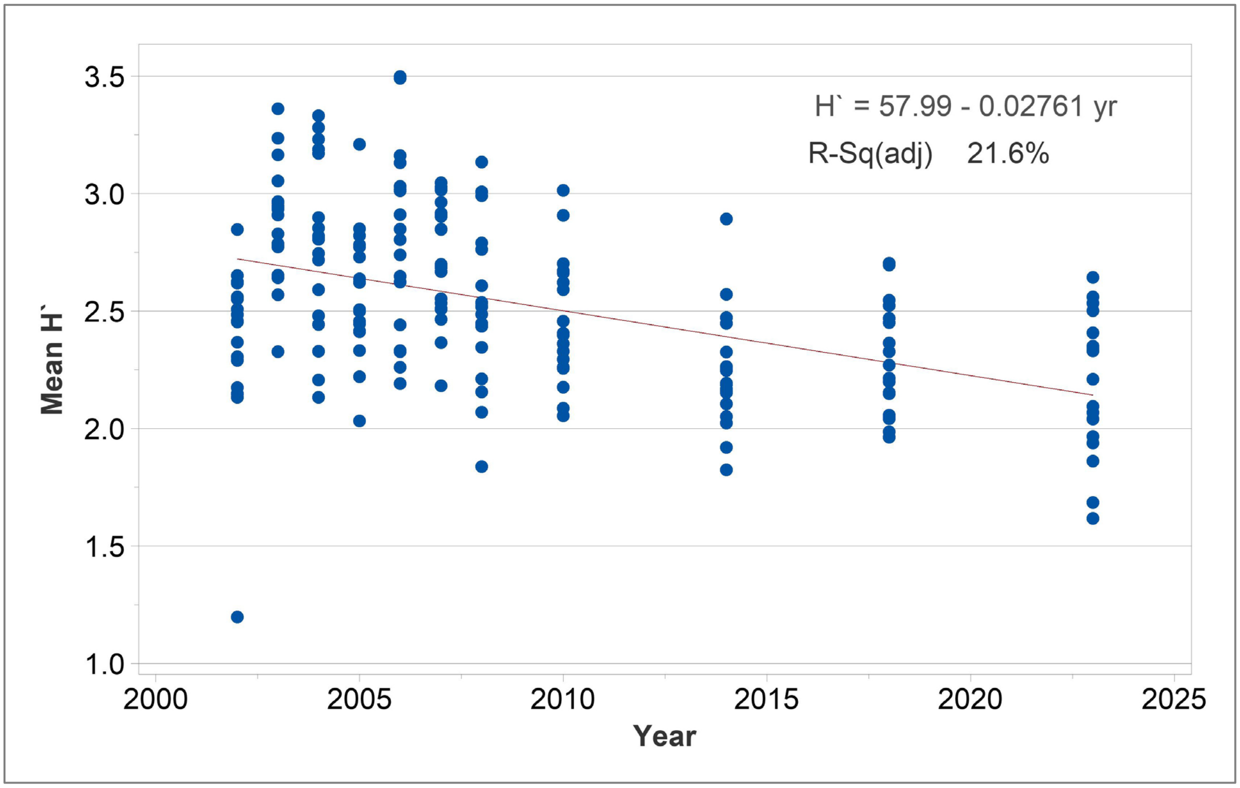

Figure 18. Mean H′ (Shannon diversity index) for vegetation in 18 core monitoring sites at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Significant linear regression is shown as a solid, red line. (H′ = 57.9 − 0.03 × year, F = 55.40, $P < 0.001$, adjusted R^2 = 21.6%)
NPS

Evenness of species among sites appears to be declining, although error bars on the 2023 mean overlap with prior means in 2014, 2018, and some overlap in 2002 (Figure 19). Values of 1 for this index indicate species are equally distributed, but values closer to 0 indicate uneven distribution of species. The current declining trend appears to signal that species distributions could be shifting among monitoring sites such that the preserve could become more heterogeneous in composition.

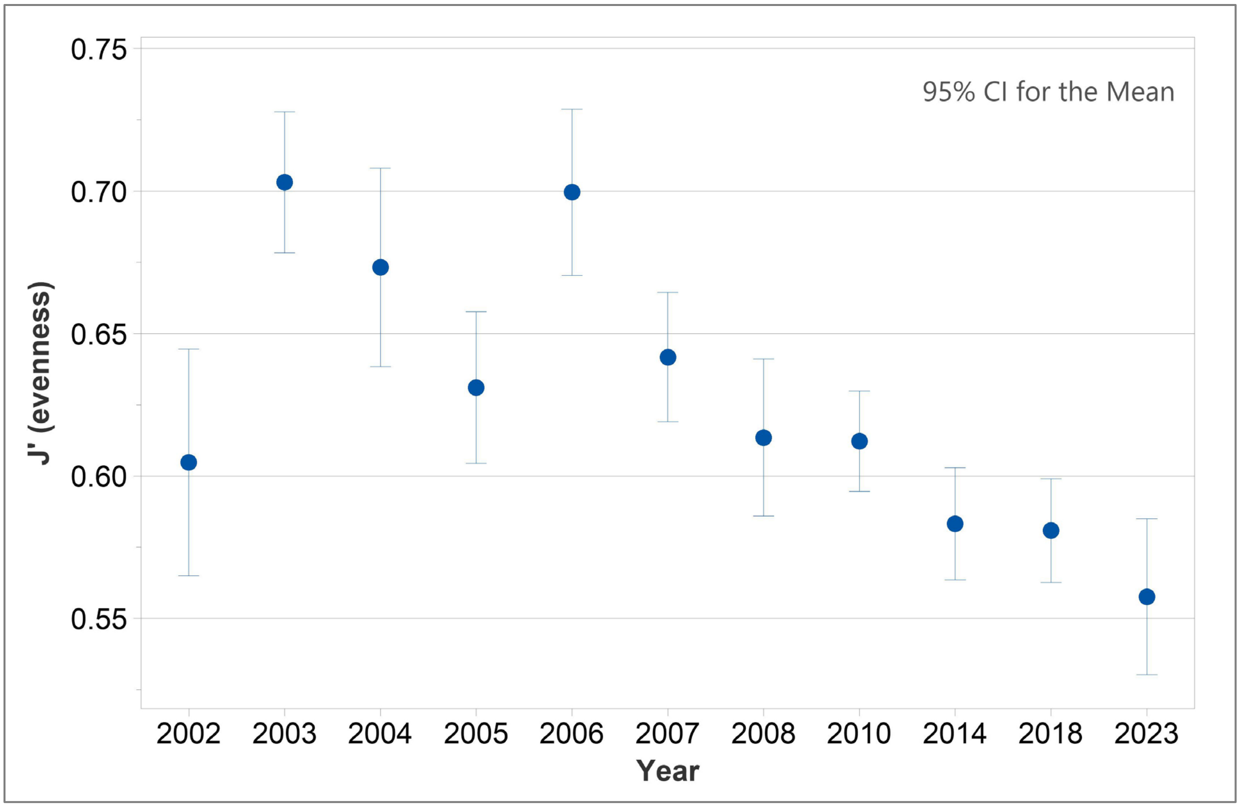

Figure 19. Mean J′ (species evenness) for vegetation in 30 core monitoring sites at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.
NPS

Beta diversity values were variable through 2008 but have been increasing from 2010 to 2023 (Figure 20). *Beta* diversity provides a measure of community heterogeneity among the monitored sites. Increasing *beta* diversity indicates that sites are becoming more distinct in terms of their species composition. Values < 1 indicate sites are very similar and values > 5 indicate sites are increasingly dissimilar (McCune and Grace 2002). Changes in *beta* diversity may be the result of the current management paradigm (fire and grazing intensity) or other factors.

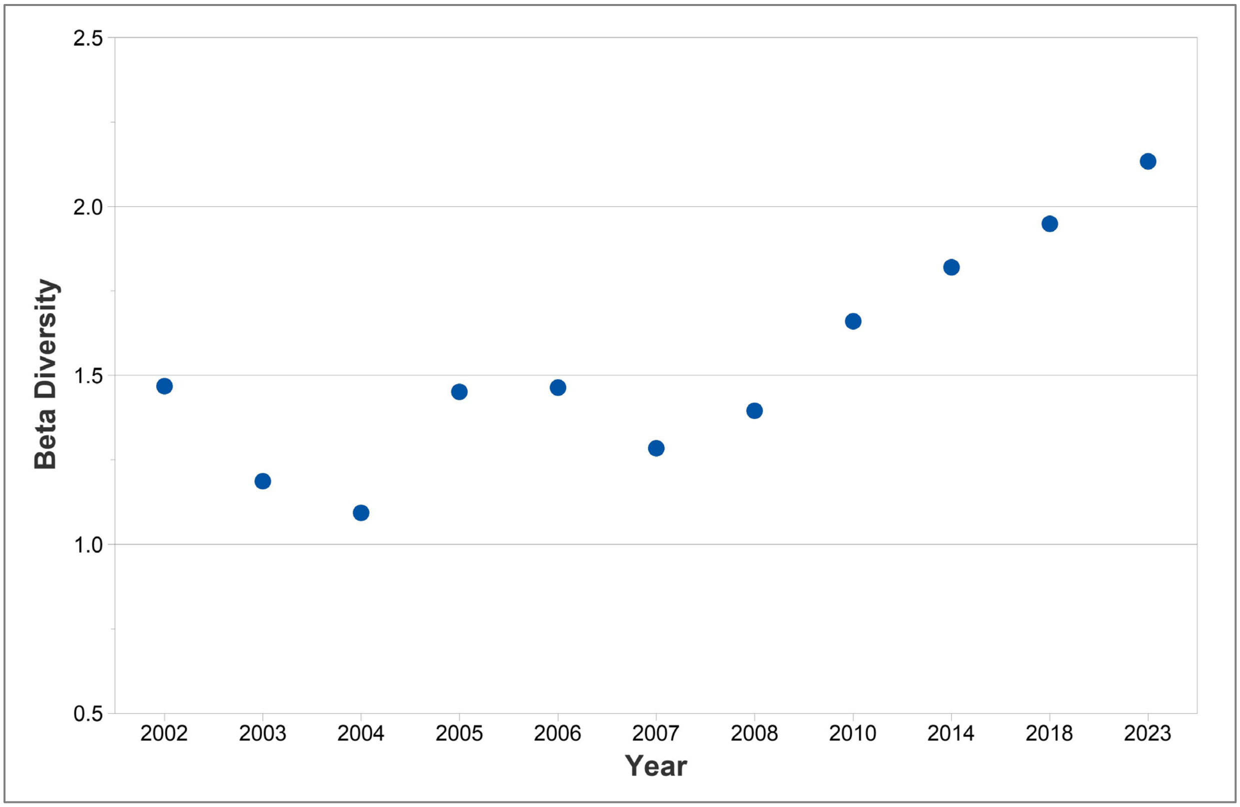

Figure 20. Mean *beta* diversity for vegetation in 30 core monitoring sites at Tallgrass Prairie National Preserve, Kansas, 2002–2023.
NPS

Although Shannon diversity (H′) has been declining, species composition of monitoring sites is becoming more heterogeneous (J′ and *beta* diversity). Disturbance intensity at the preserve began declining in 2006 with the aim of creating a heterogeneous grassland structure supporting grassland obligate species. Although there seems to be a lag in the response of diversity measures, species composition of monitoring sites is trending toward more distinct local communities. More in-depth analysis could include analyzing pasture-level diversity or correlation of community diversity with time since fire, stocking rates by pasture, or climatic variables. The influence of shifting sample sizes on *beta* diversity and evenness has also not been investigated. Other unknown factors, including climatic changes, may have influenced the trend as well.

Observer Error

Observer error for the ground flora community was measured for the second time in 2023 (Table 3). Observer error has multiple components. In this study, we quantified overlooking error (i.e., when a species was present but not noticed by an observer), misidentification error (i.e., when a species was noticed but incorrectly identified), and cautious error (i.e., when one observer lumped an observation to genus level and the other observer identified it to species). The greatest source of error came from overlooking of species by one of the observers (16.2%). Misidentification error and cautious error were low (1.3% and 0.2%, respectively). Total pseudoturnover is a function of all three types of error and averaged 17.7% for the sampling event, resulting in 82.3% agreement (Table 3), which exceeded our goal of 80% based on a survey of the literature (Morrison 2016; Morrison et al. 2020). We have mitigated for sample frame relocation using flagging, but trampling from the first plot read may affect the second reader's ability to recognize the plants. We do not have a quantitative assessment of the contribution of this source of error.

Table 3. Observer error rates for 2018 and 2023 at Tallgrass Prairie National Preserve, Kansas (N = 30).

Error Component	Mean Error Rate 2018 (%)	Mean Error Rate 2023 (%)
Overlooking	18.6	16.2
Misidentification	1.4	1.3
Cautious	0.6	0.15
Total Pseudoturnover	20.5	17.7

Our rate of pseudoturnover was less in 2023 than in 2018. Reduction in the error rate may be related to reducing relocation error of the plots via flag placement or from the team composition. We included a botanical taxonomy expert on the team in 2023, and two of the team members were present in both surveys.

We also assessed the amount of agreement in the cover classes for species that both observers recorded. Overall, on average there was 75.7% agreement in cover classes; 21.7% of observations were within one cover class and 2.6% of observations differed by > 1 cover class. This level of agreement falls within the range of what we have measured across five network parks (range: 73–85% agreement; Morrison et al. 2020; Leis and Morrison 2024; Leis and Morrison 2022).

If pseudoturnover is taken into account, the change in species composition based on the Sørensen Index is considerably less compared to the uncorrected value (37.1% dissimilarity − 17.7% pseudoturnover = 19.4% actual change); however, compositional change increased by 3.3% over the amount calculated for the 2002–2018 period. Mean annualized turnover was 1.1% of species/year in 2023. This was a similar rate of turnover to other parks with tallgrass prairie (range of 0.67–1.1% species/year; Leis and Morrison 2024). However, this correction for pseudoturnover can only be considered an approximation since observer error was measured at the plot scale rather than at the site scale, and no pseudoturnover estimates are available for 2002 when different observers were involved. Nevertheless, the increase in compositional change coincides with declines in species richness and Shannon diversity.

Ground cover class assignments agreed for 73.9% of the observations and were within 1 cover class for 25.0% of the observations. Only 1.1% of observations differed by > 1 cover class (two bare ground observations differed by two classes). This level of agreement met our goals. We compared litter depth estimates for only N = 28 plots because of missing data. Litter depth measurements proved more difficult to replicate than other measurements. More than one third (35.7%) of observations were in exact agreement, while 39.3% of observations were within 1 cm. A quarter

(25.0%) of observations differed by more than 1 cm, with a maximum difference of 7 cm in one plot. Lesser litter depths proved to be more easily replicated between observers. While trampling during the first plot read could contribute to the discrepancies, we have not evaluated sources of error.

Protocol Changes

We expected a small decline in species observed when the revisit design changed in 2014 (James et al. 2009; Appendix B). In prior years, sites were visited twice in a sample year, but beginning in 2014, sites were only visited once. It is also possible that some detected species were not phenologically at the same level of cover as when previously recorded. For example, warm season grasses tend to peak in late July to early August, so there could be phenological reasons for the lower cover levels in June. It is also not clear what effect climate related phenology shifts have on the data collection. It is unclear whether the change in sample design contributed to the patterns we observed for species cover and diversity metrics in 2014–2023 as compared to prior years.

Management Implications

Tallgrass Prairie National Preserve manages tallgrass prairie for a variety of species and ecological objectives, including maintaining populations of greater prairie-chicken, an iconic umbrella species. The preserve continues to have characteristic tallgrass prairie species with a small shift in native species composition. Vegetation structure has changed over time but is currently meeting objectives, including for heterogeneity, a key indicator for endemic species like the greater prairie-chicken. Continued evaluation of fire frequency and grazing intensity will be critical to achieving ecological goals, including conserving the greater prairie-chicken. A focus on achieving the desired fire return intervals to control woody plants may help maintain the desired plant community structure over time. However, as climate change progresses there could be a reduction in available burn days because of increased temperatures, moisture levels, and potentially windier conditions.

The Flint Hills of Kansas and the Great Plains region are experiencing expansion of woody plants, particularly trees (Briggs et al. 2002; Morford et al. 2022). While the preserve has managed to avoid tree invasion in our monitoring sites, it is of note that a small cohort of tree seedlings was found in 2023 where they had largely been absent for the entire prior monitoring record. Continued efforts to keep tree growth in check on the preserve as well as in neighboring properties will be critical going forward.

Development of grazing plans with prescribed stocking rates that are consistent with the preserve's ecological and cultural objectives will also be important. Plans could also identify alternative herbivores or expansion of bison to be deployed in target areas for specific objectives. Recently, management units were subdivided into smaller compartments, which may facilitate focal grazing options. For example, multispecies grazing could reduce problematic woody plant growth in draws or riparian areas. Consideration of coat color or drought tolerance for livestock breeds may be helpful in reducing heat stress. Earlier phenology of vegetation should factor into grazing period planning as well. For example, areas with abundant cool season grass invaders could receive early targeted grazing. During drought periods, already diminished water sources may experience increased pressure.

Because of the natural variability in climate metrics and complexity of the disturbance history and vegetation, further investigation of climate related trends may be helpful, especially in planning for the future. Climate change planning for the near future should focus on the effects of increasing temperatures and phenological shifts as well as intermittent water deficit. Climate planning should consider increasing wildfire risk, temperature effects on cattle and bison, and water availability for animals. Timing of prescribed fire to avoid high risk time periods and implementing protective measures may help support a healthy prairie community.

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Appendix A

Table 4 shows all species observed in prairie monitoring plots at Tallgrass Prairie National Preserve.

Table 4. All species observed in the prairie sites (N = 30) at Tallgrass Prairie National Preserve, Kansas. Values for mean cover, standard error (SE) and occurrence are for 2023 (most recent monitoring event) only. N = native, I = introduced.

Species	Common Name	Guild	Origin	2023 Mean Cover (%)	SE	Occurrence (% Sites Observed) 2023
Acalypha virginica	Virginia threeseed mercury	forb	N	< 0.01	< 0.01	10
Achillea millefolium	common yarrow	forb	N	0.18	0.03	83
Agrostis hyemalis	winter bentgrass	grass	N	0.04	0.02	23
Ambrosia artemisiifolia	annual ragweed	forb	N	0.09	0.05	33
Ambrosia psilostachya	Cuman ragweed	forb	N	0.86	0.3	100
Amorpha canescens	leadplant	forb	N	7.62	0.94	100
Andropogon gerardii	big bluestem	grass	N	23.68	2.54	100
Antennaria neglecta	field pussytoes	forb	N	0.03	0.02	27
Apocynum cannabinum	Indianhemp	forb	N	< 0.01	< 0.01	7
Arnoglossum plantagineum	Tuberous Indian plantain	forb	N	0.02	0.01	30
Artemisia ludoviciana	white sagebrush	forb	N	0.84	0.24	100
Asclepias sp.	milkweed	forb	N	< 0.01	< 0.01	7
Asclepias stenophylla	slimleaf milkweed	forb	N	0.01	0.01	10
Asclepias tuberosa	butterfly milkweed	forb	N	< 0.01	< 0.01	10
Asclepias verticillata	whorled milkweed	forb	N	0.03	0.01	40
Asclepias viridiflora	green comet milkweed	forb	N	0.03	0.01	33
Asclepias viridis	green antelopehorn	forb	N	0.14	0.02	87
Astragalus crassicarpus	groundplum milkvetch	forb	N	0.01	< 0.01	13
Astragalus lotiflorus	lotus milkvetch	forb	N	< 0.01	< 0.01	3
Baptisia australis	blue wild indigo	forb	N	0.11	0.02	67
Baptisia bracteata var. *leucophaea*	longbract wild indigo	forb	N	0.05	0.01	43
Bouteloua curtipendula	sideoats grama	grass	N	0.43	0.11	97
Bouteloua dactyloides	buffalograss	grass	N	0.04	0.01	40
Bouteloua gracilis	blue grama	grass	N	0.04	0.02	30
Bouteloua hirsuta	hairy grama	grass	N	0.1	0.04	37

[A] Five species of *Carex* were observed in the monitoring plots. In order of declining frequency, the five species were *C. meadii, C austrina, C. inops, C. brevior, and C. gravida*.

[B] *Cyperus mesocharus* and a hybrid species of *Cyperus* were identified.

Table 4 (continued). All species observed in the prairie sites (N = 30) at Tallgrass Prairie National Preserve, Kansas. Values for mean cover, standard error (SE) and occurrence are for 2023 (most recent monitoring event) only. N = native, I = introduced.

Species	Common Name	Guild	Origin	2023 Mean Cover (%)	SE	Occurrence (% Sites Observed) 2023
Brickellia eupatorioides	false boneset	forb	N	0.24	0.03	83
Bromus arvensis	Japanese chess	forb	I	0.26	0.2	10
Callirhoe alcaeoides	light poppymallow	forb	N	0.22	0.03	80
Calylophus serrulatus	yellow sundrops	forb	N	< 0.01	< 0.01	3
Calystegia sepium	hedge false bindweed	forb	N	0.01	0.01	7
Carex sp. [A]	sedge	grass-like	N	0.55	0.06	100
Ceanothus herbaceus	Jersey tea	woody	N	0.55	0.21	57
Cerastium brachypodum	shortstalk chickweed	forb	N	< 0.01	< 0.01	3
Chamaecrista fasciculata	partridge pea	forb	N	0.01	0.01	3
Chamaesyce sp.	sandmat	forb	N	0.04	0.01	37
Chamaesyce prostrata	prostrate sandmat	forb	N	< 0.01	< 0.01	3
Cirsium altissimum	tall thistle	forb	N	0.05	0.01	50
Cirsium undulatum	wavyleaf thistle	forb	N	0.11	0.03	63
Croton capitatus	hogwort	forb	N	0.01	< 0.01	10
Croton monanthogynus	prairie tea	forb	N	0.01	0.01	13
Cyperus sp. [B]	flatsedge	grass-like	N	< 0.01	< 0.01	3
Cyperus lupulinus	Great Plains flatsedge	grass-like	N	0.07	0.02	50
Dalea aurea	golden prairie clover	forb	N	< 0.01	< 0.01	3
Dalea candida	white prairie clover	forb	N	0.04	0.01	53
Dalea multiflora	roundhead prairie clover	forb	N	0.01	< 0.01	10
Dalea purpurea	purple prairie clover	forb	N	0.15	0.03	60
Delphinium carolinianum	Carolina larkspur	forb	N	0.03	0.01	30
Desmanthus illinoensis	Illinois bundleflower	forb	N	0.01	0.01	7
Desmodium sessilifolium	sessileleaf ticktrefoil	forb	N	0.05	0.02	37
Dichanthelium oligosanthes	Heller's rosette grass	grass	N	< 0.01	< 0.01	3
Dichanthelium oligosanthes var. *scribnerianum*	Scribner's rosette grass	grass	N	0.44	0.04	100

[A] Five species of *Carex* were observed in the monitoring plots. In order of declining frequency, the five species were *C. meadii, C austrina, C. inops, C. brevior, and C. gravida*.

[B] *Cyperus mesocharus* and a hybrid species of *Cyperus* were identified.

Table 4 (continued). All species observed in the prairie sites (N = 30) at Tallgrass Prairie National Preserve, Kansas. Values for mean cover, standard error (SE) and occurrence are for 2023 (most recent monitoring event) only. N = native, I = introduced.

Species	Common Name	Guild	Origin	2023 Mean Cover (%)	SE	Occurrence (% Sites Observed) 2023
Dichanthelium villosissimum var. *praecocius*	whitehair rosette grass	grass	N	0.02	0.01	17
Digitaria cognata	fall witchgrass	grass	N	0.01	0.01	17
Draba cuneifolia	wedgeleaf draba	forb	N	< 0.01	< 0.01	3
Echinacea angustifolia	blacksamson echinacea	forb	N	0.03	0.01	17
Eleocharis sp.	spikerush	grass-like	N	0.23	0.11	27
Elymus canadensis	Canada wildrye	grass	N	0.04	0.03	17
Elymus virginicus	Virginia wildrye	grass	N	< 0.01	< 0.01	7
Eragrostis spectabilis	purple lovegrass	grass	N	0.06	0.06	10
Erigeron philadelphicus	Philadelphia fleabane	forb	N	0.02	0.01	20
Erigeron strigosus	prairie fleabane	forb	N	0.07	0.02	43
Escobaria missouriensis	Missouri coryphanthe	forb	N	< 0.01	< 0.01	3
Euphorbia corollata	flowering spurge	forb	N	< 0.01	< 0.01	3
Euphorbia dentata	toothed spurge	forb	N	< 0.01	< 0.01	3
Euphorbia spathulata	warty spurge	forb	N	< 0.01	< 0.01	10
Euthamia gymnospermoides	Texas goldentop	forb	N	0.07	0.05	23
Evolvulus nuttallianus	shaggy dwarf morning-glory	forb	N	0.01	0.01	13
Geranium carolinianum	Carolina geranium	forb	N	0.01	0.01	10
Geum canadense	white avens	forb	N	< 0.01	< 0.01	3
Hieracium longipilum	hairy hawkweed	forb	N	0.02	0.01	13
Hordeum pusillum	little barley	grass	N	0.01	0.01	7
Hybanthus verticillatus	babyslippers	forb	N	0.07	0.03	27
Hymenopappus scabiosaeus	Carolina woollywhite	forb	N	0.05	0.02	30
Juncus sp.	rush	grass-like	N	0.01	0.01	17
Juncus interior	inland rush	grass-like	N	0.03	0.02	10
Juncus tenuis	poverty rush	grass-like	N	0.01	0.01	13
Koeleria macrantha	prairie Junegrass	grass	N	0.1	0.03	57
Lactuca serriola	prickly lettuce	forb	I	0.01	< 0.01	10

[A] Five species of *Carex* were observed in the monitoring plots. In order of declining frequency, the five species were *C. meadii, C austrina, C. inops, C. brevior, and C. gravida*.

[B] *Cyperus mesocharus* and a hybrid species of *Cyperus* were identified.

Table 4 (continued). All species observed in the prairie sites (N = 30) at Tallgrass Prairie National Preserve, Kansas. Values for mean cover, standard error (SE) and occurrence are for 2023 (most recent monitoring event) only. N = native, I = introduced.

Species	Common Name	Guild	Origin	2023 Mean Cover (%)	SE	Occurrence (% Sites Observed) 2023
Lepidium densiflorum	common pepperweed	forb	N	0.04	0.01	27
Lespedeza capitata	roundhead lespedeza	forb	N	0.14	0.03	63
Lespedeza violacea	violet lespedeza	forb	N	1.78	0.55	73
Lespedeza virginica	slender lespedeza	forb	N	0.04	0.03	10
Liatris aspera	tall blazing star	forb	N	< 0.01	< 0.01	3
Liatris punctata	dotted blazing star	forb	N	0.05	0.02	33
Linum sulcatum	grooved flax	forb	N	0.04	0.01	33
Lithospermum incisum	narrowleaf stoneseed	forb	N	< 0.01	< 0.01	7
Lomatium foeniculaceum	desert biscuitroot	forb	N	0.03	0.01	13
Medicago lupulina	black medick	forb	I	< 0.01	< 0.01	7
Mimosa nuttallii	Sensitive brier	forb	N	0.11	0.04	43
Mirabilis sp.	four o'clock	forb	N	< 0.01	< 0.01	3
Mirabilis albida	white four o'clock	forb	N	< 0.01	< 0.01	7
Monarda fistulosa	wild bergamot	forb	N	0.06	0.04	20
Muhlenbergia cuspidata	plains muhly	grass	N	0.02	0.01	13
Myosotis verna	spring forget-me-not	forb	N	< 0.01	< 0.01	3
Nothoscordum bivalve	crowpoison	forb	N	0.01	0.01	13
Oenothera macrocarpa	bigfruit evening primrose	forb	N	< 0.01	< 0.01	7
Oenothera speciosa	pinkladies	forb	N	0.01	0.01	20
Oligoneuron rigidum var. *rigidum*	Stiff goldenrod	forb	N	0.03	0.02	20
Onosmodium bejariense var. *bejariense*	Western false gromwell	forb	N	0.09	0.05	17
Opuntia macrorhiza	twistspine pricklypear	forb	N	< 0.01	< 0.01	3
Oxalis sp.	woodsorrel	forb	N	0.12	0.02	77
Oxalis violacea	violet woodsorrel	forb	N	0.08	0.02	63
Packera plattensis	Platte groundsel	forb	N	0.05	0.02	43
Panicum virgatum	switchgrass	grass	N	0.55	0.19	93
Pascopyrum smithii	western wheatgrass	grass	N	< 0.01	< 0.01	7

[A] Five species of *Carex* were observed in the monitoring plots. In order of declining frequency, the five species were *C. meadii, C austrina, C. inops, C. brevior, and C. gravida*.

[B] *Cyperus mesocharus* and a hybrid species of *Cyperus* were identified.

Table 4 (continued). All species observed in the prairie sites (N = 30) at Tallgrass Prairie National Preserve, Kansas. Values for mean cover, standard error (SE) and occurrence are for 2023 (most recent monitoring event) only. N = native, I = introduced.

Species	Common Name	Guild	Origin	2023 Mean Cover (%)	SE	Occurrence (% Sites Observed) 2023
Pediomelum argophyllum	Silvery scurf-pea	forb	N	0.02	0.01	13
Pediomelum esculentum	Breadroot scurf-pea	forb	N	0.05	0.01	40
Penstemon cobaea	cobaea beardtongue	forb	N	< 0.01	< 0.01	7
Physalis pumila	dwarf groundcherry	forb	N	0.12	0.04	57
Physalis virginiana	Virginia groundcherry	forb	N	0.14	0.03	80
Plantago rhodosperma	redseed plantain	forb	N	< 0.01	< 0.01	3
Poa pratensis	Kentucky bluegrass	grass	I	0.15	0.04	50
Psoralidium tenuiflorum	slimflower scurfpea	forb	N	2.91	0.69	97
Ratibida columnifera	upright prairie coneflower	forb	N	0.02	0.01	27
Rhus glabra	smooth sumac	woody	N	0.16	0.11	13
Rosa arkansana	prairie rose	woody	N	0.04	0.02	17
Ruellia humilis	fringeleaf wild petunia	forb	N	0.34	0.02	100
Rumex crispus	curly dock	forb	I	< 0.01	< 0.01	3
Salvia azurea	azure blue sage	forb	N	0.46	0.09	97
Schizachyrium scoparium	little bluestem	grass	N	19.28	1.62	100
Scutellaria parvula	small skullcap	forb	N	< 0.01	< 0.01	7
Silene antirrhina	sleepy silene	forb	N	0.01	0.01	10
Silphium laciniatum	compassplant	forb	N	0.01	0.01	3
Sisyrinchium campestre	prairie blue-eyed grass	forb	N	0.04	0.01	40
Solanum carolinense	Carolina horsenettle	forb	N	< 0.01	< 0.01	7
Solanum rostratum	buffalobur nightshade	forb	N	< 0.01	< 0.01	3
Solidago canadensis	Canada goldenrod	forb	N	1.1	0.54	43
Solidago missouriensis	Missouri goldenrod	forb	N	0.14	0.02	83
Sorghastrum nutans	Indiangrass	grass	N	1.55	0.32	97
Spermolepis inermis	Red River scaleseed	forb	N	0.01	< 0.01	13
Sphenopholis obtusata	prairie wedgescale	grass	N	0.1	0.03	53
Spiranthes cernua	nodding lady's tresses	forb	N	< 0.01	< 0.01	3
Sporobolus compositus	composite dropseed	grass	N	3.97	0.79	100

[A] Five species of *Carex* were observed in the monitoring plots. In order of declining frequency, the five species were *C. meadii, C austrina, C. inops, C. brevior, and C. gravida*.

[B] *Cyperus mesocharus* and a hybrid species of *Cyperus* were identified.

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Species	Common Name	Guild	Origin	2023 Mean Cover (%)	SE	Occurrence (% Sites Observed) 2023
Sporobolus heterolepis	prairie dropseed	grass	N	< 0.01	< 0.01	3
Stenaria nigricans var. *nigricans*	diamondflowers	forb	N	< 0.01	< 0.01	3
Symphoricarpos orbiculatus	coralberry	woody	N	0.2	0.14	17
Symphyotrichum ericoides var. *ericoides*	Squarrose white wild aster	forb	N	1.72	0.35	100
Symphyotrichum oblongifolium	Aromatic wild aster	forb	N	0.77	0.29	70
Symphyotrichum pilosum var. *pilosum*	Awl wild aster	forb	N	0.02	0.02	10
Symphyotrichum sericeum	Western silvery wild aster	forb	N	0.04	0.02	23
Teucrium canadense	Canada germander	forb	N	0.06	0.05	7
Torilis arvensis	spreading hedgeparsley	forb	I	< 0.01	< 0.01	3
Tradescantia bracteata	longbract spiderwort	forb	N	0.02	0.01	23
Tradescantia ohiensis	bluejacket	forb	N	< 0.01	< 0.01	3
Tragia betonicifolia	betonyleaf noseburn	forb	N	0.01	0.01	3
Tragia ramosa	branched noseburn	forb	N	0.04	0.02	13
Tridens flavus	purpletop tridens	grass	N	0.05	0.03	30
Triodanis leptocarpa	slimpod Venus' looking-glass	forb	N	< 0.01	< 0.01	3
Triodanis perfoliata	clasping Venus' looking-glass	forb	N	0.02	0.01	17
Triosteum perfoliatum	feverwort	forb	N	< 0.01	< 0.01	7
Verbena simplex	narrowleaf vervain	forb	N	< 0.01	< 0.01	3
Verbena stricta	hoary verbena	forb	N	0.02	0.01	20
Vernonia baldwinii	Baldwin's ironweed	forb	N	1.36	0.4	100
Veronica arvensis	corn speedwell	forb	I	< 0.01	< 0.01	3
Viola sp.	violet	forb	N	0.03	0.01	20
Viola pedatifida	prairie violet	forb	N	0.08	0.02	63
Vulpia octoflora	sixweeks fescue	grass	N	0.02	0.01	23
Zigadenus nuttallii	Nuttall's deathcamas	forb	N	0.01	0.01	7

[A] Five species of *Carex* were observed in the monitoring plots. In order of declining frequency, the five species were *C. meadii, C austrina, C. inops, C. brevior, and C. gravida*.

[B] *Cyperus mesocharus* and a hybrid species of *Cyperus* were identified.

Appendix B. Supplemental Species Assessments

We selected a group of individual species that may be sensitive to changes in disturbance and/or climate to analyze further. Taking a deeper look at these species may provide additional insights into how the preserve is meeting its goals and objectives. This suite of species includes matrix grasses and disturbance-sensitive forbs.

Grasses

Big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*) are matrix grasses that are critical forage for ungulates in the tallgrass prairie (Bragg 1995). Climate change may shift the range of big bluestem (Yurkonis and Harris 2019), so we documented it here as a baseline for future observations (Figure 21). We included little bluestem in contrast to observe whether it would compensate for changes in big bluestem. Both species exhibited annual variation with little bluestem cover becoming more like big bluestem in more recent years. In earlier years, it was a smaller component of the vegetation community. Big bluestem had greater mean cover in all years except for three: 2007, 2008, and 2018. It is not clear what those years may have in common in terms of stimulating little bluestem; however, disturbance intensity was reduced from 2006 to present over prior years. The reduction in disturbance may have led to greater growth of these plants. Additionally, wetter conditions (Figure 3) or some combination of disturbance intensity (see fire and grazing sections) and precipitation may be benefiting the grasses.

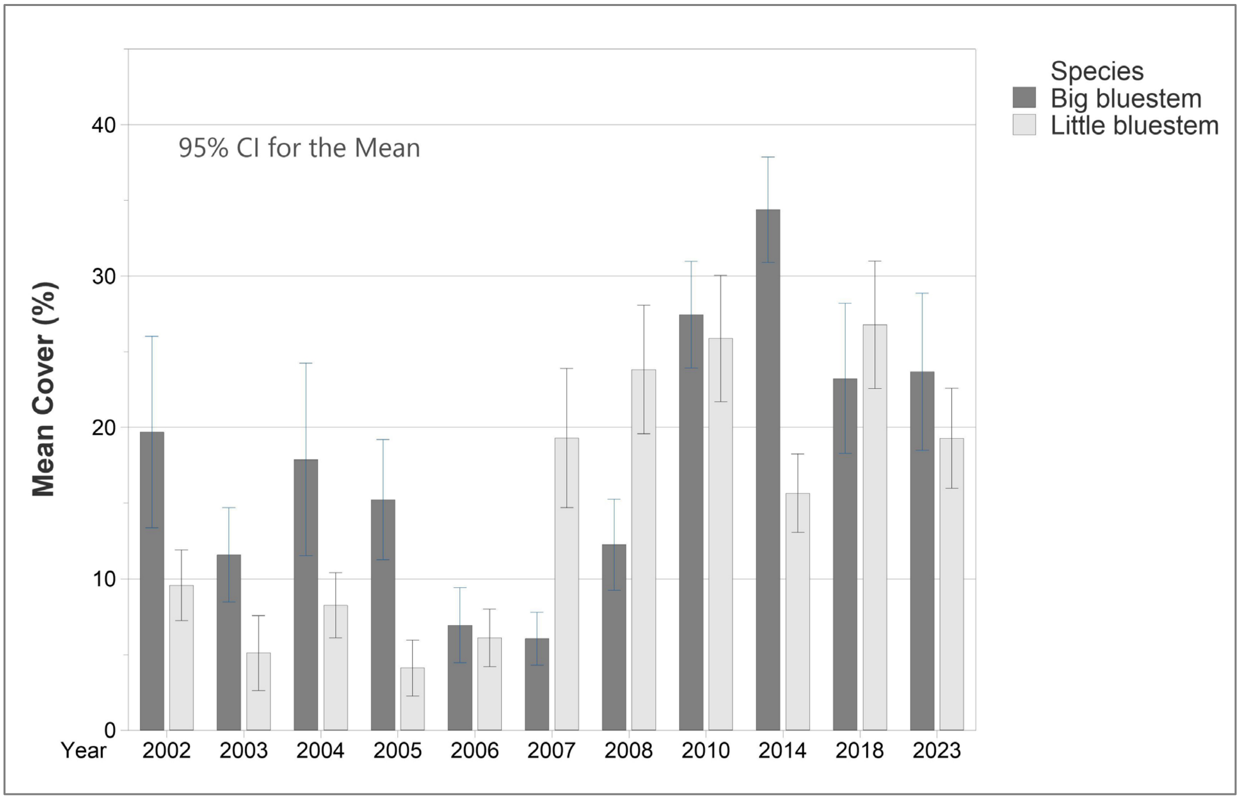

Figure 21. Mean percent cover for two grass species: big bluestem (*Andropogon gerardii*) and little bluestem (*Schizachyrium scoparium*) at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.
NPS

Forbs

We also analyzed two forbs that are often known as "increasers" (species that increase with disturbance intensity), goldenrod (*Solidago* spp.), and ironweed (*Verbena bladwinii*) to better understand what changes in composition might be occurring (Figures 22 and 23). Although there are multiple species of goldenrod present, the most abundant species was likely *S. altissima,* which has a coefficient of conservatism value of 1 indicating that it does not have high fidelity to a community type and can be found widely (Jog et al. 2006; Freeman 2017; see Leis and Morrison 2018). Goldenrod was absent in some years (2002, 2005, 2007, 2008). Mean goldenrod was greatest in 2018–2023 but is highly variable across the preserve (Figure 22). It was observed in 3.3% of sites in 2003, but in 43% of sites in 2023. More than half of the sites where goldenrod was recorded in 2023 were in Big Pasture. Mean site cover for goldenrod remains small (1.4 % cover) despite the increase in distribution. Interestingly, the increase occurred coincidentally with the reduction in overall disturbance intensity.

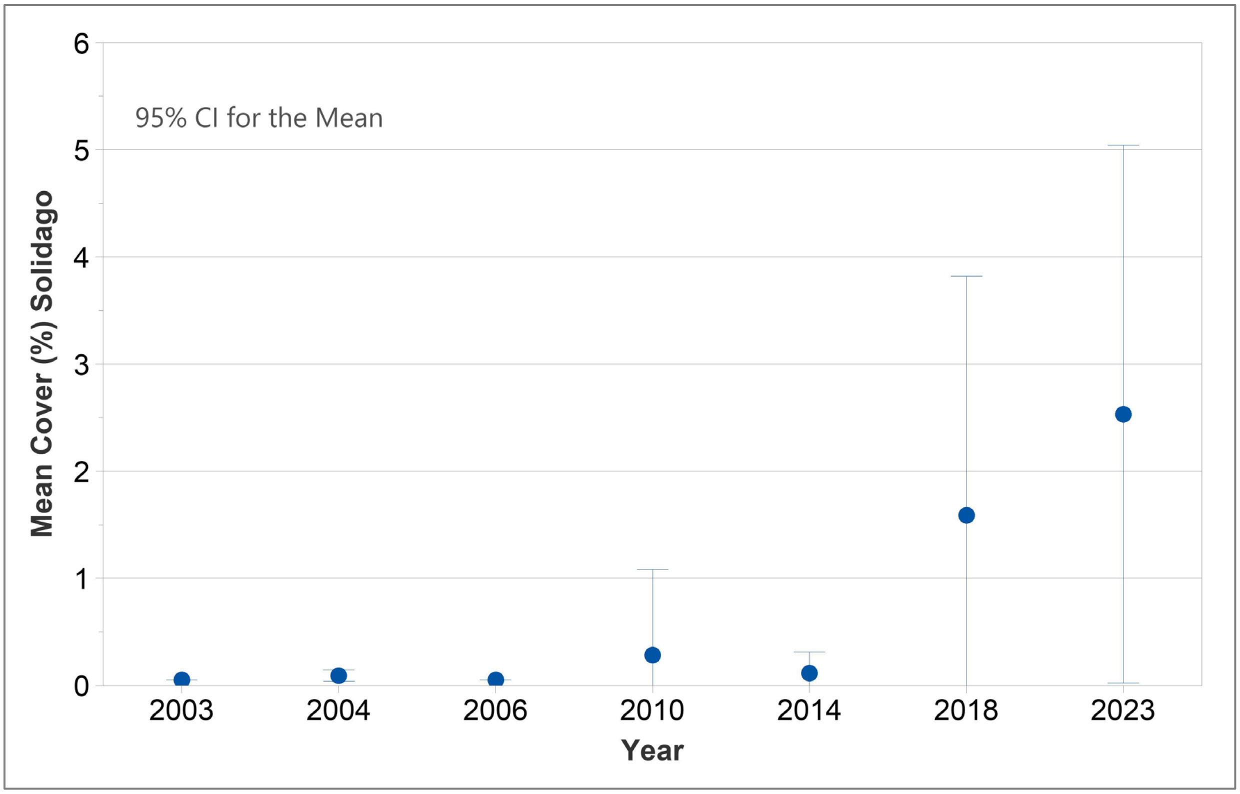

Figure 22. Mean percent cover for goldenrod (*Solidago* spp.) at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.

NPS

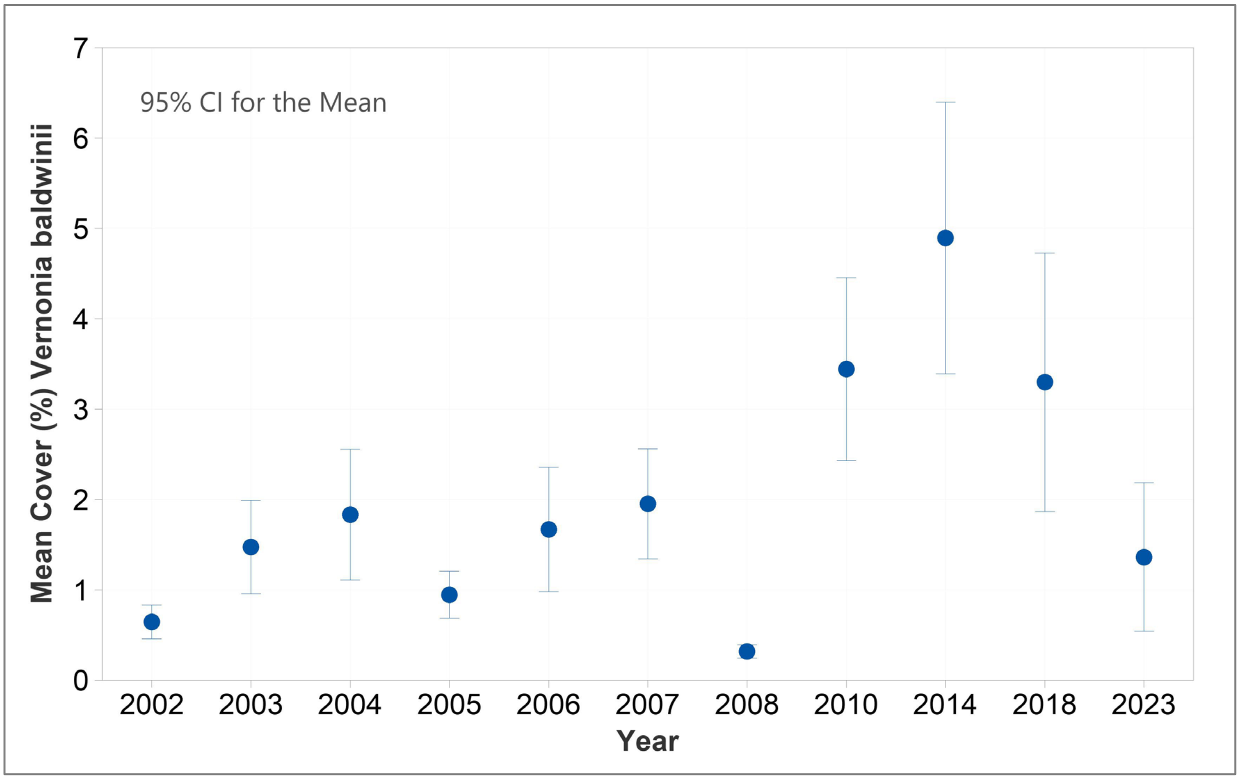

Figure 23. Mean percent cover for Baldwin's ironweed (*Vernonia baldwinii*) at Tallgrass Prairie National Preserve, Kansas, 2002–2023. Error bars are 95% confidence intervals.
NPS

Ironweed has a coefficient of conservatism of 2 indicating that it is a ubiquitous plant. Ironweed has been shown to increase with grazing intensity. However, in a study at Konza Prairie Biological Station, bison affected plant biomass but cattle did not (Damhoureyeh and Harnett 1997). Grazing reduces competition from other more palatable plants, allowing ironweed to thrive. Ironweed was not affected by fire frequency in the Konza Prairie Biological Station study. The pattern of abundance for ironweed in our study varied through time but was greatest from 2010 to 2018 (Figure 23). Crusher Hill Pasture had the greatest mean cover of ironweed among the pastures in 2014 (7.26% ± 1.73 SE). Overall, it is currently a small component of the flora with mean cover of 1.36% (0.4 SE, Appendix A), but is found in most monitoring sites. Increased cover of ironweed beginning in 2010 coincided with the shift in protocol design, but since the levels have not been sustained, there may be some other driving factor. Field crews reported a late frost a week prior to the spring 2008 sample event in addition to prescribed fires being completed late (completion date: April 29) that could have affected plant development. Analysis of fire and grazing levels did not provide any consistent stimulus for change.

Appendix C. Archival Photography

Figures 24 through 29 show a sampling of archival photos from monitoring events in the early monitoring record juxtaposed with photographs from the 2023 monitoring event at Tallgrass Prairie National Preserve, Kansas. We did not include images from all sites in order to maintain a manageable file size, so selected images represent a range of communities from some of the original 18 core sites across the preserve. These photos aide in visualizing changes in vegetation cover over time. Plot code includes site number, transect (A, B) and location (S = start, 0 m; F = finish, 50 m).

Figure 24. Tallgrass Prairie National Preserve plot 25 AS in Windmill Pasture in 2003 (left) and in 2023 (right). This site had a greater mean woody cover in 2023 than most other sites. Note the expansion of trees in the background.
NPS

Figure 25. Tallgrass Prairie National Preserve plot 42 BS in Crusher Hill Pasture in 2003 (left) and in 2023 (right). Tree seedlings were observed in this site for the first time in 2023. Note increased height of plants in recent photo.
NPS

Figure 26. Tallgrass Prairie National Preserve plot 30 BS in Red House Pasture in 2003 (left) and in 2023 (right). This site often gets excluded from fire because of its position between crossing trails.
NPS

Figure 27. Tallgrass Prairie National Preserve plot 34 BS in Red House Pasture in 2003 (left) and in 2023 (right). Plant structure in recent photo appears to be greater in heigh and density, covering rocks.
NPS

Figure 28. Tallgrass Prairie National Preserve plot 55 AS in Big Pasture in 2003 (left) and in 2023 (right). Note the cows trailing on the ridgeline in the left photo. The tapeline is less visible in 2023, indicating more tall and dense plants.

NPS

Figure 29. Tallgrass Prairie National Preserve plot 65 AF in Big Pasture in 2003 (left) and in 2023 (right). A single tree seedling was observed in this site three times, including in 2023. Note the increased tree heights and extent in background of recent photo.

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