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Status and Trend of Upland Vegetation and Soils at Dinosaur National Monument, 2010–2020



Pinyon-juniper woodlands interface with sagebrush communities in Dinosaur National Monument.
NPS/AMY WASHUTA

Status and trend of upland vegetation and soils at Dinosaur National Monument, 2010–2020

Science Report NPS/SR—2024/101

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Abstract

In 2009, the Northern Colorado Plateau Network began long-term monitoring of pinyon-juniper woodlands, high-elevation sagebrush, and low-elevation sagebrush shrublands in Dinosaur National Monument. Sagebrush communities have been impacted by fire and grazing in the past, and some areas have converted to an alternative grassland state, which can be dominated by native or exotic species. Pinyon-juniper woodlands are varied in structure but comprise the majority of plant communities throughout the monument. The primary goals of this monitoring are to track the status of these ecosystems and determine trends, or how they are changing over time. This report summarizes ten years of monitoring data and addresses the following questions: (1) What is the status and trend of vegetation and soil indicators for high-elevation sagebrush, low-elevation sagebrush, and pinyon-juniper woodland communities?, and (2) How do climate, grazing, and fire history influence observed status and trend? Results showed that in sagebrush communities, native cool-season perennial grasses decreased in cover while exotic annuals rapidly increased. Cover of big sagebrush (*Artemisia tridentata*) was on the lower end of known reference conditions but slowly increasing. Biological soil crust and soil stability were both increasing in sagebrush communities indicating improving conditions, but bare soil was also increasing in high-elevation sagebrush. In pinyon-juniper woodlands located in the south-central area of the monument, tree cover was stable but sapling density was decreasing. Cover of exotic species was low in pinyon-juniper woodlands. Managers should consider actions to control the rapid increase of exotic species (primarily cheatgrass) and protect native cool-season grasses in high- and low-elevation sagebrush communities.

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Executive Summary

In 2009, the Northern Colorado Plateau Network (NCPN) began long-term monitoring of pinyon-juniper woodlands, high-elevation sagebrush, and low-elevation sagebrush shrublands in Dinosaur National Monument (Dinosaur NM). Sagebrush communities have been impacted by fire and grazing in the past, and some areas have converted to an alternative grassland state, which can be dominated by native or exotic species. Pinyon-juniper woodlands are varied in structure but comprise the majority of plant communities throughout the monument. The primary goals of this monitoring are to track the status of these ecosystems and determine trends, or how they are changing over time. This report summarizes ten years of monitoring data and addresses the following questions:

1) What is the status and trend of vegetation and soil indicators for high-elevation sagebrush, low-elevation sagebrush, and pinyon-juniper woodland communities?
2) How do climate, grazing, and fire history influence observed status and trend?

Key findings for analysis of status and trend are:

- In both high- and low-elevation sagebrush communities, native cool-season perennial grasses decreased in cover while exotic annuals rapidly increased. The dominant exotic annual was cheatgrass (*Bromus tectorum*).
- Cover of big sagebrush (*Artemisia tridentata*) at both high and low elevations was on the lower end of known reference conditions but slowly increasing. Sagebrush at high elevation also had increasing density of mountain big sagebrush (*A. tridentata spp. vaseyana*) individuals, indicating some successful recruitment.
- Soil surface indicators in high- and low-elevation sagebrush were mostly stable or improving, with biological soil crust cover and soil stability both increasing. However, bare soil cover was highest and increasing in high-elevation sagebrush, which could lead to increased erosion.
- Pinyon-juniper woodlands in the south-central area of the monument had stable tree cover but decreasing density of saplings.
- Exotic species cover in pinyon-juniper woodlands was low (<10%) and stable over the monitoring period; however, exotic frequency detections were increasing. There is potential for invasion from adjacent sagebrush sites with high cover of exotics.
- Water balance variables, such as actual evapotranspiration and deficit, were better predictors of vegetation and soil change over time than precipitation alone.
- A grazing covariate was only influential in models for soil stability and bare soil cover and did not have a clear directional effect on the response. There are many confounding factors that limit interpretation of this driver, including variable timing and intensity of grazing on allotments, lack of compliance with grazing permits, and other ungulate activity.
- While fire has played an important role in structuring current plant communities, time since fire was only an influential covariate in the tree cover and soil stability models. This

suggests it is not influencing status or trend at the time scale of the 10-year monitoring period or the spatial scale of the monitoring design.

To put the analysis of status and trend for indicators into a management context for easier interpretation, the condition of sagebrush cover, live perennial grass cover, exotic annual cover, tree cover, biological soil crust cover, and soil stability were evaluated, and these data are presented in Table 1. Known reference conditions based on soil map units, ecological site descriptions, and published literature were used to evaluate these indicators following symbology used in the NPS Natural Resource Condition Assessment program's State of the Park reports (https://www.nps.gov/subjects/science/nrca.htm). Reference conditions could not be determined for live cover, exotic frequency, sagebrush density, sapling density, or bare soil cover. Indicators were assigned a condition of "Good," "Warrants Moderate Concern," or "Warrants Significant Concern" based on comparison to reference conditions.

Based on the results presented in this report, the NCPN recommends the following for the management of high- and low-elevation sagebrush and south-central pinyon-juniper woodlands:

- Prioritize control of exotic annuals, particularly cheatgrass, which are rapidly increasing in high- and low-elevation sagebrush. Since control and restoration of cheatgrass-invaded areas at a landscape scale is often not feasible, managers could focus on preventing further spread through limiting disturbance by fire, particularly in areas of low resilience that have not already been invaded (Davies et al. 2011; Rodhouse et al. 2021).
- Concurrent with exotic control, native cool-season grasses that are declining in cover should be protected. Strategies to accomplish this could include limiting disturbance by fire, reducing the intensity of grazing, or altering the timing of grazing to protect the spring growth period of cool-season species.
- Where high- and low-elevation sagebrush are protected, consider active restoration of cool-season grasses. Restoration efforts could include seeding or planting of native species, in combination with surface modifications, to stabilize soil and retain soil water.
- Monitor the status of pinyon and juniper saplings, which are decreasing in density and have been identified as vulnerable to drought in the region.

Table 1. Natural resource condition summary for key vegetation and soil indicators at Dinosaur National Monument, 2010–2020. Status shows the range of model-predicted means for the given indicator over the 10-year monitoring period. Reference conditions were determined based on information given in soil surveys, ecological site descriptions, and/or scientific literature. Assessment colors indicate whether the current condition is Good (green), Warrants Moderate Concern (yellow), or Warrants Significant Concern (red), and are based on comparison to reference conditions. In some cases, no reference condition is available for the given indicator, and the author's professional judgment was used to determine the assessment color. Assessment arrows indicate whether the condition is improving or declining. The outer ring of the assessment circle indicates the level of confidence in the assessment, where a solid line is medium confidence and a dashed line is low confidence. The level of confidence is based on uncertainty in the data and the applicability of the assessment point, and no assessments were given high confidence due to the relatively short dataset, wide credible intervals, and lack of well-defined reference conditions.

	High-elevation Sagebrush			Low-elevation Sagebrush			Pinyon-juniper Woodland		
Indicator	**Status**	**Reference Condition**	**Assessment**	**Status**	**Reference Condition**	**Assessment**	**Status**	**Reference Condition**	**Assessment**
Big sagebrush cover	11–15%	10–20%[a]		8–10%	5–30%[a]		2%	0–30%[a]	
Perennial grass cover	7–17%	19–55%[ab]		7–15%	14–75%[ab]		2–4%	20–55%[a]	
Exotic annual species cover	2–12%	<10%[c]		12–42%	<10%[c]		4–10%	<10%[c]	
Tree cover	1–2%	0%		2%	0%		22–25%	5–35%	

[a] NRCS, 2007

[b] NRCS Ecological Site Description

[c] Jones et al., 2021; Sherrill and Romme, 2012

Table 1 (continued). Natural resource condition summary for key vegetation and soil indicators at Dinosaur National Monument, 2010–2020. Status shows the range of model-predicted means for the given indicator over the 10-year monitoring period. Reference conditions were determined based on information given in soil surveys, ecological site descriptions, and/or scientific literature. Assessment colors indicate whether the current condition is Good (green), Warrants Moderate Concern (yellow), or Warrants Significant Concern (red), and are based on comparison to reference conditions. In some cases, no reference condition is available for the given indicator, and the author's professional judgment was used to determine the assessment color. Assessment arrows indicate whether the condition is improving or declining. The outer ring of the assessment circle indicates the level of confidence in the assessment, where a solid line is medium confidence and a dashed line is low confidence. The level of confidence is based on uncertainty in the data and the applicability of the assessment point, and no assessments were given high confidence due to the relatively short dataset, wide credible intervals, and lack of well-defined reference conditions.

	High-elevation Sagebrush			Low-elevation Sagebrush			Pinyon-juniper Woodland		
Indicator	**Status**	**Reference Condition**	**Assessment**	**Status**	**Reference Condition**	**Assessment**	**Status**	**Reference Condition**	**Assessment**
Biological crust cover	1–6%	N/A		5–12%	N/A		9–17%	N/A	
Soil stability rating	3.5–4.5	N/A		4.2–5.2	N/A		4.1–5.3	N/A	

[a] NRCS, 2007

[b] NRCS Ecological Site Description

[c] Jones et al., 2021; Sherrill and Romme, 2012

1 Introduction

1.1 Purpose and Scope

Long-term monitoring of vegetation and soils in upland communities of Dinosaur National Monument (Dinosaur NM) was initiated in 2009 by the Northern Colorado Plateau Network (NCPN). The monitoring of upland vegetation and soil indicators is one of the protocols selected by the NPS Inventory & Monitoring Network because they provide insight into the overall health or condition of the park's natural resources. More specifically, the upland vegetation and soils protocol monitors "vital signs" that include plant communities, soil stability, biological soil crusts, and hydrologic function to understand long-term change and the impact of selected drivers and stressors. The NPS Organic Act (16 U.S.C. 1,2,3, and 4; 39 Stat. 535) directs park managers to preserve these ecosystems "unimpaired for the enjoyment of future generations", requiring information on current status and long-term trends. This report presents the first 10-year status and trend analysis for upland vegetation and soil indicators at Dinosaur NM.

When establishing the long-term monitoring program at Dinosaur NM, resource managers identified big sagebrush and pinyon-juniper communities as priority ecosystems. Sage-steppe shrublands are considered one of the most threatened ecosystems in the United States because of extensive habitat fragmentation and type conversion to pinyon-juniper woodlands or grasslands dominated by invasive annual species (Noss et al. 1995). Intact sagebrush ecosystems provide habitat for over 350 plants and animals of conservation concern and provide forage for domestic livestock (Davies et al. 2011). Sagebrush conversion varies depending on alterations to fire regimes; fire suppression may allow for encroachment of pinyon-juniper woodlands into sagebrush while more frequent fire, especially at lower elevations, often leads to invasion by exotic annual grasses (Miller and Rose 1999; Perryman et al. 2002; Davies et al. 2011; Sherrill and Romme 2012). At Dinosaur NM, sagebrush communities are dominated by *Artemisia tridentata,* with mountain big sagebrush (*A. tridentata spp. vaseyana*) found at higher elevations (>6500 ft) and Wyoming big sagebrush (*A. tridentata spp. wyomingensis*) generally below 6500 ft.

Across the Colorado Plateau, pinyon-juniper woodlands range from dense old-growth forests with sparse understory to open savanna-like systems with low tree density and greater cover of grasses and shrubs (Miller 2005). While pinyon pine and juniper trees are sometimes considered invaders into shrubland ecosystems, these woodlands provide their own set of ecosystem services, including providing plant materials for cultural use as well as habitat for sensitive plant and animal species throughout their range (Hartsell et al. 2020). Pinyon-juniper woodlands may be expanding in some parts of their range; however, they are also experiencing dieback and mortality related to drought stress in other areas (Miller et al. 2019; Kannenberg et al. 2021). At Dinosaur NM, Floyd et al. (2017) found that fire rotations in pinyon-juniper woodlands have shortened, suggesting contraction rather than expansion of this community.

As the climate has warmed, sagebrush shrublands and pinyon-juniper woodlands at Dinosaur NM have experienced greater maximum and minimum temperatures, warmer winters with loss of snowpack, and possibly an increase in extreme precipitation events (Monahan and Fisichelli 2014;

Jones et al. 2021). The warming temperature trends are projected to continue, and more winter and spring precipitation is likely to fall as rain rather than snow (Jones et al. 2021). Warmer winter temperatures have led to a longer growing season characterized by earlier spring green-up (Thoma 2022), which impacts vegetation community dynamics.

Both high- and low-elevation sagebrush shrublands and pinyon-juniper woodlands at Dinosaur NM are subject to fires (natural and prescribed), grazing (livestock and native ungulates), and climatic stress, with potentially different responses to these disturbances. The purpose of this report is to address the following questions:

1) What is the status and trend of vegetation and soil indicators for high-elevation sagebrush, low-elevation sagebrush, and pinyon-juniper woodland communities? Specific indicators evaluated in this report include plant cover (total live, perennial grasses, big sagebrush, exotic annuals, trees), surface cover (biological soil crust, bare soil), soil stability, canopy and basal gaps, big sagebrush density, sapling density, and exotic frequency.
2) How do climate, grazing, and fire history influence observed status and trend?

Data used in this report include the NCPN 2010–2020 upland vegetation and soils monitoring data. Although 2010 and 2011 were pilot years, no significant changes in methods occurred during that time that would have excluded these data from the analysis.

1.2 Study Area

Dinosaur NM was established in 1915 and expanded in 1938 and 1960 to its current extent of 85,447 ha (211,144 ac). While the initial proclamation protected extraordinary fossil resources, the later expansions included the Green and Yampa River corridors and surrounding land for the purpose of natural resource protection and recreation use. Approximately two-thirds of the monument is located in northwestern Colorado and the rest is in northeastern Utah (Figure 1).

Elevations in Dinosaur NM range from 1,433 to 2,743 m (4,700 to 9,000 ft). The climate is semiarid, with winter temperatures ranging from about -12 to 1°C (11 to 33°F) and summer temperatures from 14 to 33°C (57 to 91°F). Precipitation averages approximately 294 mm (11.6 in) annually and is generally evenly distributed throughout the year, with slightly higher input in spring (April, May) and fall (September, October) (Jones et al. 2021).

A diversity of geologic substrates combined with complex topography in Dinosaur NM results in highly heterogeneous soils and plant communities. However, the more common plant communities within the monument include native grassland, mixed mountain shrubland, sagebrush/rabbitbrush shrubland, and pinyon-juniper woodlands (Coles et al. 2008). Pinyon-juniper woodlands cover the largest land area in the monument but range from open grass savanna-type woodlands to dense forests.

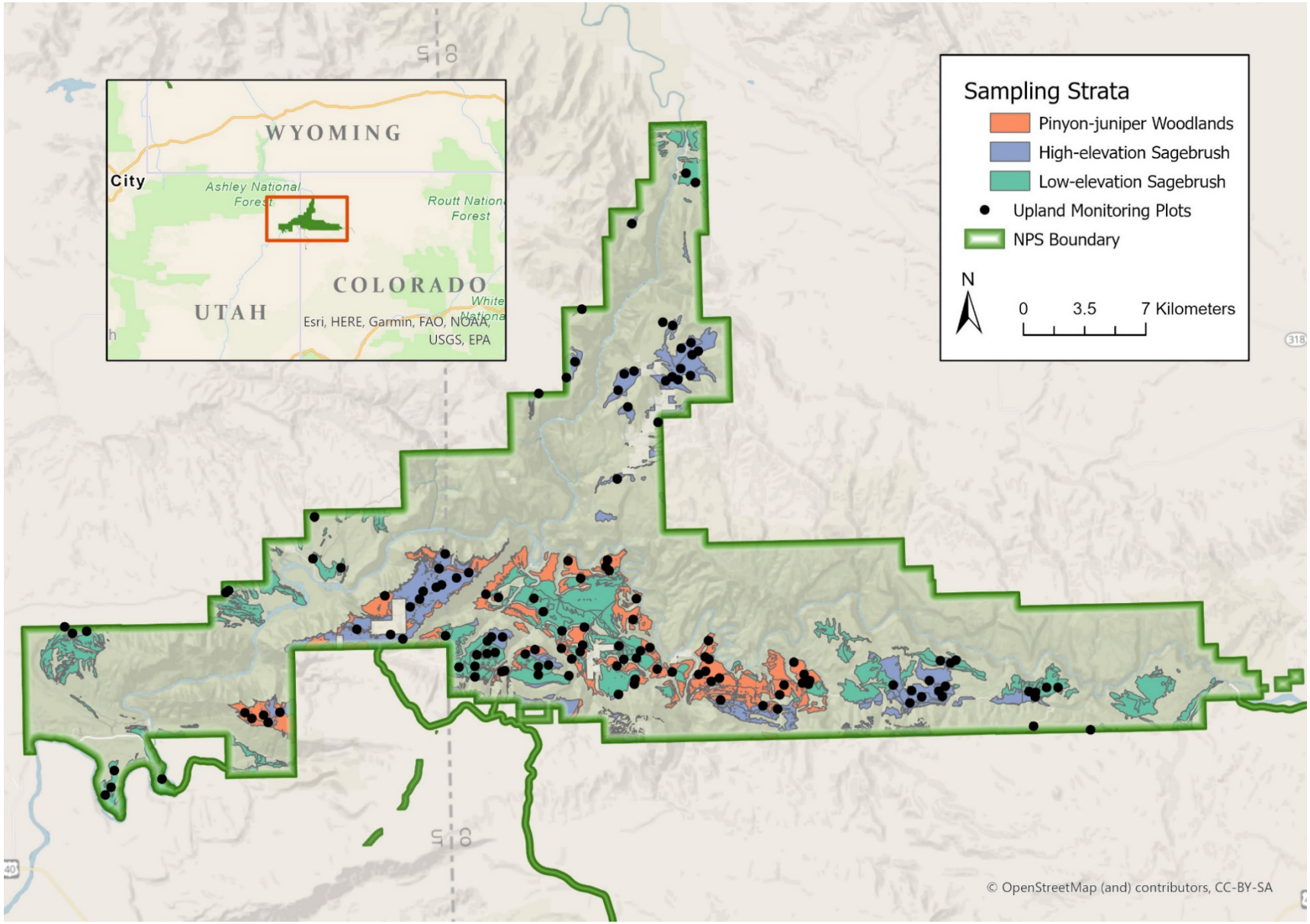


Figure 1. Location of upland vegetation monitoring plots and sampling strata at Dinosaur National Monument.

Disturbance by fire is a natural part of ecosystem processes in pinyon-juniper woodlands and sagebrush shrublands. However, cheatgrass invasion and increased human use may have altered fire regimes in Dinosaur NM beyond what they were in the past. Estimated historic fire rotation for pinyon-juniper woodlands at Dinosaur NM is approximately 550 years (Floyd et al. 2017), 2500–5000 years for Wyoming big sagebrush shrublands, and 458–729 years in mountain big sagebrush shrublands (Arendt and Baker 2013). Fire frequency has increased in all of these ecosystems in recent years (Jones et al. 2021). Arendt and Baker (2013) estimate the more recent fire rotations for Dinosaur NM to be 188 years for pinyon-juniper woodlands, 53 years for Wyoming big sagebrush shrublands, and 166 years for mountain big sagebrush shrublands. This is of particular concern in sagebrush systems where fire contributes to the conversion of shrublands to grasslands, often dominated by invasive species, which can further increase fire frequency and intensity.

There are ten active grazing allotments covering 23,544 ha (58,178 ac, approximately 28% of the monument) in Dinosaur NM. Livestock grazing in the region began in the 1880s, and much of Dinosaur NM was heavily grazed by cattle and sheep through the 1920s. Current grazing allotments vary in timing (spring, summer, or year-round) and intensity of use, and prior use is not well documented. Some allotments are contiguous with adjacent federal or private land, and cattle may move freely between the different management agencies. Cattle trespass is known to have occurred regularly, further confounding the effect of grazing within the monitored area. Additional grazing pressure by native ungulates, particularly elk in fall and winter, has been noted by park staff and likely impacts plant communities, but there is little quantification of this use.

2 Methods

2.1 Study design

Pinyon-juniper woodlands are abundant in the park. Sagebrush systems include grasslands, which are an early successional stage after fire disturbance and are susceptible to invasion by exotic species. In 2009, pinyon-juniper woodlands, high-elevation sagebrush shrublands, and low-elevation sagebrush shrublands were the strata targeted for long-term monitoring by natural resource managers at Dinosaur NM (see Figure 1, Figure 2). Resource managers were particularly interested in comparing high- and low-elevation sagebrush areas in defined, permitted allotments grazed by domestic livestock to those that were ungrazed. The steps for determining plot locations included: 1) filtering the land area based on accessibility (excluding slopes ≥50%, hiking distances >4 km, and privately owned land); 2) developing a sampling frame based on suitable ecological sites for resources of interest; 3) generating a grid of spatially balanced points within the sample frame; and 4) visiting potential plot locations in the field to determine suitability for inclusion in the study. Additional details on sampling design can be found in Witwicki et al. (2017).

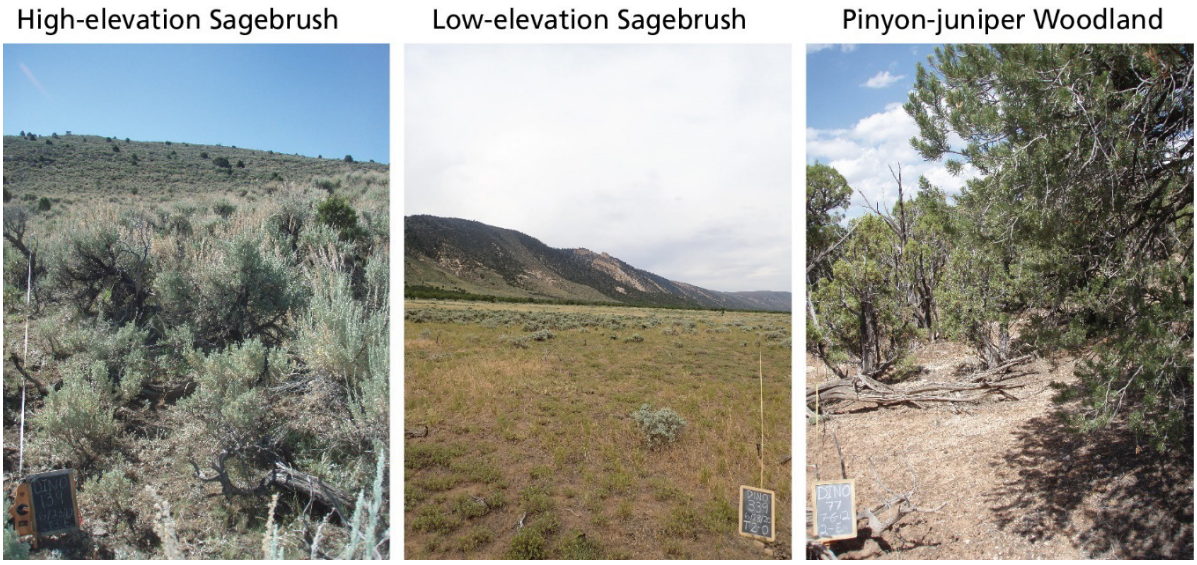


Figure 2. Sampling strata at Dinosaur National Monument. (NPS).

The Dinosaur NM vegetation map (Coles et al. 2008), NRCS soil map units (NRCS 2007), and field reconnaissance were used to determine which ecological sites were associated with sagebrush communities. These resulted in 11 Wyoming big sagebrush ecological sites and 3 mountain big sagebrush ecological sites found to be suitable for inclusion in the sampling frame (see Appendix B). The ecological sites were further characterized by grazing allotment and elevation to address management interests. The cutoff elevation used to separate Wyoming big sagebrush and mountain big sagebrush was 2,000 m (6,562 ft) (Goodrich and Neese 1986). At the time of development of the sampling design, ecological site descriptions did not exist for pinyon-juniper communities at Dinosaur NM, so they were not used to develop a sampling frame. Instead, discussion with park staff

helped to identify a geographic area that would represent pinyon-juniper communities and be logistically suitable for sampling.

Potential monitoring sites from pinyon-juniper and sagebrush sampling frames were selected at a 10 m grid size using Reversed Randomized Quadrat-Recursive Raster (RRQRR) (Theobald et al. 2007), which ensured a spatially balanced sampling grid across the landscape. The sampling frame area for low-elevation sagebrush was 7,122 ha (17,598 ac), high-elevation sagebrush was 5,008 ha (12,375 ac), and pinyon-juniper woodlands was 5,076 ha (12,543 ac). Using the ordered list of points generated by RRQRR, 40 plots were established for pilot monitoring in 2009. These included eight plots each in five monitoring strata: high-elevation grazed sagebrush, low-elevation grazed sagebrush, high-elevation ungrazed sagebrush, low-elevation ungrazed sagebrush, and pinyon-juniper woodlands.

Pilot monitoring took place from 2009 to 2011. A variance and power-for-trend assessment was completed following pilot data collection (Witwicki 2012). The findings of this analysis led to the establishment of 135 plots, of which 36 are subject to annual monitoring, with a rotation through all plots every 5 years (see Appendix C). Twelve plots were scheduled to be monitored in each stratum per year. The actual number of plots monitored each year varied due to staffing as a result of funding inconsistencies and a flat budget. While sagebrush plots were generally sampled consistently among years, no pinyon-juniper plots were monitored in 2013, 2016, 2017, or 2020 (Table 2). Sagebrush plots were prioritized for monitoring when necessary because they were a more immediate management concern. Data included in this report encompass the years from 2010 to 2020. Pilot data from 2009 were excluded because sampling took place in October and the data were inconsistent with subsequent years, which were sampled in June and July.

Table 2. Sampling effort by year at Dinosaur National Monument. Following pilot monitoring, which ended in 2011, a total of 12 plots per year were scheduled to be monitored in each stratum.

Stratum	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
High-elevation Sagebrush	14	14	12	12	12	12	12	12	12	12	11
Low-elevation Sagebrush	8	8	12	12	11	12	12	12	12	12	12
Pinyon-juniper Woodlands	8	8	12	--	12	12	--	--	6	12	--

2.2 Vegetation and soil measurements

Each plot had three parallel, 50-meter transects spaced 25 meters apart (Figure 3). Transects ran across the slope and were permanently marked by rebar at the end points of each transect. To minimize impacts to biological soil crusts, crew members walked in washes or on rocks or litter whenever possible and stepped in the same disturbed locations when it was not possible to avoid soil crusts. The field methods for upland monitoring in NCPN parks (Witwicki et al. 2017) were based largely on rangeland monitoring procedures recommended by Herrick et al. (2005).

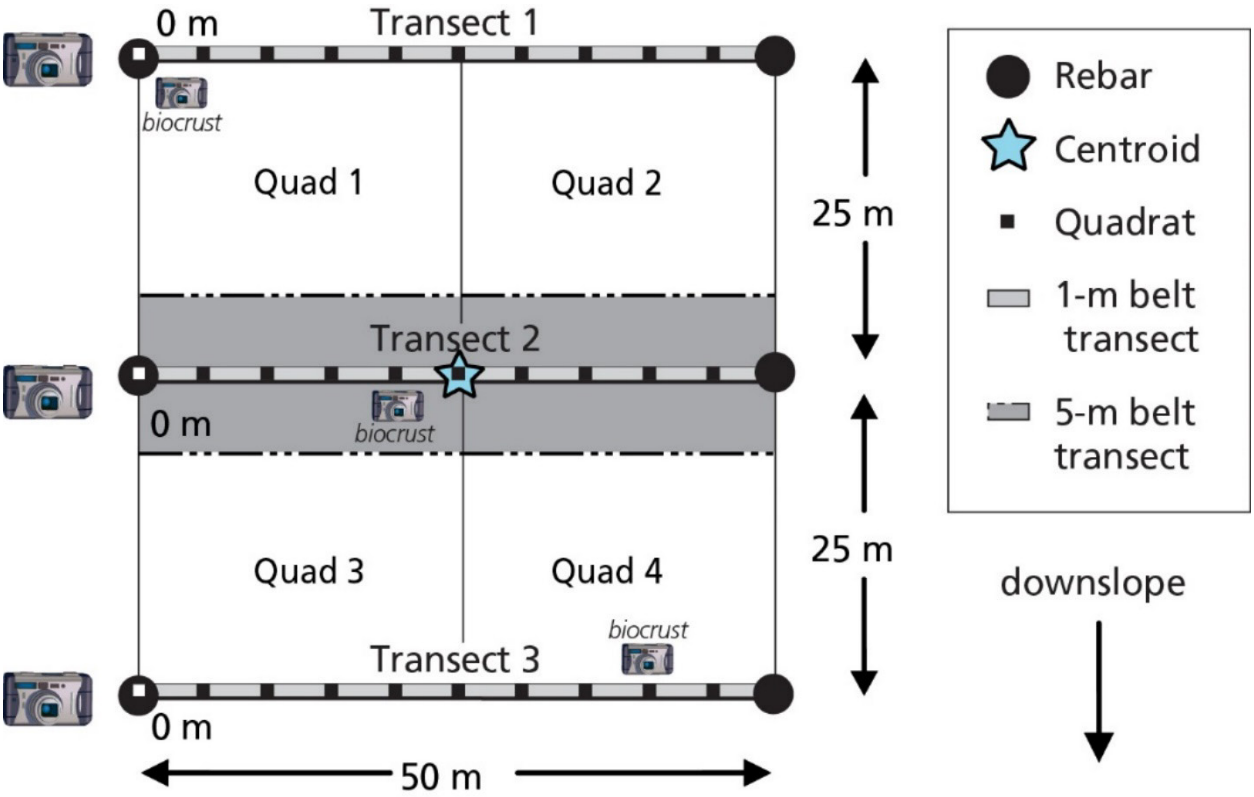


Figure 3. Locations of transects and monitoring measurements at upland plots, including 1-m2 quadrats for sampling frequency of exotics and one-meter belt transects for measuring shrubs. Soil stability was collected at six randomized locations along each transect. Sapling density was collected in the five-meter belt centered on Transect 2. Repeat photos were taken of each transect (larger camera symbols show transect photo locations) and of the soil surface on each transect (smaller camera symbols labeled "biocrust" show soil surface photo locations).

A point-intercept method was used to record the surface percent cover of plant species and ground cover attributes at 0.5-meter intervals along each transect (Figure 3). Beginning in 2011, the frequency of exotic species was recorded using 1×1-meter quadrats placed every five meters along each transect. Counts of shrubs by species and height class were recorded in a one-meter belt. Counts of tree saplings were recorded in a five-meter belt centered on the middle transect. Canopy and basal gap measures were acquired along each of the transects using a line-intercept method to record all gaps 20 cm or greater. Six soil surface samples were taken along each of the transects to assess soil-aggregate stability. Samples were rated from 1 to 6 based on their structural integrity following immersion in water, with higher ratings corresponding to greater soil stability.

Quality assurance and quality control (QA/QC) procedures were performed for all vegetation and soil data and included domain checking in data entry, checks for missing values, and logic checks.

2.3 Analysis

2.3.1 Data summaries

2.3.1.1 Vegetation and soil data

To provide context for the Bayesian status and trend analysis, vegetation and surface percent cover was summarized across all plots and years and presented in section 3.1. The Bayesian analysis included data at the transect level. Point-intercept data for functional groups of vegetation and surface features was summarized as the number of hits along the 50-m transect. Live cover included photosynthetic and live woody tissue. Hits of cyanobacteria, moss, and lichen at the soil surface were summed and analyzed as biological soil crust. Bare soil was defined as loose soil at the surface.

Exotic frequency was calculated as the number of 1-m^2 quadrats where any exotic species was detected, out of 10 quadrats along the 50-m transect. Shrubs and saplings were summed for analysis and transformed post-hoc to the number of individuals per hectare. Sapling density was only calculated for pinyon-juniper woodlands because this measurement was not taken in high- and low-elevation sagebrush plots until 2016. All shrubs were counted in a one-meter belt; however, only *A. tridentata* shrubs over 10 cm in height were included in the final analysis. *A. tridentata* is the focal species in the high- and low-elevation sagebrush shrublands, and the smallest size class was excluded to reduce noise in the data from large seedling recruitment events that did not result in established shrubs. Soil stability data were modeled as counts of categorical class ratings from 1 to 6.

Bayesian trend models for canopy and basal gap data were unsuccessful, so these data are summarized as the proportion of a transect in 5 size classes: 0–20 cm, 21–50 cm, 51–100 cm, 101–200 cm, and >200 cm. Gap sizes of less than 20 cm were not measured in the field, so the 0–20 cm category could include gaps smaller than 20 cm and/or plant canopy or plant base, for canopy and basal gaps, respectively. Distributions of gaps by size class were averaged over plots and years.

2.3.1.2 Climate data

Climate data for each plot were downloaded from the Daymet daily surface weather data at a 1-km grid resolution from 1980 to 2020 (Thornton et al. 2020). These data were used to calculate the 30-year average temperature and precipitation, defined as the period from 1980 to 2010. Then, the difference in temperature over the monitoring period relative to the long-term average was calculated, as was precipitation as a percentage of the long-term average.

Actual evapotranspiration (AET) and water deficit (D) at the plot level were calculated using a modified Thornthwaite water balance model (Tercek et al. 2021) and precipitation (P), AET, and D were summed over the cool season (October–May) and monsoon season (July–September) in the year prior to monitoring. These water balance metrics were used as covariates in Bayesian models as described below.

All data summaries and figures were created in R 4.1.2 (R Core Team 2022).

2.3.2 Bayesian hierarchical models

2.3.2.1 Bayesian models

Bayesian hierarchical modeling was used to analyze these data and determine status and trend for each stratum (pinyon-juniper woodlands, high-elevation sagebrush, and low-elevation sagebrush); see Appendix A for additional background on Bayesian data analysis. Bayesian hierarchical modeling provides a flexible approach that can address challenges presented by the long-term monitoring data, including: 1) complex revisit designs that have data that are missing by design; 2) non-normally distributed response variables; and 3) censored or truncated data (Zachmann et al. 2021).

The general form for the Bayesian models is as follows:

[Model parameters | data] α [data | likelihood model] [priors]

where "|" is "conditional on" and α is "proportional to."

All models took the form of a general linear model that included a time slope (β_1), intercept (β_0), and ecologically informative covariates. Intercepts were allowed to vary among plots in all models (i.e., plot-level random effect) but slopes were held constant. This accounted for within-plot variance since data were included at the transect level. A deterministic model represented the ecological process being modeled. For example, change in cover over time is well represented by an inverse logit model. For each response type (e.g., cover), a likelihood model appropriate to the distribution of the data was chosen. In all models, vague priors were used due to a lack of comparable data in the literature and because pilot data were included in the trend analysis.

All Bayesian analyses were run in R 4.1.2 (R Core Team 2022) using JAGS (Plummer 2019) as the Markov Chain Monte Carlo (MCMC) sampler. For each model, three chains with a burn-in and number of iterations were used, which varied by model.

2.3.2.2 Covariates

Water availability is the primary limiting factor to plant growth on the Colorado Plateau, and more specifically, the cool-season grasses abundant throughout Dinosaur NM are known to respond to cool-season P (Comstock and Ehleringer 1992; Witwicki et al. 2016). However, other functional groups (e.g., exotic annuals, shrubs) may respond to P over different time periods, and climate conditions can influence vegetation production for a year or more (Thoma et al. 2016). The influence of P on plant growth is often mediated by soil properties such as water holding capacity or the timing and intensity of rainfall. Therefore, water balance covariates like AET and D may better predict plant response than P alone (Thoma et al. 2019). Biological soil crusts can benefit from wet periods in the spring (Darrouzet-Nardi et al. 2015). Cool (October–May) and monsoon (July–September) season P, AET, and D data from the 12-month period prior to sampling were used as covariates in Bayesian models. Water balance covariates were checked for correlation prior to use in models, and covariates with a Pearson's r value ≥0.8 or ≤-0.8 were excluded from use in the same model.

Across all years, timing of monitoring occurred between June 1 and July 31. However, plant phenology is strongly influenced by heat accumulation, and a cool or warm spring could lead to phenological differences not accounted for by the water balance covariates. Therefore, cumulative growing degree-days (GDD) from January to April in the models were also included. GDD were calculated following McMaster and Wilhelm (1997),

$$cumGDD_{mont\ hly} = \sum_{1}^{d} \frac{T_{max} + T_{min}}{2} - T_{base}$$

where $cumGDD_{monthly}$ is the cumulative GDD over a single month, d is the number of days in a month, T_{max} is the maximum temperature on a given day, T_{min} is the minimum temperature on a given day, and T_{base} = 4.4°C, the temperature below which grass physiological activity ceases (Wang 1960).

To examine the impact of grazing and fire on plant and soil response at Dinosaur NM, covariates were also tested for grazing and burning in model sets. For grazing, a binary covariate was used to indicate whether a plot was currently part of an active grazing allotment or not. The burned covariate was the interaction between whether a plot had burned at any time in the recorded past and the time since fire occurred.

All covariates were centered and standardized so that relative effects of coefficients could be directly compared in the models. For each model, a set of covariates was tested that included an ecologically driven combination of the nine covariates described above: cool-season AET, monsoon-season AET, cool-season D, monsoon-season D, cool-season P, monsoon-season P, cumulative GDD, grazed, and burned. For example, the model comparison set for live plant cover included each covariate alone and each of the water balance covariates in combination with cumulative GDD, grazed, and burned.

2.3.2.3 Annual status

To model annual status of each indicator, actual standardized values of covariates were used in the model to create a figure that expressed the year-to-year variation of the indicator due to the covariate(s). When cumulative GDD was included in the model, this was held at its mean across all years of sampling since the removal of variation associated with differences in timing of plot visits was desired. The annual status figures utilized data from all plots sampled in all years of monitoring, with modeled predictions of values for plots that were not sampled in a particular year. Status was reported as the model-predicted mean of an indicator for each year using this version of the model. Values within the 95% credible intervals (CI) should also be considered as possible values for the mean when evaluating these results. Credible intervals are probabilities and can be interpreted as having 95% confidence that the true value of the mean of the indicator falls within the lower and upper values for the given interval. These figures highlight shorter-term variation caused by differences in weather.

2.3.2.4 Trend

In this report, trend is defined as “the directional change in an indicator over time apart from year to year, stochastic variation that is caused by changes in weather, observers, instrumentation,

disturbances, or other short-term influences on status" (Zachmann et al. 2021, pg. 3). Trend figures model directional change over time under average conditions. In these models, covariates were held at their mean (that is, the mean over the sampling period), essentially removing the interannual variation in the indicator to observe the remaining trend. To determine if an indicator was trending up, down, or remaining constant, the distribution of the change in response was observed per year (i.e., the time slope in the model, β_1). The results showed that there was no evidence of trend if the 95% CI of the change in response contained zero. If the distribution centered on a value greater or less than zero with the 95% CI not including zero, the conclusion indicated that there was evidence of a trend, and this was reported as significant.

When results showed a significant trend, this indicated that the covariates in the model did not fully explain the directional change over time. For example, a water balance covariate like cool-season AET might be a large source of variation in perennial grass cover and explain the change in status over the period of monitoring. When cool-season AET is held at its mean and there is still a significant trend, this suggests that an underlying mechanism not included in the model is driving the observed trend (e.g., competition with an invasive species). When a covariate is largely responsible for shorter-term changes in status over the period of monitoring, this information can be used to guide management and understand how the indicator may respond to projected changes in climate.

2.3.2.5 Model selection and evaluation

To select the final model, convergence using the Gelman-Rubin diagnostic with a threshold of 1.1 was checked for first (Gelman and Rubin 1992). In addition, the models passed posterior predictive checks for Bayesian p-values (P_B), which describe the ability of the model to predict the data. Following Hobbs and Hooten (2015), when the P_B of the test statistic is either ≤ 0.1 or ≥ 0.9 the chosen model does not adequately represent the data. Models not meeting these criteria were rejected. Of the models converging and passing posterior predictive checks, the best fit model was chosen by minimizing posterior predictive losses (Hobbs and Hooten 2015). When the 95% CI of the posterior distribution of the covariate coefficient did not include zero, this indicated that the covariate had a significant directional effect on the response. Though some covariates had nominal direction (the 95% CI contained 0), they still contributed ecological meaning to the model and were retained.

3 Results

3.1 Vegetation and soil summary

Previous reporting presents summary data for 40 plots that were visited in 2011 and includes all measures of vegetation and soil surface features (Witwicki 2013). This section updates that information for functional groups using all plots and an additional 9 years of monitoring data; however, the 2011 report data may be referenced for information on percent cover of dominant species, volume of fuels, and litter depth, which are not shown here.

Across all plots, pinyon-juniper woodlands have moderate tree cover (ranging from 12% to 49%) with open understories (Figure 4). They have the lowest percent cover of annual grasses, which is primarily the non-native cheatgrass (*Bromus tectorum*). Shrub and perennial grass cover in pinyon-juniper woodlands is also relatively low, with the dominant shrubs being mountain mahogany (*Cercocarpus montanus*) and big sagebrush (*A. tridentata*), while the dominant perennial grasses are bluebunch wheatgrass (*Pseudoroegneria spicata*), muttongrass (*Poa fendleriana*), and prairie Junegrass (*Koeleria macrantha*).

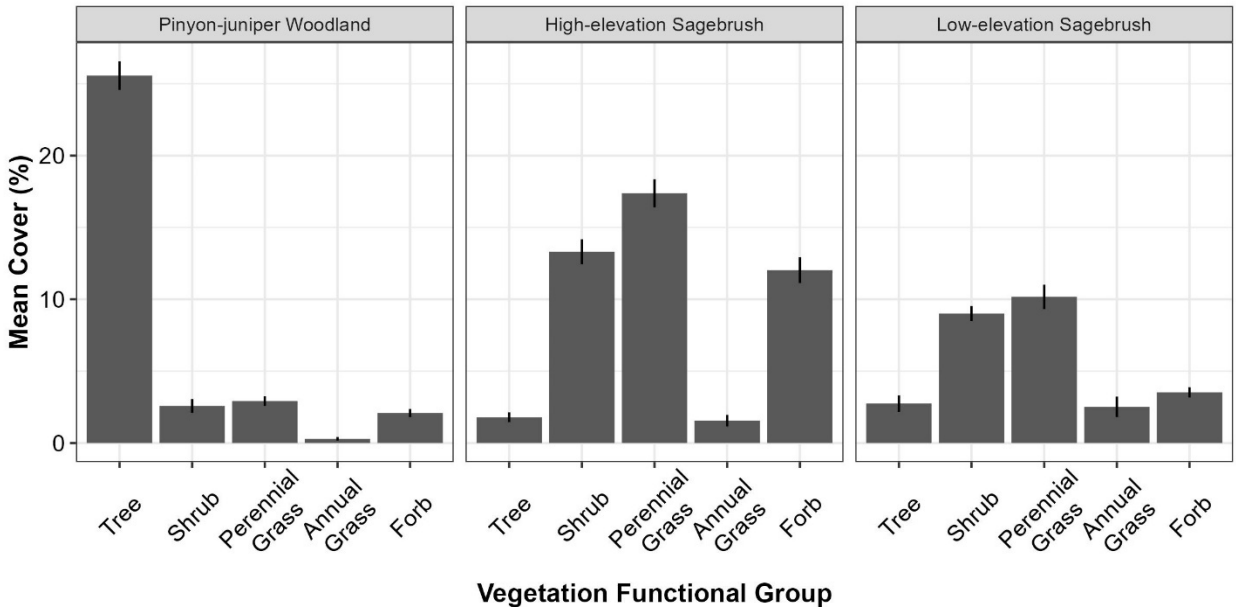


Figure 4. Percent cover of vegetation functional groups. Cover is the mean of all upland monitoring plots across all years of monitoring, shown with ±1 standard error of the mean.

High- and low-elevation sagebrush communities have a similar distribution of functional groups, but with generally greater cover in the cooler, moister higher elevations (Figure 4). The dominant perennial grasses in both high- and low-elevation sagebrush communities are needle-and-thread grass (*Hesperostipa comata*) and western wheatgrass (*Pascopyrum smithii*). Sandberg bluegrass (*Poa secunda,* native) and Kentucky bluegrass (*Poa pratensis,* non-native) are also common in high-elevation sagebrush communities. While most of the dominant perennial grasses are cool-season species (C3 photosynthetic pathway), low-elevation sagebrush also has relatively high cover of sand

dropseed (*Sporobolus cryptandrus*), a warm-season (C4) grass. High-elevation sagebrush communities have a much greater presence of perennial forbs, including arrowleaf balsamroot (*Balsamorhiza sagittata*), silvery lupine (*Lupinus argenteus*), and hairy false goldenaster (*Heterotheca villosa*). The dominant shrub in both high- and low-elevation communities is *A. tridentata,* but high-elevation sagebrush communities also have mountain mahogany (*Cercocarpus montanus*) and antelope bitterbrush *(Purshia tridentata*), while low-elevation sagebrush communities include greasewood (*Sarcobatus vermiculatus*), spineless horsebrush (*Tetradymia canescens*), and spiny hopsage (*Grayia spinosa*). Rabbitbrush (*Ericameria nauseosa*) is common in both communities.

Cheatgrass is widespread across Dinosaur NM; however, in low-elevation sagebrush communities, Russian thistle (*Salsola tragus*) has similarly high cover. Non-native annual forbs like tall tumblemustard (*Sisymbrium altissimum*) and desert madwort (*Alyssum desertorum*) are present in high-elevation sagebrush (although fairly low cover, ~ 1%), and redstem stork's bill (*Erodium cicutarium*) is additionally found in low-elevation sagebrush communities.

At the soil surface, litter made up the highest percent cover in all strata (Figure 5). Undifferentiated crust, which is a less developed stage of biological soil crust, had the second highest percent cover. However, undifferentiated crust is not a good indicator of either soil stability or susceptibility to erosion since it may change in either direction, i.e., development into biological soil crust or degradation to bare soil. Therefore, it was not included in the analysis of surface indicators.

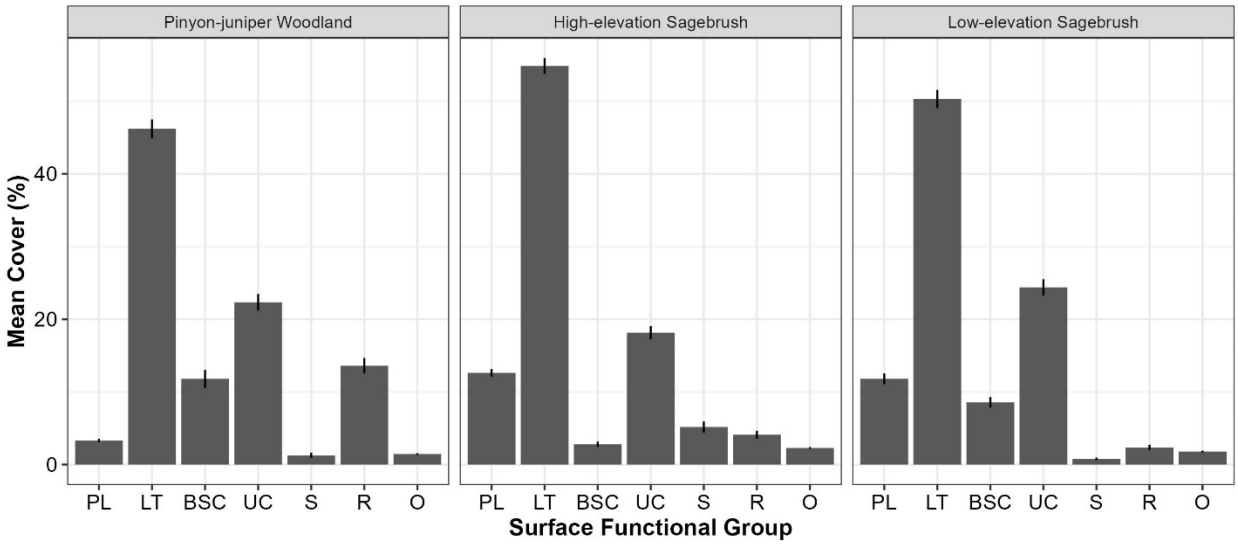


Figure 5. Percent cover of functional groups at the surface. PL = Plant, LT = Litter, BSC = Biological soil crust, UC = Undifferentiated crust, S = Bare soil, R = Rock, and O = Other. Cover is the mean of all upland monitoring plots across all years of monitoring, with ±1 standard error of the mean.

Biological soil crust cover was greatest in pinyon-juniper woodlands. These woodlands have an open understory with low plant cover and are generally subject to less disturbance from livestock or wildlife grazing, although some plots are within grazing allotment boundaries. They also have a

relatively high percent cover of rock, including bedrock, which is not susceptible to erosion. High- and low-elevation sagebrush communities had similar amounts of plant cover, but high-elevation sagebrush had lower biological soil crust with a higher percent cover of bare soil compared to the low-elevation sagebrush (Figure 5). Bare soil is highly susceptible to erosion. The "other" category for surface functional groups included washes and occasional disturbances by humans, wildlife, or livestock.

3.2 Climate

Temperatures over the monitoring period were above the long-term (1980–2010) average of 7.4°C (Figure 6) in 9 out of the 12 monitoring years. The average annual temperature during the monitoring period (red line on Figure 6) was 0.5°C warmer than the long-term average. This is consistent with the observed climate warming trend in the region (Monahan and Fisichelli 2014). 2012 was particularly hot, with the average annual temperature over 2°C greater than the long-term average. Annual P was generally closer to the long-term average and there was little (~ 1 cm) difference between the long-term and monitoring period averages (Figure 6). However, 2012, 2018, and 2020 were noticeably dry years, all less than 75% of normal. The period from 2014 to 2016 included the wettest years of the monitoring period.

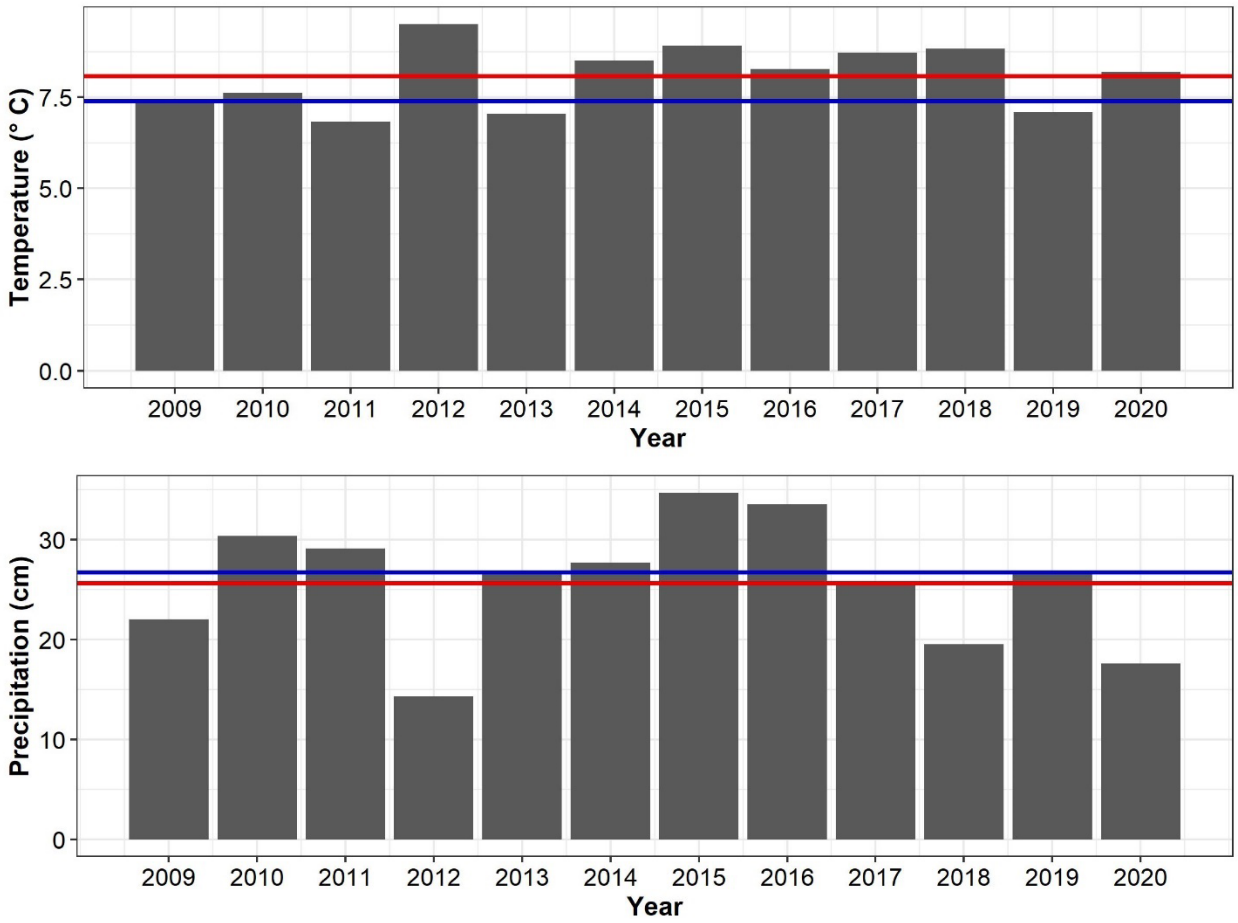


Figure 6. Mean annual temperature and precipitation during the monitoring period. Blue lines show the long-term average of each metric from 1980 to 2010, and red lines show the average during the monitoring period from 2009 to 2020..

Despite wetter individual years, drought conditions were prevalent over most of the monitoring period. The Standardized Precipitation Evapotranspiration Index (SPEI) shown in Figure 7 indicates longer term drought conditions. The SPEI is a standardized index of precipitation minus potential evapotranspiration (P – PET) accumulated over a specified timescale, in this case five years. The SPEI standardizes relative to the reference period (1980–2020) and transforms to a Z-score (number of standard deviations above or below the mean) for interpretation. The drought beginning in 2012 lasted multiple years but was less intense (SPEI value closer to zero) than the drought beginning around 2018 and continuing through the end of 2020. Generally, the wet periods were shorter in duration and intensity than the dry periods, which were extreme compared to historical conditions.

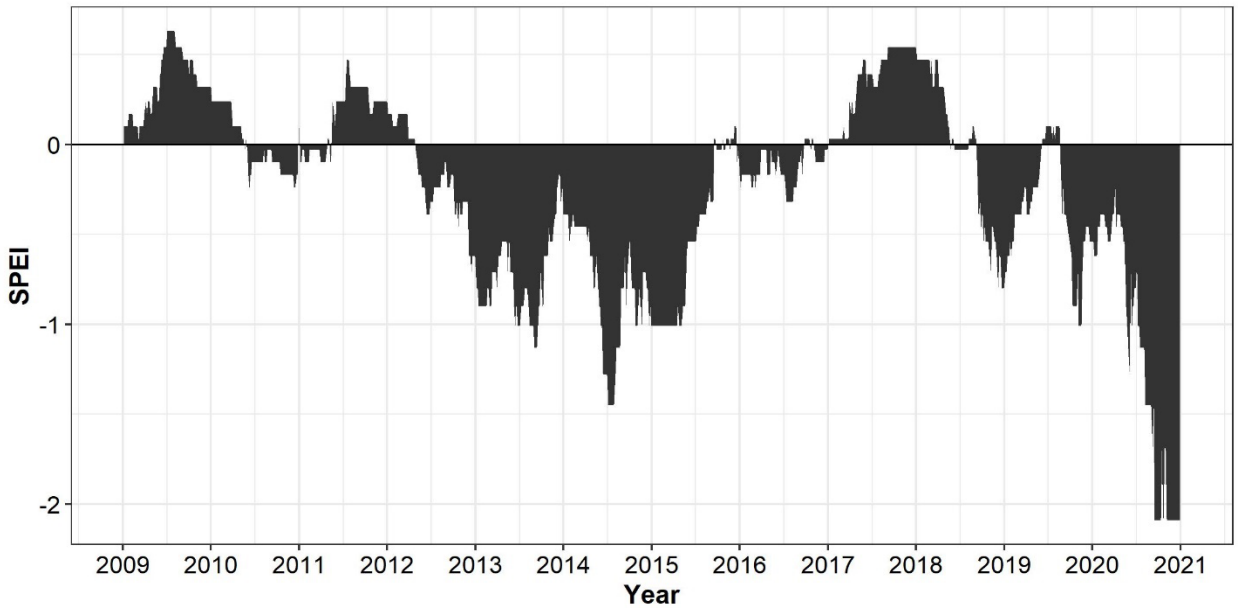


Figure 7. 5-year Standardized Precipitation Evapotranspiration Index (SPEI) for Dinosaur National Monument from 2009 to 2020. The SPEI is the standardized index of precipitation (P) minus potential evapotranspiration (PET) accumulated over 5 years and is presented on the y-axis as a Z-score (number of standard deviations from the mean), which compares to the distribution of values from the 1980 to 2020 reference period. An SPEI value of 2 or more is considered extremely wet and -2 or less indicates extreme drought. Data were accessed from Climate Engine (app.climateengine.com, accessed 24 May 2023, Abatzoglou, 2013).

Figure 8 shows the actual values of water balance covariates (P, AET, and D) used in the Bayesian trend analysis over the monitoring period. Because covariates were summarized over time periods prior to monitoring, summaries are presented of water balance in the 12-month period prior to the sampling year rather than calendar year summaries. For example, if the sample year was 2017, cool-season AET is the sum of cool-season AET from October 2016 to May 2017. Cool- and monsoon-season P varied across the monitoring period, with high cool-season P in sample years 2011, 2016, and 2019 and high monsoon-season P in 2014–2018 (Figure 8). Precipitation contributes to AET and D; however, the latter water balance metrics also incorporate temperature, slope, aspect, and soil properties that impact plant water use. For example, high temperatures and below-average P in 2012 (see Figure 6) likely contributed to high monsoon-season D in sample year 2013 (Figure 8). Monsoon-season D was generally greater than cool-season D, due to lower water input and greater evaporative demand during those months.

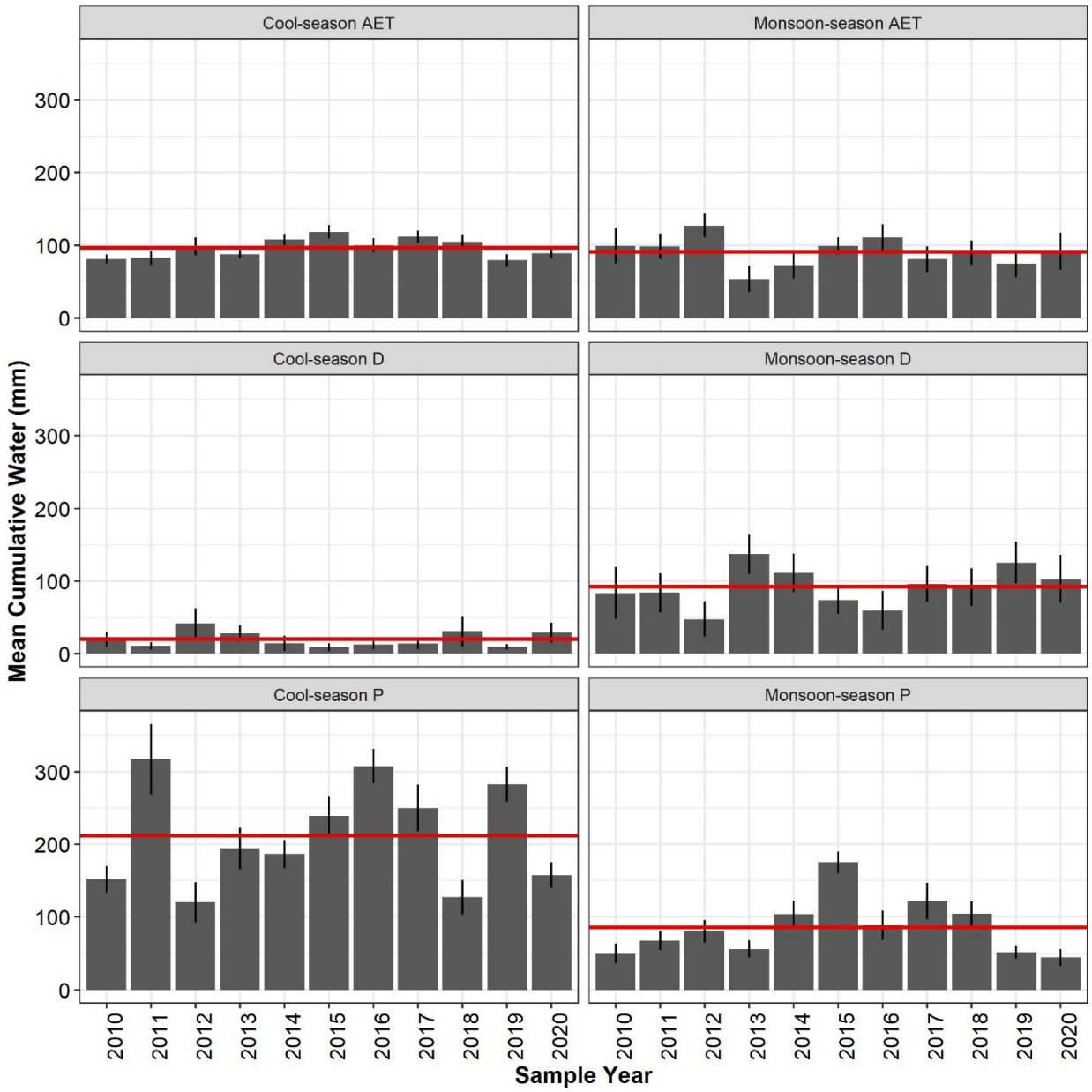


Figure 8. Water balance covariates used in Bayesian models over the monitoring period. Covariates include actual evapotranspiration (AET), water deficit (D), and precipitation (P) over cool (Oct–May) and monsoon (Jul–Sept) seasons. Sample year indicates the year sampling took place, and water balance covariates are summed over the time period prior to sampling. The bars indicate the cumulative water balance metric averaged across plots (n=135) ± 1 standard deviation. Red lines show the mean value over the monitoring period for each metric.

3.3 Bayesian data analysis

Final specifications for Bayesian hierarchical models are shown in Table 3. The 11 models presented in this report all converged and passed posterior predictive checks, indicating model fit and performance were within an acceptable range. Bayesian p-values represent how well the model can predict a distribution of a statistic of the observed data: ideal values are close to 0.5, with extremes of

>0.9 or <0.1 indicating a poor fit (Hooten and Hobbs 2015). Attempted models for canopy and basal gaps did not pass posterior predictive checks, meaning that any inference from those model results would be faulty. Therefore, status and trend for canopy and basal gaps are not included in this report. The summary of canopy and basal gap size classes are presented instead.

Water balance (AET, D) variables outperformed P alone as covariates, and cool- and/or monsoon-season D was in 6 of 11 final models (Table 3). Cumulative GDD was present in the final model for 6 of 11 indicators. While the grazed and/or time since burned covariates were present in 3 of the final models, they never had a significant directional effect.

Table 3. Bayesian model specifications for indicators. All models take the form of a general linear model with a site-level random effect, i.e., intercepts in the model were allowed to vary by site. The likelihood describes the form of the response variable. The deterministic model represents the ecological process being modeled. In addition to having varying site means, the variance of sites could either be fixed or variable, indicated by a hierarchical variance structure. The Gelman diagnostic indicates model convergence and BP (Bayesian p-value for the model mean and variance) is an evaluation of model fit.

Indicator Category	Indicator	Likelihood	Deterministic Model	Variance Structure	Gelman Diagnostic	B_P (mean, variance)	Burn-in and Iterations of MCMC Sampler	Covariates (direction of effect)[1]
Vegetation Indicators	Live cover	Beta binomial	Inverse logit	Fixed	1.01	0.5, 0.61	20,000 and 10,000	• **Cool-season D (-)[2]** • **Monsoon-season D (-)[2]** • **GDD (-)[2]**
	Perennial grass cover	Zero-inflated beta binomial	Inverse logit	Fixed	1.01	0.51, 0.53	20,000 and 10,000	• Monsoon-season AET • **Cool-season AET (+)[2]** • **GDD (-)[2]**
	Exotic annual cover	Zero-inflated beta binomial	Inverse logit	Fixed	1.06	0.51, 0.62	30,000 and 10,000	• **Monsoon-season P (+)[2]** • GDD
	Exotic frequency	Beta binomial	Inverse logit	Fixed	1.01	0.47, 0.38	20,000 and 10,000	• **Monsoon-season D (+)[2]** • Grazed • GDD
	Big sagebrush cover	Zero-inflated beta-binomial	Inverse logit	Fixed	1.01	0.51, 0.75	20,000 and 10,000	• GDD
	Big sagebrush density	Negative-binomial-simple	Linear	Hierarchical	1.08	0.3, 0.49	20,000 and 10,000	• Cool-season D • Monsoon-season D
	Tree cover	Zero-inflated beta binomial	Inverse logit	Fixed	1.05	0.53, 0.56	20,000 and 10,000	• Monsoon-season P • Burned
	Sapling density	Negative-binomial-simple	Exponential	Fixed	1	0.51, 0.51	20,000 and 10,000	• **Monsoon-season P (+)[2]** • **Cool-season P (+)[2]**
Soil Surface Indicators	Biological soil crust cover	Zero-inflated beta-binomial	Inverse logit	Fixed	1.07	0.52, 0.76	30,000 and 10,000	• Monsoon-season D, GDD
	Bare soil cover	Zero-inflated beta-binomial	Inverse logit	Fixed	1.03	0.54, 0.5	20,000 and 10,000	• Cool-season D, Grazed

[1] D = water deficit, P = precipitation, AET = actual evapotranspiration, GDD = cumulative growing degree-days.

[2] Indicates that the covariate had a significant effect with the direction of influence on the response variable indicated in parentheses as positive (+) or negative (-) (also shown in bold font)

Table 3 (continued). Bayesian model specifications for indicators. All models take the form of a general linear model with a site-level random effect, i.e., intercepts in the model were allowed to vary by site. The likelihood describes the form of the response variable. The deterministic model represents the ecological process being modeled. In addition to having varying site means, the variance of sites could either be fixed or variable, indicated by a hierarchical variance structure. The Gelman diagnostic indicates model convergence and BP (Bayesian p-value for the model mean and variance) is an evaluation of model fit.

Indicator Category	Indicator	Likelihood	Deterministic Model	Variance Structure	Gelman Diagnostic	B_P (mean, variance)	Burn-in and Iterations of MCMC Sampler	Covariates (direction of effect)[1]
Soil Surface Indicators (continued)	Soil stability rating	Ordinal-latent-normal	Linear	Hierarchical	1.02	0.5, 0.57	20,000 and 10,000	• **Cool-season D (-)[2]** • **Monsoon-season D (-)[2]** • Grazed • Burned

[1] D = water deficit, P = precipitation, AET = actual evapotranspiration, GDD = cumulative growing degree-days.

[2] Indicates that the covariate had a significant effect with the direction of influence on the response variable indicated in parentheses as positive (+) or negative (-) (also shown in bold font)

3.3.1 Vegetation indicators

Total live vegetation cover was greater in 2010 than by the end of the monitoring period in 2020 in all strata (Figure 9A). Live cover responded negatively to both cool- and monsoon-season D in the year prior to monitoring: when D increased, live cover decreased. It also responded negatively to cumulative GDD. Live cover decreased early in the monitoring period in 2013, following a high D and drought in 2012, but increased with wetter conditions from 2014 to 2016 (see Figure 6). After accounting for interannual variation in D, there was no evidence of an increasing or decreasing trend over time (Table 4, Figure 9B). This indicates that the change in total live vegetation cover is explained by both cool- and monsoon-season D. However, the functional groups that comprised the total live cover had varied responses.

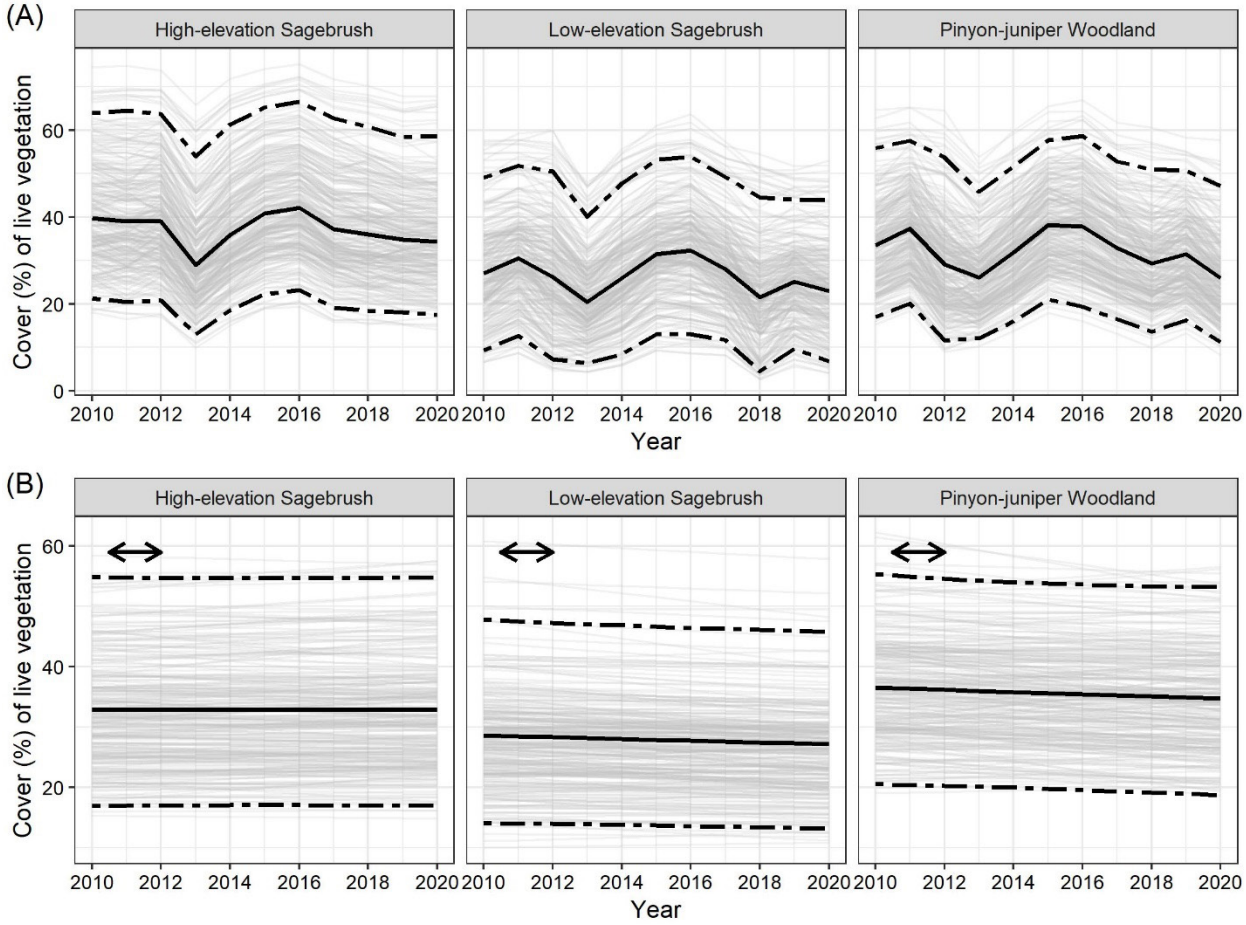


Figure 9. Model-predicted live vegetation cover. Lines describe the posterior predicted mean distribution: the light gray lines are draws from the distribution, the solid black line is the median of the distribution, and the dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in cover predicted by cool- and monsoon-season deficit (D). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

Table 4. Status and trend of all indicators. Ten-year minimum and maximum status are the model-predicted mean values of the indicator with the 95% credible interval for the year with the lowest and highest values. Direction of trend over time indicates whether a significant increase or decrease occurred while holding climate covariates at their mean; 'None' means that no significant trend was detected. A trend is considered significant if the 95% credible interval of the posterior distribution of the time slope coefficient (β1) does not include zero. Annual average rate of change is the slope of significant trend lines in the same units as the indicator, shown with the 95% credible interval.

Indicator	Stratum	10-year Minimum Status (95% CI)	10-year Maximum Status (95% CI)	Direction of Trend over Time	Annual Average Rate of Change (95% CI)
Live cover (%)	High-elevation sagebrush	30 (13, 54)	43 (23, 67)	None	--
	Low-elevation sagebrush	21 (6, 40)	33 (13, 54)	None	--
	Pinyon-juniper woodlands	26 (11, 48)	38 (21, 58)	None	--
Perennial grass cover (%)	High-elevation sagebrush	7 (2, 18)	17 (7, 39)	Decreasing	-0.5 (-0.6, -0.1)
	Low-elevation sagebrush	7 (<1, 18)	15 (<1, 37)	Decreasing	-0.3 (-0.5, 0)
	Pinyon-juniper woodlands	2 (0, 6)	4 (0, 13)	None	--
Exotic annual cover (%)	High-elevation sagebrush	2 (<1, 8)	12 (<1, 47)	Increasing	1 (0.2, 4)
	Low-elevation sagebrush	12 (<1, 40)	42 (4, 82)	Increasing	3 (2, 5)
	Pinyon-juniper woodlands	4 (0, 16)	10 (0, 34)	None	--
Exotic frequency (proportion)	High-elevation sagebrush	0.44 (0.01, 0.99)	0.83 (0.14, 0.99)	Increasing	0.04 (0.01, 0.08)
	Low-elevation sagebrush	0.85 (0.18, 0.99)	0.97 (0.82, 0.99)	Increasing	0.01 (<0.001, 0.06)
	Pinyon-juniper woodlands	0.47 (0.01, 0.99)	0.76 (0.07, 0.99)	Increasing	0.02 (0.01, 0.06)
Big sagebrush cover (%)	High-elevation sagebrush	11 (0, 29)	15 (12, 38)	Increasing	0.4 (0.1, 1)
	Low-elevation sagebrush	8 (<1, 21)	10 (<1, 26)	Increasing	0.2 (0.1, 0.6)
	Pinyon-juniper woodlands	2 (0, 13)	2 (0, 15)	None	--
Big sagebrush density (shrubs/ha)	High-elevation sagebrush	2025 (<1, 7245)	2121 (<1, 7377)	Increasing	11 (7, 23)
	Low-elevation sagebrush	1579 (<1, 3845)	1658 (<1, 3936)	None	--
	Pinyon-juniper woodlands	139 (<1, 957)	284 (<1, 1271)	Decreasing	14 (-23, 0)
Tree cover (%)	High-elevation sagebrush	1 (0, 10)	2 (0, 15)	None	--
	Low-elevation sagebrush	2 (0, 22)	2 (0, 23)	None	--
	Pinyon-juniper woodlands	22 (13, 37)	25 (15, 41)	None	--
Sapling density (trees/ha)	Pinyon-juniper woodlands	224 (14, 852)	1206 (69, 4329)	Decreasing	-67 (-93, -4)

Table 4 (continued). Status and trend of all indicators. Ten-year minimum and maximum status are the model-predicted mean values of the indicator with the 95% credible interval for the year with the lowest and highest values. Direction of trend over time indicates whether a significant increase or decrease occurred while holding climate covariates at their mean; 'None' means that no significant trend was detected. A trend is considered significant if the 95% credible interval of the posterior distribution of the time slope coefficient (β1) does not include zero. Annual average rate of change is the slope of significant trend lines in the same units as the indicator, shown with the 95% credible interval.

Indicator	Stratum	10-year Minimum Status (95% CI)	10-year Maximum Status (95% CI)	Direction of Trend over Time	Annual Average Rate of Change (95% CI)
Biological soil crust cover	High-elevation sagebrush	1 (0, 7)	6 (0, 28)	Increasing	0.4 (0.2, 2)
	Low-elevation sagebrush	5 (0, 17)	12 (0, 38)	Increasing	0.7 (0.2, 2)
	Pinyon-juniper woodlands	9 (<1, 27)	17 (<1, 43)	Increasing	0.6 (0.4, 1)
Bare soil cover	High-elevation sagebrush	3 (0, 15)	6 (0, 26)	Increasing	0.2 (<0.1, 1)
	Low-elevation sagebrush	<1 (0, 2)	<1 (0, 5)	None	--
	Pinyon-juniper woodlands	<1 (0, 5)	1 (0, 7)	None	--
Soil stability rating	High-elevation sagebrush	3.5 (1.3, 5.1)	4.5 (2.4, 5.7)	Increasing	0.004 (<0.01, 0.02)
	Low-elevation sagebrush	4.2 (2.6, 5.2)	5.2 (3.9, 5.8)	Increasing	0.007 (0.002, 0.02)
	Pinyon-juniper woodlands	4.1 (2, 5.8)	5.3 (3.5, 6)	Increasing	0.008 (0.002, 0.02)

Cool-season AET had a significant positive relationship with perennial grass cover (see Table 3), which was expected since perennial grasses at Dinosaur NM are primarily cool-season (C3) species. Cumulative GDD had a significant negative relationship with perennial grass cover, indicating that cooler temperatures in late winter and early spring favored the growth of perennial grasses. In high- and low-elevation sagebrush, lower than average cool-season AET contributed to lower perennial grass cover in 2013, and again in 2019 and 2020 (see Figure 8, Figure 10A). After controlling for interannual variation in AET, perennial grasses showed a decreasing trend over time in high- and low-elevation sagebrush (see Table 4, Figure 10B). This indicates that perennial grass cover is decreasing regardless of climate during the monitoring period, and additional unmodeled drivers (e.g., competition, cumulative effects of long-term drought) may be playing a role. However, the average annual rate of change was low for both high- and low-elevation sagebrush (≤0.5%, see Table 4). Perennial grass cover in pinyon-juniper woodlands was generally low (<5%) and varied little between years (see Table 4, Figure 10A).

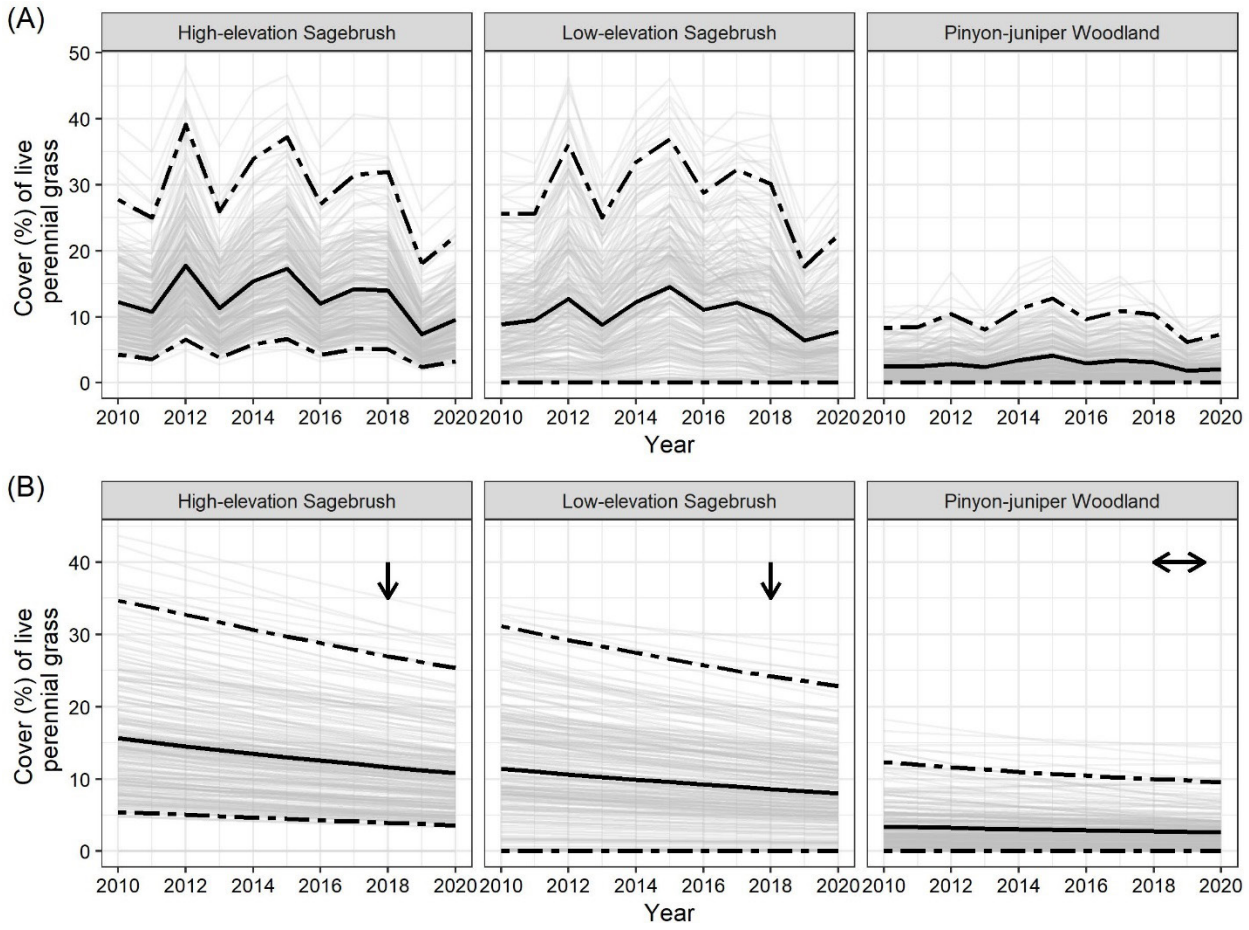


Figure 10. Model-predicted live perennial grass cover. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in cover predicted by cool- and monsoon-season actual evapotranspiration (AET). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

While perennial grasses decreased, cover of exotic annuals increased greatly in high- and low-elevation sagebrush. Modeled percent cover in high-elevation sagebrush increased from 2% in 2010 to 12% in 2020, and in low-elevation sagebrush increased from 12% to 42% (Figure 11A). Predicted cover in low-elevation sagebrush had wide credible intervals, with mean exotic annual cover falling between <1% and 82% (see Table 4, Figure 11A). Cover of exotic annuals in pinyon-juniper woodlands varied little over the monitoring period and remained at or below 10% in all years (Figure 11A). Monsoon-season P had a significant positive effect on exotic cover (see Table 3). High- and low-elevation sagebrush had a trend of increasing exotic cover over time at relatively high rates of 1% and 3% per year, respectively (see Table 4, Figure 11B). Pinyon-juniper woodlands did not show a trend of increasing or decreasing exotic cover (see Table 4, Figure 11B).

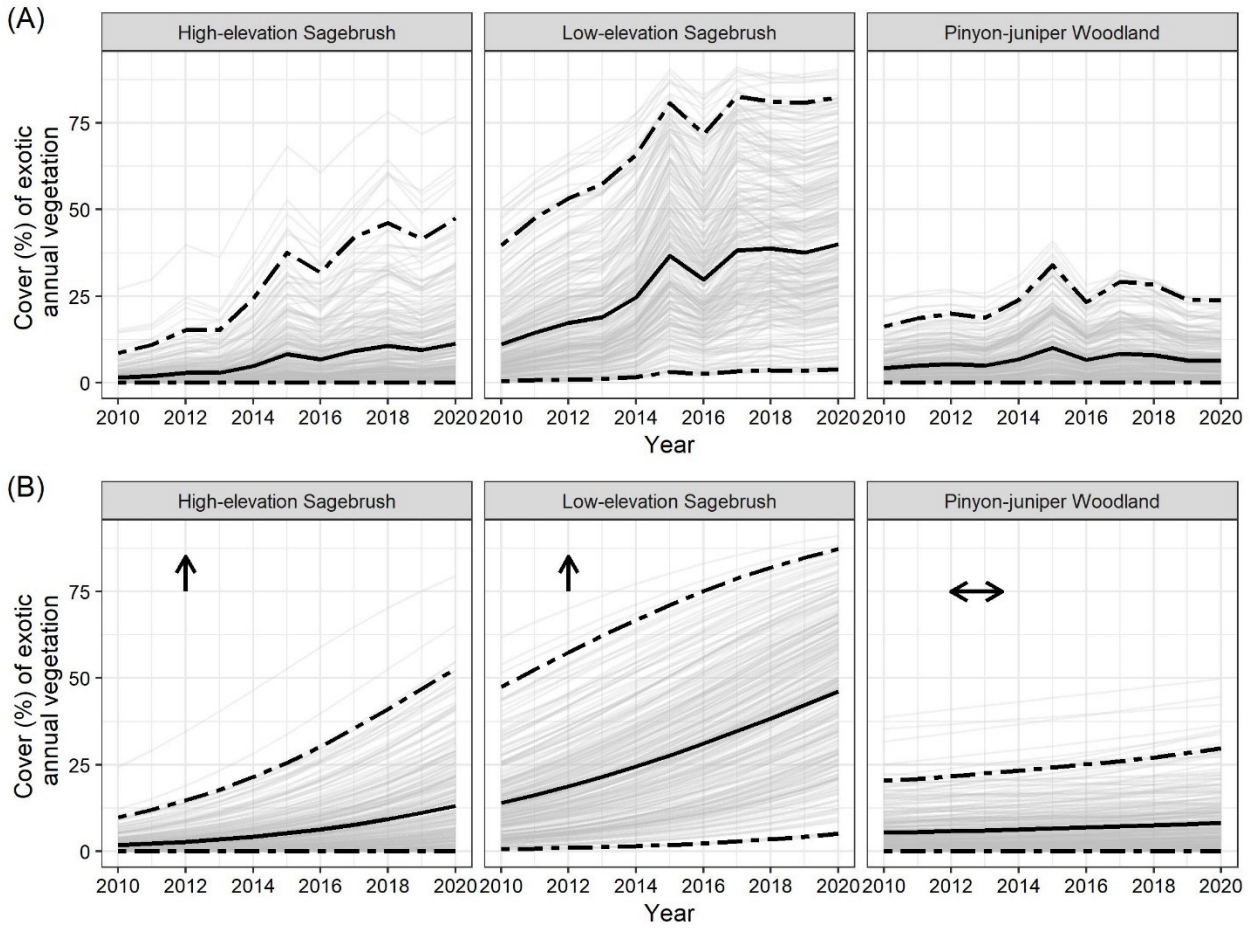


Figure 11. Model-predicted exotic annual vegetation cover. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in cover predicted by monsoon-season precipitation (P). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

Frequency of exotic species in 1-m^2 quadrats was also measured, which is a more sensitive but less precise method of detecting changes in plant cover. Exotic species' frequency responded positively

to monsoon-season D. Grazing and cumulative growing degree day covariates were also included in the final model but did not have significant effects. The proportion of quadrats where exotic species were detected increased in all strata over the 2011–2020 monitoring period (Figure 12A). However, in low-elevation sagebrush, exotic frequency started relatively high at 0.85. Results were consistent with the increases in exotic annual cover shown in the line-point intercept data but also showed that pinyon-juniper woodlands have an increasing presence of exotic species despite lower overall cover. While exotic frequency is clearly increasing, both pinyon-juniper and high-elevation sagebrush have very wide confidence intervals, highlighting the high variability between plots (Figure 11A). Cheatgrass (*B. tectorum*) was the dominant exotic species detected, but others include tall tumblemustard (*S. altissimum*), desert madwort (*A. desertorum*), and redstem stork's bill (*E. cicutarium*). After controlling for interannual variation in monsoon-season D, the frequency of exotics had an increasing trend over time in all strata (see Table 4, Figure 11B).

Since big sagebrush (*A. tridentata*) is the dominant shrub and a species of management concern at Dinosaur NM, this was the only species modeled for shrubs. None of the water balance covariates were influential in the final big sagebrush cover model, indicating that longer-term water balance dynamics or other factors may be influencing big sagebrush trends over time. As a result, modeled big sagebrush cover did not vary between years; however, it increased slightly between 2010 and 2020 in high- and low-elevation sagebrush (Figure 13A). Big sagebrush cover did show a significant positive trend over time in both high-elevation sagebrush and low-elevation sagebrush communities, but at low average annual rates of change (≤0.4%, see Table 4, Figure 13B).

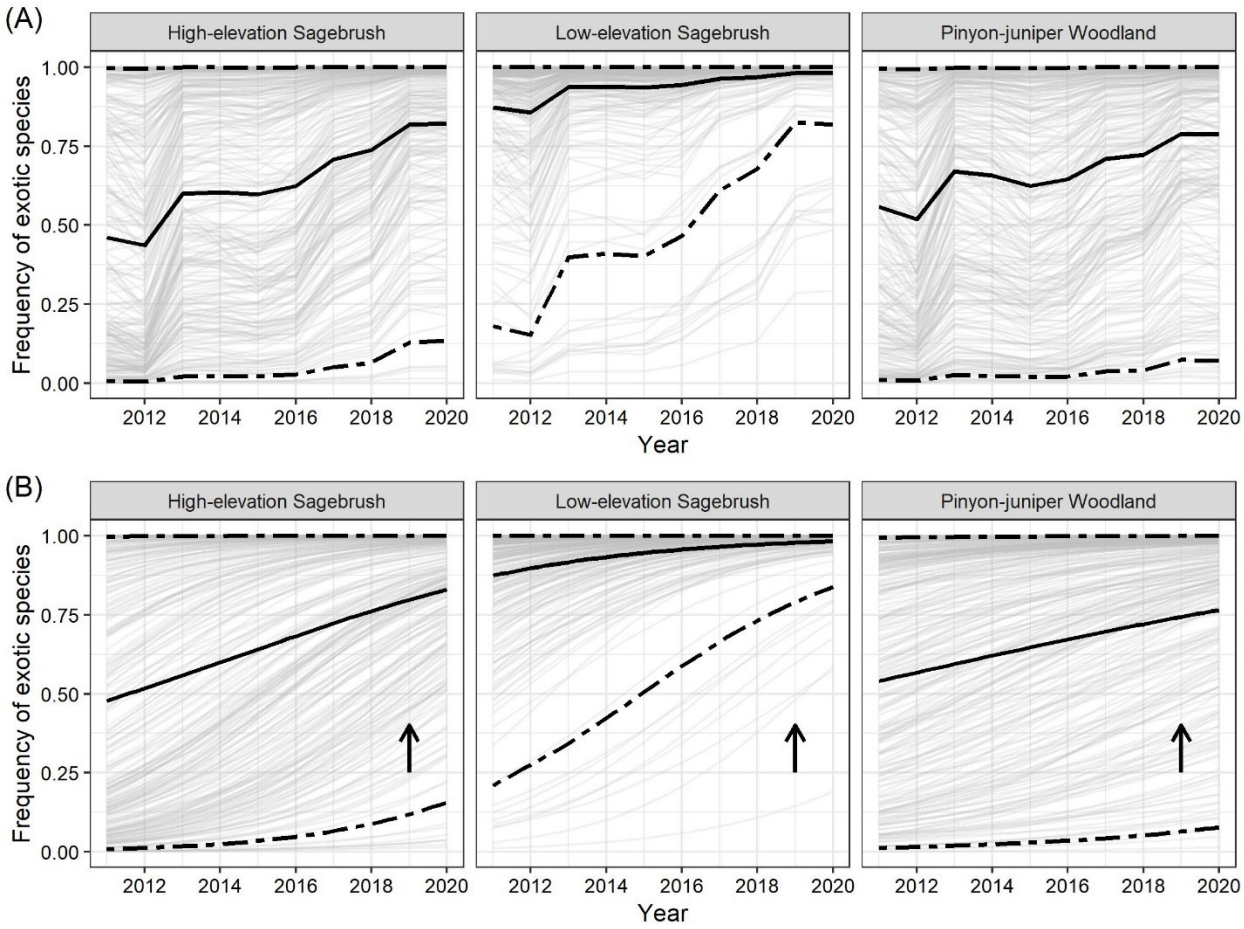


Figure 12. Model-predicted exotic frequency. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in frequency predicted by monsoon-season deficit (D). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

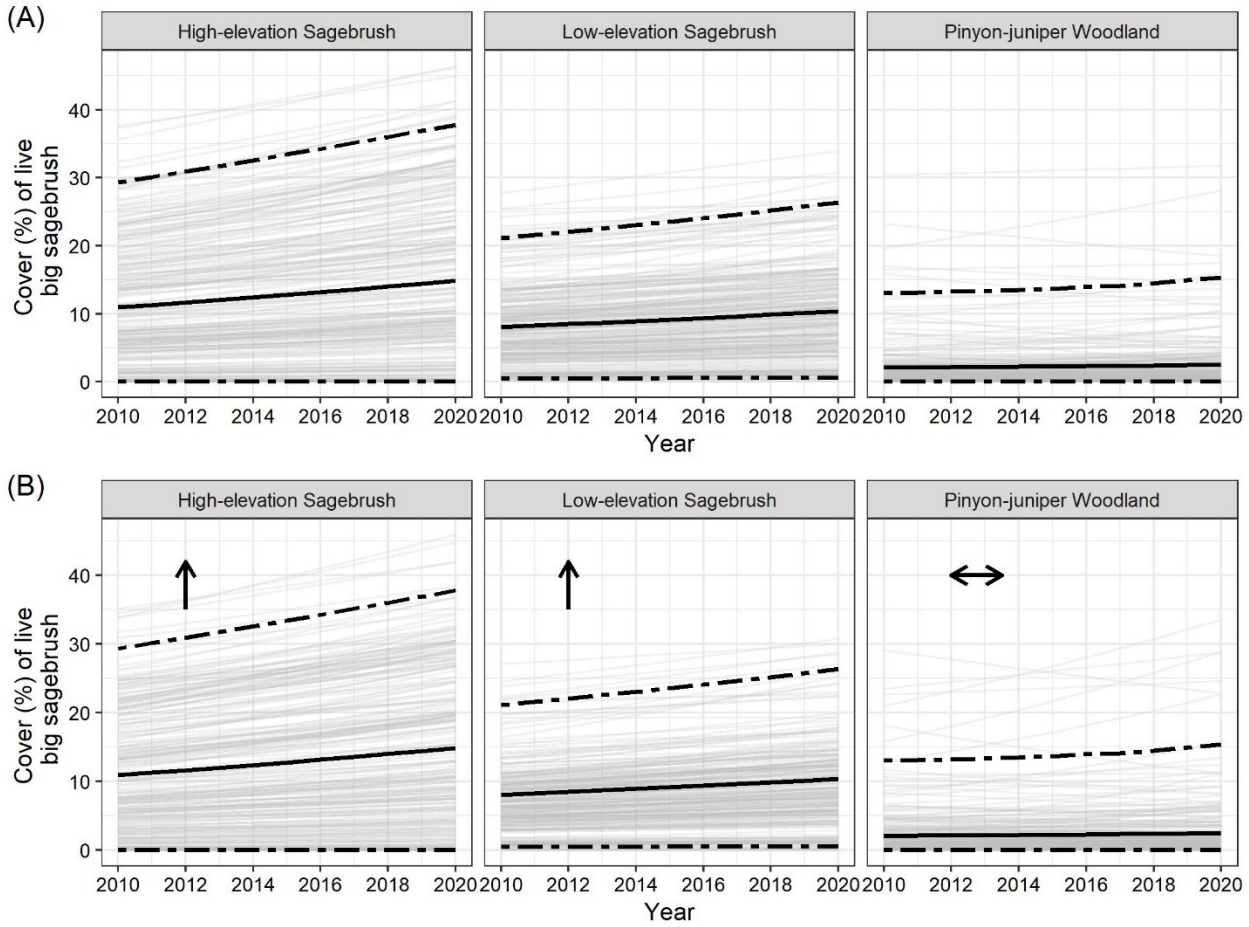


Figure 13. Model-predicted big sagebrush cover. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in cover predicted by cumulative growing degree-days. (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

In the model for big sagebrush density, the smallest size class was excluded. Although recruitment events were observed during the monitoring period, these often have high mortality and increase uncertainty around adult big sagebrush density predictions. Seedlings that persist into adulthood would be captured in counts during subsequent years. Counts of big sagebrush in a one-meter belt showed that density of big sagebrush in all strata was generally stable and varied little between years (Figure 14A). Big sagebrush density was very low in pinyon-juniper woodlands (139–284 shrubs/ha, see Table 4). Cool- and monsoon-season D were included in the big sagebrush density model but did not have a directional impact on the response (see Table 3). High-elevation sagebrush had an increasing trend in big sagebrush density at a rate of 11 shrubs/ha/year (Table 4, Figure 14B). Pinyon-juniper woodlands showed a decreasing trend at a rate of 14 shrubs/ha/year (see Table 4, Figure 14B).

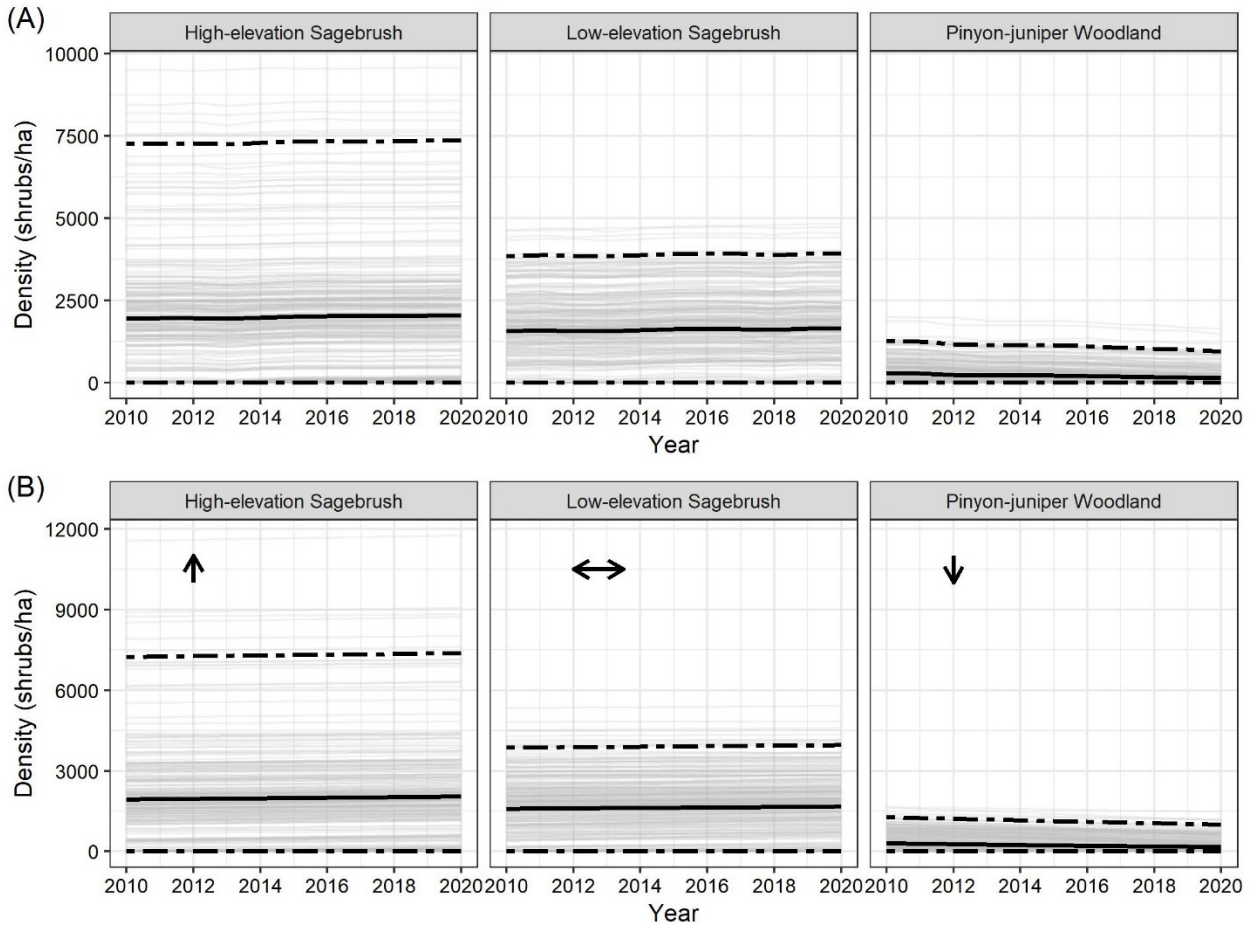


Figure 14. Model-predicted big sagebrush density. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in density predicted by cool- and monsoon-season deficit (D). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

Cover of trees was primarily pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) and was greatest in pinyon-juniper woodlands (see Figure 4). However, while low-elevation sagebrush had low mean tree cover, it was highly variable, and there were some plots with over 20% cover. Although monsoon-season P was included in the final model, it did not have a significant relationship with tree cover (see Table 3). This was expected, especially in long-lived and slow-growing species like pinyon pine and Utah juniper. Time since fire was included in the final model but also did not have a directional relationship (see Table 3). Tree cover varied little between years, and there was no evidence of an increasing or decreasing trend over time (Figure 15B).

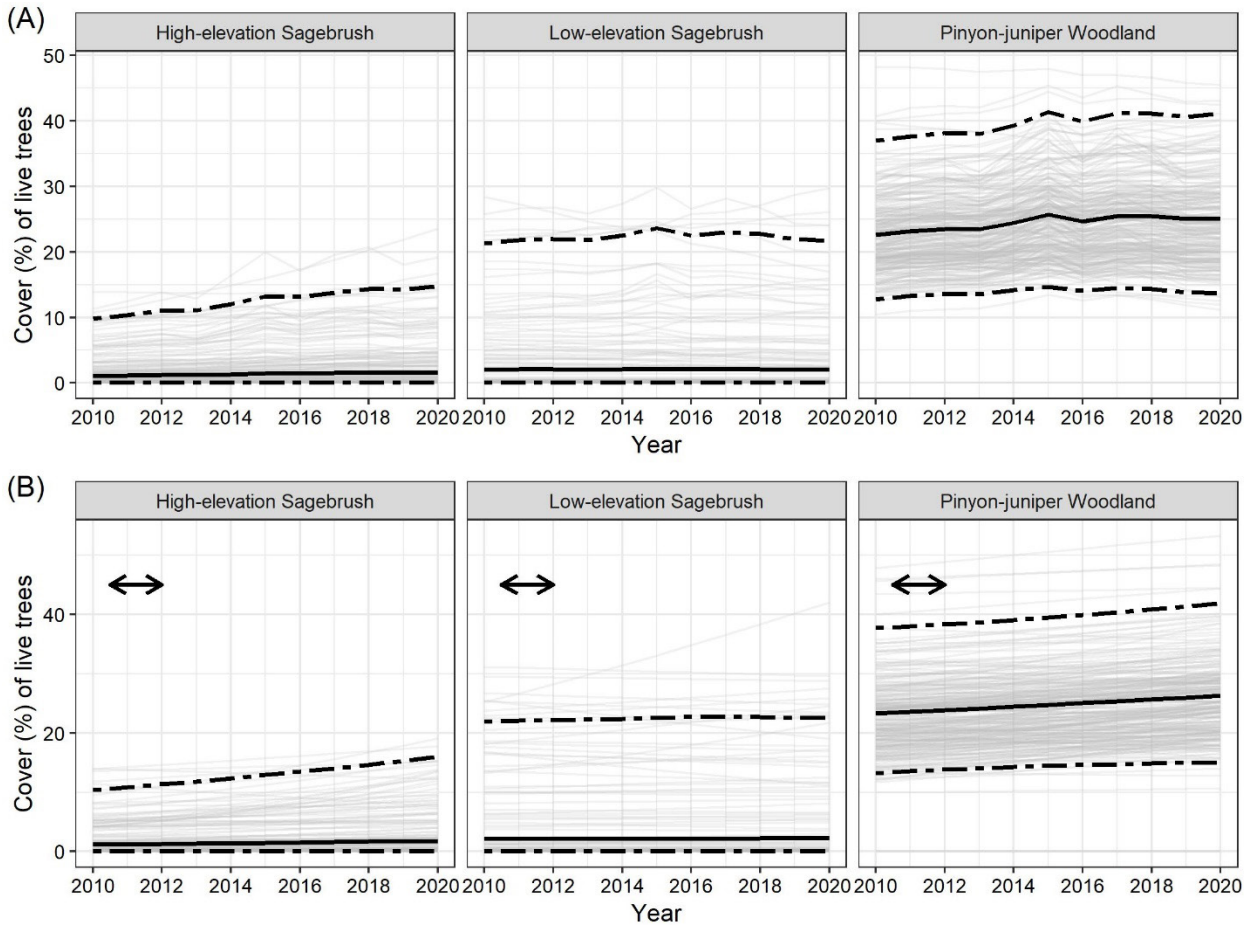


Figure 15. Model-predicted tree cover. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in cover predicted by monsoon-season precipitation (P). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

Sapling density was only evaluated in pinyon-juniper woodlands due to methodological differences in density measurements in high- and low-elevation sagebrush. Methods were standardized in 2016, so sapling density in all strata may be evaluated in future years. The density of pinyon pine and Utah juniper saplings responded positively to cool- and monsoon-season P and was more variable among years than tree cover (see Table 3, Figure 16A). After controlling for interannual variation due to P, sapling density showed a decreasing trend over time (Figure 16B).

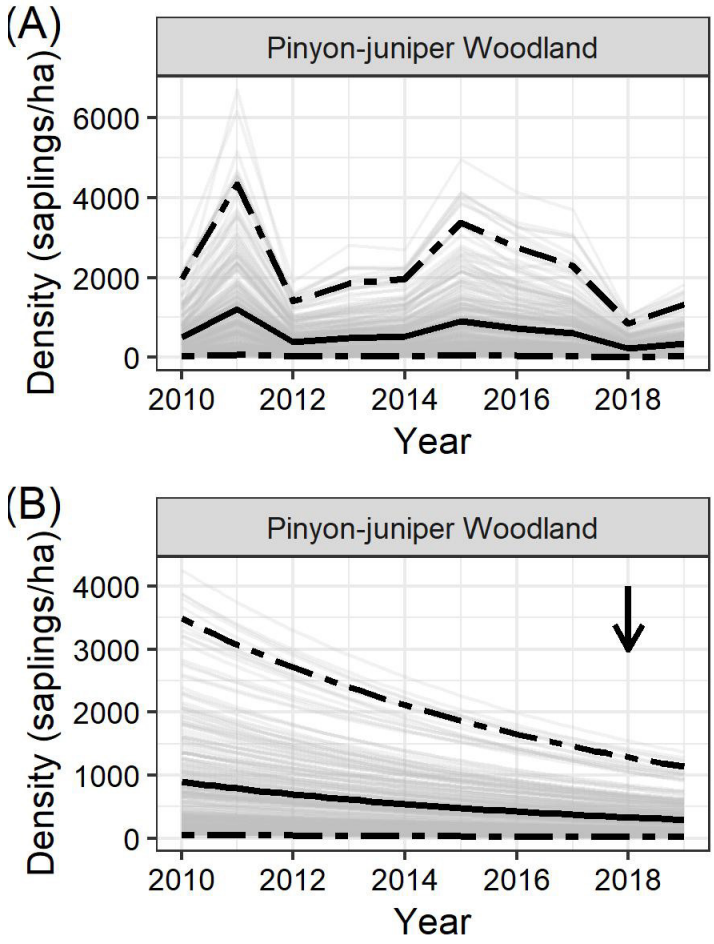


Figure 16. Model-predicted sapling density. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in density predicted by cool- and monsoon-season precipitation (P). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

3.3.2 Soil surface indicators

Biological soil crust was at its highest percent cover in 2020 in all strata (Figure 17A). Crust cover was highest in pinyon-juniper woodlands and lowest in high-elevation sagebrush (Figure 17A). Biological soil crust cover varied little between years, and although monsoon-season D and grazing were included in the final model, they did not have significant directional effects (Table 3). Trend models showed that biological crust cover increased significantly over time in all strata, although the annual average rate of change was less than 1% (see Table 4, Figure 17B).

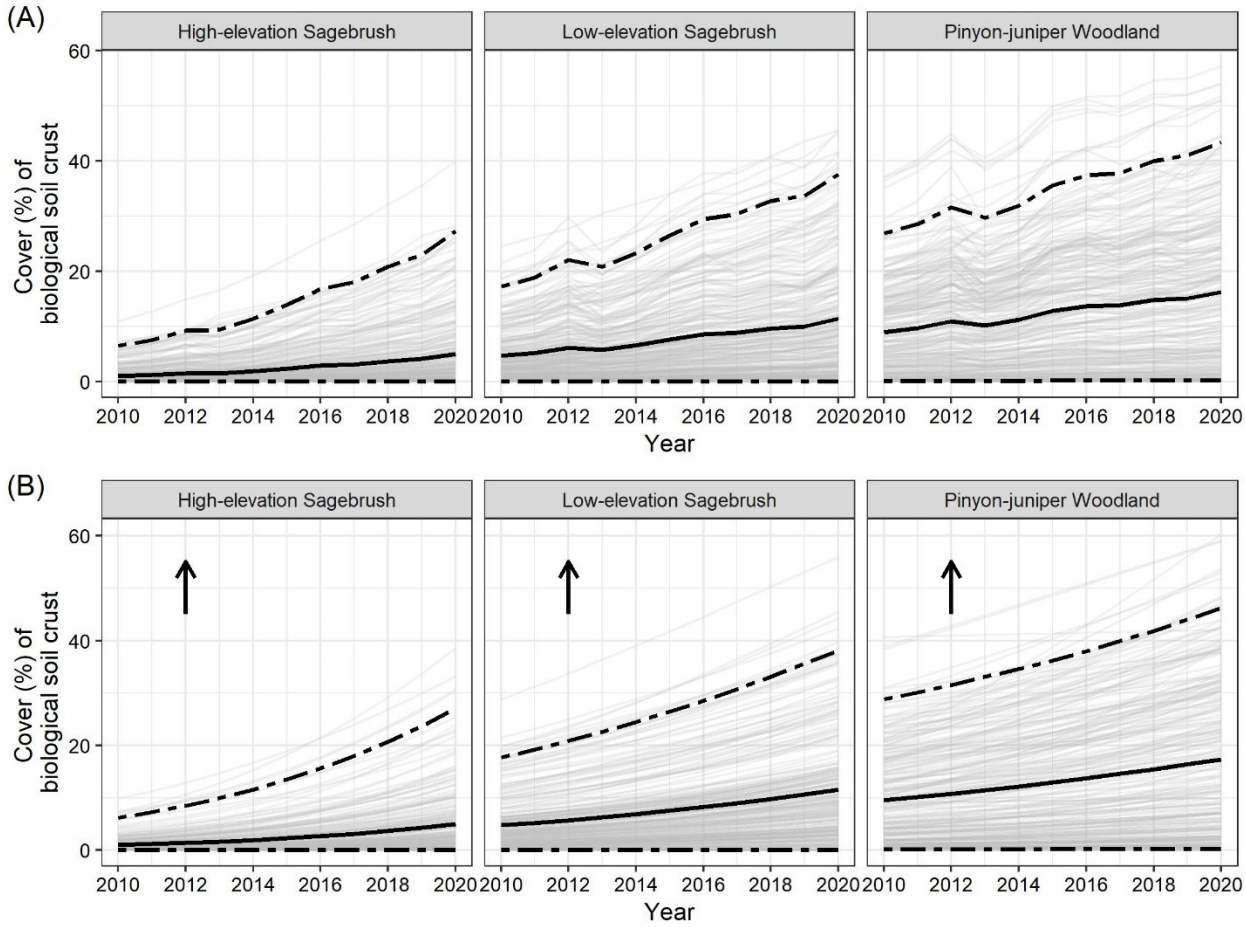


Figure 17. Model-predicted biological soil crust cover. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in cover predicted by monsoon-season deficit (D). (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

Bare soil cover was very low in low-elevation sagebrush and pinyon-juniper woodlands and varied little between years (Figure 18A). In high-elevation sagebrush, bare soil cover was slightly greater, although the model-predicted mean remained at or below 6% (see Table 4). Bare soil cover was greatest in high-elevation sagebrush in 2020 (Figure 18A). Cool-season D and grazing were in the final model for bare soil, but they did not have a significant effect on the response. In addition to the higher cover, high-elevation sagebrush had an increasing trend over time in bare soil cover at a rate of 0.2% per year (see Table 4, Figure 18B). Low-elevation sagebrush and pinyon-juniper woodlands did not have a significant trend over time (Figure 18B).

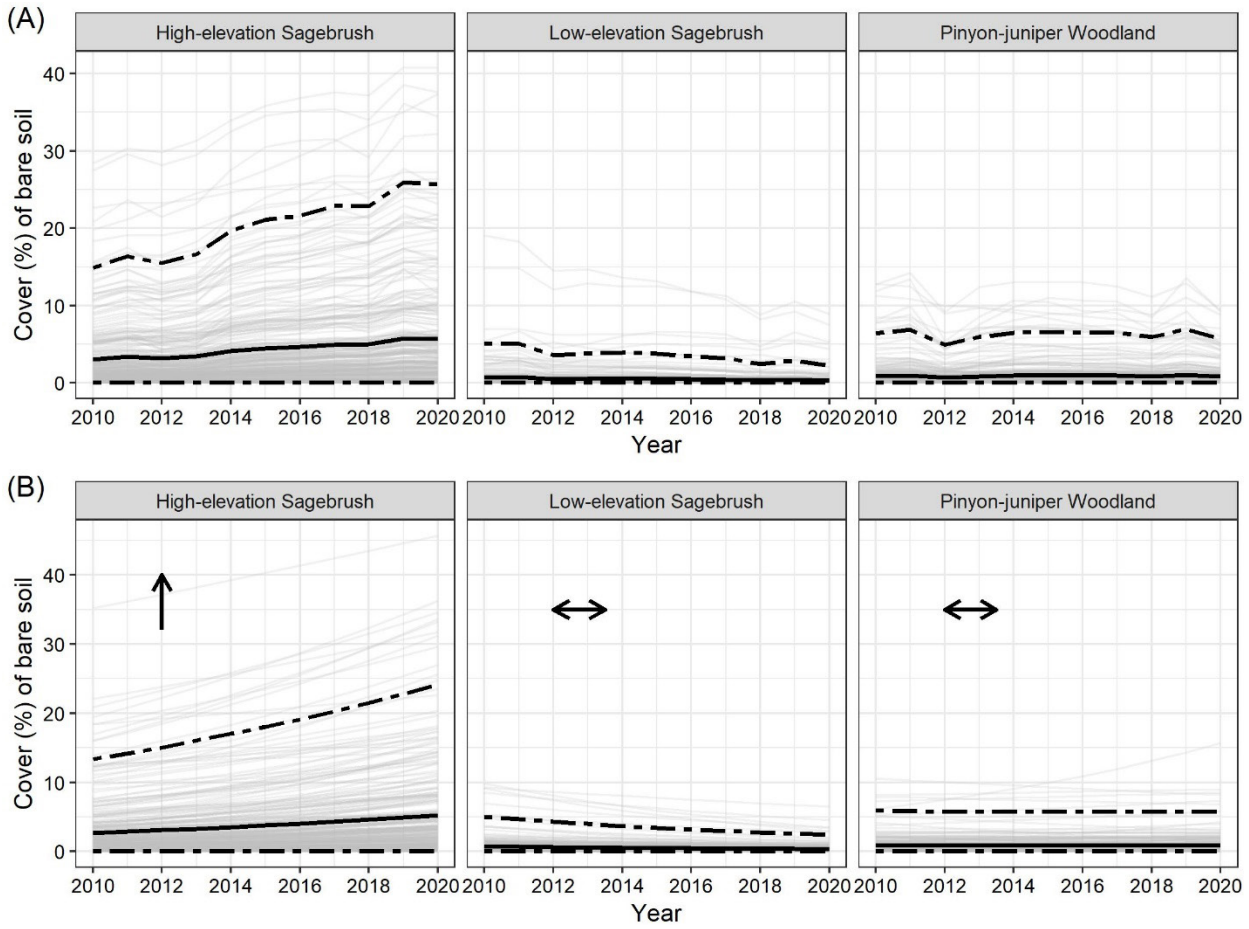


Figure 18. Model-predicted bare soil cover. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in cover predicted by cool-season deficit (D) and grazing. (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

Soil stability ratings (categorical ratings from 1 to 6, where higher ratings indicate greater stability) were highest in 2020 in all strata and varied between years (Figure 19A). There was a significant negative relationship between monsoon- and cool-season D: as D decreased, soil stability increased (see Table 3). When D was high in 2013 following an exceptionally hot, dry year in 2012, soil stability had a noticeable decrease across all strata (Figure 19A). Grazing and burning were also informative in the model but did not have a directional effect on the response. After accounting for interannual variation due to cool- and monsoon-season D, all strata exhibited a significant increasing trend over time (Figure 19B). Soil stability was increasing slowly at a rate of 0.004, 0.007, and 0.008 in high-elevation sagebrush, low-elevation sagebrush, and pinyon-juniper woodlands, respectively (see Table 4).

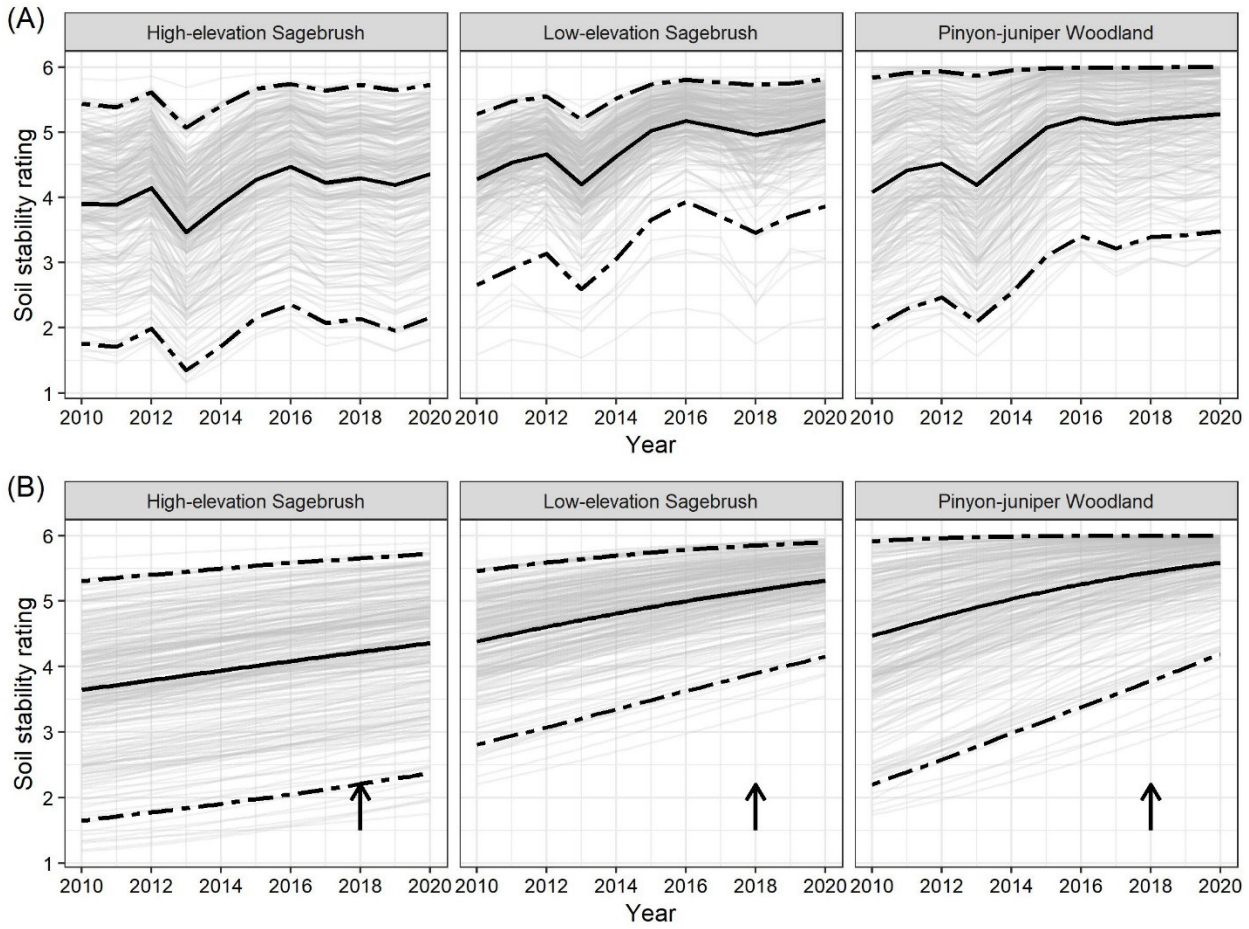


Figure 19. Model-predicted soil stability. Lines describe the posterior predicted mean distribution: light gray lines are draws from the distribution, the solid black line is the median of the distribution, and dashed black lines are the upper and lower boundaries of the 95% credible interval. (A) Interannual variation in stability ratings predicted by cool- and monsoon-season deficit (D), grazing, and burning. (B) Trend over time with climate covariates held at their mean. Black arrows indicate a stable, increasing, or decreasing trend.

3.3.3 Wind and water erosion indicators

Attempts to model status and trend over time of mean canopy and basal gap size were unsuccessful, meaning that the model did not adequately pass posterior predictive checks. Therefore, no inferences could be made on status and trend of canopy and basal gaps. A summary of canopy gap data shows that high-elevation sagebrush had, on average, a greater proportion of canopy gaps in the smallest size class (Figure 20). In contrast, low-elevation sagebrush had the greatest proportion of canopy gaps in the largest size class. However, variation among plots and years is high, particularly in the largest and smallest size classes. A small number of plots with very large gaps (up to 5000 cm, the length of a transect) may have influenced the higher proportion in the largest size class in low-elevation sagebrush.

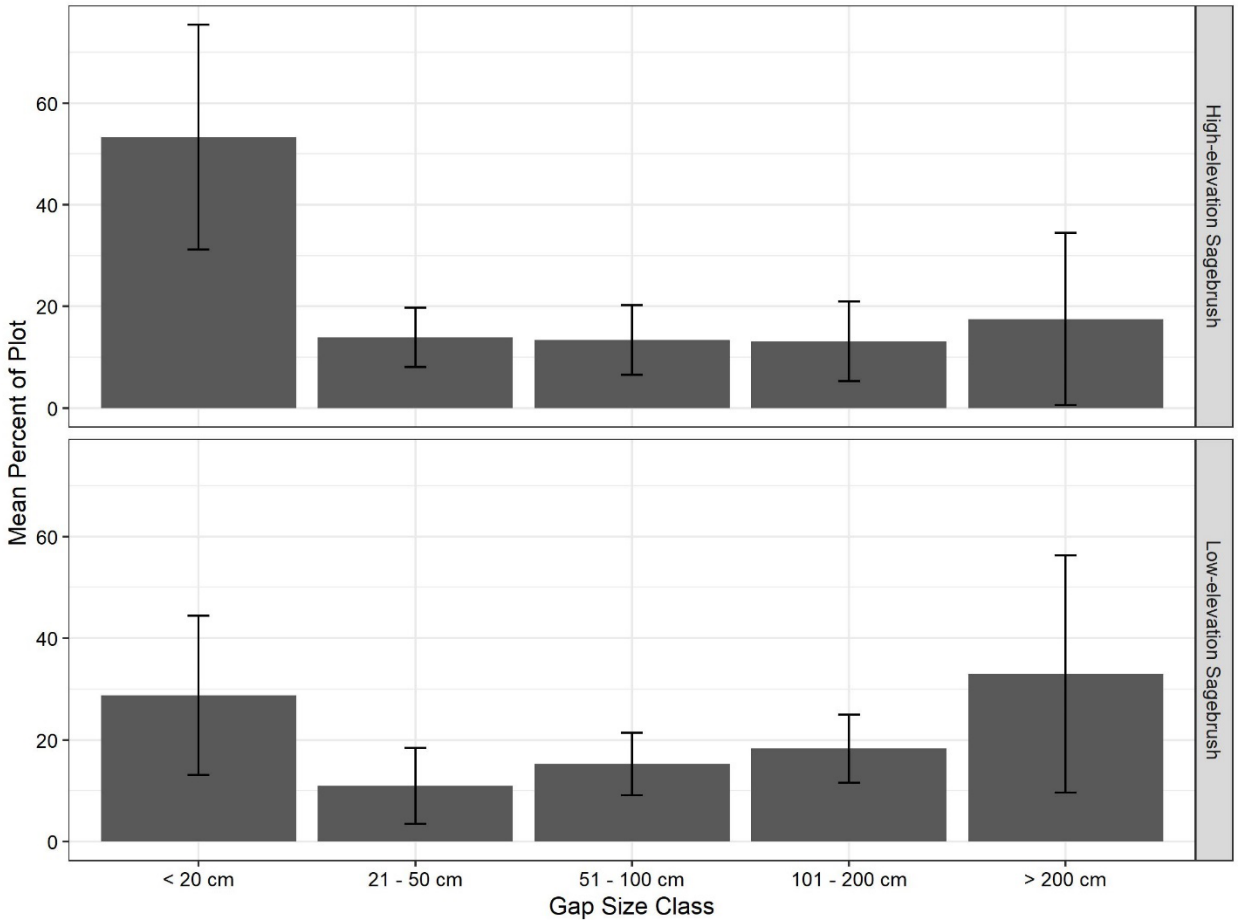


Figure 20. Distribution of canopy gaps. Bars represent the mean percentage of the plot that has gaps in the given size class, averaged over plots and years for each stratum. Error bars are ±1 standard deviation from the mean.

Basal gaps had the greatest proportion of gaps in the smallest gap size class for high-elevation sagebrush and the lowest proportion in the largest size class (Figure 21). In low-elevation sagebrush, the opposite pattern was observed; the greatest proportion of gaps were in the largest size class. Gaps generally decreased from the smallest to largest size class for high-elevation sagebrush and increased from the smallest to largest size class for low-elevation sagebrush. Similar to canopy gaps, there was wide variation among plots and years.

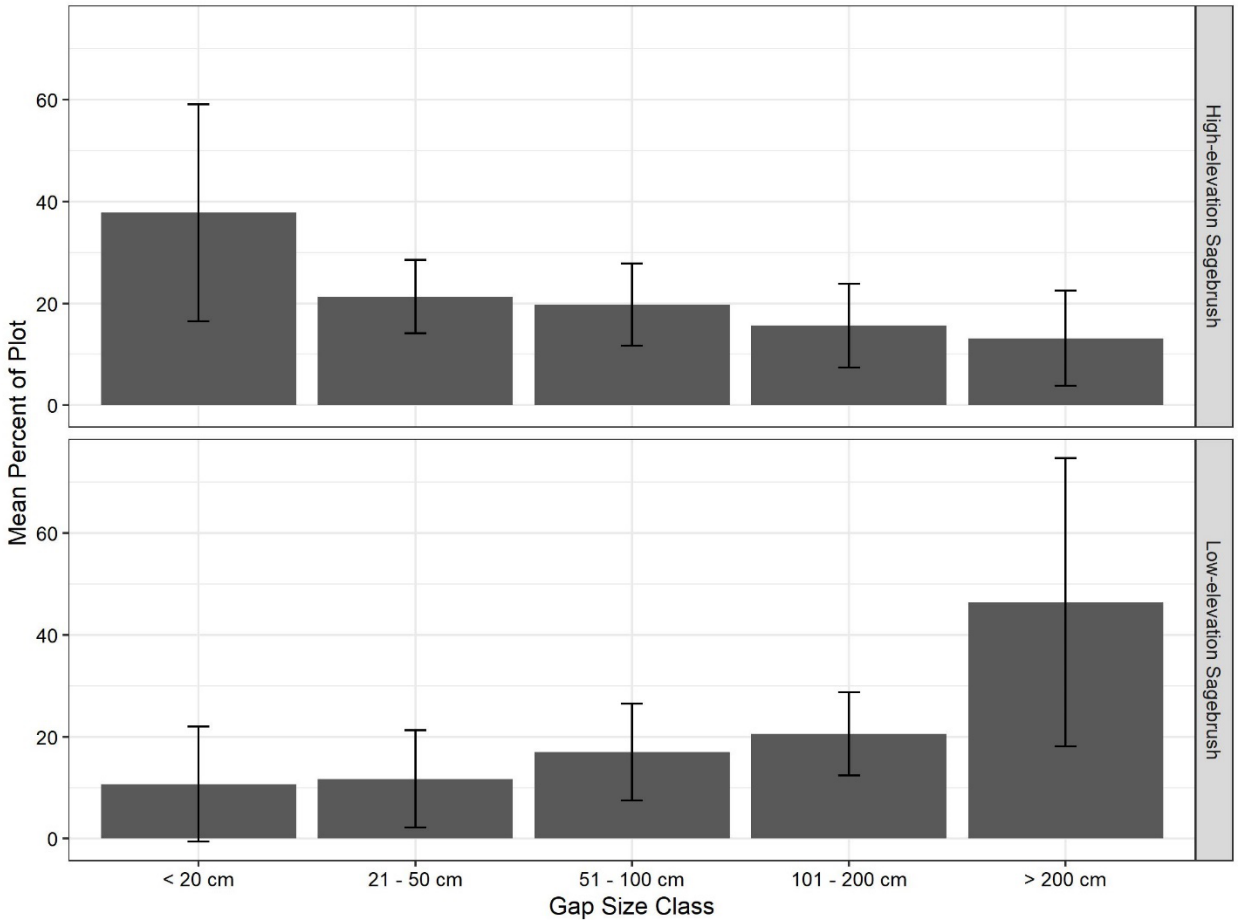


Figure 21. Distribution of basal gaps. Bars represent the mean percentage of the plot that has gaps in the given size class, averaged over plots and years for each stratum. Error bars are ±1 standard deviation from the mean.

4 Discussion

Changes in vegetation functional groups, particularly exotic annuals, perennial grasses, and big sagebrush, suggest that plant community change is taking place in sagebrush ecosystems at Dinosaur NM. There was a rapid, increasing trend in exotic cover and frequency. Indicators of pinyon-juniper woodland condition were generally stable, but a decreasing trend in the density of saplings may be of concern. Soil surface indicators were mostly stable or improving; however, comparisons to reference conditions are limited.

4.1 High- and low-elevation sagebrush vegetation

The rapid increase in cover and frequency of exotic annual plants in both high- and low-elevation sagebrush communities that took place over the monitoring period is of significant concern. This result is consistent with change detected by remote sensing over a 20-year period, where production in areas dominated by annual exotics increased rapidly relative to other vegetation groups (Thoma 2022). The dominant exotic annual species is cheatgrass (*B. tectorum*), which has spread following fires in the past, particularly low-severity fires accompanied by dry conditions afterwards, and now has been detected in over 25% of the park (Sherrill and Romme 2012; Bishop et al. 2019). Cheatgrass is a winter annual that germinates when adequate soil moisture is available in the fall (Munson et al. 2011), and the annual exotic group responded strongly in years with greater monsoon P (see Table 3). Additional species of concern include Russian thistle (*S. tragus*), which has high cover in low-elevation sagebrush, and desert madwort (*A. desertorum*), which has low cover but was detected with high frequency in high-elevation sagebrush. The increasing trend in exotic species cover after accounting for interannual variation due to cumulative growing degree-days and cool- and monsoon-season P suggests that an additional mechanism, such as the strong competitive ability of exotic species, may be driving longer term change.

While exotic plants are increasing in cover and frequency, native perennial grasses are decreasing in high- and low-elevation sagebrush communities. The range in mean cover of these grasses (7 to 17% for high-elevation sagebrush and 7 to 15% for low-elevation sagebrush) is primarily below the known reference conditions of 19 to 55% for high-elevation sagebrush and 14 to 75% for low-elevation sagebrush (see Table 1, Jablonski et al. 2020). The ecological site descriptions for high- and low-elevation sagebrush describe disturbance-mediated alternative states for these communities as having decreased perennial bunchgrasses due to overgrazing (Appendix B). While past grazing has almost certainly contributed to current conditions, grazing was not an influential covariate in the perennial grass model, meaning there is not a clear effect of grazing on perennial grass response at the stratum level. Competitive interactions with invading species such as cheatgrass may be contributing to the poor condition.

Perennial grasses did respond positively to increased cool-season AET, which was expected since the perennial grasses monitored are primarily cool-season species. Climate projections suggest that P will increase at Dinosaur NM, with increases more likely to occur in winter and spring (Monahan and Fisichelli 2014). This may benefit native cool-season grasses and cheatgrass alike. While trends were not evaluated specifically in cool-season versus warm-season (C4) perennial grasses in this analysis,

some studies suggest a shift towards warm-season grass dominance in the region (Witwicki et al. 2016; Palmquist et al. 2021), and it may be useful to evaluate warm-season grasses in low-elevation sagebrush in the future.

Cover of big sagebrush in high- and low-elevation communities is within known reference conditions but on the lower end of potential cover (see Table 1). In high-elevation sagebrush communities, there are wide credible intervals showing that some sites have high mountain big sagebrush cover, up to 40%. While trend analysis did show increasing trends in big sagebrush cover in high- and low-elevation sagebrush, annual rates of increase were low (<0.5%, see Table 4). Big sagebrush cover was higher at the end of the monitoring period, which could be a result of shrubs growing larger or more individuals establishing. Models of the density of big sagebrush showed a small but significant increasing trend in high-elevation sagebrush, indicating the establishment of new individuals in that stratum. Despite warming temperatures, studies suggest that soil moisture will continue to adequately support big sagebrush through increased cool-season P regionally, particularly in higher elevation sites (Bradford et al. 2019; Palmquist et al. 2021). Sagebrush and other woody species may also benefit from CO2 enrichment (Barger et al. 2011), which is a potential driver of change not captured in the models. However, sagebrush communities are still vulnerable to fire, and projected increased fire frequency due to climate warming may negatively impact the stability of sagebrush ecosystems (Davies et al. 2011; Westerling 2016). This is of particular concern at Dinosaur NM, where there is already an increase in the presence and cover of exotic annuals and sagebrush communities may undergo an irrevocable transition into exotic annual grassland communities due to fire. Rodhouse et al. (2021) suggest a framework to protect, restore, or triage sagebrush communities based on inherent resistance, resilience, and current conditions. In this framework, sites with low resilience that are already invaded may be triaged while sites with low resilience but intact plant communities would be prioritized for protection.

4.2 High- and low-elevation sagebrush soil surface

Soil surface indicators in high- and low-elevation sagebrush were mostly stable or improving; however, bare soil had an increasing trend in high-elevation sagebrush, and low-elevation sagebrush had a relatively large proportion of basal gaps in the largest size class. Low cover of bare soil, high soil stability and biological soil crust cover, and smaller canopy and basal gaps generally indicate lower potential for erosion. The results are consistent with recent Rangeland Health Assessments of grazing allotments, which found soil and site stability to be primarily close to reference conditions (Jablonski et al. 2020). However, quantitative measures could not be found for comparison. Bare soil cover was generally low (see Figure 18, see Table 4). However, it is important to exercise caution when comparing this metric to other monitoring protocols, such as the NRCS Ecological Site Descriptions or the BLM's Assessment, Inventory, and Monitoring, which may refer to undifferentiated crust as "bare ground." In high-elevation sagebrush, bare soil cover is increasing and may become a concern in the future.

Canopy and basal gaps in high-elevation sagebrush were, on average, in the smallest size class, which indicates low potential for erosion, but the trend could not be determined. The high proportion of large basal and canopy gaps in low-elevation sagebrush indicates greater potential for erosion.

However, the impact of these large gaps may be mitigated by the relatively high soil stability with an increasing trend and the stable trend of bare soil cover. Although soil stability was in good condition, it did respond significantly to cool- and monsoon-season D and showed a noticeable decrease in 2013 following the hot, dry weather in 2012, indicating the potential for deteriorating conditions with a warming climate.

Biological soil crust cover was lowest in high-elevation sagebrush; however, these sites generally have greater and denser vegetative cover and accumulated litter than low-elevation sagebrush sites, which can limit development of biological soil crust (Belnap et al. 2001). Recent research indicates that biological crust communities may decline under warming temperatures (Ferrenberg et al. 2015; Rodriguez-Caballero et al. 2018), but an increasing trend in biological soil crust cover in high- and low-elevation sagebrush was found. This result was surprising given the generally warm and dry conditions during the study period (see Figure 6). Lack of disturbance may allow biological crust cover to increase over time (Belnap et al. 2001). While a grazing covariate was included in the final model for biological soil crust cover, it did not have a directional effect. Further, the limitations of the grazing covariate mean that it may not adequately represent overall disturbance in high- or low-elevation sagebrush communities. A recent study found differences in moss and lichen responses to warming and plant community changes, with lichen decreasing due to warming temperatures and moss increasing with warming temperatures and after cheatgrass invasion (Finger-Higgens et al. 2022). Further investigations at Dinosaur NM could look at the moss and lichen components of biological soil crust and their individual responses to temperature and plant invasion.

4.3 Pinyon-juniper woodlands

The pinyon-juniper community at Dinosaur NM was not associated with a specific ecological site; rather, it represents multiple soil map units and was limited by geographic region to facilitate sampling. Therefore, reference conditions may be variable within the area of inference, which is limited to the south-central portion of Dinosaur NM. Results do not imply condition, status, or trend throughout the pinyon-juniper woodlands at Dinosaur NM, which are widespread. Model-predicted tree cover in pinyon-juniper woodlands ranged from 22% to 25% over the monitoring period, which is on the upper end of the known reference conditions (see Table 1). Historically, pinyon-juniper woodlands have expanded or contracted in response to multiple hypothesized causes, including fire or fire exclusion, drought, disease, livestock grazing, and periods of wet or dry climate (Romme et al. 2009). Previous studies at Dinosaur NM suggest that pinyon-juniper woodlands have contracted and the fire rotation is shorter than it was historically (Arendt and Baker 2013; Floyd et al. 2017).

The results do not show an increase or decrease in tree cover in either pinyon-juniper woodlands or sagebrush shrublands (see Figure 15), suggesting that tree cover is stable in the area monitored. However, the results did show that sapling density was decreasing in pinyon-juniper woodlands (see Figure 16). Sapling density had a significant relationship with cool- and monsoon-season P (see Table 3) and was at its lowest point in 2018, a year with relatively low P. The majority of saplings monitored were Utah juniper (*J. osteosperma*), which corresponds with observations from field crews of juniper dieback, particularly in smaller trees. Regionally, browning and dieback of juniper

have occurred due to ongoing extreme drought stress (Kannenberg et al. 2021). Future monitoring will include an additional measure of tree crown health with the goal of detecting dieback.

The understory of pinyon-juniper woodlands in the monitored area had sparse perennial grass and big sagebrush cover, both with stable trends (see Figure 13). Big sagebrush density had an increasing trend at a rate of 14 shrubs/ha/year, indicating some successful recruitment (see Table 4). Exotic cover was much lower (<10%) and varied less between years than in high- and low-elevation sagebrush communities (see Figure 11). While there was no trend in exotic annual cover, the frequency of exotic annual detections did have an increasing trend (see Table 4). Frequency measurements are less precise than cover but are more sensitive and may detect change earlier. Pinyon-juniper woodlands are adjacent to and sometimes interspersed with more highly invaded areas, so concern that exotic annuals may increase in these areas in the future is warranted.

Reference conditions for soil surface indicators for the pinyon-juniper soil map units monitored were not available; however, biological soil crust cover, bare soil cover, and soil stability all appear to be in good condition. Biological soil crust cover was higher in pinyon-juniper woodlands than adjacent sagebrush shrublands (see Figure 17), where there is greater open area underneath tree canopies for crusts to develop (Belnap et al. 2001). Soil stability was also relatively high, and both biological soil crust cover and soil stability showed an increasing trend. Bare soil cover was low and stable. The relatively good condition of soils in these pinyon-juniper woodlands may reflect a lack of disturbance in this area.

4.4 Indicator responses to grazing, fire, and climate

Managers were concerned about the impacts of grazing and fire on sagebrush communities. Therefore, grazing and fire covariates were tested in the model sets to evaluate their impact on plant community and soil dynamics. The results showed that time since fire was included in the tree cover model, grazing was included in the exotic frequency and bare soil models, and both time since fire and grazing were included in the soil stability model. While grazing and time since fire did not have significant directional effects, their inclusion suggests that they do play a role in determining the response of those indicators over time and in a variable climate. Grazed and ungrazed allotments in the sagebrush ecosystems were determined to not be comparable due to the confounding effects of trespass cattle, poor understanding of prior use, and the unquantified use by native ungulates. Approximately one-third of plots have experienced fire at some point in the past 40 years. Fire history has certainly impacted plant community composition locally, but there is no evidence that it is influencing short-term vegetation dynamics at the stratum level.

Water balance covariates (AET and D) were generally better predictors of vegetation response than P alone, particularly cool- and/or monsoon-season D, which was a significant covariate for four indicators and contributed to improved models for an additional three (see Table 3). Water balance metrics take into account temperature and soil properties, which mediate plant responses to climate, and have been shown to explain vegetation responses well in the Colorado Plateau (Thoma et al. 2016, 2019; Witwicki et al. 2016). Water deficit has increased in the western US, including the study area, over the past 40 years (Tercek et al. 2021), and that trend is expected to continue as temperatures rise. This will likely have a negative impact on indicators that have a negative

relationship with water deficit, which are live cover and soil stability. Big sagebrush cover was an exception, which did not have a water balance or P covariate included in the final model; however, big sagebrush growth is likely responding to longer-term water dynamics because of its deeper root systems and slower growth (Comstock and Ehleringer 1992). Big sagebrush density did have a cool- and monsoon-season deficit in the final model, but unlike cover, density may fluctuate with recruitment events driven by short-term weather events.

5 Management Recommendations

Based on the results presented in this report, the NCPN recommends the following for management of high- and low-elevation sagebrush and south-central pinyon-juniper woodlands of Dinosaur NM:

- Prioritize control of exotic annuals, particularly cheatgrass, which are rapidly increasing in high- and low-elevation sagebrush. Since control and restoration of cheatgrass-invaded areas at a landscape scale is often not feasible, managers could focus on preventing further spread through limiting disturbance by fire, particularly in areas of low resilience that have not already been invaded (Davies et al. 2011; Rodhouse et al. 2021).
- Concurrent with exotic species control, native cool-season grasses that are declining in cover should be protected. Strategies to accomplish this could include limiting disturbance by fire, reducing the intensity of grazing, or altering the timing of grazing to protect the spring growth period of cool-season species.
- Where high- and low-elevation sagebrush are protected, consider active restoration of cool-season grasses. Restoration efforts could include seeding or planting of native species in combination with surface modifications to stabilize soil and retain soil water.
- Monitor the status of pinyon and juniper saplings, which are decreasing in density and have been identified as vulnerable to drought in the region.

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Appendix A. Additional Information on Bayesian Trend Models

This appendix presents a brief overview of Bayesian data analysis. However, a thorough understanding of Bayesian data analysis is not needed to understand the results presented in this report.

A.1 Overview of Bayesian data analysis

According to Kruschke (2015), Bayesian data analysis involves the following steps:

1) *Define a descriptive model for the relevant data.* In this case, this is understanding the changes in indicators of upland vegetation and soil condition over time, and understanding the relationships between indicators of upland vegetation and soil condition and the climate conditions that are driving changes. Linear models with slopes and intercepts were developed, like those used in frequentist statistics. However, there is often more random variation built into Bayesian models than into their frequentist counterparts. Bayesian models also contain a *likelihood*, which describes the distribution of the response data (e.g., normal, lognormal, binomial, Poisson, negative binomial). The *deterministic model* that describes the data is related to the distribution of the likelihood.
2) *Specify a prior distribution on the parameters. The prior must pass muster with skeptical scientists evaluating the analysis.* In some cases, previous analyses help to inform the values of priors. In the data analyzed in this report, vague priors were used that allow for equal credibility across a wide range of possible values for each parameter, meaning there will be effectively no biasing of the resulting posterior distribution.
3) Use Bayesian inference to re-allocate credibility across parameter values. Assuming that the model is a reasonable description of the data, interpret the posterior distribution with respect to the relevant issue. The posterior distribution describes values of the slope, intercept, and other parameters that are credible, given the data. This range of values can be described through a 95% credible interval, which can be interpreted as a 95% confidence that the true value of the mean of the indicator falls within the lower and upper values for the interval.
4) Check that the posterior predictions mimic the data with reasonable accuracy by conducting a posterior predictive check. If not, consider a different descriptive model. To assess model fit, the field data were compared with data simulated from the Bayesian model, and test statistics were used to determine if they were sufficiently similar.

A.2 Bayesian trend modeling

Bayesian modeling provides an eloquent way to model the status and trends of data with complex sampling design features that are challenging with traditional frequentist statistical approaches (Hobbs and Hooten 2015). For example, hierarchical Bayesian models can easily accept data from the NCPN's five-year repeating sampling schedule (Witwicki et al. 2017, appendices A and B) and non-normally distributed data from categories, counts, proportions, and binomial distributions. In addition, Bayesian models provide a cohesive approach for determining status at a given point in

time and trend over time using the same model, unlike frequentist methods. This approach also allowed us to examine the influence of important covariates at any hierarchical level in the model (plot, strata, park) that influence vegetation and soils, such as climate (Zachmann et al. 2021).

These models are hierarchical because they allow parameters (slopes and intercepts) to vary at multiple levels—in this case, at the plot and stratum levels. Inferences were made at the stratum scale based on the statistical distribution of slopes and intercepts at the plot scale. These models take into account that there can be wide variation in an indicator among plots. These differences were incorporated to inform stratum-level trends, rather than averaging over them.

Because these models are Bayesian, they predict the plot response for all years—even years when data were not collected. This means plots are modeled during years when they are not visited, thereby preventing the possibility that inferences on trends could be influenced by annual variation in the set of plots observed in a particular year. Additionally, plots observed more frequently have greater weight in inference than plots observed infrequently, a straightforward method to account for the difference in visitation frequency among plots due to the repeating five-year sampling schedule.

A.3 Equations

The general formula for Bayesian models is

$$[\text{Model parameters} \mid \text{data}] \propto [\text{data} \mid \text{likelihood model}] [\text{priors}]$$

where " | " is "as conditional on" and α is "is proportional to".

For a simple model where intercepts are allowed to vary among sites within a single stratum and slopes are the same for all sites with no other covariates, the model can be written as

$$[\beta_0, \beta_1, \sigma_{\beta 0}^2, \mu_{\beta 0} \mid y] \propto [y_i \mid \beta_0 + \beta_1 x_i, \sigma^2] [\beta_0 \mid \mu, \sigma_{\beta 0}^2] [\beta_1] [\mu_{\beta 0}] [\sigma_{\beta 0}^2]$$

where β_0 is the intercept, β_1 is the common slope, $\sigma_{\beta 0}^2$ is the variance of the intercept, and $\mu_{\beta 0}$ is the mean of the intercepts. The expression $[y_i \mid \beta_0 + \beta_1 x_i, \sigma^2]$ reads the probability of the observation conditional on the model parameters and error variance. As such, it represents a general expression for the likelihood. Specific distributions (e.g., binomial, beta, Poisson) are chosen depending on the values of the observations (i.e., their "support").

The likelihood model varies based on the distribution of the indicator. In the case of the shrub-density model, the simplified likelihood can be written as:

$$y \sim \text{Negative binomial}(e^{ß\,0 + ß1\,x + ß2\,x2}) \backslash kappa$$

where β2 is the row vector of covariates in a covariance matrix, $x2$ is the corresponding column vector of coefficients, and kappa is a dispersion parameter.

A.4 Variance structures

Hierarchical models were built with different variance structures to determine which best fit these data. The following three types of variance structures were examined:

1) **Hierarchical site:** The variance of observations at a given site is drawn from a hyperdistribution of variances shared by all of the sites in the sample. In other words, there is an underlying distribution of the variances of observations within sites that gives rise to the variance of observations at a specific site.
2) **Fixed site:** As above, variances vary by site, but the variance of observations at each site is estimated completely independently of one another. There's not a shared hyperdistribution of variances.
3) **Fixed stratum:** Variance is forced to be the same for all sites within a stratum.

A.5 Model checking and selection

All models were checked for convergence using the Gelman-Rubin diagnostic (Gelman and Rubin 1992). A properly converged model will not meaningfully change the posterior distribution with more iterations in the Markov Chain Monte Carlo (MCMC) process, which ensures the model output provides reliable inference (Hobbs and Hooten 2015). In many cases, models would only converge under the most appropriate variance structure for the data.

To assess model fit, the field data were compared with data simulated from the Bayesian model to compute the probability that test statistics summarizing the real and simulated data were sufficiently similar. The Bayesian p value (P_B) tests whether the distribution of the data that would arise from the model is more extreme than the distribution of the observed data. Ideal values should be close to 0.5, with extremes of >0.9 or <0.1 indicating a poor fit (Hooten and Hobbs 2015).

No evidence was found of any spatial or temporal autocorrelation in the residuals, perhaps because of the addition of climate covariates in the models.

To choose the "best" model among those with either varying intercepts (β0), or varying slopes and intercepts (β0 β1), and with different combinations of covariates, the posterior predictive loss was minimized. Posterior predictive loss (Gelfand and Ghosh 1998) contains a goodness-of-fit term and a penalty term based on the number of model parameters. Its goal is to minimize loss by comparing replicated data created at each iteration of the MCMC chain with observed data. Generally, the results showed that models with random intercepts only (β0) converged more often than models with random slopes and random intercepts (β0 β1), so these types of models were focused on for most of the trend analyses.

Appendix B. Description of big sagebrush ecological sites selected for upland monitoring in Dinosaur National Monument (Table 5).

Table 5. Descriptions for big sagebrush ecological sites selected for upland monitoring in Dinosaur National Monument.

Ecological Site Name	Ecological Site Number	Soil Properties	Geological Properties	Vegetative Properties	Disturbance-mediated Alternative States
Rolling Loam	R034XY298CO	Deep sandy loams, loams, and gravelly loams	Eolian deposits, alluvium, residuum, and/or colluvium	Dominated by Wyoming big sagebrush, western wheatgrass, and needle-and-thread grass	Decreased bunchgrasses from overgrazing. Pinyon-juniper, cheatgrass, and exotic annuals may invade.
Sandy Foothills	R034XY310CO	Shallow to moderately deep loamy sands	Eolian deposits over residuum weathered from shale or just residuum	Dominated by Wyoming big sagebrush and needle-and-thread grass	Decreased bunchgrass and increased sagebrush from overgrazing. Decreased antelope bitterbrush from excessive browsing. Cheatgrass, Russian thistle, annual mustards, and juniper may invade.
Sandyland	R034XY330CO	Deep loamy sands	Alluvium and/or residuum	Dominated by Wyoming big sagebrush, Indian ricegrass, and needle-and-thread grass	Decreased bunchgrass and increased sagebrush from overgrazing. Cheatgrass, Russian thistle, annual mustards, and juniper may invade.
Semidesert Clay Loam	R034XB328CO	Deep clay loams on footslopes and toeslopes	Alluvium and/ or colluvium over residuum weather from shale	Dominated by Wyoming big sagebrush, western wheatgrass, bluebunch wheatgrass, and Sandberg bluegrass	Decreased palatable grasses and shrubs and increased big sagebrush from overgrazing. Cheatgrass, Russian thistle, annual mustards, and juniper may invade.
Semidesert Gravelly Sandy Loam (Wyoming big sagebrush)	R034XY206UT	Deep gravelly loams	Alluvium, colluvium, and/or reworked eolian from sandstone or limestone	Dominated by Wyoming big sagebrush and galleta grass	Decreased bunchgrasses and increased shrubs from overgrazing. Increased rabbitbrush and decreased sagebrush from fire. Cheatgrass and juniper may invade.

Table 5 (continued). Descriptions for big sagebrush ecological sites selected for upland monitoring in Dinosaur National Monument.

Ecological Site Name	Ecological Site Number	Soil Properties	Geological Properties	Vegetative Properties	Disturbance-mediated Alternative States
Semidesert Loam	R034XY327CO	Deep loams on footslopes and toeslopes	Alluvium derived from sandstone and shale	Dominated by Wyoming big sagebrush and galleta grass	Decreased bunchgrasses and palatable shrubs from overgrazing. Bulbous bluegrass, greasewood, cheatgrass, and Russian thistle may invade.
Semidesert Loam (Wyoming big sagebrush)	R034XY212UT	Deep sandy loams	Alluvium and eolian deposits	Dominated by Wyoming big sagebrush and Indian ricegrass	Decreased bunchgrasses and increased sagebrush from overgrazing. Decreased sagebrush from fire or fall/ winter grazing by sheep. Increased grasses and forbs after fire. Cheatgrass, annual weeds, and juniper may invade.
Semidesert Sand (fourwing saltbush)	R034XY214UT	Deep sand on footslopes and toeslopes	Eolian deposits	Dominated by Indian ricegrass, sand sagebrush, fourwing saltbush, and needle and thread	Decreased Indian ricegrass, needle and thread, and fourwing saltbush and increased snakeweed, rabbitbrush, sandhill muhly, and threeawn from overgrazing. Cheatgrass and annual weeds may invade.
Semidesert Sandy Loam	R034XB326CO	Deep sandy loams	Eolian deposits, alluvium, residuum, and/or colluvium	Dominated by Indian ricegrass, needle-and-thread grass, and Wyoming big sagebrush	Decreased bunchgrasses and winterfat from overgrazing. Cheatgrass, Russian thistle, annual mustards, and juniper may invade.
Semidesert Sandy Loam (fourwing saltbush)	R034XY216UT	Deep sandy loams	Eolian and alluvial deposits	Dominated by Indian ricegrass and needle-and-thread grass	Decreased Indian ricegrass, winterfat, and fourwing saltbush from overgrazing. Decreased sagebrush and fourwing saltbush from fire. Increased snakeweed, rabbitbrush, and galleta grass from overgrazing and fire.

Table 5 (continued). Descriptions for big sagebrush ecological sites selected for upland monitoring in Dinosaur National Monument.

Ecological Site Name	Ecological Site Number	Soil Properties	Geological Properties	Vegetative Properties	Disturbance-mediated Alternative States
Semidesert Shallow Loam (Wyoming big sagebrush)	R034XY225UT	Shallow cobbly sandy loams	Slope alluvium and/ or colluvium over residuum from sandstone and limestone	Dominated by Wyoming big sagebrush and needle-and-thread grass	Decreased bunchgrasses and increased sagebrush, galleta grass, and snakeweed from overgrazing. Cheatgrass may invade.
Mountain Loam (mountain big sagebrush)	R047XC430UT	Deep loams	Alluvium, eolian deposits, and/or colluvium	Dominated by mountain big sagebrush and western wheatgrass	Decreased bunchgrasses and antelope bitterbrush and increased sagebrush, rabbitbrush, and western wheatgrass from overgrazing. Decreased sagebrush and increased rabbitbrush and western wheatgrass from fire.
Mountain Shallow Loam (mountain big sagebrush)	R047XC446UT	Shallow, extremely flaggy, sandy loams on shoulders and summits	Slope alluvium and/ or colluvium derived from sandstone	Dominated by mountain big sagebrush and bluebunch wheatgrass	Decreased bunchgrasses and increased sagebrush and western wheatgrass from overgrazing. Decreased sagebrush and increased rabbitbrush and western wheatgrass from fire.

Appendix C. Rotating Panel Sampling Design

Dinosaur National Monument

Vegetation types:

- High-elevation big sagebrush
- Low-elevation big sagebrush
- Pinyon-juniper

Revisit design (Table 6, Table 7):

- 5 years to rotate through all plots
- 36 plots sampled/year; 135 plots total
- 1/3 of plots are visited 2 years in a row
- 12 plots sampled/year for each vegetation type
- 45 unique plots/vegetation type
- Pilot data collected 2009–2011 (Year 1 = 2012)

Table 6. Dinosaur National Monument revisit design for each vegetation type.

Panel↓/Year→	1	2	3	4	5	6	7	8	9	10
1	3	3	n/a	n/a	n/a	3	3	n/a	n/a	n/a
2	n/a	3	3	n/a	n/a	n/a	3	3	n/a	n/a
3	n/a	n/a	3	3	n/a	n/a	n/a	3	3	n/a
4	n/a	n/a	n/a	3	3	n/a	n/a	n/a	3	3
5	3	n/a	n/a	n/a	3	3	n/a	n/a	n/a	3
6	6	n/a	n/a	n/a	n/a	6	n/a	n/a	n/a	n/a
7	n/a	6	n/a	n/a	n/a	n/a	6	n/a	n/a	n/a
8	n/a	n/a	6	n/a	n/a	n/a	n/a	6	n/a	n/a
9	n/a	n/a	n/a	6	n/a	n/a	n/a	n/a	6	n/a
10	n/a	n/a	n/a	n/a	6	n/a	n/a	n/a	n/a	6

Table 7. Panels sampled each year at Dinosaur National Monument.

Panel year	Panels sampled	Actual year
1	1, 5, 6	2012, 2017, 2022
2	1, 2, 7	2013, 2018, 2023
3	2, 3, 8	2014, 2019, 2024
4	3, 4, 9	2015, 2020, 2025
5	4, 5, 10	2016, 2021, 2026

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