ROOTING FOR THE LITTLE GUYS:

UNDERSTORY VEGETATION RESPONSE TO FUELS REDUCTION TREATMENTS IN THE JEMEZ MOUNTAINS, NEW MEXICO

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ABSTRACT

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MEAGAN K. DREHER

The increasing frequency and intensity of wildfires in Southwestern ponderosa pine and mixed-conifer forests pose significant threats to biodiversity and ecosystem resilience. In response, forest and fire managers have implemented fuels reduction treatments, including thinning, mastication, and prescribed burns. This research aims to understand the long-term effects of these treatments on understory communities, focusing on two sites in the Monument Canyon Research Natural Area in the Jemez Mountains, New Mexico. Mastication and prescribed burning treatments were applied to one site in the Monument Canyon Research Natural Area, while the other site was left untreated. My research compares the treated and untreated sites using datasets from the University of Arizona and the Pueblo/Four Winds Fire Ecology Program. I aimed to answer the following questions: 1) Were the sites similar prior to treatment? 2) Is there a treatment effect in 2022 and 2023? 2.a) What species are driving these differences? 3) What was the recovery trajectory in the treated site from 2009-2023? Results reveal pre-treatment site similarity and significant post-treatment differences among functional groups, particularly in forbs, abiotic, and biotic substrates. Lifeforms such as graminoids and shrubs endured notable shifts over time. Indicator species analyses identify key contributors to community changes in the treated site. These long-term shifts in species and lifeform cover highlight the importance of extended monitoring for understanding post-treatment ecological dynamics. This study contributes valuable insights into the enduring effects of treatments on biodiversity, with implications for ecological management in fire-prone landscapes.

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Chapter 1 Introduction

Managers across the Southwestern United States have been faced with seemingly never-ending issues in conservation and forest management. Due to a combination of climate change and the hundred-year prolonged period of fire suppression in the 1900s, wildfires are increasing in size and severity; fire regimes are changing, and we are encountering novel patterns in landscape and resource response to disturbances (Johnstone et al. 2016). In response, land managers across the southwestern U.S. are implementing fuels reduction treatments in this fire-prone environment and attempting to tackle the issue of increasing fire danger (Sample et al. 2022). There are several ways to introduce fuels treatments into a forest ecosystem depending on management goal, such as pest management or prevention, disturbance preparation, forest health, etc. There are three common ways in which fuels treatments are implemented in Southwestern mixed-conifer forests: thinning, mastication, and prescribed burning.

I am interested in understory vegetation communities and their response to fuels reduction treatments. The understory is a vital component of any forested ecosystem and comprises most of the biodiversity in the forest. The understory also has important roles in nutrient cycling, wildlife habitat, and is an important determinant of site productivity (Jang 2021). There are several papers about understory vegetation response to fuels treatments and to wildfire in the scientific literature, with a notable gap: there are few long-term studies monitoring the changes and responses over time (Abella & Springer 2015). Much of the literature suggests that the understory nativeness and species richness remain high after a disturbance such as thinning, mastication or burning, when compared to controls that have not been treated (Rossman et al. 2018, Kane et al. 2010, Wolk & Rocca 2009).

To assess long term trends in understory vegetation response to fire and fuel-related disturbance, we compare datasets from two studies. The study site is in the Jemez Mountains of New Mexico. Anthropogenic and natural fire events have always played a role in the landscape ecology of the Jemez Mountains, from the long history of indigenous occupancy through the fire-suppression era in the 1900s (Swetnam et al. 2016). In recent history in the Jemez Mountains, fires have become more severe and larger, significantly altering landscape and species compositions (Dewar et al. 2021). Recent notable fire history consists of several large and high-severity fires, such as the South Fork fire in 2010, the Las Conchas fire in 2011, the Thompson Ridge fire in 2013 (Bandelier Story Map), and most recently the Cerro Pelado fire in 2022. Our study site, Monument Canyon Research Natural Area (RNA) is split into two sites: San Juan Mesa and Cat Mesa. San Juan Mesa was treated by thinning, mastication, and prescribed burning in 2006 and 2012 respectively, and Cat Mesa remained untreated. We aim to answer questions about community differences between the two sites through time to evaluate long-term treatment effects.

The second chapter of this thesis is a comprehensive systematic review paper aimed to update Abella & Springer (2015) and their review on understory vegetation responses to fuels reduction treatments in the Western U.S. We ask the same questions of Abella & Springer (2015), plus one about climate. The review questions include: 1) How does understory vegetation respond to fuels treatments, including thinning and prescribed burning, in the western forests of the United States? 1.a) Do different treatment methods or disturbances (thinning, prescribed burning, combination, or wildfire) drive the responses among understory vegetation? 1.b) Does the temporal scale of studies present different results in long-term response of understory vegetation to treatments? 1.c) How do plant functional groups and traits alter the responses to treatments and disturbances? 1.d) Does treatment severity and forest type (dry or wet) affect understory responses to treatments? 1.e) Does climate variability, including drought or precipitation trends, and topographic variability, such as slope and aspect, impact how the understory responds to disturbance?

The third chapter evaluates the effects of fuels reduction treatments on understory vegetation communities in the Monument Canyon RNA. We used two datasets to answer these questions, one establishing a baseline for comparison before treatment to assess post-treatment effects (UA dataset) and the other evaluating temporal shifts in the treated site and comparing the two sites in 2022 and 2023 (P4WFEP dataset). Our research questions are: 1. Were the sites similar prior to treatment? 2. Are there treatment effects in 2022 and 2023? 3. What was the treated site's recovery trajectory from 2009-2023?

The fourth and final chapter in this thesis is a brief fact sheet where we explain the management implications of our findings. The management implications will be distributed to the Southwest Fire Consortium to provide information to stakeholders, fire managers and fire ecologists in the Southwest.

Literature Cited

Abella, S. R., & Springer, J. D. (2015). Effects of tree cutting and fire on understory vegetation in mixed conifer forests. *Forest Ecology and Management*, *335*, 281–299. https://doi.org/10.1016/j.foreco.2014.09.009

Dewar, J. J., et al. "Valleys of Fire: Historical Fire Regimes of Forest-Grassland Ecotones across the Montane Landscape of the Valles Caldera National Preserve, New Mexico, USA." *Landscape Ecology*, no. 2, Springer Science and Business Media LLC, Jan. 2021, pp. 331–52. *Crossref*, doi:10.1007/s10980-020-01101-w.

"Fire in the Jemez Mountains." *ArcGIS StoryMaps*, Esri, 14 June 2021, https://storymaps.arcgis.com/stories/e8dcc55831b348c6841ec7bd592559e8.

Fornwalt, P. J., Rocca, M. E., Battaglia, M. A., Rhoades, C. C., & Ryan, M. G. (2017). Mulching fuels treatments promote understory plant communities in three Colorado, USA, coniferous forest types. *Forest Ecology and Management*, *385*, 214-224.

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M. C., Meentemeyer, R. K., Metz, M. R., Perry, G. L. W., Schoennagel, T., & Turner, M. G. (2016). Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment*, *14*(7), 369-378. https://doi.org/10.1002/fee.1311

Jang, W., Crotteau, J. S., Ortega, Y. K., Hood, S. M., Keyes, C. R., Pearson, D. E., ... & Sala, A. (2021). Native and non-native understory vegetation responses to restoration treatments in a dry conifer forest over 23 years. *Forest Ecology and Management*, *481*, 118684.

Kane, J. M., Varner, J. M., Knapp, E. E., & Powers, R. F. (2010). Understory vegetation response to mechanical mastication and other fuels treatments in a ponderosa pine forest. *Applied Vegetation Science*, *13*(2), 207-220.

Rossman, A. K., Halpern, C. B., Harrod, R. J., Urgenson, L. S., Peterson, D. W., & Bakker, J. D. (2018). Benefits of thinning and burning for understory diversity vary with spatial scale and time since treatment. *Forest Ecology and Management*, *419*, 58-78.

Sample, M., Thode, A. E., Peterson, C., Gallagher, M. R., Flatley, W., Friggens, M., Evans, A., Loehman, R., Hedwall, S., Brandt, L., Janowiak, M., & Swanston, C. (2022). Adaptation Strategies and Approaches for Managing Fire in a Changing Climate. *Climate*, *10*(4), Article 4. https://doi.org/10.3390/cli10040058

Swetnam, T. W., Farella, J., Roos, C. I., Liebmann, M. J., Falk, D. A., & Allen, C. D. (2016). Multiscale perspectives of fire, climate and humans in western North America and the Jemez Mountains, USA. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *371*(1696), 20150168.

Westover, Robert Hudson (2021). Thinning the Forest for the Trees. *US Forest Service*. https://www.fs.usda.gov/features/thinning-fores*t*-trees.

Wolk, B., & Rocca, M. E. (2009). Thinning and chipping small-diameter ponderosa pine changes understory plant communities on the Colorado Front Range. *Forest Ecology and Management*, *257*(1), 85-95.

Chapter 2 Literature Review

Abstract

Understory vegetation in Western U.S. forests is affected by fuel reduction treatments in many ways. The two most common fuel reduction treatments, thinning and prescribed fire, impact the understory as either independent treatments, or they interact with each other, depending on desired management objectives. Abella & Springer (2015) provide a review of 41 studies looking at fuel reduction treatment impacts to understory vegetation.

Through this update and expansion of Abella & Springer (2015), I expected to answer the following question: How does understory vegetation respond to fuels treatments, including thinning and prescribed burning, in western forests of the United States? As I explored this question, I determined which treatment methods or disturbances (thinning, prescribed burning, thinning + burning) drive the responses of understory vegetation, compared the influence of temporal scale on observed outcomes, and examined the relationship between drought trends and understory vegetation in treated areas.

From the reviewed papers, I determined that the herbaceous understory species richness peaks in short intervals and decreases or stabilizes to a lower level in the long-term. Thinning + prescribed fire has the most prominent results compared to thin-only, burn-only, or control treatments. There are more papers that study shor*t*-term (3-5 years) impacts of fuels treatments than long-term (more than 10 years). Climatic variability and its relationship to understory changes over time, and its relationship to the response of the understory to fuels treatments is still relatively unknown. Future research needs to be conducted to add to the body of literature on long-term impacts of fuels treatments on understory vegetation so that researchers and managers

alike can understand the effects on functional plant groups, invasive species, and manage the implications that may come when treatments are paired with a changing climate.

Background

What are Fuel Treatments?

Pos*t*-Euro-American settlement fire exclusion and suppression has resulted in overstocked forests with altered disturbance regimes (Covington & Moore 2004, Moore et al. 1999). Increased forest density, combined with a warmer and dryer climate, are leading to increased fire severity and frequency (Williams et al. 2012, Sample et al. 2022).

In response to these trends, managers have adopted several practices to reduce fuels and increase fire resiliency of western forests (Agee & Skinner 2005, Hessburg et al. 2015, Stephens et al. 2018). These practices include overstory thinning and prescribed burning. There are several methods of thinning, including low, crown , or selection thinning, as well as mastication. Low thinning removes trees from smaller diameter classes, crown thinning focuses on mid-canopy trees, and selection thinning focuses on the largest trees in the stand (Agee & Skinner 2005). Mastication treatments use machines to chip or mulch prescribed trees and broadcast the mulched material on the ground (Fornwalt et al. 2017). Thinning reduces overstory density, which reduces both the horizontal and vertical continuity of fuel and the risk of widespread and highly severe fires (Reynolds et al. 2013, Drury 2020). It also opens the canopy and allows more light and water to pass through to the understory, increasing both the biodiversity and functional diversity of the forest (Jang et al. 2021). Prescribed burning is often conducted by either broadcast burning or burning organized piles of downed woody debris (branches, pole-sized and smaller trees, and brush). Burning can reduce fuel in both the live overstory and midstory trees, but can also aid in removing downed logs, pine needles, and duff, and reduce the continuity of burnable fuels when a hot fire sweeps through (Collins, Moghaddas & Stephens 2006). Often, thinning and prescribed burning are both implemented in stands or forests to maximize fuel reduction and ecological resilience (Collins, Moghaddas & Stephens 2006). Overall, the main objective of fuel treatments is to modify potential fire behavior and resulting effects.

What do we know about fuel treatments and understory vegetation responses? Updating a review

This review asks related questions to the Abella & Springer (2015) paper: 1) How does understory vegetation respond to fuels treatments, including thinning and prescribed burning, in the western forests of the United States? 1.a) Do different treatment methods or disturbances (thinning, prescribed burning, combination, or wildfire) drive the responses among understory vegetation? 1.b) Does the temporal scale of studies present different results in long-term response of understory vegetation to treatments? 1.c) How do plant functional groups and traits alter the responses to treatments and disturbances? 1.d) Does treatment severity and forest type (dry or wet) affect understory responses to treatments? 1.e) Does climate variability, including drought or precipitation trends, and topographic variability, such as slope and aspect, impact how the understory responds to disturbance?

The body of knowledge surrounding the effects on the herbaceous understory is growing, but there are still several needs for future research. The number of studies on how the understory is impacted by thinning and prescribed fire, both as independent and interactive drivers of change, has increased in the last 10-20 years. The understory is a critical component of any forested ecosystem and most of the biodiversity within a forest is found in these communities. Biological diversity is integral to the function of many wildlife, plant and mycorrhizal species and their relationships to the ecosystem around them (Peterson, Allen & Holling 1998). The understory provides nutrient cycling, wildlife habitat, increases site productivity, (Jang 2021) and is also a soil stabilizer in a post-fire regenerating ecosystem.

The Abella & Springer (2015) review paper, titled “Effects of tree cutting and fire on understory vegetation in mixed conifer forests,” is a comprehensive systematic review that concluded research efforts in April 2014. Abella & Springer reviewed 41 papers that covered research conducted in mixed conifer forests across the Western U.S., including the Pacific coast, the Intermountain region, and the Rocky Mountains. They posed five research questions (Table 2.1): 1) Do tree cutting, prescribed fire, a combination of the two, and wildfire have different effects on total understory plant abundance and species richness? 2) Are understory metrics such as plant cover and species richness related to time since tree cutting or time since fire? 3) Do responses to these disturbances vary among plant groups? 4) Do understory responses to tree cutting and fire differ between moist and dry mixed conifer forests? 5) How do understory responses vary with intensity of tree cutting or severity of fire?

Table 2.1 Summarized findings from each of Abella & Springer (2015)'s five research questions.

Question 1: Relative Influences of Treatments on Total Understory Plant Abundance and Species Richness	Cutting and prescribed fire applied individually similarly increased understory plant abundance or species richness in about half of the studies. However, applying cutting and prescribed fire together typically resulted in decreased plant abundance but increased species richness. Decreased plant abundance was noted to have been recorded in short-term (< 10 years) studies.
Question 2: Influence of Time Since Treatment	The longest-term studies (> 10 years) usually found increases in total understory plant measures after cutting or prescribed fire. Longer-term studies reported greater increases in total plant abundance, and the effect was more pronounced in studies of cutting. For prescribed fire, the two longest-term studies reported the greatest increase in total plant abundance. Species richness was also influenced by time since treatment. Longer-term studies reported the greatest relative increase in richness, and similar trends were observed for prescribed fire. After cutting + prescribed fire, shorter-term studies reported declines in richness, while longer-term studies reported increases.
Question 3: Responses Among Plant Groups	Forbs and graminoids increased in abundance more frequently than shrubs across various treatments. Shrub abundance usually decreased

	after treatments, particularly after combined cutting + fire. Non-native plant abundance increased the most after cutting + prescribed fire, however native species constituted the total plant abundance and richness measures and corresponding responses to treatments.
Question 4: Treatment Effects in Moist Versus Dry Mixed Conifer	No studies directly compared treatment responses between moist and dry mixed conifer forests. There appeared to be little relationship between treatment response and average long-term precipitation in study areas.
Question 5: Influence of Treatment Intensity or Fire Severity	Results were mixed for studies comparing cutting intensity. For prescribed fire and wildfire, high-severity burning displayed greater increases in total plant abundance and richness than low-severity burning.

Several variables influence how the plant communities respond over time to disturbance and fuels reduction efforts. For instance, climatic variability, including drought and precipitation trends, as well as topographic variability that influences local moisture conditions in microsites, may alter understory responses. In the western United States, uncertainty about ecosystem and climatic responses to fire and fuels management is still high, especially as climate change becomes a more prominent issue (Hurteau et al. 2014). Sample et al. (2022) suggest that fuels reduction treatments can be used to reduce climate change vulnerability. However, many management strategies focus on overstory vegetation: comparatively less is known about the understory community effects. Abella & Springer (2015) provide the most comprehensive review of the topic and cover understory responses to thinning and burning in mixed-conifer Western forests. Still, important topics, including climate variability and longer-term responses to fuels treatments remain unknown. Furthermore, functional plant groups and their traits may respond differently across the community after disturbance. There is a multitude of new information to apply to management of fire ecology and climate change since this 2014 review including several newly published papers about the response of understory communities to fuels treatments. New work covers mixed conifer and ponderosa pine forests in the western U.S.

Abella and Springer (2015) highlight pivotal factors shaping understory dynamics in mixed conifer forests after treatments: time since treatment, whether thinning and fire were applied together or separately, the extent of overstory removal, the intensity of the treatment, the arrangement of residual trees, methods of treating slash, and management practices (Table 2.1). Abella & Springer identified future research needs, including more research on moist vs. dry forests, long-term (more than 10 years) effects, and species response. They recommended that monitoring early and for at least four years post-treatment is vital. Native understory species can persist and thrive after thinning and prescribed fire treatments, if overstories remain open.

Table 2.2 Common diversity metrics in understory vegetation analyses and their definitions.

Diversity metric	**Definition**
Abundance, relative abundance	Describes the number of individuals found in each species, the percent cover per species, sometimes biomass per species (Pyron 2010).
Species richness	The number of species in each community (Pyron 2010).
Evenness	Describes how abundance is distributed across species in a community, i.e., evenness is highest when all species in each sample have equal abundance (Pyron 2010).
Beta diversity	The variability in species composition among sampling units for a given area (Anderson et al. 2006).

Methods

I conducted two searches, on October 22, 2022 (Google Scholar) and November 2, 2022 (Web of Science), which were refined to 2014-2022. In these searches, I included terms to find papers discussing understory response to fuels treatments in ponderosa and mixed conifer forests (see Appendix A). Google Scholar resulted in 17,700 hits (see Appendix A), which were also the most varied and least related to the search terms.

To determine which papers I would include in this review, I established the following criteria for inclusion: papers that described studies in the Western United States, including the

Pacific Northwest, Rocky Mountains, and the Southwest. I included studies that looked at only thinning, only burning, or a combination of the two. To effectively update Abella & Springer (2015), I refined my search terms for the years 2014-2022 so that I would obtain papers that were published after their review had been concluded. Lastly, the inclusion criteria included mixed conifer, conifer, or ponderosa pine forests, to get a wide breadth of results from western forests. I excluded papers that focused on pinyon-juniper forests or oak woodlands because plant communities differ between mixed conifer and ponderosa forests and lower-elevation scrub or pinyon-juniper-dominated ecosystems. For this review, I defined the understory as any plant, graminoid or shrub found on the forest floor. Lastly, I excluded gray literature.

I used CAB Abstracts, Web of Science, and Google Scholar to obtain my initial results. From these searches, I found 14 papers that met my criteria. Appendix A and B show several results.

Results

Metadata from collected papers and diversity metrics

Six of 14 papers sampled vegetation five years pos*t*--fuel treatment. Zald et al. (2020), currently classified as 'Other' in figure 2, had several treatment application intervals *and* several sampling intervals and could not be easily merged with other studies.

When I compared the number of studies that utilized specific treatment, including "thin only," "burn only," "thinning + burning," or "wildfire + thin." (Figure 2.3), I found fewer papers focused on thin-only treatments, and even less that looked at burn-only. Seven of the 14 papers used a combination of thinning and burning in their research. Thus, less information was available from 2014-2022 on the individual impacts of prescribed burning, thinning, or to wildfire followed by thinning.

Table 2.3 Summary of impacts of fuel treatments on understory vegetation communities reported from 14 studies conducted since 2014 . Positive impacts indicated with a "+", negative with "-". More + signs indicate vigorous increases in the given metric. General patterns in diversity metrics can be seen depending on treatment type.

Treatment	**Abundance**	**Cover**	**Richness**	**Non-native cover**
Thin only (3)	++	++	++,-	+,-
Burn only (2)	n/a	++	+	n/a
Thin + burn (7)	+++	+++	+++	+
Wildfire + thin (2)	++	++	++	Dependent on severity

1. Do different treatment methods or disturbances drive understory vegetation response?

Abella & Springer's paper found that understory plant diversity indices such as richness and relative cover increased after treatments, except for abundance, which was found to decrease (Table 2.1), though notably those results were from short-term studies (Abella & Springer 2015). I found similar results; treatment methods and disturbances do drive responses among understory vegetation (Table 2.3). Most of the papers I evaluated had a combined treatment of thinning followed by prescribed fire (Table 2.3; Fig 2.3). This combination elicited the most prominent results and increased the understory's diversity metrics, including abundance, cover, and species richness (Table 2.3). The combination treatments have the most significant effect on understory vegetation, though burn-only and thin-only still positively influenced observed diversity metrics.

1.a) Combination of thin and prescribed burn treatments

Abella & Springer (2015) concluded that thinning and prescribed fire treatments result in the greatest increase in richness, while abundance decreased. However, it is important to note that these trends in abundance were primarily reported in short-term studies, and specific to mixed-conifer forests with naturally more closed canopies. I found similar trends for species richness in the new reviewed literature, but I found that abundance generally increased after the combination treatment. Overall, the combination of thinning and prescribed fire results in the greatest response rate compared to the other treatments, usually measured in abundance, cover, or species richness of the understory (Table 2.3). Several papers found that thinning followed by prescribed fire will result in the highest diversity and abundance of understory vegetation in most functional plant groups, but especially graminoid and forbs (Table 2.4).

Jang et al. (2021) found that thinning followed by prescribed fire in the cu*t*-and-dry-burn, the most intense treatment, amplifies understory responses over time compared to thin-only. Korb et al. (2020) similarly found that graminoid abundance more than doubled in all treatments (thin and burn combination, burn-only, or no treatment), and shrub density increased by 250% after treatment. Korb et al. 2020 also found that plant diversity increased overall by 14% in the thin and burn combination plots compared to burn-only and continued to increase 10 years after the treatments. Strahan et al. (2015) found that understory species richness and cover increased in thin and burn treatments, compared to control and burn-only plots. According to Strahan et al. (2015), restoration treatments using the combination of thin and prescribed fire will result in the highest amount of understory abundance and diversity. Goodwin et al. (2018) also found that plant cover increased in all thin and burn treatments (between thin only, burn only, combination), but the largest increases of diversity metrics were found in the combination treatments. I found that each of these approaches had slightly different ways of expressing diversity metrics, but resulted in similar findings- that understory richness, cover and abundance tend to increase post-combination treatment.

Thomas and Waring (2015) had a different approach to evaluating the effects of thinning and burning on the understory. They studied plots in ponderosa pine forests in New Mexico that had been high-graded in the late 1800s, and then many plots underwent a second-entry harvest about 70-90 years later. The stands were then thinned again, and only a subsample of stands were burned with prescribed fire two years after the third thinning treatment. They found that the treatments are effective in increasing understory cover, mostly driven by graminoid regeneration. The variation in cover was driven by forbs in their study. Their third thin treatment was applied to small trees, and they found that the understory responds better with the removal of small trees compared to large trees, or trees from all diameter classes.

Using metrics other studies did not, Odland et al. (2021) found that understory richness increased immediately following treatment, but it did not generate understory beta diversity (Table 2.2) and shrub cover similar to historic reference conditions. They found that multiple burns without thinning created the most heterogeneity across the landscape, which was most like historical reference conditions containing low shrub levels, high local diversity, species richness, evenness, and beta diversity at least two years po*st*-second burn treatment. Odland et al. (2021) suggest that thinning and burning may promote dominance by shrubs, and even type conversion in some forests.

Zald et al. (2020) evaluated seasonal and interval burns after all sites were thinned. Their results suggested that all understory vegetation is sensitive to the seasonality of burning, but the response is subtle. Season and burn interval do not result in substantial changes in the understory composition. They found that fall burns result in different understory compositions than spring prescribed burns, but also noted that native seral species may not be able to maintain regeneration with repeated disturbance, even at low prescribed fire severities. These results are like Kerns & Day (2018).

1.b) Thin-only treatments

Studies of thin-only fuel reduction treatments found overall cover increased after treatments compared to control sites, and non-native species tended to increase for a few years and then decrease again. Native species remain dominant in all forest types, across all studies. There was no evidence of complete non-native invasion in any treatment type in any study. Faist et al. (2015) looked at thinning's effect on understory and seed banks, and found that in ponderosa forests, forbs and graminoid communities had a higher cover in treated sites than they did in control. They suggest that stand-scale responses can be attributed to the increased availability of light and nutrients. Springer et al. (2018) found that understory cover pos*t*-fire was higher in native species in the treated plots compared to no treatment. Their results suggest that hazardous fuel reduction treatments are useful and not negatively impactful on the understory vegetation populations. Fornwalt et al. (2017) evaluated understory richness and cover after a mastication treatment and found that both metrics increase after treatment. There was an increase in non-native plants in the treated areas, but still a dominance in native species, comparatively. They concluded that mastication increases diversity and density (cover) in native plant species.

1.c) Burn-only treatments

Strahan et al. (2015) found that burn-only units, compared to the combination thin and burn units, showed no significant difference in native species richness from untreated units. Goodwin et al. (2018) found that cover reduced in burn-only and untreated plots, compared to thin and prescribed fire treatments where cover increased overall. Rhoades & Fornwalt (2015) were the only paper that looked specifically at pile burning effects on understory vegetation. They found that pile openings after the burn had higher graminoid and forb cover, compared to the surrounding forest that was not burned (Table 2.4). They also did not find a significant temporal trend over the 50 years that they monitored the burn piles.

Kerns & Day (2018) conducted a study based in Oregon that not only evaluated understory response to fuels treatments, but also included seasonality and burning intervals as predictor variables. They had five fire regimes spread out between two seasons that are conducive to “good” prescribed fire, and they had five-year burning intervals in some stands, where some were just single burns. They evaluated plant cover, richness, and diversity among the understory plants. The first fall burns increased plant richness by about 14%. Reburning maintained higher exotic plant richness in two five-year intervals (after 10 years). Spring burning resulted in rhizomatous grass cover increasing by three times, but there was no difference in pattern after two burn intervals. In the spring, richness and diversity did not show significant differences compared to the control plots. In sum, their results suggest that most plant functional group responses do not change significantly among regimes or in comparison to no treatment at all, especially for perennial native herbs. Kerns & Day (2018) found that burn season impacts the plant community more than the frequency in fall burning but burn frequency in the spring season is important. This paper, as well as Zald et al. (2020), were the only two studies that evaluated seasonal differences in understory response, and it may lead to interesting management implications if the studies can be replicated or updated in other regions of western North America.

2) Does the temporal scale of studies affect results in long-term treatment response of understory vegetation?

Abella & Springer found that there was a difference between long-term studies and shor*t*-term (Table 2.1). Long-term studies showed more significant increases among plant abundance and species richness. Regardless of treatment type, there were more increased metrics when looking at long-term results compared to shor*t*-term. Notably, in the combined treatments of cutting and prescribed fire, shor*t*-term papers resulted in a trend of decreased species richness, while long-term reported that it increased.

Many papers that I reviewed were written around three to five years post-treatment, and fewer than half of the papers were long-term studies (Figure 2.2).

Understory richness, following thinning and burning treatments, significantly peaks within three to five years after treatment and then steadily declines to levels like pre-treatment (Odland et al. 2021, Jang et al. 2021, Strahan et al. 2015, Goodwin et al. 2018, Dodge et al. 2019). This is contrary to results from shor*t*-term studies that took place from two to six years after treatment, where the general conclusion is that the combination of thinning and burning (not considering interval burns) increases abundance and richness of native plants. The studies that evaluated long-term results found that the species richness decreases in the long-term. However, a 10-year study conducted by Korb et al. (2020) found no significant effects on herbaceous plant and shrub richness by the end of their study, which suggests that there is not an effect of treatment over time. Plant cover and richness reverted to the pre-treatment conditions in Jang et al. (2021), which they presume is because regenerating trees reoccupied the space that was left empty after the burn. They also found that there was a strong non-native plant response 15-years post-treatment. Odland et al. (2021) suggests that understory plant diversity responses or trends need a long period after fire, or perhaps several fire events, to become fully transparent, and Jang et al. (2021) suggests that, as predicted by successional theory, the cover of most understory plant groups should peak about 3-5 years after treatments, but then in the long-term, plant richness declines to pre-treatment level as regenerating trees take up space in the forest again, with the result that key limiting resources (light, water, nutrients) may become less available in the forest understory. Species richness and cover both follow this pattern (Jang et al. 2021). In forests where prescribed burning is cautiously applied, the understory restoration may require more time and repeated burning (Odland et al. 2021).

Non-native species responses in the understory tend to follow similar trends across papers, usually resulting in a peak 3-5 years after treatment, and then steadily decreasing (Jang et al. 2021, Strahan et al. 2015, Springer et al. 2018). Goodwin et al. (2018) found that there was no long-term invasive species establishment.

There were also differences in long-term study results indicating that invasive plants do not persist eventually (Korb et al. 2020), compared to Jang et al. 2021 who noted that the non-native and invasive response 15 years post-treatment remained strong. It is possible that, had the shorter-term papers conducted research beyond their temporal constraints, they would have entirely different results. Faist et al. (2015) also suggests the need for longer-term studies to assess the long-term impacts of treatment on understory vegetation communities. These differences highlight the importance of implementing more long-term monitoring efforts in management and scientific research.

3) Do results differ among functional plant groups (shrubs, forbs, graminoids, grass-like plants, trees, invasive plant species)?

Abella & Springer (2015) indicate that forbs and graminoids increased in abundance more frequently than shrubs across various treatments (Table 2.1). Shrub abundance usually decreased after treatments, particularly after combined cutting + fire. Non-native plant abundance increased the most after cutting + prescribed fire, while native species constituted the total plant abundance and richness measures and corresponding responses to treatments. I found that overall, treatment usually elicits a positive response from all plant functional groups, including shrubs, graminoids, forbs, and invasive species (Table 2.4). There was not enough information in the sources I gathered about how pollination traits within functional groups are affected by fuels treatments, though Neil & Puettman (2013) indicate that reducing overstory density from thinning increases insec*t*-pollinated forbs. Most cover and richness in all functional groups decreases several years after treatment, often to pre-treatment conditions (Table 2.4). Severity also plays a role in the functional group responses.

Table 2.4 Plant functional groups and how they respond to treatments. Separated between short-term studies (1-5 years post-treatment) and long-term studies (10+ years).

Shrub	Mixed results. SHORT TERM STUDY: - Higher cover in high-severity vs. low severity (Dodge et al. 2019) - No difference in seed bank density across treatments (Faist et al. 2015) - Higher cover in treated site vs. control (Faist et al. 2015) LONG TERM STUDY: - Thinning only understory or overstory, or especially in combination with burning, stimulates cover after 15 years, but burn-only decreases cover (Goodwin et al. 2018) - Immediate peak in cover 3-5 years post-treatment, then decrease to pre-treatment levels over time (Jang et al. 2021)
Graminoid	Mixed results. SHORT TERM STUDY: - Higher cover in low-severity than high severity (Dodge et al. 2019) - Higher graminoid seed bank density in high-severity (Faist et al. 2015) - Higher cover in treated site vs. control (Faist et al. 2015, Kerns & Day 2018, Strahan et al. 2015) - Greater cover in treated (masticated) site than control (Fornwalt et al. 2017) - Thinning had a positive effect on the percent of aerial cover (Thomas & Waring 2015). LONG TERM STUDY: - Immediate peak in cover 3-5 years post-treatment, then decrease to pre-treatment levels over time (Jang et al. 2021) - Cover doubled across treatment and controls over 10 years, overall increasing herbaceous cover (Korb et al. 2020). - After pile burning, openings exhibited higher cover compared to the surrounding forest (Rhoades & Fornwalt, 2015)
Forb	Mixed results. SHOR*T*-TERM STUDY: - No difference in seed bank density across treatments (Faist et al. 2015) - Higher cover in treated site vs. control, and cover differs among forest types of post-treatment (Faist et al. 2015)

	- Greater cover in treated site than control (Fornwalt et al. 2017, Kerns & Day 2018, Strahan et al. 2015) - Thin & Burn combination treatments elicited the most significant species richness changes in forbs (Strahan et al. 2015) - Annual forb cover was notably higher in untreated units than treated units (Springer et al. 2018) - No significant differences in forb cover observed among treated stands, but still higher cover in treated (Thomas & Waring 2015) LONG-TERM STUDY: - High-severity burned area had higher cover than low-severity (Dodge et al. 2019) - Immediate increase in cover pos*t*-thin and burn treatment, but over time decreased to pre-treatment levels. Continuous increase after 10 years in thin-only treatment sites. After 15 years no difference between treatments, cover decreased to pre-treatment levels (Goodwin et al. 2018) - Immediate peak in cover 3-5 years post-treatment, then decrease to pre-treatment levels over time. However, showed greatest response to restoration treatments (Jang et al. 2021) - After pile burning, openings exhibited higher cover compared to the surrounding forest (Rhoades & Fornwalt, 2015)
Substrate	- Moss/lichen/fungi cover decreased with higher severity, unaffected by thinning (Dodge et al. 2019 - Litter cover reestablished to pre-treatment conditions 10 years post-treatment (Goodwin et al. 2018) - Much higher cover of bare mineral soil in pile burn openings, while litter and duff were lower in the scars. Water infiltration was much higher in the pile burn openings (Rhoades & Fornwalt, 2015)
Invasive	SHORT TERM STUDY: - Increased (particularly non-native graminoids) in treated site (Faist et al. 2015) - Cover stimulated by mastication, but levels remained low, always less than 10% of total species richness in a site (Fornwalt et al. 2017) - Initial strong increases, but with subsequent reburning effects decreased (Kerns & Day 2018) - Cover differences not significant between untreated and treated units (Springer et al. 2018) - Non-native species richness highest in thin & burn treatments compared to burn-only and untreated (Strahan et al. 2015) LONG TERM STUDY: - Higher cover in high-severity areas than low-severity (Dodge et al. 2019, Jang et al. 2021) - No non-native plant establishment 15 years post-treatment (Goodwin et al. 2018), or similarly, lack of treatment effect in invasive plant cover 10 years post-treatment (Korb et al. 2020) - Immediate strong increase post-treatment, level out in the long-term (Jang et al. 2021)

	- More exotic graminoids across treatment over time. Suggests that species respond positively to treatments (Zald et al. 2020)

4) Does vegetation or forest type (wet/dry forest) affect understory responses to fuels reduction treatments?

Abella & Springer (2015) identified a gap in the literature, as no studies directly compared treatment responses between moist and dry mixed conifer forests. I encountered the same gap in knowledge, other than one paper by Jang et al. 2021 that simulates the comparison between wet and dry forest types by conducting burning in two different intervals, a wet burn (WB) and a dry burn (DB). The wet burn was implemented where the duff and large wood moisture content was 50% and 100%, respectively, whereas the dry burn was implemented when the duff moisture content was 16% and large wood fuel moisture was 30%.

Jang et al. 2021 shows that dry burn treatments (DB) exhibited distinct impacts on the understory vegetation. Native and non-native forb cover displayed significant variation among years, with peak values observed 3–5 years after restoration treatments and subsequent declines. The treatment effects became apparent pos*t*-cutting and burning, with noticeable differences during peak cover years. Native forb cover was notably higher in treated units relative to control units, and this difference persisted even in the final sampling years. In contrast, non-native forb cover exhibited stronger treatment effects, with significantly elevated cover in burned units, particularly during post-treatment years 2 to 5. Both native and non-native forb species richness followed a similar temporal pattern, peaking during years 3–5 after treatment, and their richness tended to be higher in the burned (DB) treatments compared to unburned (NB) and untreated

(CO) units. Non-native graminoids showed heightened responses to dry burns (DB), especially non-native annual graminoids, indicating a pronounced effect of fire severity on non-native species.

Jang et al. 2021 also suggests that wet burn treatments (WB) displayed distinctive responses in the understory vegetation as well. Like dry burns, native and non-native forb cover exhibited significant year-to-year variation, peaking during the 3–5 years after restoration treatments before declining. Treatment effects became evident after cutting and burning, with differences in peak cover years. Native forb cover remained significantly higher in treated units compared to control units, even in the final years of sampling. Non-native forb cover showed stronger treatment effects, particularly during post-treatment years 2 to 5, with significantly elevated cover in burned units. Both native and non-native forb species richness displayed the same temporal pattern as their cover, peaking during years 3–5 after treatment. The richness of both forb groups tended to be higher in burned (WB) treatments when compared to unburned (NB) and untreated (CO) units. While native and non-native graminoid cover and richness also followed similar temporal patterns as observed in dry burns, the treatment effects on graminoids were more evident in non-native species, particularly non-native annual graminoids.

Overall, the responses of the understory vegetation to wet burns (WB) demonstrated parallel patterns to those of dry burns (DB), with differences primarily stemming from the timing and intensity of treatment effects. Fire severity, coupled with treatment type, emerged as a key driver in shaping pos*t*-fire vegetation dynamics (Jang et al. 2021).

Kerns and Day (2018) conducted a study on burn intervals as well, though it was not a different forest type, simply different seasons that simulated dry vs. more moist conditions.

5) Does treatment severity or intensity affect understory responses to fuels reduction treatments?

Abella and Springer (2015) deduced that understory responses to fuels treatments are affected by treatment severity.

Evidence in more recent studies supports the assertion of Abella and Springer (2015) that treatment intensity directly affects understory responses to fuels treatments. Intensity, or treatment severity, influences the effectiveness of treatments in altering the understory vegetation composition and cover. Thinning and burning treatments, for instance, were found to be effective at increasing total understory cover when compared to untreated stands (Dodge et al. 2019, Jang et al. 2021). Several papers that included severity in their analyses noted that the treatment intensity directly affects understory species composition. In high-severity sites, there are shrub or subshrub indicator species that establish post-treatment and indicate the severity of the fire or intensity of treatment (Korb et al. 2019). Furthermore, when high severity prescribed burns cause higher tree mortality, increases in perennial graminoid cover are seen (Springer et al. 2018). The severity of treatments can influence the distribution and prevalence of native and non-native species. While native species richness and cover might respond positively to treatments (Faist et al. 2015, Springer et al. 2018), non-native species responses can vary (Kerns & Day 2018, Jang et al. 2021). The effectiveness of using fuels treatments to control non-native species might depend on the combination of treatment severity and post-treatment conditions. Diversity metrics and indicator species are commonly affected by burn severity. Specifically, beta diversity and evenness decrease as severity increases (Table 2.2; Korb et al. 2019).

Severity effects also interact with treatment intervals and seasonality, such that under low severity, repeated prescribed burns, outcomes may not elicit enough change in the understory composition or initiate seed germination (Kerns & Day 2018). The success of these treatments in promoting understory vegetation growth could be attributed to their intensity and the resulting reduction in overstory canopy cover. The impact of treatment severity on understory vegetation responses can vary over time. Some species show consistent fidelity to certain treatments, while others display changing trends in response (Jang et al. 2021). Temporal variability is particularly evident in species with interannual variability in their importance values, which may indicate pulses of response following burning events.

Overall, the relationship between treatment intensity and understory responses is complex and can be influenced by multiple factors, including burn frequency, stand structure, and interannual variability (Zald et al. 2020). While some trends are consistent, the dynamics of species' responses can be influenced by a combination of treatment factors and site-specific conditions. In summary, the severity of fuel treatments plays a crucial role in shaping the understory vegetation responses. Different species exhibit varying degrees of sensitivity to treatment severity, and these responses can change over time.

6) Does climate variability, including drought or precipitation trends, impact the understory vegetation in treated areas?

There was not enough information to characterize a clear relationship between climate variability and understory changes over time, or understory responses to fuels treatments. Of the three papers that evaluated climatic variability, only two considered precipitation and temperature (Korb et al. 2020, Strahan et al. 2015), and one used climate to communicate natural and environmental site factors, such as elevation, slope, and aspect (Springer et al. 2018).

The results in Korb et al. 2020 reveal a significant interaction between treatment and time for diversity, as well as significant time effects for species richness, diversity, and plant cover. Although there were no treatment effects on understory cover, both treatments and controls showed a doubling of graminoid cover over the 10-year study period, leading to increased herbaceous cover. This increase in graminoids was attributed to their ability to rapidly colonize disturbed areas and tolerate dry conditions. The complexity of understory dynamics in these forests was highlighted, suggesting that factors like climate and competition might be as influential as thinning and prescribed fire in determining understory responses. The study also noted that the understory cover increases might be attributed to below-average precipitation and above-average maximum annual temperature during the sampling years. Korb et al. (2020) concluded that further research is recommended to better understand the interactions between understory vegetation responses, climate effects, and restoration treatments in the context of a warmer and drier climate.

Strahan et al. (2015) indicates that the response of understory vegetation five years after restoration treatments is specific to the site and is linked positively to the amount of annual precipitation. Consequently, the range of variation in how understory responds to restoration should be expected to increase based on the location of treatment implementation along this precipitation gradient. These results indicate that in situations where water availability becomes more limited, the responses to restoration can become more mixed in terms of their effects on different aspects of vegetation. This mixed response might stem from a temporary loss of species due to the disturbances caused by the restoration treatments themselves, potentially leading to unintended outcomes. The study raises the possibility that restoration-related disturbances could have intensified effects, especially in more arid ponderosa pine forests. The concept of a "pivot point" in terms of the relationship between precipitation and species richness is considered but not fully explored within the scope of this study (Munson et al. 2013). This "pivot point" might represent a critical threshold in terms of water availability, beyond which the response of understory vegetation to restoration shifts from negative to positive richness outcomes (Munson et al. 2013). However, this specific concept is not thoroughly investigated in the current study. Strahan et al. 2015 states that the net change in native cover tended to be greater at sites with higher precipitation levels. Overall, the study's findings show that the complexity of environmental gradients and site-specific factors should be considered when planning restoration efforts, suggesting that climate plays a role in understory vegetation changes over time.

Discussion

Future studies are needed on the climate effects on understory vegetation communities. I recommend that more long-term (more than 10 years) studies are conducted to look at the effects of climate on understory vegetation and the effects of fuels reduction treatments. There are not enough papers on both climate and long-term fuels reduction efforts to draw general conclusions. There also remains to be a gap in the literature about cool/wet forest understory composition compared to warm/dry forest understory compositions, and how they respond to fuels treatments. A challenge for such studies is that they would require control sites without treatments to isolate the treatment effect and differentiate it from the effects of climate variability. A before-after-control-impact (BACI) research design is needed in such cases, since climate variation would be common to all study plots (Osenberg et al. 2006, Wienk et al. 2004, Rockweit et al. 2017).

Managers will need to continue to incorporate land management practices that increase resiliency as ecosystems and communities are drying, receiving less annual precipitation, and burning more frequently and severely (Guiterman et al. 2022, Falk et al. 2022). Understanding these dynamics is essential for continued effective land management and conservation strategies, as they provide insights into how different treatments can influence the composition, cover, and diversity of understory plant communities.

There are several takeaways for managers from the findings in these reviewed papers. Long-term restoration management need to strike a balance between fuel mitigation goals and understory vegetation biodiversity and all related metrics, because there is still some uncertainty about the effects of fuel treatments on plant communities overall. More studies are needed that focus on measuring long-term response, to ameliorate these uncertainties. Managers need information on non-native and invasive plant responses that can be used when developing forest restoration treatments (Jang et al. 2021). Almost all the evaluated papers found that there was an increase in non-native plants after any treatment type, at least within 3-5 years post-treatment.

As fires are growing in severity and frequency, type conversions are another high priority of concern for managers (Korb et al. 2020). Forests may not have understory vegetation that can be restored to historical reference conditions, because many places are converting to shrubland. As results from Zald et al. (2020) suggested, some of the native seral species that make up significant proportions of the forests in the western U.S. may not be able to continue to respond to the higher frequency of wildfires, which would aid in the type conversion. The potential for ecotype conversion may severely impact plants that are shade-tolerant and rely on the tall ponderosa or mixed conifer trees to thrive as forests convert to shrubland.

Conclusion

Understory responses to fuels treatments in the western United States vary by treatment type, severity, vegetation type, and sampling interval. I updated and expanded the Abella and Springer (2015) review to gather newly available information about these relationships. They posed five research questions: 1) Do tree cutting, prescribed fire, a combination of the two, and wildfire have different effects on total understory plant abundance and species richness? 2) Are understory metrics such as plant cover and species richness related to time since tree cutting or time since fire? 3) Do responses to these disturbances vary among plant groups? 4) Do understory responses to tree cutting and fire differ between moist and dry mixed conifer forests? 5) How do understory responses vary with intensity of tree cutting or severity of fire? My findings in this update are like Abella & Springer (2015) (Table 2.1).

1) There are differences among treatments, most generally that the combination of thinning and burning elicits the strongest responses over time in terms of richness and abundance. Burning and thinning still affect the understory in terms of changing the composition over time and increasing diversity metrics, but thinning + burning had the strongest effects. 2) Temporal differences affect the results in the literature. There were several studies that found results from 3-5 years post-treatment, and compared to studies that were 10-20 years post-treatment, the effects are different. Long-term research shows that though abundance of both native species and non-native species increases immediately post-treatment, it levels out in the long-term to pre-treatment conditions. 3) Functional plant groups, including graminoids, forbs, shrubs, and non-native species respond differently. Graminoids and forbs are easily affected by treatment, and shrub abundance changes in the long-term. 4) There is not enough information from the studies I reviewed to draw general conclusions about the difference between cool/wet and warm/dry forest effects on the understory vegetation communities. Almost all the studies reviewed here were conducted in warm/dry vegetation types. 5) The influence of treatment severity on understory responses to fuels treatments is evident, shaping vegetation composition and cover dynamics. Thinning and burning treatments have demonstrated effectiveness in enhancing total understory cover, while the interplay between severity and post-treatment conditions influences native and non-native species distribution. Severity also affects diversity metrics and indicator species, with

implications for beta diversity and evenness (Table 2.2). Overall, treatment severity emerges as a critical factor, with its complex interactions guiding the trajectory of understory vegetation responses over time.

I had a sixth question: does climate variability, including drought or precipitation trends, and topographic variability, such as slope and aspect, impact how the understory responds to disturbance? The influence of climate variability on understory changes and treatment responses is not well-established due to limited data. Among the studies, Korb et al. 2020 found complex interactions between treatment, time, and climate for diversity and cover, highlighting the role of climate in influencing understory dynamics beyond treatments. Strahan et al. 2015 suggested that understory responses are site-specific and linked to annual precipitation, implying varying outcomes along a precipitation gradient. The studies emphasize the need to consider both climate and site-specific factors in restoration planning, indicating potential climate involvement in understory changes.

This review illuminates the intricate relationship between treatment severity and understory responses to fuels treatments in the western United States. Thinning and burning treatments, especially in combination, exhibit pronounced effects on richness and abundance, while temporal variation influences outcomes over the long term. Sensitivity to treatment varies among functional plant groups, emphasizing complexity. Yet, the influence of cool/wet versus warm/dry forests remains relatively unexplored due to limited research in cool/wet vegetation types.

Treatment severity emerges as a pivotal factor, shaping understory composition, cover dynamics, and diversity metrics. Notably, thinning and burning treatments show promise in enhancing total understory cover, with severity's interplay guiding native and non-native species distribution. Although climate variability's impact on understory responses requires further study, findings from Korb et al. 2020 and Strahan et al. 2015 underscore the importance of considering climate and site-specific factors in restoration planning. In summary, the intricate interplay of treatment severity, climate variability, and site characteristics collectively defines the trajectory of understory vegetation responses, highlighting the complexities of ecosystem management and restoration.

Literature Cited

Abella, S. R., & Springer, J. D. (2015). Effects of tree cutting and fire on understory vegetation in mixed conifer forests. *Forest Ecology and Management*, *335*, 281–299. https://doi.org/10.1016/j.foreco.2014.09.009

Agee, J. K., & Skinner, C. N. (2005). Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*, *211*(1–2), 83–96. https://doi.org/10.1016/j.foreco.2005.01.034

Collins, B. M., Moghaddas, J. J., & Stephens, S. L. (2007). Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, *239*(1), 102–111. https://doi.org/10.1016/j.foreco.2006.11.013

Dodge, J. M., Strand, E. K., Hudak, A. T., Bright, B. C., Hammond, D. H., & Newingham, B. A. (2019). Shor*t*- and long-term effects of ponderosa pine fuel treatments intersected by the Egley Fire Complex, Oregon, USA. *Fire Ecology*, *15*(1), 40. https://doi.org/10.1186/s42408-019-0055-7

Drury, S. (2020). Fuel Continuity. In S. L. Manzello (Ed.), *Encyclopedia of Wildfires and Wildland-Urban Interface (WUI) Fires* (pp. 1–3). Springer International Publishing. https://doi.org/10.1007/978-3-319-51727-8_239-1

Faist, A. M., Stone, H., & Tripp, E. A. (2015). Impacts of Mastication: Soil Seed Bank Responses to a Forest Thinning Treatment in Three Colorado (USA) Conifer Forest Types. *Forests*, *6*(9), 3060–3074. https://doi.org/10.3390/f6093060

Fornwalt, P. J., Rocca, M. E., Battaglia, M. A., Rhoades, C. C., & Ryan, M. G. (2017). Mulching fuels treatments promote understory plant communities in three Colorado, USA, coniferous forest types. *Forest Ecology and Management*, *385*, 214–224. https://doi.org/10.1016/j.foreco.2016.11.047

Goodwin, M. J., North, M. P., Zald, H. S. J., & Hurteau, M. D. (2018). The 15-year post-treatment response of a mixed-conifer understory plant community to thinning and burning treatments. *Forest Ecology and Management*, *429*, 617–624. https://doi.org/10.1016/j.foreco.2018.07.058

Guiterman, C. H., Gregg, R. M., Marshall, L. A. E., Beckmann, J. J., van Mantgem, P. J., Falk, D. A., Keeley, J. E., Caprio, A. C., Coop, J. D., Fornwalt, P. J., Haffey, C., Hagmann, R. K., Jackson, S. T., Lynch, A. M., Margolis, E. Q., Marks, C., Meyer, M. D., Safford, H., Syphard, A. D., … Stevens, J. T. (2022). Vegetation type conversion in the US Southwest: Frontline observations and management responses. *Fire Ecology*, *18*(1), 6. https://doi.org/10.1186/s42408-022-00131-w

Hessburg, P. F., Churchill, D. J., Larson, A. J., Haugo, R. D., Miller, C., Spies, T. A., North, M. P., Povak, N. A., Belote, R. T., Singleton, P. H., Gaines, W. L., Keane, R. E., Aplet, G. H., Stephens, S. L., Morgan, P., Bisson, P. A., Rieman, B. E., Salter, R. B., & Reeves, G.

H. (2015). Restoring fire-prone Inland Pacific landscapes: Seven core principles. *Landscape Ecology*, *30*(10), 1805–1835. https://doi.org/10.1007/s10980-015-0218-0

Hurteau, M. D., Bradford, J. B., Fulé, P. Z., Taylor, A. H., & Martin, K. L. (2014). Climate change, fire management, and ecological services in the southwestern US. *Forest Ecology and Management*, *327*, 280–289. https://doi.org/10.1016/j.foreco.2013.08.007

Jang, W., Crotteau, J. S., Ortega, Y. K., Hood, S. M., Keyes, C. R., Pearson, D. E., Lutes, D. C., & Sala, A. (2021). Native and non-native understory vegetation responses to restoration treatments in a dry conifer forest over 23 years. *Forest Ecology and Management*, *481*, 118684. https://doi.org/10.1016/j.foreco.2020.118684

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M. C., Meentemeyer, R. K., Metz, M. R., Perry, G. L., Schoennagel, T., & Turner, M. G. (2016). Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment*, *14*(7), 369–378. https://doi.org/10.1002/fee.1311

Kerns, B., & Day, M. (2018). Prescribed fire regimes subtly alter ponderosa pine forest plant community structure. *ECOSPHERE*, *9*(12). https://doi.org/10.1002/ecs2.2529

Korb, J. E., Stoddard, M. T., & Huffman, D. W. (2020). Effectiveness of Restoration Treatments for Reducing Fuels and Increasing Understory Diversity in Shrubby Mixed-Conifer Forests of the Southern Rocky Mountains, USA. *Forests*, *11*(5), 508. https://doi.org/10.3390/f11050508

Moore, M. M., Casey, C. A., Bakker, J. D., Springer, J. D., Fulé, P. Z., Covington, W. W., & Laughlin, D. C. (2006). Herbaceous Vegetation Responses (1992–2004) to Restoration Treatments in a Ponderosa Pine Forest. *Rangeland Ecology & Management*, *59*(2), 135–144. https://doi.org/10.2111/05-051R2.1

Moore, M. M., Wallace Covington, W., & Fulé, P. Z. (1999). Reference Conditions and Ecological Restoration: A Southwestern Ponderosa Pine Perspective. *Ecological Applications*, *9*(4), 1266–1277. https://doi.org/10.1890/1051-0761(1999)009[1266:RCAERA]2.0.CO;2

Munson, S. M., Muldavin, E. H., Belnap, J., Peters, D. P. C., Anderson, J. P., Reiser, M. H., Gallo, K., Melgoza-Castillo, A., Herrick, J. E., & Christiansen, T. A. (2013). Regional signatures of plant response to drought and elevated temperature across a desert ecosystem. *Ecology*, *94*(9), 2030–2041. https://doi.org/10.1890/12-1586.1

Odland, M. C., Goodwin, M. J., Smithers, B., Hurteau, M. D., & North, M. P. (2021). Plant community response to thinning and repeated fire in Sierra Nevada mixed-conifer forest understories. *Forest Ecology and Management*, *495*, 119361. https://doi.org/10.1016/j.foreco.2021.119361

Peterson, G., Allen, C. R., & Holling, C. S. (n.d.). *Ecological Resilience, Biodiversity, and Scale*. 13.

Pyron, M. (2010). *Characterizing Communities | Learn Science at Scitable*. Nature Education Knowledge. https://www.nature.com/scitable/knowledge/library/characterizing-communities-13241173/

Relative species abundance. (2022). In *Wikipedia*. https://en.wikipedia.org/w/index.php?title=Relative_species_abundance&oldid=1092830985#cite_note-Hubbell01-1

Reynolds, R. T., Sanchez Meador, A. J., Youtz, J. A., Nicolet, T., Matonis, M. S., Jackson, P. L., DeLorenzo, D. G., & Graves, A. D. (2013). *Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency* (RMRS-GTR-310; p. RMRS-GTR-310). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-310

Rhoades, C. C., & Fornwalt, P. J. (2015). Pile burning creates a fifty-year legacy of openings in regenerating lodgepole pine forests in Colorado. *Forest Ecology and Management*, *336*, 203–209. https://doi.org/10.1016/j.foreco.2014.10.011

Rockweit, J. T., Franklin, A. B., & Carlson, P. C. (2017). Differential impacts of wildfire on the population dynamics of an old-forest species. *Ecology*, *98*(6), 1574–1582. https://doi.org/10.1002/ecy.1805

Sample, M., Thode, A. E., Peterson, C., Gallagher, M. R., Flatley, W., Friggens, M., Evans, A., Loehman, R., Hedwall, S., Brandt, L., Janowiak, M., & Swanston, C. (2022). Adaptation Strategies and Approaches for Managing Fire in a Changing Climate. *Climate*, *10*(4), Article 4. https://doi.org/10.3390/cli10040058

Springer, J. D., Huffman, D. W., Stoddard, M. T., Sánchez Meador, A. J., & Waltz, A. E. M. (2018). Plant community dynamics following hazardous fuel treatments and mega-wildfire in a warm-dry mixed-conifer forest of the USA. *Forest Ecology and Management*, *429*, 278–286. https://doi.org/10.1016/j.foreco.2018.06.022

Stephens, S. L., Collins, B. M., Fettig, C. J., Finney, M. A., Hoffman, C. M., Knapp, E. E., North, M. P., Safford, H., & Wayman, R. B. (2018). Drought, Tree Mortality, and Wildfire in Forests Adapted to Frequent Fire. *BioScience*, *68*(2), 77–88. https://doi.org/10.1093/biosci/bix146

Strahan, R. T., Stoddard, M. T., Springer, J. D., & Huffman, D. W. (2015). Increasing weight of evidence that thinning and burning treatments help restore understory plant communities in ponderosa pine forests. *Forest Ecology and Management*, *353*, 208–220. https://doi.org/10.1016/j.foreco.2015.05.040

Thomas, Z., & Waring, K. M. (2015). Enhancing resiliency and restoring ecological attributes in second-growth ponderosa pine stands in northern New Mexico, USA. *Forest Science*, *61*(1), 93–104. CAB Abstracts with Full Text.

Walker, R. F., Fecko, R. M., Frederick, W. B., Miller, W. W., & Johnson, D. W. (2012). Influences of Thinning, Chipping, and Fire on Understory Vegetation in a Sierran Mixed Conifer Stand. *Journal of Sustainable Forestry*, *31*(6), 493–517. https://doi.org/10.1080/10549811.2011.622225

Wienk, C. L., Sieg, C. H., & McPherson, G. R. (2004). Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. *Forest Ecology and Management*, *192*(2–3), 375–393. https://doi.org/10.1016/j.foreco.2004.02.004

Williams, A. P., Allen, C. D., Macalady, A. K., Griffin, D., Woodhouse, C., Meko, D. M., Swetnam, T. W., Rauscher, S. A., Seager, R., Grissino-Mayer, H. D., Dean, J. S., Cook, E. R., Gangodagamage, C., Cai, M., & McDowell, N. G. (2012). *Temperature as a potent driver of regional forest drought stress and tree mortality*. *3*, 292–297. https://doi.org/10.7916/d8-b9ec-8z87

Wolk, B., & Rocca, M. E. (2009). Thinning and chipping small-diameter ponderosa pine changes understory plant communities on the Colorado Front Range. *Forest Ecology and Management*, *257*(1), 85–95. https://doi.org/10.1016/j.foreco.2008.08.014

Zald, H. S. J., Kerns, B. K., & Day, M. A. (2020). Limited Effects of Long-Term Repeated Season and Interval of Prescribed Burning on Understory Vegetation Compositional Trajectories and Indicator Species in Ponderosa Pine Forests of Northeastern Oregon, USA. *Forests*, *11*(8), Article 8. https://doi.org/10.3390/f11080834

Figures & Appendices

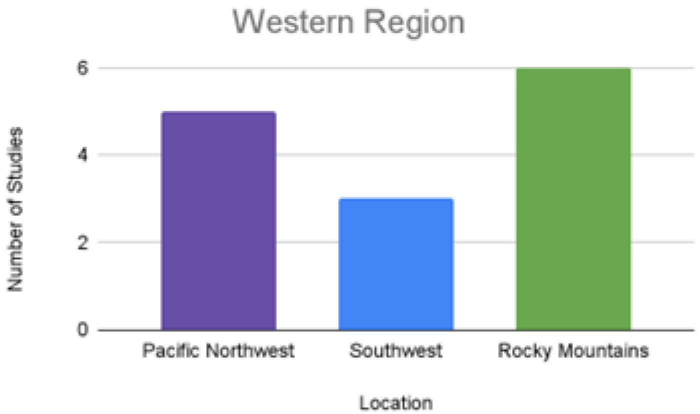


Figure 2.1 The number of studies in each western region of the United States, including three western regions: The Pacific Northwest, including California, Washington, and Oregon; the Southwest, including Arizona and New Mexico; and the Rocky Mountain region, including Colorado, Idaho, Wyoming, and Montana.

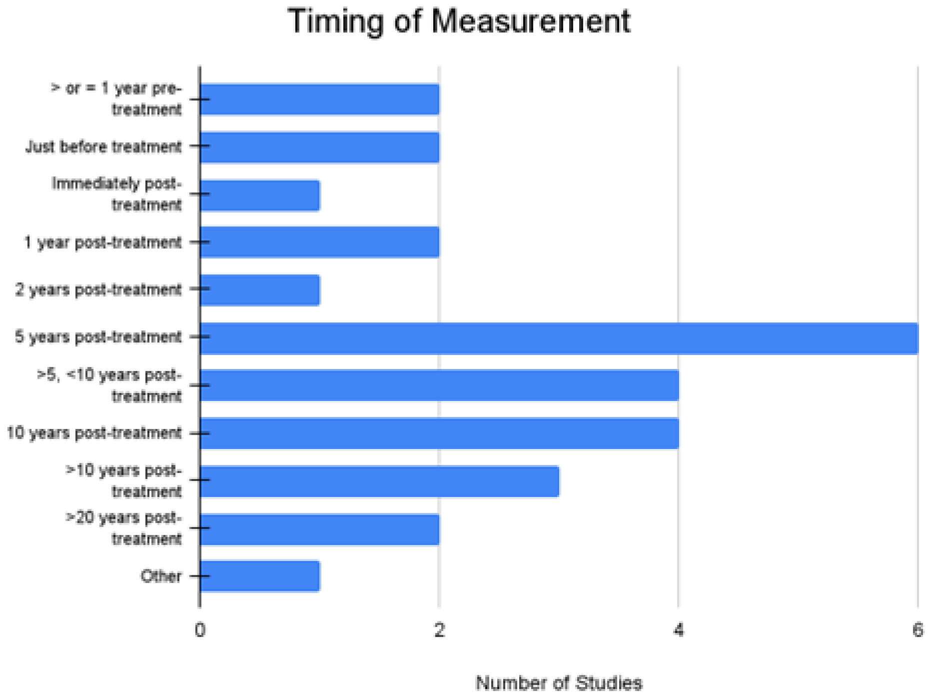


Figure 2.2 The number studies reviewed in this chapter that measured understory at various time intervals. A bar chart showing when studies were measured, and how many of the sixteen studies measured there. Studies that sampled more than once are represented at each relevant period.

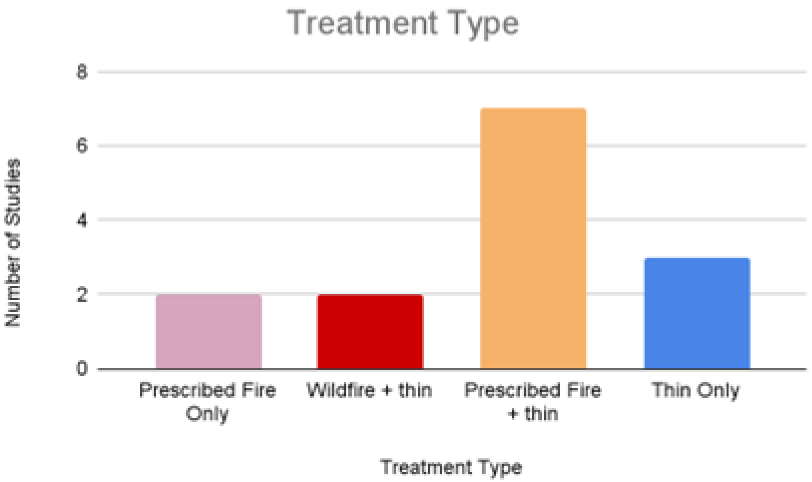


Figure 2.3 A number of studies using particular treatment types prescribed fire only, combination of wildfire and thinning, prescribed fire and thinning, or thinning only.

Appendix 2.A: All search terms, databases, and outcomes from search.

Date	Database Used	Search Terms	# Results
11/2/22	Web of Science	Understory OR understorey OR understories AND fuels treatments OR thinning OR masticat* OR prescribed burn OR fuels reduction AND mixed conifer OR “mixed conifer forest”. 2014-2022.	113
10/22/22	Google Scholar	understory or understorey OR understories OR vegetation AND "fuels treatments" OR thin OR burn OR thinning OR "prescribed burn" OR masticat* AND ponderosa OR "ponderosa pine" OR conifer OR "mixed conifer". 2014-2022.	17700
10/19/22	EBSCOhost CAB Abstracts	"climate" OR "climate change" OR climat* OR "climate variability" AND understory OR understorey OR understories AND thinning OR thin OR masticat* OR "fuels treatment" OR "fuels treatments" OR reduction	240

Chapter 3 Understory vegetation responses to fuels reduction treatments in the Jemez Mountains, New Mexico, USA

Introduction

The historical fire regime in warm-dry mixed-conifer forests of the Southwest consisted of frequent, low-severity surface fires, which maintained lower overstory densities in forests compared to today (Margolis et al. 2013). However, a century of fire exclusion, land use changes, and climate change are increasing the risks of widespread and uncharacteristically high-severity wildfire (Fitzgerald et al. 2005, Reynolds et al. 2013, Singleton et al. 2019). In response, forest and fire managers across the Southwest are implementing fuels reduction treatments as an effort to mitigate fire hazard, by reducing high fuel densities and removing small ladder fuels in dense forests (Sample et al. 2022, Reynolds et al. 2013). Mechanical treatments can include thinning and mastication, and prescribed burning is another treatment aimed to achieve these objectives (Agee & Skinner 2005). Mechanical thinning is the removal of trees at varying size classes with the goal of reducing fire potential and severity, reducing overstory densities, and more (Dodge et al. 2019). Mastication is the chipping or mulching of overstory trees or downed logs, aiming to reduce ladder fuels and increase openings in the overstory (Fornwalt et al. 2017). Prescribed burns, including both pile burns and broadcast burns, are a management method of inducing lower severity burns across a predetermined landscape, usually with the goal of reducing surface and ladder fuels (Agee & Skinner 2005).

Fuel reduction treatments promote heterogeneity in the landscape and increase diversity, which helps maintain biological diversity, resilience to climate effects, and regeneration after disturbance (Margolis et al. 2013, Neill & Puettman 2013, Sample et al. 2022). Climate- and fire-forward management will continue to allow managers to be proactive in preserving species diversity, key structural elements, and resilience (Sample et al. 2022, Guiterman et al. 2022).

In a systematic review on plant community responses to fuel reduction treatments, Abella & Springer (2015) found that in general, understory vegetation communities respond to fuel reduction treatments by increasing species richness, cover, and diversity. In our literature review, we found similar results- that all native lifeforms, including graminoids, forbs, and shrubs increase after treatment. A notable gap in the available literature, noted by Abella & Springer (2015) and continuing in our review in 2023, was that long-term studies in mixed-conifer systems are particularly absent. There are several studies available that research the effects of thinning and burning treatments on understory communities up to 5 years after a treatment. Studies analyzing effects on the understory more than 10 years after a fuels reduction treatment were limited.

From our review combined with Abella & Springer (2015), we found that non-native plants peak from 2-5 years after a thin + burn treatment, and in the long-term (more than 10 years) decrease to pre-treatment levels (Jang et al. 2021, Goodwin et al. 2018, Strahan et al. 2015). We also found that native graminoids and forbs particularly respond well in the long-term to fuels reduction treatments, with an immediate decrease after treatment followed by gradual increase in cover through time (Korb et al. 2020, Jang et al. 2021).

Forest types in the Southwest span from ponderosa pine zones in mid-high elevations, to spruce-alpine fir zones in higher elevations. Mixed-conifer forests span one million hectares across the Southwest (Dieterich 1983) and these forests range from cool, mesic wet forests and warm, dry forests. Warm-dry forests are generally more fire resistant, less shade tolerant, and between ponderosa pine to dry mixed conifer ecozones (Reynolds et al. 2013).

Santa Fe National Forest, where our research takes place, is in a warm-dry mixed conifer forest. Mixed-conifer forest species diversity is high, and because of the complexity of these ecosystems, including topographic and microclimate variety, overstory dominance, and fire regimes, the effects of fuel treatments and wildfire on understory communities varies (Springer et al. 2018).

Our study site is in the Jemez Mountains in north-central New Mexico. Monument Canyon Research Natural Area (RNA) is divided into two subsites; one was masticated in 2006-2007 and burned in 2012, while the other is a control and has never been treated. The mastication and burn were experimental in design to assess overstory changes, but the understory data has never been analyzed previously.

We evaluate differences between the control and the treated subsites pre- and post-treatments by analyzing two datasets, one from the University of Arizona (UA) and one from the National Park Service Pueblo/Four Winds Fire Ecology Program (P4WFEP) . With this research, we hope to answer the following questions: 1. Were the two sites similar prior to treatment? 2. Is there a treatment effect in 2022 and 2023? 2.a. What species are driving these changes? 3. What is the community recovery trajectory in the treated site from 2009-2023?

Methods

Study site:

Monument Canyon Research Natural Area (RNA) is in the Jemez Mountains on land managed by the Santa Fe National Forests in north-central New Mexico, USA. The land is within the historical and contemporary use area of the Jemez Pueblo, holding enduring significance for the Jemez Pueblo communities, and was used for hunting, growing crops, and seasonal occupancy (Swetnam et al. 2016, Roos et al. 2021). Recent studies reveal profound demographic shifts following European contact, underlining the intricate interplay between human society and ecological shifts through time (Swetnam et al. 2016; Liebmann et al. 2016).

The Jemez Mountains are a remnant volcanic caldera from early Pleistocene eruptions that created a large feature known today as Valles Caldera, and the surrounding mountain range (Roos et al. 2021). Soils are volcanic, primarily Lower Bandelier tuff (Otowi member) (QBO) and El Cajete pumice (QEC), with Holocene alluvium in drainages (Kelley et al. 2003, Goff et al. 2005, Marshall & Falk 2019). The Monument Canyon Research Natural Area encompasses an area of 256 hectares of the caldera rim and is divided by a central erosional valley into two distinct subsites, Cat Mesa and San Juan Mesa (Figure 3.4).

The elevation of Monument Canyon RNA ranges from 8,000-8450 ft (2438-2575 m). Monument Canyon RNA was established in the 1930s and is the second-oldest RNA in the country (New Mexico Forest and Watershed Restoration Institute 2011). (“Monument Canyon CFRP Field Inventory Summary / Fall 2011 New Mexico ...”) The Monument Canyon RNA supports dry mixed-conifer and Ponderosa pine forest communities on the southwestern side of the mountain range.

The mean annual temperature from 2001 to 2010 was 8.3°C (46.94°F), while the mean annual precipitation was 552 mm (21.73 inches)(Wang et al. 2016). The mean annual temperature from 2011-2020 was 8.9°C (48.02°F), and the mean annual precipitation was 630 mm (24.8 inches) (Wang et al. 2016). The mean average temperature in 2022 was 8.7°C (47.66 °F), and the annual precipitation was 732mm (28.82 in) (Wang et al. 2016). Temperatures from 2001-2010 ranged from an average coldes*t*-month minimum of –1.1°C (30.02°F) , and an average warmes*t*-month maximum of (19.4°C (66.9 °F) (Wang et al. 2016). Decadal averages in 2011-2020 show that the mean coldes*t*-month temperatures are –1.2°C (29.84°F). There is a significant impact of the 21st-century drought in the Southwest, which has led to substantial reductions in snowpack, river flow, and groundwater availability (Williams et al. 2020). This prolonged drought event, which began in the early 2000s, is notably drier and warmer compared to the 20th century conditions (Williams et al. 2020).

Data:

Field data were derived from sampling conducted by two research teams, the NPS Pueblo/Four Winds Fire Ecology Program (P4WFEP) and the University of Arizona. The National Park Service's Fire Monitoring Handbook (FMH) was adapted by the P4WFEP to monitor transect lines and conduct frequent monitoring of overstory, understory, fuels, fire weather, and more (USDI National Park Service 2003). The selection of plots from P4WFEP and UA sites adhered to a random and strategically dispersed approach, respectively.

The P4WFEP installed 25 plots on San Juan Mesa in 2009, and 18 plots on Cat Mesa in 2018 (Figure 3.4) in anticipation of planned fuels treatments. Monitoring was conducted frequently after installation (Table 3.1, Table 3.2). UA installed 16 plots in both Cat Mesa (8) and San Juan Mesa (8) in 2000 (Figure 3.5, Figure 3.6). Some of the P4WFEP plots that were installed several years later coincided with UA locations (Figure 3.4).

We used earlier (2004) collections by UA to provide pre-treatment references in the two sites. In addition to treatment effects, the earlier sampling data in the UA plots facilitated a comparison of change due to climate variability. 2022 and 2023 are the only years that were monitored across both sites for the P4WFEP datasets, and 2023 was the only year in which all UA and P4WFEP plots were monitored in both sites (Figure 3.5). Because plot design differed, along with the variation in sampling events for each site and dataset, we used different subsets of plots to answer specific questions.

For both datasets, we collected "tree cover" from each plot by estimating the percent cover of a given tree species in a quadrat, or by intercepting a species on the transect line. As it is not a reflection of canopy cover or closure, and it is also not a presence/absence cover estimate with stems, we refer to it in the text as basal area tree cover. It is not the basal area often collected by foresters; it is the average cover of tree species that were collected in plot observations.

Sampling Design:

Pueblo/Four Winds Fire Ecology Program's point-intercept transect protocol:

Field sampling for P4WFEP's poin*t*-intercept transect protocol was based on herbaceous vegetation transects, shrub density, and tree canopy closure. Plots on San Juan Mesa consisted of plots 50m by 20m with point transects running along both 50m edges. (Figure 3.2). The plot design was the same for Cat Mesa, but the plots were 25m by 20m with point transects running along each of the 25m sides.. Rebar posts were permanently installed at the four corners of the plot and the origin so that the transect tapes could be monitored with spatial consistency every time.

A 2m-tall sampling rod was used to conduct vegetation monitoring. The vegetation or substrate touching the pole was identified to species, or at least to genus if the species could not be identified. The height at which the vegetation contacted the pole was also recorded in meters; only the tallest vegetation height was recorded. If the sampling rod touched several species, the taxonomic identification of all other species or substrates touching the sampling rod was recorded. Substrates, such as litter, moss, lichen, rock, bare soil, and wood, were recorded only if there was not a live plant species intercepted with the sampling rod. The monitor walked the length of the transect on each side, measuring and recording the intercepted vegetation every

0.3m. If a species was not intercepted along the transect, but was present on the plot, then the monitor recorded the new species (in these cases, height or frequency were not recorded).

Shrub density was measured on the inside of the transect line by tallying each shrub species, regardless of size, within 5m of the tape on the inside of both transect lines.

Shrub species observed, or "species present but not intercepted," were recorded along a belt on the inside and outside of each transect by 10m. If a species was not intercepted along the transect, but was present on the plot, then the monitor recorded the new species (in these cases, height or frequency were not recorded).

Canopy closure, the proportion of the sky obscured by vegetation when measured from a single point, was measured using a spherical densiometer along the center line, from 0P to 50P for San Juan Mesa, or 0P to 25P for Cat Mesa (Figure 3.2), at designated points and then averaged to get canopy closure per plot.

University of Arizona quadrat monitoring protocol:

The University of Arizona plots used a nested quadrat design. Each sample site included eight 10m by 10m plots, each with five 1m by 1m quadrats (Figure 3.3). In each plot location, understory vegetation cover was estimated aerially, using a 10 cm by 10 cm reference guide that represented 1% cover.

Data analysis and statistical methods:

The P4WFEP entered and queried all their data into Feat/FIREMON Integrated (FFI – Lutes et al. 2009); and UA’s data were entered into Excel databases from original paper datasheets. We cleaned and formatted their 2023 data for analysis in both Microsoft Excel and R.

For both UA and P4WFEP plots, percent cover per species and per plot were calculated by averaging the cover measured in the quadrats, or on the transect lines.

In the UA dataset, some plots were monitored more than once over the summer season; we utilized data from the sampling events that occurred in July or August to maintain consistency with species phenology and monitoring over time. Percent cover, as well as lifeform (shrub, graminoid, forb, grass-like, tree, or substrate), and nativity (native or non-native) were recorded in the data summary.

The two datasets utilized different collection methodologies, and consequently, we approached them as separate analytical entities. The UA dataset from 2004-2005 (Figure 3.1, 3.3, 3.4) served as the foundation for establishing pre-treatment conditions, and then the same methods were implemented in 2023 to obtain a pre- and post-treatment analysis. We used the P4WFEP dataset to compare plant communities over time and for a compositional assessment of control and treated sites in 2022 and 2023.(Figure 3.1, 3.4).

Excel was used primarily for cleaning, formatting, and organization, while R was used for all statistical tests. Packages used in R included “ggplot2”, “vegan”, “tidyverse”, “dplyr”, and “readxl”, “gridExtra”, “car”, “rstatix”, and “indicspecies”.

Our data from both datasets did not conform to assumptions of normality or sphericity. Transformations were not successful in normalizing the data. Nonparametric tests will be implemented in future research, and we recognize that with our data not conforming to assumptions it gives us greater probability of Type I errors. In other words, there is a higher chance that we reject the null hypothesis and conclude a significant treatment effect when it is not correct to do so.

Question 1: Were the two sites similar prior to treatment?

We hypothesized that the two sites were not significantly different in functional group cover and diversity before treatment in 2004 and 2005, because the overall study area has a common geological origin, soils, fire history, and human presence over multiple centuries. We analyzed the pre-treatment data from UA to provide context for any statistically significant differences found due to treatments.

We organized plant species by lifeform (forb, shrub, tree, graminoid, abiotic and biotic substrate) and calculated the mean percent cover of each lifeform for each year and site. Abiotic substrates included bare ground cover and rock cover. Biotic substrates included litter, duff, coarse woody debris, lichen, moss, fungi, etc.

To assess if the sites were similar before treatment on a broader, community scale, we conducted multi-response permutation procedure (MRPP) tests. MRPPs test differences between the two communities in given years by analyzing percent cover matrices for species and lifeforms. MRPP is a non-parametric method to test multivariate differences in communities (McCune & Mefford 1999, Antoninka et al. 2009). We used the Bray-Curtis distance because it is common in ecological community data, does not assume normality, and does not interpret zeros as similarity (Minchin 1987, Antoninka et al. 2009). We reported test statistics *p*-value and A-value. The *p*-value indicates the probability that observed differences are due to chance alone, and the accepted *p*-value to indicate differences between communities is less than 0.05. The A-value describes within-group similarity, with 0 indicating difference within a sample is equal to difference expected by chance; positive values indicating higher differences than expected by chance, and negative values indicating lower within-sample differences than expected by chance (Carter & Feeney 2012, Antoninka et al. 2009). In ecological community data, A-values over 0.1

are generally accepted (McCune & Grace 2002, Antoninka et al. 2009). Lastly, we ran non-metric multidimensional scaling ordinations (NMDS) to visualize potential differences in the two communities each year. The MRPP significance values can be attributed to the visualized differences in the NMDS ordinations. We ran NMDS ordinations between both species and lifeforms. We also conducted repeated measures ANOVAs and post hoc paired *t*-tests to evaluate the differences in average percent cover of each lifeform between Cat Mesa and San Juan Mesa in 2004 and 2005.

Question 2: Are there treatment effects in 2022 and 2023?

Both the UA and P4WFEP datasets were used to answer the question of treatment effect. UA was used to compare the control and treated sites in 2023, as well as the difference from pre-to post-treatment (2004 and 2005 to 2023). The P4WFEP dataset compared treatment effects in 2022 and 2023, the only two years where both sites were monitored together. We hypothesized that in both dataset analyses there would be a significant difference in percent cover of forbs, graminoids, shrubs, and biotic substrates between the control and treated sites after treatment, as previous studies suggest that these lifeforms may increase after treatment (Jang et al. 2021, Fornwalt et al. 2017, Strahan et al. 2015). We also hypothesized that there would be a significant short-term increase in non-native species cover after treatment and a gradual decrease through time (Jang et al. 2021, Zald et al. 2020, Goodwin et al. 2018).

We analyzed the two datasets separately, but conducted the same tests for both: MRPPs, NMDS ordinations, repeated measures ANOVAs, and post-hoc paired *t*-tests, like the methods described in Question 1. We also compared nativeness in the sites from both datasets- with the UA dataset, we used boxplots to visualize the pre- and post-treatment non-native species cover

(2004, 2005, 2023) in both sites. With the P4WFEP dataset we compared nativeness between our control and treated sites in 2022 and 2023.

Question 2.a. Which species are driving differences between the sites after treatment?

To identify which species were potentially driving differences, or lack thereof, in certain years and lifeforms we conducted indicator species analysis (ISA) in San Juan Mesa in 2023 (Caceres et al. 2012). We ran the ISA with species matrices containing the 2023 data from both sites. From the ISA, we created a 60% strength threshold to parse out more significant species and then cross-checked the strength value with associated *p*-values. If a species had a high strength value and a high *p*-value, we did not present them as indicator species. If the *p*-value was significant (<0.05), we included the species as an indicator. After identifying the indicator species from both datasets, we created boxplots to visualize their cover through time and assess potential treatment effects in San Juan Mesa.

Question 3. What was the treated site's recovery trajectory from 2009-2023?

We used the P4WFEP dataset from 2009-2023 to address this temporal analysis. We hypothesized that graminoid, forb, and shrub cover would decrease immediately post-burn, and over time increase significantly. Relevant literature suggests that graminoid, forb, and shrub populations immediately decline, but gradually increase after treatments through time (Korb et al. 2020, Jang et al. 2021, Goodwin et al. 2018). We further hypothesized that non-native species cover would decline immediately post-fire, peak two to five years after the burning treatment, and decrease in the long-term (Jang et al. 2021, Abella & Springer 2015). We also hypothesized that biotic substrate (coarse woody debris, litter, etc.) would decrease immediately after the burn, but recover over time .

We conducted repeated measures ANOVAs and post-hoc paired *t*-tests. In the repeated measures ANOVA, Year was the main effect as the temporal analysis was only in the treated site. We assessed if there were differences in functional group cover by year in San Juan Mesa.

Results

Question 1: Were the sites similar prior to treatment?

Our results indicate that Cat Mesa and San Juan Mesa were compositionally similar before treatment in 2004 and 2005. Multi-response permutation procedure testing (MRPP), non-metric multi-dimensional scaling (NMDS) ordinations, repeated measures ANOVAs, and post-hoc paired *t*-tests indicate that the sites were similar prior to treatment in percent cover of species and functional groups.

The MRPP from pre-treatment analyses indicate that the sites were not significantly different in 2004 by lifeform or species, while 2005 results indicate a significant difference in lifeform, but no difference in species (Table 3.3). The NMDS support the MRPP, as there are no distinct patterns or distances between the two communities, except in the 2005 lifeform NMDS plot (Fig. 3.7). Repeated measures ANOVA tests by lifeforms (forb, graminoid, shrub, tree, abiotic substrate and biotic substrate) indicate that the two sites were compositionally similar before treatment in every lifeform except for graminoids (Site $p = 0.01$, Year $p = 0.02$, Site:Year $p = 0.00$, Table 3.4). Graminoid cover was significantly higher in Cat Mesa, the control site, before treatment. Because of the significant difference in graminoid cover, we ran a post-hoc paired *t*-test to assess the direction of effect, which showed that there was a difference in graminoid cover between the sites in 2004 and 2005 ($p = 0.03$). We created boxplots to visualize differences in lifeform cover by site. Although there is interannual variation in each lifeform, mean lifeform percent cover in 2004 and 2005 were not significantly different, aside from graminoids (Fig 3.8).

Question 2. Is there a treatment effect in 2022 and 2023?
UA:

In 2023, NMDS ordinations from the UA dataset indicate compositional differences between Cat Mesa and San Juan Mesa (Figure 3.10). In the NMDS, there are distinct distances and patterns between the two communities, indicating differences on a lifeform and species scale. MRPP results from the UA dataset indicate that the control and treated sites are significantly different on a species scale (A = 0.18, $p = 0.00$), and on a functional group scale (A = 0.07, $p = 0.04$). The MRPP values support these NMDS ordinations by quantitatively indicating that Cat Mesa and San Juan Mesa are significantly different in 2023, with data from the UA dataset (Figure 3.10).

Repeated measures ANOVA tests from 2004, 2005 and 2023 indicate that there was a significant difference in percent cover in the Site:Year interaction term in forbs, graminoids, shrubs, abiotic and biotic substrates, and trees (Table 3.5). There was not a significant difference in graminoid cover between sites in 2023, but since graminoids were statistically different with higher cover in the control in 2005, the pre-treatment difference is likely driving the post-treatment Site:Year interaction. Graminoid cover in the treated site exhibited significant changes over time, with a notable difference from our post-hoc *t*-test between 2004 and 2023 ($p = 0.04$) and a highly significant difference between 2005 and 2023 ($p = 0.01$) (Table 3.6, Figure 3.10). There were also significant differences in the within-Year effect in forbs, abiotic and biotic substrates, indicating that there were significant changes in cover for those lifeforms from 2004-2023 (Table 3.5).

Conversely, the shrub population demonstrated no significant difference between pre-treatment years and 2023 (p = 0.07 and p = 0.07, respectively) (Table 3.6). The pattern in shrub cover is similar to that of graminoids: shrub cover increased on the treated site, resulting in higher cover than the control (Figure 3.10). There was an average of 0% shrub cover in San Juan in 2004 and 2005, and the average increased in 2023 (Figure 3.9). For forbs, while there was no significant difference between 2004 and 2023 (p = 0.17), there was a highly significant change that emerged between 2005 and 2023 (p = 0.00). Tree cover and substrate composition showed no significant variations between the pre-treatment and post-treatment years.

Tree cover increased in both sites from 2004-2023 (Fig 3.10). Biotic substrates decreased significantly in 2023 compared to pre-treatment sampling years. Cat Mesa biotic substrate cover stayed consistent from 2004-2023, while abiotic substrate cover increased significantly in San Juan Mesa post-treatment (Figure 3.10).

Non-native species were not recorded on the sites in 2004 and 2005, and in 2023 non-native species were recorded on both sites, but with higher abundance in San Juan Mesa (Figure 3.13).

P4WFEP:

The 2022 and 2023 NMDS ordinations also indicate differences between the two communities, using both lifeform and species scale observations in 2022 and 2023 (Figure 3.11). The MRPP also supported the results from the P4WFEP dataset (p = 0.00 for 2022 and 2023 in both lifeform and species, Table 3.8). The repeated measures ANOVA and post-hoc paired t-tests indicate similar results; there are significant differences between the two sites on lifeform scales (p = <0.00 for forb, graminoid, basal area tree, and substrate cover, p = 0.01 for shrub cover). Forb, graminoid and shrub cover were higher in the treated site than the control, and substrate and tree cover was higher in the control site (Figure 3.12).

We assessed native cover between the control and treated sites in 2022 and 2023 and found that in Cat Mesa, the control, there were very low cover values for non-native species recorded in both years. In San Juan Mesa, the treated site, there were more non-native species present (Figure 3.13).

Question 2.a) What species are driving differences in the treated site?

The UA indicator species for 2023 included moss, bare ground, *Bromus ciliatus* (fringed brome), *Carex* species, *Conyza canadensis* (horseweed), moss, *Robinia neomexicana* (New Mexico locust), *Penstemon barbatus* (Beardlip penstemon), *Senecio wootonii* (Wooton's ragwort), *Verbascum thapsus* (Mullein), and coarse woody debris (Figures 3.14-3.21). Litter was an indicator species for the UA dataset, so we included its change through time from both datasets (Figure 3.22).

Moss cover in both datasets increased through time, including from pre- to post-treatment and in the temporal analysis, from post-burn through 2023 (Figure 3.14). From the UA dataset, *Robinia neomexicana* increased from pre- to post-treatment, and in the P4WFEP the cover of *Robinia* increased through time after treatment as well, with the highest variance and cover in 2022-2023 (Figure 3.15). Bare ground cover increased from pre-post treatment in the UA dataset and decreases through time after treatment in the P4WFEP dataset (Figure 3.16). Coarse woody debris increased significantly from pre- to post-treatment in the UA dataset, and in the P4WFEP dataset, it was highest post-thin pre-burn, decreased two years post-burn and then gradually increased through time (Figure 3.17). *Senecio wootonii* increased through time in both datasets, though between 2022 and 2023 there was interannual variability (Figure 3.18). Native grasses such as *Koeleria macrantha* , *Poa fendleriana* and *Elymus elymoides* gradually increased over time after an initial decrease in cover post-burn (Figure 3.19).

Non-native indicator species, *Conyza canadensis* and *Verbascum thapsus* show low cover values immediately after treatment, a peak from 2014-2017, two to five years after treatment, and decreased in cover in 2022-2023 (Figure 3.20, Figure 3.21).

Lastly, litter decreases pre- to post-treatment from the UA dataset, as well as gradually decreases through time after treatment from the P4WFEP dataset (Figure 22).

Question 3. What was the community composition of the treated site through time?

We ran repeated measures ANOVA tests on the changes of lifeforms including forbs, graminoids, shrubs, basal area trees, and substrates in San Juan Mesa with Year as the main effect (Table 3.9). We found that there were significant differences between the years for all the lifeforms, which indicates that the cover of each lifeform significantly changed from 2009-2023 (p = <0.00 for forb, graminoid, shrub, and substrate cover, p = 0.05 for basal area tree cover). Post-hoc paired t-tests supported these significant results with significant differences between the majority of tested years (Table 3.10).

Significant differences in forb cover were observed in nearly all inter-year comparisons with an increasing trend from 2013 to 2022 and a decrease in 2023, indicating their sensitivity to temporal changes in the ecosystem (Figure 3.23a). Graminoids showed significant differences between 2009 and 2022 with an increasing trend in 2013 and after (Figure 3.23b). Shrubs displayed significant variation in cover between 2009 and 2022, showing an increasing trend from 2013 to 2022 with a decrease in 2023 (Figure 3.23c). Trees showed significant differences in cover between 2009 and 2013, 2009 and 2014, and 2013 and 2014 (Figure 3.23d). The lifeforms were not significantly different between 2022 and 2023. The boxplots help visualize our findings from both the repeated measures ANOVA results and the post-hoc paired *t*-tests.

Non-native species cover shifted through time (Fig 3.24). Non-native species cover was low in 2009 and 2013, peaked in 2014 with high variance in 2017, and has decreased to low values in 2022 and 2023 (Figure 3.24).

Discussion

Climate change continues to affect ecological communities in the southwestern U.S., resulting in uncharacteristically severe wildfires and immense periods of drought. These issues threaten biodiversity and ecosystem services, paired with shifting land use and management patterns. As forests undergo treatments to mitigate these stressors, there arises a need to understand the intricate dynamics of treatments. Long-term responses of understory communities to fuels reduction treatments in the Southwest are relatively understudied. This this research adds to the important body of knowledge that managers may use to make informed decisions in the future.

Were the two sites similar prior to treatment?

We hypothesized that the sites would be similar before treatment. Repeated measures ANOVA tests, MRPP tests, and NMDS ordinations support the conclusion that Cat Mesa and San Juan Mesa prior to treatment were compositionally similar, providing a crucial baseline for post-treatment comparison. Graminoids between sites were statistically different in both 2004 and 2005. Graminoid cover in Cat Mesa was higher before treatment.

Treatment effects in 2022 and 2023, after treatment

Analysis of lifeform cover, species richness, and substrate composition from the UA dataset in 2023 reveals significant differences between Cat Mesa and San Juan Mesa. Forbs, abiotic and biotic substrates exhibited notable disparities on treated sites, indicating a substantial impact of the prescribed burn and thinning treatments on these functional groups. Notably, graminoid cover, significantly higher on Cat Mesa pre-treatment, increases and becomes more abundant (though not significantly so) after treatment on San Juan Mesa, indicating a positive treatment effect.

The P4WFEP dataset corroborates the U of A dataset and showed significant differences in plant communities between sites for both 2022 and 2023. Cat Mesa had higher substrate and basal area cover, while San Juan Mesa exhibited higher values of forbs, shrubs, and graminoids. NMDS ordinations visually confirmed site differences, emphasizing persistent variations after treatments. These findings highlight long term variations in community composition and ecological dynamics after the mastication and prescribed fire treatments. Both analyses comparing communities after treatment resulted in similar findings: that diversity and functional group cover are different between sites after treatment. Both datasets also identify similar indicator species.

It is also notable that non-native species were not recorded on the sites pre-treatment but were found on both sites in 2023. San Juan Mesa, as predicted, had higher levels of non-native species cover than Cat Mesa after treatment.

Which species are driving these differences?

Indicator species analyses identified the key species driving observed shifts in community composition. A few of these indicator species suggest fire response traits that warrant further investigation. *Robinia neomexicana* is a seral sprouting species following thinning or fire disturbances, forming dense clusters from the root crown and rhizomatous roots (Pavek 1993). The species was not recorded on the landscape in 2004, was rare in 2005, but increased three-fold on some plots in 2023. The mean cover did not increase significantly, but variance showed some plots had over 8% total *Robinia* cover. This could indicate a treatment effect that speaks to the possibility of shrub conversion across these landscapes after higher-severity burns. Initial field observations lend to *Robinia* having highest cover in plots that endured the highest fire severity from the burn treatment, though this has not yet been validated. It would be an interesting additional study to examine fire response of plants in context of their exposure to the severity gradient of the prescribed burn in San Juan Mesa.

Recovery trajectories and long-term effects of treatment in San Juan Mesa

The investigation into the post-thin/pre-burn (2009) and post-both treatments (2013, 2014, 2017, 2022, 2023) periods provides a temporal perspective on community composition changes. Forbs, graminoids, shrubs, and trees exhibited varying responses over time, with some significant differences in cover between specific years. Indicator species analyses identified key species that played a role in the recovery trajectory, with native species gradually increasing over time. The recovery trajectory suggests a complex interplay of ecological dynamics, with both native and non-native species responding differently to the treatments.

Indicator species and their respective cover trajectories were revealed through time (Figures 3.14-3.19). *Verbascum thapsus* is a ubiquitous non-native species that often survives or germinates after a thinning or fire disturbance (Gucker 2008). The predicted trajectory of all non-native plant species from 2009-2023 was confirmed (Fig. 3.20- 3.21), which was an immediate decrease, a peak 2-5 years after treatment, and a long-term gradual decrease in cover. There was low non-native cover after fire in 2013; and in 2014-2017, 2-5 years after treatment, non-native species cover peaked, and then stabilized 10-11 years after treatment in 2022 and 2023. This prolonged decrease and stabilization of non-native species cover speaks to the importance of long-term monitoring.

Native perennial graminoids, such as *Elymus elymoides* and *Koeleria macrantha,* squirrel tail and prairie junegrass, were also indicator species. Their cover declined immediately after treatment, and then increased through time (Figure 3.19). Substrate ISA analysis indicates that litter decreased post-thin but pre-burn (2009), peaked in 2013 just after the prescribed burn, and has decreased gradually from 2014-2023 (Figures 3.16, 3.17, 3.22). The immediate increase in bare ground after treatment indicated that treatment caused more erosion on the treated site, but it also showed a decrease in bare ground through time, which indicates that treatment stimulates plant cover through time, potentially assisting in erosion prevention (Figure 3.16). The immediate increase and gradual decrease in bare ground cover was attributable to postfire erosion, while the steady increase in moss may be attributed to overall biotic recovery and erosion control (Grover et al. 2020). Coarse woody debris was at its highest in 2009, post-thin/pre-burn, but after fire decreased to less than 5% cover on average. The cover of coarse woody debris in 2022 and 2023 is increasing 10 years post-treatment, which may suggest that repeated interval burning is helpful for the landscape in maintaining biotic substrates as time goes on.

Implications for ecological management

The study provides valuable insights into the long-term effects of prescribed burn and thinning treatments on community composition and ecological dynamics. Our results suggest that prescribed burn and thinning treatments positively impact understory vegetation communities, leading to increased diversity and plant cover, even in the longer term. This has important implications for ecological management, particularly in areas where the goal is to enhance biodiversity and ecosystem resilience. Land managers can consider implementing similar fuels reduction treatments to achieve these positive outcomes. Although the treated and control sites were not significantly different in certain functional groups, our data indicate an increase through time of each biotic functional group and indicator species in the treated site. Understanding the differential responses of various lifeforms and species can inform future management strategies.

This complex research with varying datasets, sampling methods, and data collection events highlights the importance of long-term monitoring, and to encourage land managers and scientists to increase communication and collaborative work. Our findings forge a narrative of recovery after treatments by piecing together two different datasets . We used the UA dataset to establish a baseline of comparison for the two sites, and the P4WFEP dataset to evaluate temporal shifts through the treated site. Both datasets provided site comparisons post-treatment. A review of current literature indicates that there are few long-term studies evaluating the effects of the understory over time. Our study indicates that monitoring data can, in some cases, be compiled by combining data from multiple monitoring episodes to provide a longer-term perspective. Our temporal comparisons, over a decade after treatments, give us a glimpse of the effects of treatment on functional groups and certain species.

Limitations and future research

The primary limitation to this study is also what makes it so unique: our aim was to draw comparisons between the two datasets, but the inherent differences in methodologies and sampling years limited our ability to conduct direct statistical analyses for a comprehensive 19-year comparison. The lack of statistical comparability in some variables adds a layer of complexity to our interpretation, emphasizing the need for cautious consideration of the findings. The only years from the P4WFEP dataset that we could compare both sites (2022 and 2023) was limiting because it prevented site comparisons immediately post-treatment or assess temporal shifts through time. These differences in method and sampling effort demonstrate the benefits of pre-survey planning for both land managers and scientists . Installing and implementing landscape scale monitoring systems is challenging for land managers due to limitations in staffing, seasonal windows, and policies. However, ideally future long-term monitoring projects across sites or regions would aim to monitor treated and control sites simultaneously at annual, biannual, or on a consistent alternate year schedule. Regardless of methodological differences and sampling variability, we can use this data to contribute to the broader discussion of long-term effects of treatment, and of the efficacy of fuels reduction treatments in promoting ecosystem resilience and biodiversity.

In any field sampling effort, there is likely to be inherent sampling error, with the possibility of overestimations, misidentified plant species, or other field sampling errors. The protocols in place for both methods were sufficient in limiting sampling bias as much as possible, such as the standardized quadrats and transect layouts with rebar in the corners and origins, detailed written protocol, and ample training of technicians, but individual bias is always possible.

Several additional analyses could further develop knowledge of plant community response to fuel treatments. Analysis of the influence of fire severity could increase our understanding of how patchwork severity may impact species and the communities on a plot level. An analysis of the relationship of topographic variability to potential response of plants to fuel treatment could explain some among-plot variation. On San Juan Mesa, plots with steeper slopes burned at hotter severity in the prescribed burn (pers comm D. Falk), which may have led to higher levels of bare ground cover or non-native cover. Further research into the effects of interannual climate variation would add explanatory power, such as the impact of the ongoing drought conditions; interannual climate variability influence between functional group cover in San Juan Mesa in 2022 and 2023, and differences with pre-treatment conditions.

Plant functional traits and fire response traits would be crucial to further understanding the temporal analyses from the indicator species, namely *Robinia neomexicana*, non-native species, and several of the native graminoids. Understanding how they respond to treatments and using their response traits as a parameter in a model would be very helpful to add to the body of understanding.

Keystone species analyses and further evaluating potential impacts of treatment on the species-scale would allow us to get a glimpse at the impact treatments have on rare plants or culturally important plants. More continuous long-term ecological data will increase our understanding of how ecosystems are responding to ongoing stress.

Conclusion

In the face of escalating climate change impacts, including severe wildfires and prolonged drought in the Southwestern U.S., this study examined the long-term effects of prescribed burn and thinning treatments on understory vegetation communities. The significance of this study lies in its contribution to the broader understanding of ecosystem responses to such treatments, providing valuable insights for ecological management.

Our analyses of the Cat Mesa and San Juan Mesa communities found significant post-treatment differences between treated and untreated sites, especially in forbs, shrubs, and graminoids, indicating the impact of thinning and prescribed burning treatments. These findings suggest that prescribed burn and thinning treatments can be instrumental in enhancing biodiversity and ecosystem resilience over the long term. More generally, this study underscores the need for continued collaboration between land managers and scientists to share data in the interests of creating long-term understanding of ecosystem changes. of the analysis of two datasets from the UA and P4WFEP offers a nuanced narrative of recovery after treatments. Our work encourages a broader discussion on the efficacy of fuels reduction treatments and the necessity for sustained, collaborative efforts in monitoring and managing forest ecosystems.

As managers navigate the complexities of managing forested landscapes, our findings advocate for a balanced approach that considers both the immediate and long-term impacts of prescribed burn and thinning treatments. By fostering collaborative efforts between managers and scientists, and embracing a commitment to sustained monitoring, we may lay the foundation for sustainable management, sustainable forests, and ecosystem resilience.

Tables & Figures

Table 3.1 Cat Mesa plot information with years monitored, and respective dataset (UA or P4WFEP)

Plot	Dataset	Years monitored
127	UA	2004, 2005, 2006, 2023
137	UA	2004, 2005, 2006, 2023
147	UA	2004, 2005, 2006, 2023
149	UA	2004, 2005, 2006, 2023
154	UA	2004, 2005, 2006, 2023
156	UA	2004, 2005, 2006, 2023
164	UA	2004, 2005, 2006, 2023
166	UA	2004, 2005, 2006, 2023
FPIPO1T09109	P4WFEP	2018, 2022, 2023
FPIPO1T09127	P4WFEP	2018, 2022, 2023
FPIPO1T09136	P4WFEP	2018, 2022, 2023
FPIPO1T09137	P4WFEP	2018, 2022, 2023
FPIPO1T09145	P4WFEP	2018, 2022, 2023
FPIPO1T09146	P4WFEP	2018, 2022, 2023
FPIPO1T09147	P4WFEP	2018, 2022, 2023
FPIPO1T09148	P4WFEP	2018, 2022, 2023
FPIPO1T09153	P4WFEP	2018, 2022, 2023
FPIPO1T09154	P4WFEP	2018, 2022, 2023
FPIPO1T09155	P4WFEP	2018, 2022, 2023
FPIPO1T09156	P4WFEP	2018, 2022, 2023
FPIPO1T09157	P4WFEP	2018, 2022, 2023
FPIPO1T09158	P4WFEP	2018, 2022, 2023
FPIPO1T09162	P4WFEP	2018, 2022, 2023
FPIPO1T09163	P4WFEP	2018, 2022, 2023
FPIPO1T09164	P4WFEP	2018, 2022, 2023
FPIPO1T09165	P4WFEP	2018, 2022, 2023

Table 3.2 San Juan Mesa plot information, years monitored, and respective dataset.

Plot	Dataset	Years monitored
111	UA	2004, 2005, 2006, 2023
113	UA	2004, 2005, 2006, 2023
115	UA	2004, 2005, 2006, 2023
121	UA	2004, 2005, 2006, 2023
123	UA	2004, 2005, 2006, 2023

125	UA	2004, 2005, 2006, 2023
131	UA	2004, 2005, 2006, 2023
141	UA	2004, 2005, 2006, 2023
FPIPO1T0901	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0902	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0903	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0904	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0905	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0906	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0907	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0908	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0909	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0910	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0911	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0912	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0913	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0914	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0915	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0916	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0917	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0918	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0919	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0920	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0921	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0922	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0923	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0924	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023
FPIPO1T0925	P4WFEP	2009, 2013, 2014, 2017, 2022, 2023

Table 3.3 MRPP tests from the UA dataset between Cat Mesa and San Juan Mesa in 2004 and 2005, by lifeform and species. Both Cat Mesa and San Juan Mesa have 8 plots, for a total of n = 16.

MRPP Test	**Year**	**Significance** (*p*-value)	**A-value**
Lifeform	2004	0.21	**0.02**
	2005	**0.02**	0.11
Species	2004	0.21	**0.02**
	2005	0.51	**0.00**

Table 3.4 Repeated measures ANOVA by lifeform for 2004-2005 using the UA dataset. The reported values are p-values from the test, showing between sites, within years, and the interaction between site and year for each lifeform.

Lifeform	Site (between) *p-value*	Year (within) *p*-value	Site: Year (interaction) *p*-value
Forb	0.80	0.24	0.21
Graminoid	**0.01**	**0.02**	**0.00**
Shrub	0.15	0.89	0.89
Tree	0.28	0.86	0.83
Abiotic substrate	0.41	0.30	0.10
Biotic	0.93	0.06	0.56

*Table 3.5 Summary statistics with mean and standard error from the UA dataset by each lifeform. Repeated measures ANOVA results showing the F-statistic; between the two sites, within years 2004, 2005, and 2023, and the interaction of Site:Year. Asterisks indicate significance- * = p <0.05, ** = p<0.01, *** = p<0.001. Sample size is n = 8 plots in Cat Mesa, n = 8 plots in San Juan Mesa.*

	Cat Mesa			San Juan Mesa			ANOVA		
Lifeform	2004	2005	2023	2004	2005	2023	**Site**	**Year**	**Site:Year**
Forb	2.11 (±0.62)	2.19 (±0.53)	1.90 (±0.89)	3.70 (±2.24)	1.34 (±0.73)	6.61 (±0.82)	2.19	**3.98 ***	**4.97 ***
Graminoid	9.27 (±0.23)	5.81 (±1.44)	3.51 (± 0.85)	2.28 (±1.02)	1.49 (±0.49)	5.79 (± 1.30)	3.99	2.45	**12.34 *****
Shrub	0.23 (±0.18)	0.21 (±.12)	0.21 (±0.06)	0.00 (± 0.00)	0.00 (± 0.00)	2.80 (±1.30)	2.56	4.56	**4.58 ***
Tree	0.16 (±0.08)	0.14 (± 0.08)	2.38 (±1.02)	1.79 (± 1.28)	1.93 (±1.79)	1.31 (±0.57)	0.42	1.01	**2.76 ***
Abiotic	6.42 (± 3.81)	6.12 (±3.19)	4.15 (± 2.36)	2.94 (±1.60)	4.23 (± 1.89)	29.5 (± 7.94)	2.44	**8.41 ****	**11.70 *****
Biotic	80.4 (± 2.88)	86.2 (± 2.81)	87.4 (±3.12)	79.6 (±8.22)	90.2 (±2.24)	58.3 (±7.15)	4.48	**4.74 ***	**6.40 ****

Table 3.6 From the UA dataset. Post-hoc paired t-tests evaluating the differences in percent cover of lifeforms by year in San Juan Mesa, the treated site.

Lifeform	Year comparison	Significance value
Graminoid	2004 vs. 2023	**0.04**
	2005 vs. 2023	**0.01**
Forb	2004 vs. 2023	0.17
	2005 vs. 2023	**0.00**
Shrub	2004 vs. 2023	0.07
	2005 vs. 2023	0.07
Tree	2004 vs. 2023	0.64
	2005 vs. 2023	0.67

Substrate	2004 vs. 2023	0.83
	2005 vs. 2023	**0.01**
Species Richness	2004 vs. 2023	**0.00**
	2005 vs. 2023	**0.00**

Table 3.7 Summary statistics from the P4WFEP dataset. Means for each lifeform in each year data was collected with the associated standard error.

	Cat Mesa			San Juan Mesa					
Lifeform	2018	2022	2023	2009	2013	2014	2017	2022	2023
Forb	1.29 (± 0.39)	2.14 (± 0.54)	1.87 (±0.63)	3.04 (±0.59)	2.78 (± 0.49)	14.1 (± 1.81)	23.2 (±2.39)	26.7 (±2.52)	9.71 (±0.99)
Graminoid	7.33 (±0.84)	17.84 (±2.14)	11.01 (±1.42)	8.00 (±1.77)	3.97 (±0.86)	8.06 (±1.52)	18.2 (± 2.98)	40.4 (± 3.96)	27.7 (±2.46)
Shrub	1.55 (±0.27)	1.07 (± 0.50)	0.67 (±0.30)	1.20 (±0.37)	0.51 (± 0.17)	0.84 (± 0.29)	2.24 (± 0.53)	5.61 (± 1.27)	4.53 (±1.30)
Tree	17.68 (± 3.07)	17.37 (± 3.16)	15.53 (± 2.61)	8.11 (± 1.19)	3.41 (± 0.87)	2.30 (±0.74)	5.95 (±2.55)	6.81 (± 2.09)	5.40 (± 1.00)
Substrate	75.20 (±2.54)	55.06 (±2.68)	67.87 (± 2.85)	77.59 (± 2.26)	88.47 (± 1.50)	74.91 (±2.35)	57.33 (± 2.44)	40.24 (± 2.83)	45.64 (± 2.77)

Table 3.8 MRPP test results to evaluate if Cat Mesa and San Juan Mesa were significantly different on a community scale looking at lifeform and species differences in 2022 and 2023, using P4WFEP's dataset. The MRPP indicates that the control and treated sites are significantly different in 2022 and 2023.

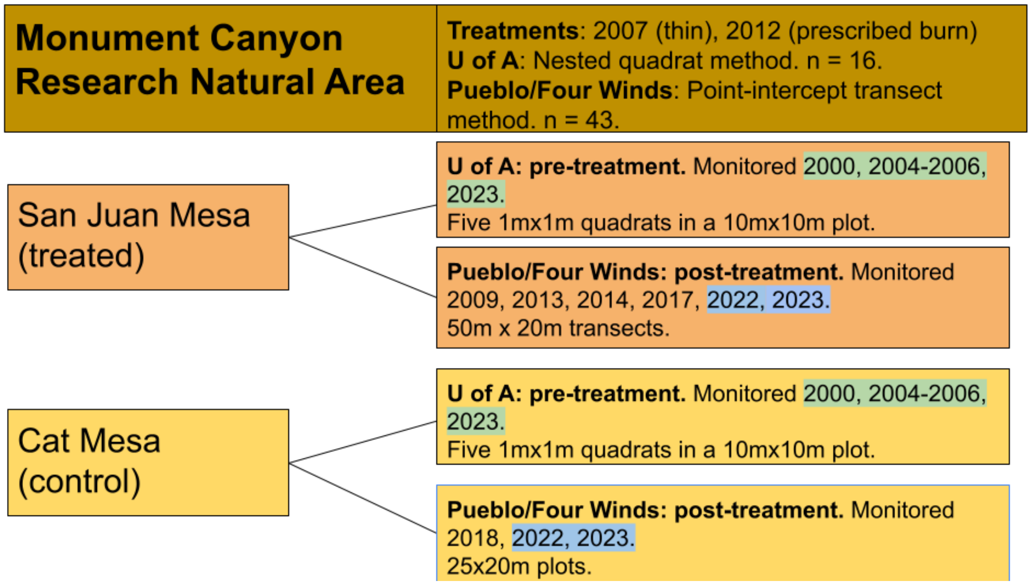
MRPP Test	Year	Significance (*p*-value)	A-value
Lifeform	2022	0.001	0.26
	2023	0.001	0.22
Species	2022	0.001	0.15
	2023	0.001	0.11

Table 3.9 From the P4WFEP dataset. Repeated measures ANOVAs by lifeform with year as the effect. The years that San Juan Mesa was measured were 2009, 2013, 2014, 2017, 2022 and 2023. Sample size is n = 25 in San Juan Mesa. All lifeforms were significantly different with Year as the main effect.

Lifeform	**F-value**	**Year (*p*-value)**
Graminoid	11.878	**2.41e-09**
Forb	55.623	**1.2e-29**
Shrub	11.878	**2.41e-09**
Tree	2.282	**0.05**

Table 3.10 Post hoc paired t-tests to identify differences in the repeated measures ANOVA tests. Only significant interactions between years were recorded. The p-values were adjusted for repeated measures with the Bonferroni method.

Lifeform	Year comparison	Adjusted *p*-value
Forb	2009 vs. 2014	**0.00**
	2009 vs. 2017	**0.00**
	2009 vs. 2022	**0.00**
	2009 vs. 2023	**0.00**
	2013 vs. 2014	**0.00**
	2013 vs. 2017	**0.00**
	2013 vs. 2022	**0.00**
	2013 vs. 2023	**0.00**
	2014 vs. 2017	**0.01**
	2014 vs. 2022	**0.00**
	2017 vs. 2023	**0.00**
	2022 vs. 2023	**0.00**
Graminoid	2009 vs 2022	**0.01**
	2013 vs. 2017	**0.00**
	2013 vs. 2022	**0.00**
	2014 vs. 2017	**0.00**
	2014 vs. 2022	**0.00**
	2017 vs. 2022	**0.01**
Shrub	2009 vs. 2022	**0.01**
	2013 vs. 2017	**0.00**
	2013 vs. 2022	**0.00**
	2014 vs. 2017	**0.00**
	2014 vs. 2022	**0.00**
	2017 vs. 2022	**0.01**
Tree	2009 vs. 2013	**0.00**
	2009 vs. 2014	**0.00**
	2013 vs. 2014	**0.01**
	2014 vs. 2023	0.05



Monument Canyon Research Natural Area

Treatments: 2007 (thin), 2012 (prescribed burn)
U of A: Nested quadrat method. n = 16.
Pueblo/Four Winds: Point-intercept transect method. n = 43.

San Juan Mesa (treated)

U of A: pre-treatment. Monitored 2000, 2004-2006, 2023.
Five 1mx1m quadrats in a 10mx10m plot.

Pueblo/Four Winds: post-treatment. Monitored 2009, 2013, 2014, 2017, 2022, 2023.
50m x 20m transects.

Cat Mesa (control)

U of A: pre-treatment. Monitored 2000, 2004-2006, 2023.
Five 1mx1m quadrats in a 10mx10m plot.

Pueblo/Four Winds: post-treatment. Monitored 2018, 2022, 2023.
25x20m plots.

Figure 3.1 The two datasets, and the years monitored with each method & site. The green highlighted years indicate consistent monitoring between sites for the UA dataset, and the blue highlighted years show when P4WFEP monitored across both sites.

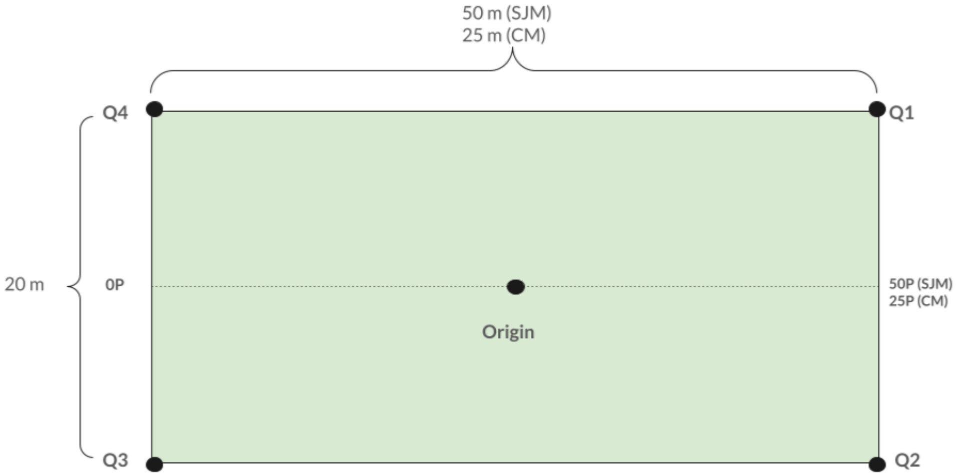


Figure 3.2 P4WFEP point-intercept transect plot layout. SJM = San Juan Mesa, 50m long transects; CM = Cat Mesa, 25m long transects. Shrub density was collected within 5m along the outside of each transect line, and species observed but not intercepted is collected within 10m along the inside and outside of each line.

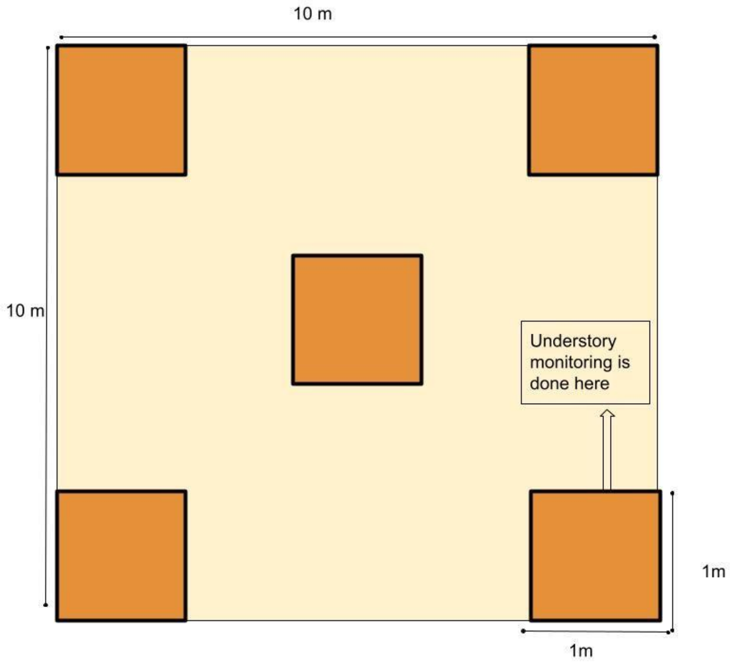


Figure 3.3 UA's 10m x 10m plot. There are five 1m x 1m quadrats in each corner and one in the center of the plot. Understory sampling is in each of the quadrats.

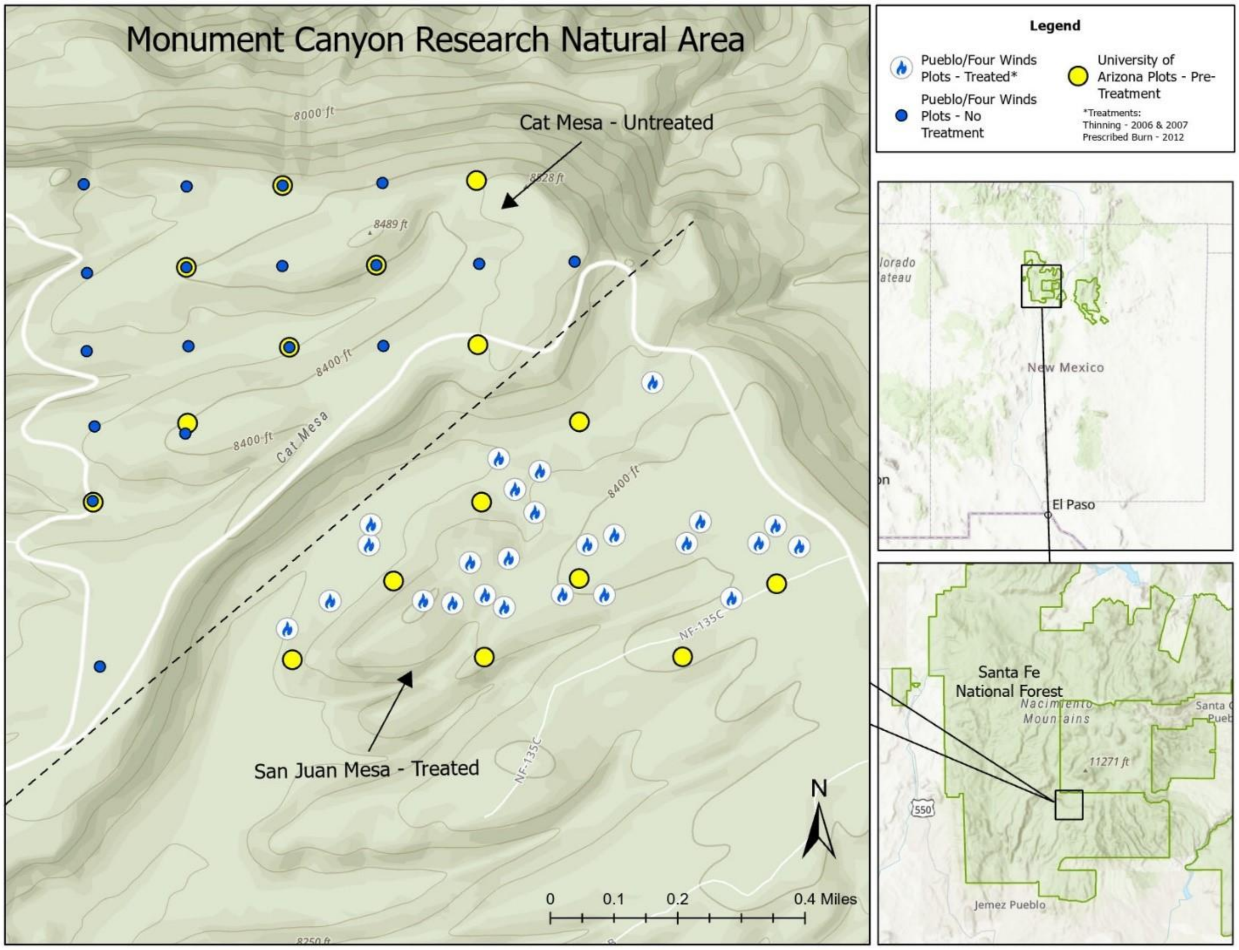


Figure 3.4 Monument Canyon RNA, including regional locator maps. Yellow dots represent the UA plot locations, blue dots represent P4WFEP plots in Cat Mesa, blue fire symbols represent the thinned & burned P4WFEP plots in San Juan Mesa.

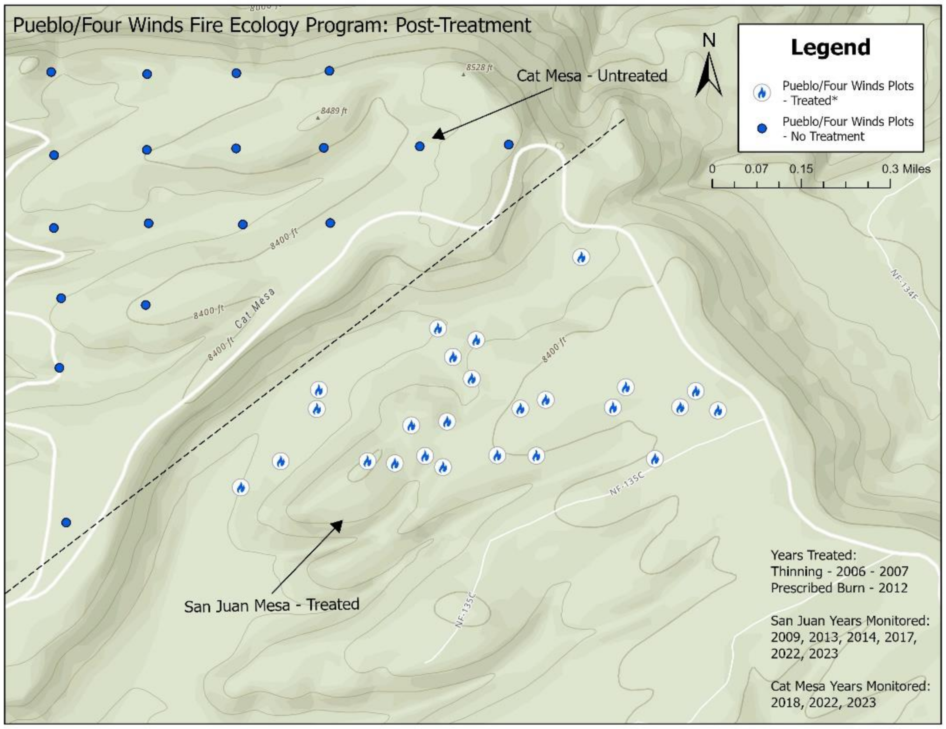


Figure 3.5 P4WFEP plots on Monument Canyon RNA. Cat Mesa (n = 18) was monitored in 2018, 2022, and 2023; San Juan Mesa (n = 25) was monitored in 2009, 2013, 2014, 2017, 2022, and 2023.

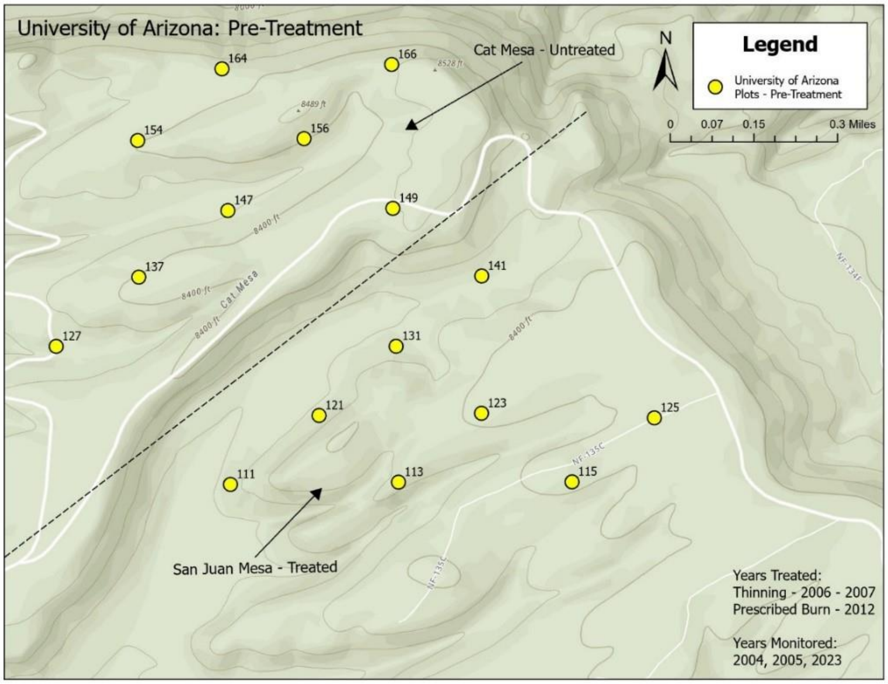
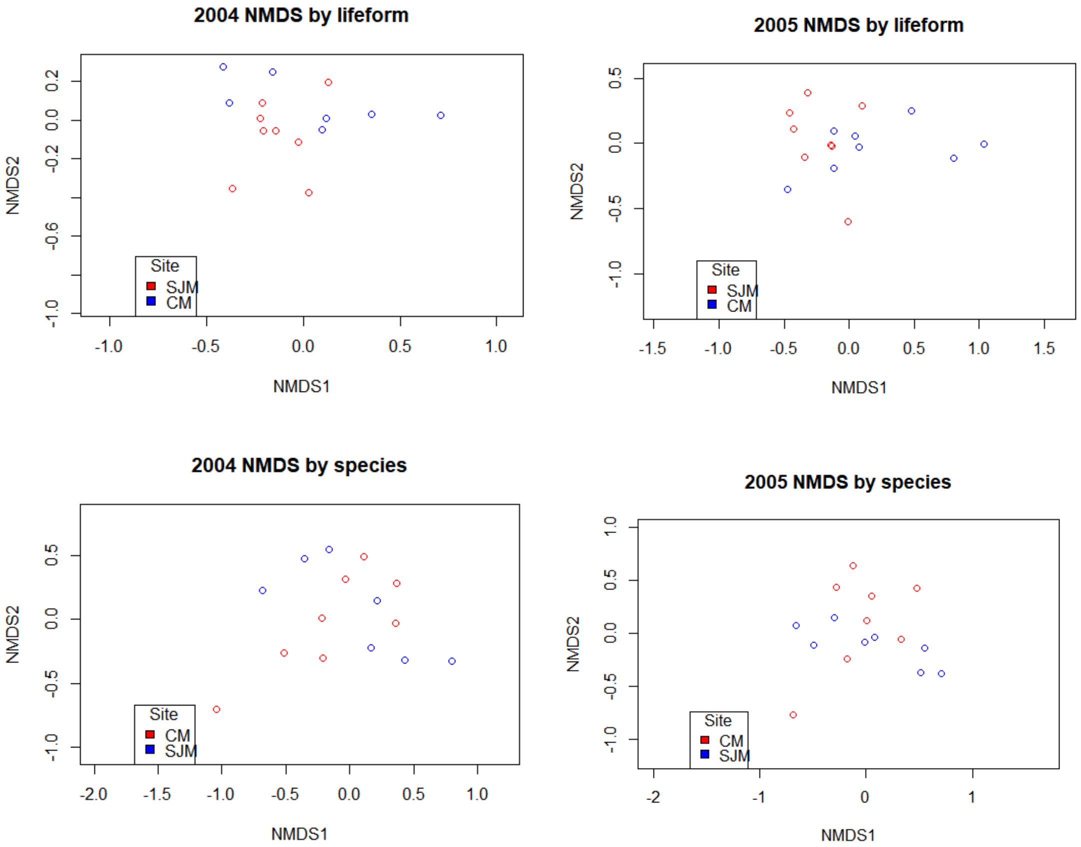


Figure 3.6 UA plots on Monument Canyon RNA. Cat Mesa (n = 8) and San Juan Mesa (n = 8) were both monitored in 2004, 2005, and 2023.



2004 NMDS by lifeform

2005 NMDS by lifeform

2004 NMDS by species

2005 NMDS by species

Figure 3.7 NMDS ordinations by lifeform and species in 2004 and 2005, comparing Cat Mesa and San Juan Mesa. There are no distinct distances or patterns between the two communities, indicating that in 2004 and 2005 the sites were similar.

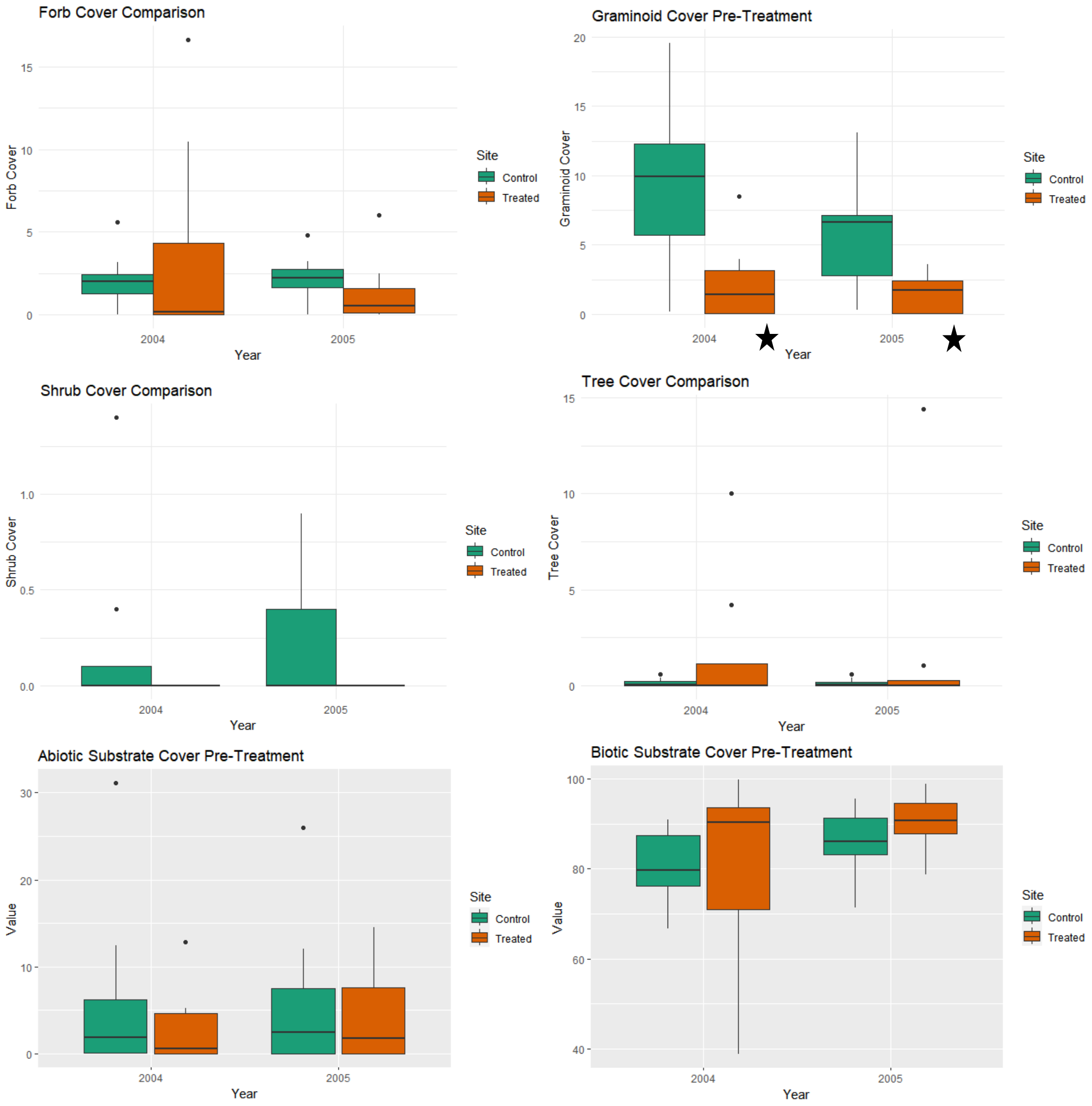


Figure 3.8 Site comparison of forb, graminoid, shrub and tree lifeforms, and abiotic and biotic substrate cover , in 2004 and 2005, showing average cover and variance in Cat Mesa and San Juan Mesa. No significant differences are seen between sites except for graminoids. The line in the boxes show the median, while the bottom and top of the boxes indicate the 25th and 75th percentiles. The lines indicate variance, and minimum to maximum values. All following boxplots have the same percentiles.

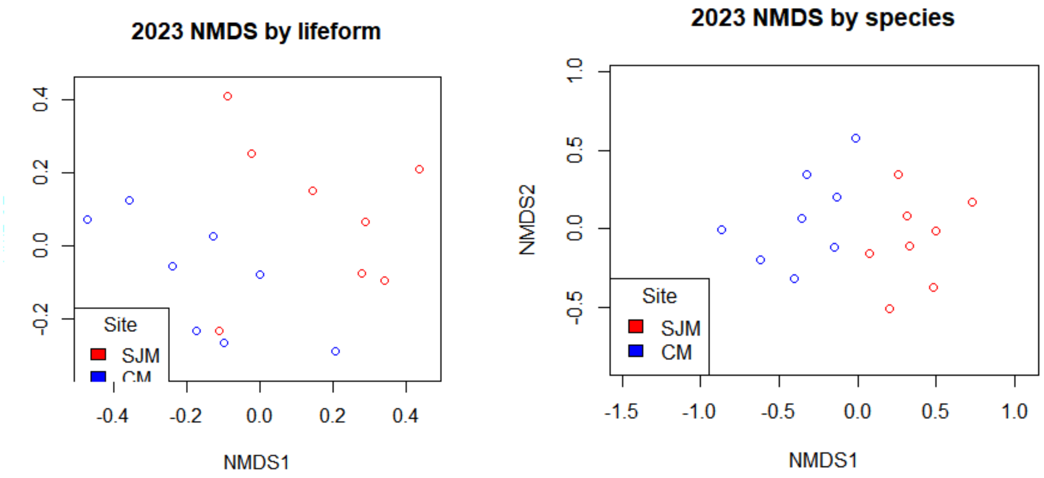
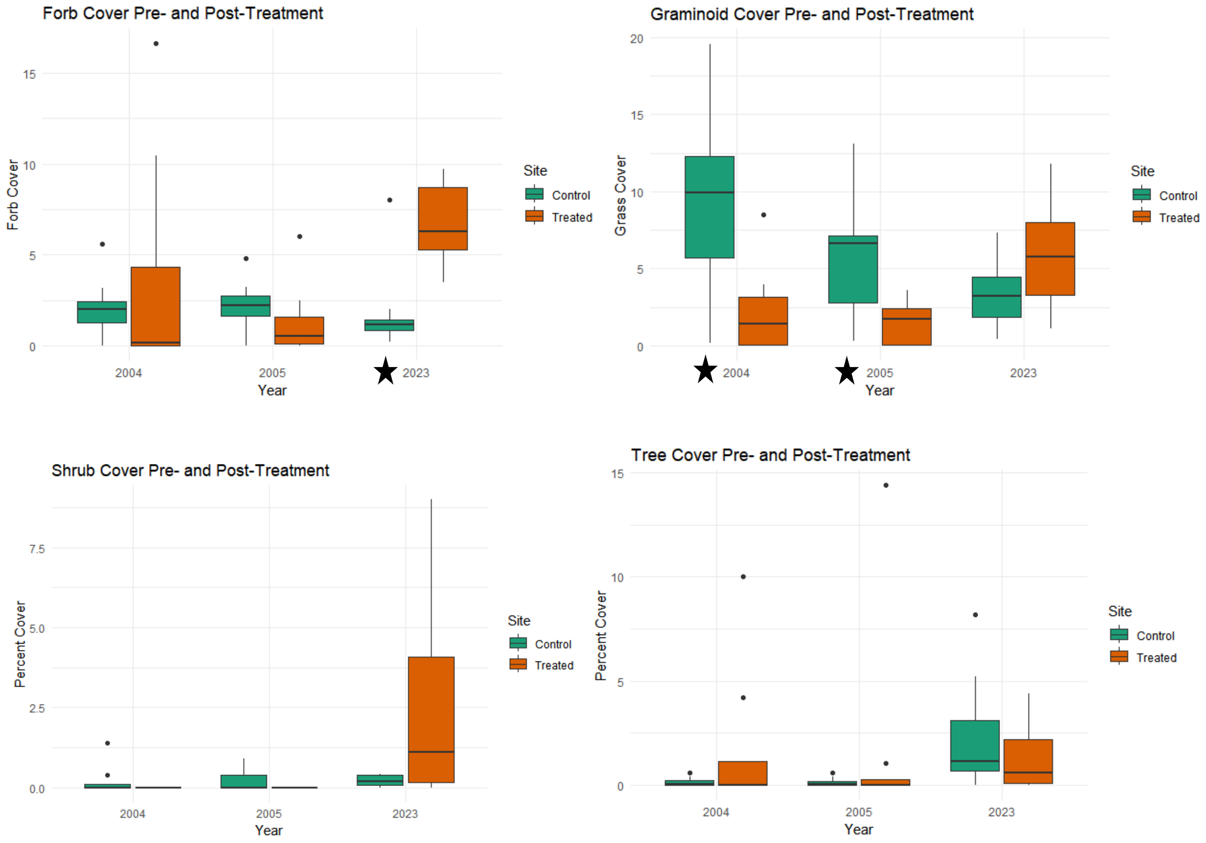


Figure 3.9 NMDS ordinations plotted by lifeform and species in 2023. Data analyzed from the UA dataset. There are distinct patterns and distances between the two sites, indicating that in 2023 there are significant differences between the two sites, and therefore a treatment effect.



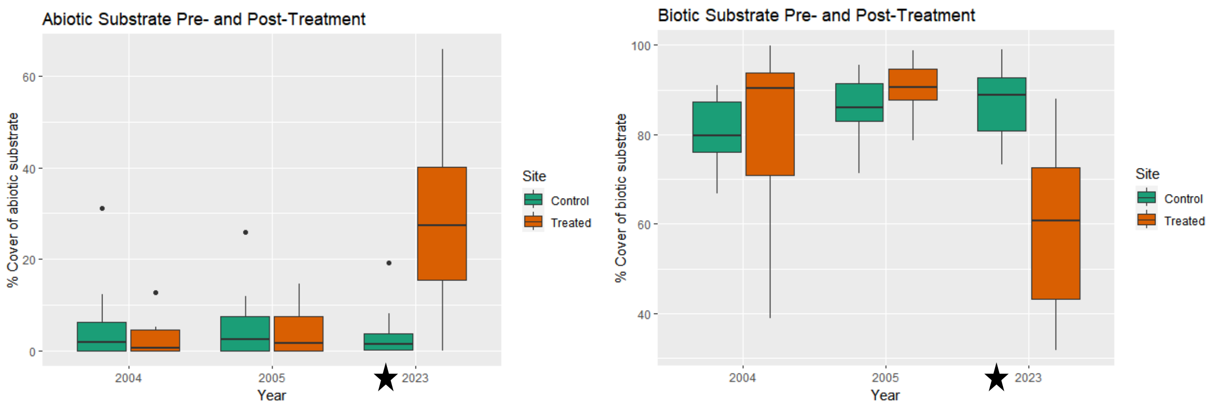


Figure 3.10 Boxplots showing lifeform cover from 2004, 2005, and 2023 to visualize differences in lifeforms. Asterisks indicate significant differences ($p < 0.05$) between the control and treatment within years. All lifeforms show significant differences in the interaction term Site:Year.

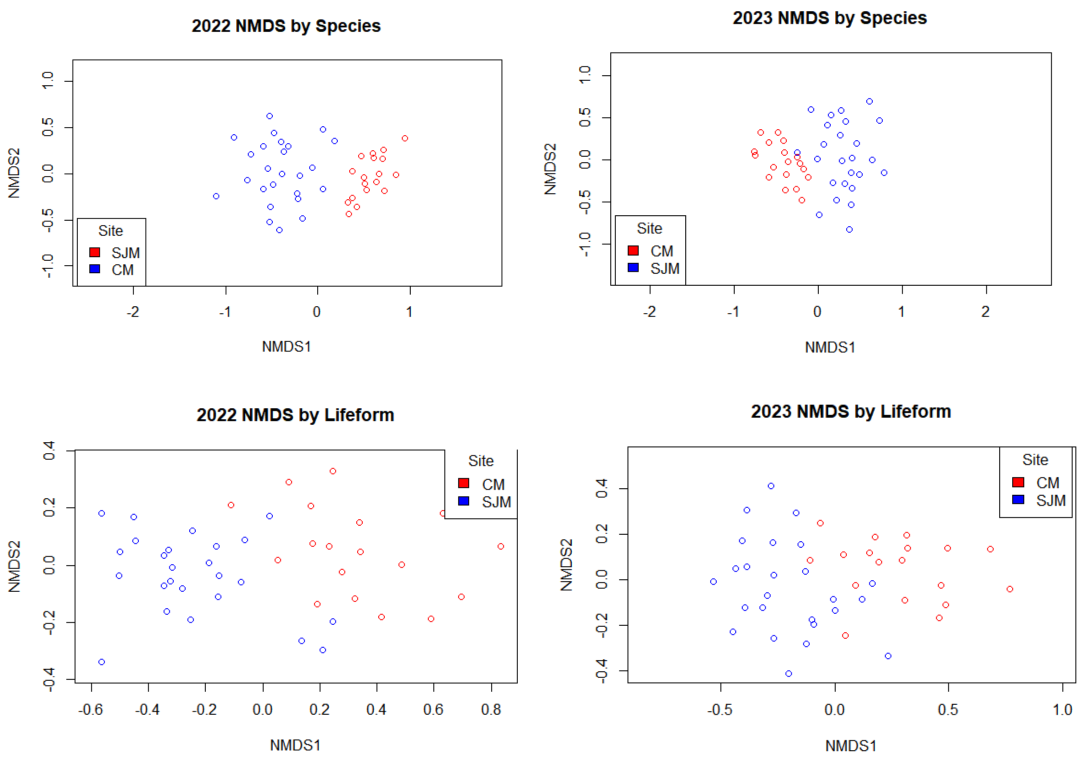


Figure 3.11 NMDS ordinations comparing Cat Mesa and San Juan Mesa by lifeform and species in 2022 and 2023. Data analyzed from the P4WFEP dataset.

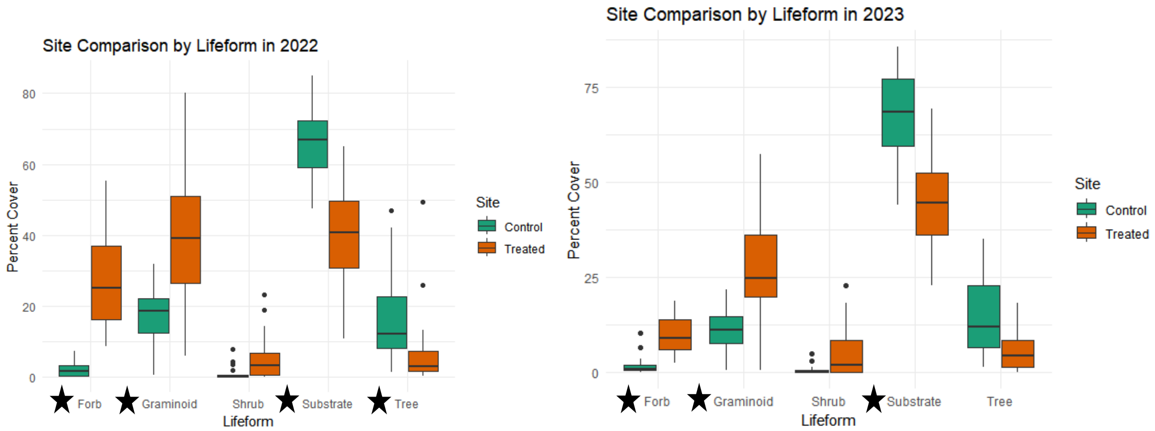


Figure 3.12 Lifeform cover comparisons between the control and treated sites in 2022 and 2023 from the P4WFEP dataset. Forbs, graminoids, and shrubs are higher in the treated site in 2022 and 2023, and substrates and tree cover is higher in the control site.

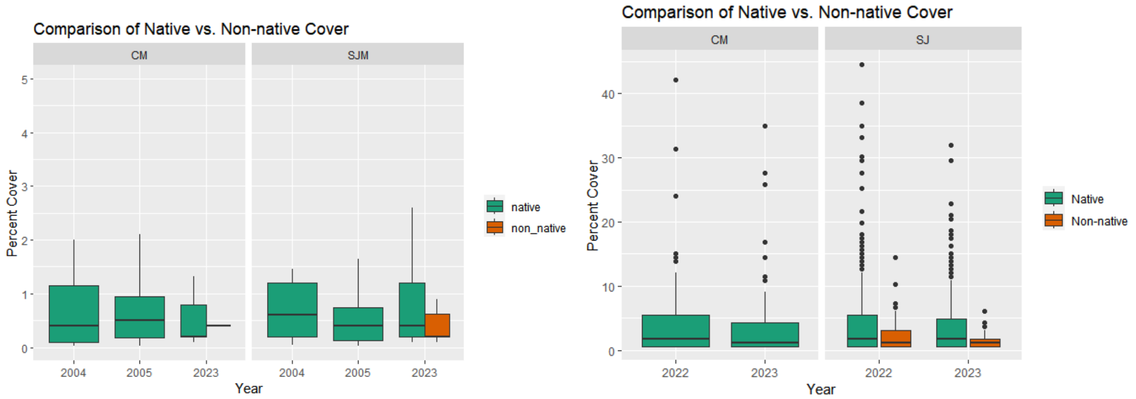


Figure 3.13 The left figure is from the UA dataset, and the right is P4WFEP. The x-axis on the UA dataset comparison is pre- to post-treatment (2004-2005, 2023) and for P4WFEP it was 2022-2023. The UA dataset shows that non-native species were not recorded on either site pre-treatment, and in 2023 there are more non-native species on San Juan Mesa than Cat Mesa. Outliers were removed from this boxplot to maintain scale in the visualization. The P4WFEP dataset shows that in the control site there were no recorded non-native species in 2022 or 2023, and in the treated site they are present.

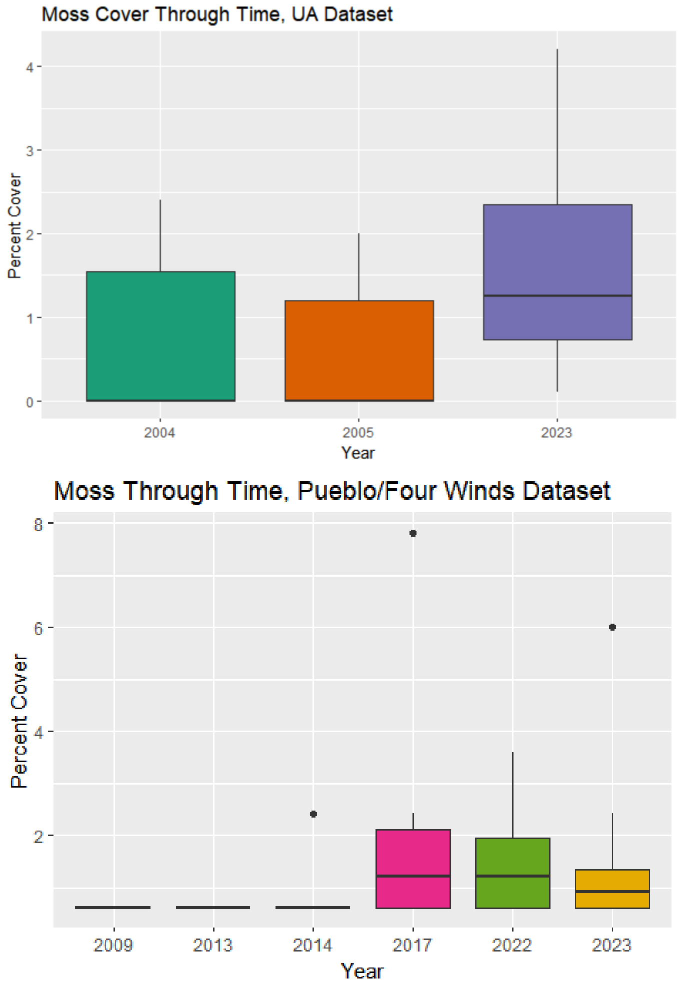


Figure 3.14 Moss cover through time in San Juan Mesa, the treated site, visualizing trends in recovery from the UA dataset and P4WFEP dataset. Moss cover increased 11 years after treatment in the UA dataset, and from 2017-2023 there was also a significant increase in moss cover according to the P4WFEP dataset. This indicates post-treatment, moss cover increases through time.

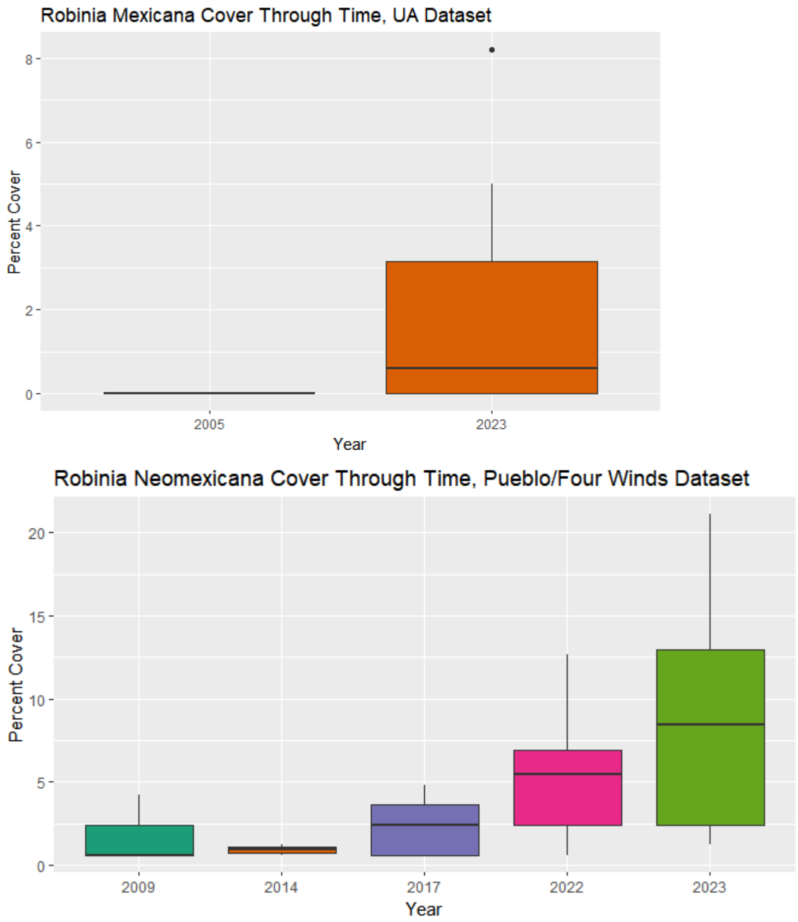
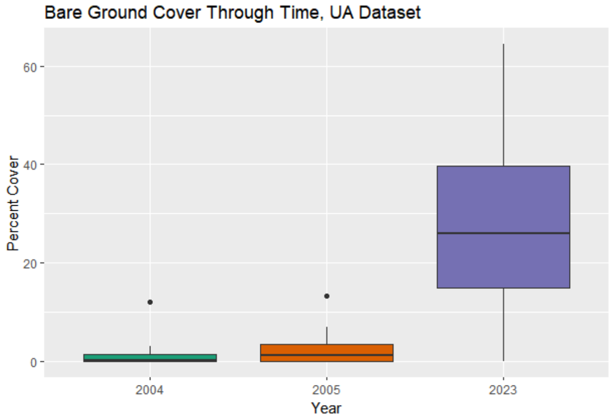


Figure 3.15 Percent cover of Robinia neomexicana in the treated site, San Juan Mesa, through time from both the UA and P4WFEP datasets. In the UA boxplot, Robinia was not recorded on San Juan Mesa in 2004, had a mean of about 0% in 2005, and in 11 years after treatment in 2023 it increased, specifically in variance. The variance in 2023 shows a plot with over 8% cover. In the P4WFEP boxplot, there is an immediate decrease after the burn and a gradual increase through time, indicating positive fire response.



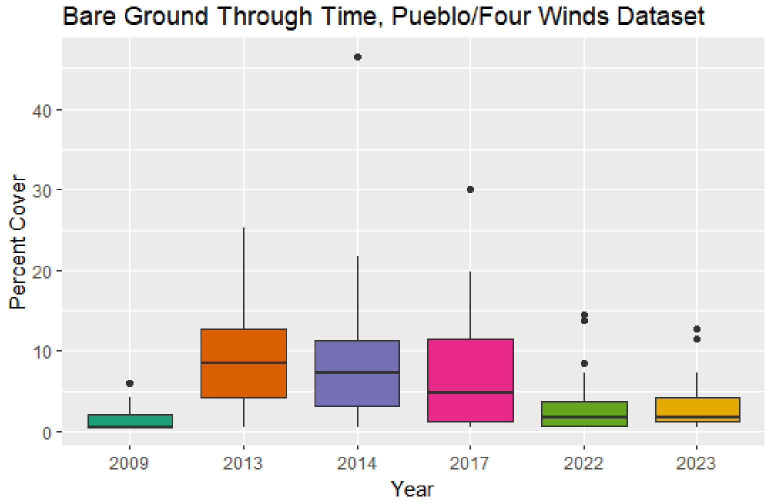
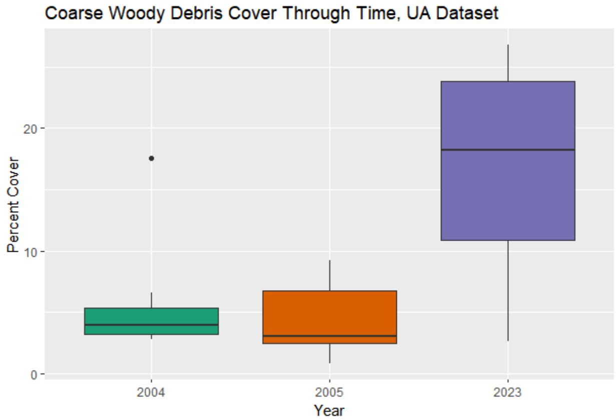


Figure 3.16 Percent cover of bare ground cover from both UA and P4WFEP datasets. Bare ground pre-treatment in 2004 and 2005 (UA) show much lower percent cover averages than in 2023, indicating higher erosion levels after treatment. In the P4WFEP dataset, there is an immediate increase in bare ground after treatment in 2013-2014, and a gradual decrease through time, indicating that long-term treatment effects have a decrease bare ground cover.



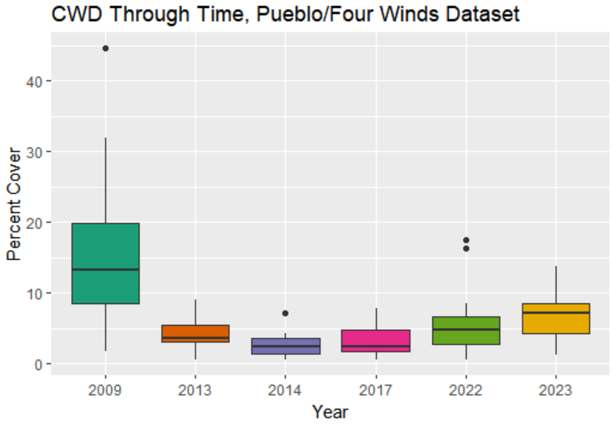


Figure 3.17 Coarse woody debris percent cover on San Juan Mesa from both UA and P4WFEP datasets. CWD shows an increase in 2023, 11 years after treatment, from the UA dataset, and in

the P4WFEP dataset there was higher CWD in post-thin/pre-burn year 2009, a general decrease immediately after the burn, and an increase through time.

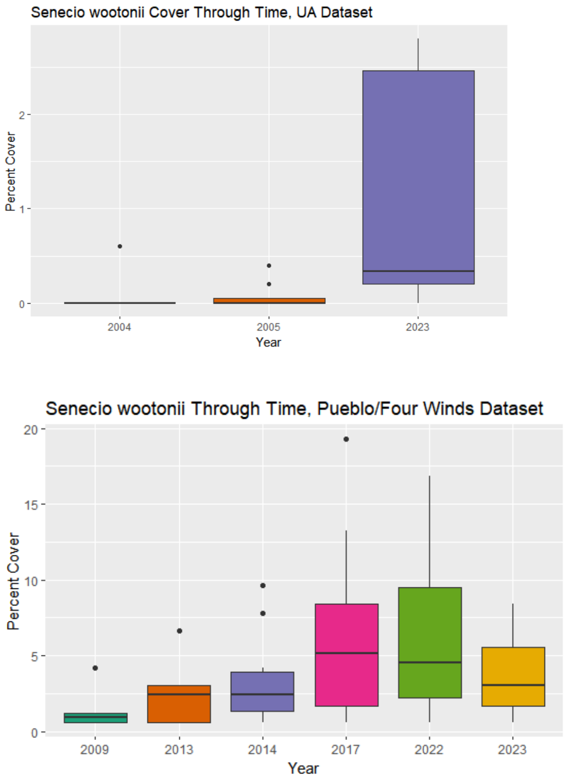


Figure 3.18 Senecio wootonii cover through time in San Juan Mesa from the P4WFEP dataset. Cover increases through time after the treatments.

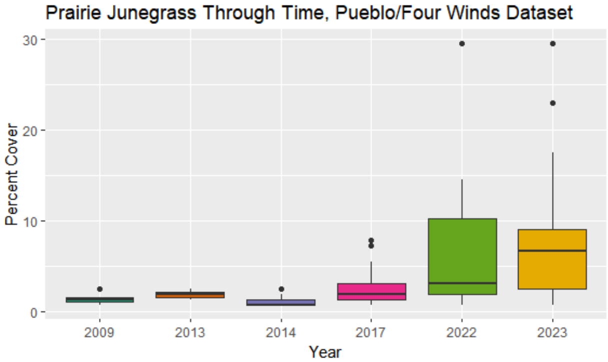


Figure 3.19 Koeleria macrantha, prairie junegrass, cover through time in San Juan Mesa from the P4WFEP dataset. Average cover and variance increase through time after the treatment.

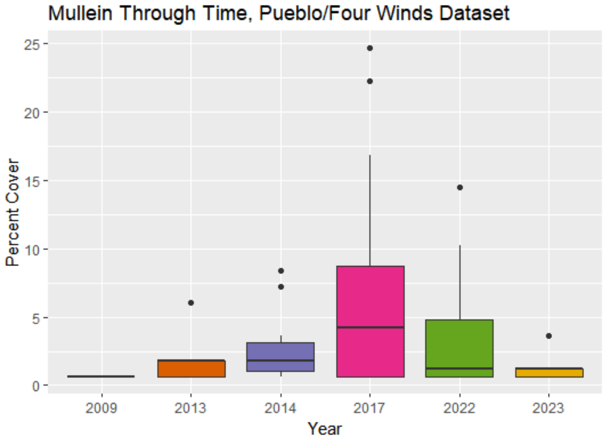


Figure 3.20 Verbascum thapsus, or common mullein, cover through time on San Juan Mesa from the P4WFEP dataset. Mullein peaks in 2017, five years after the burn, and gradually decreases in 2022 and 2023.

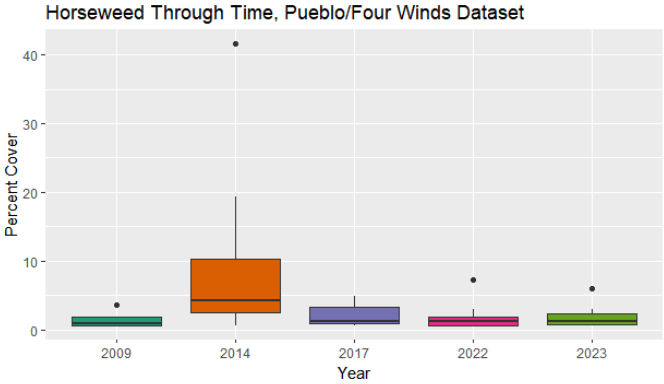
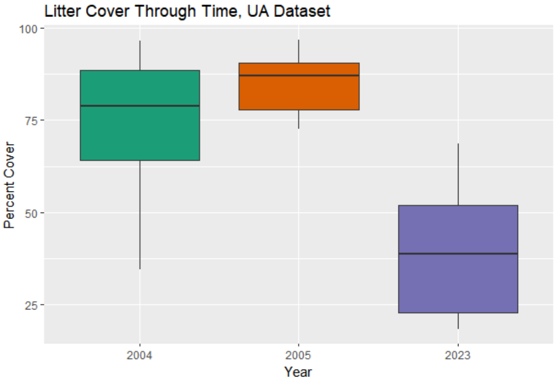


Figure 3.21 Conyza canadensis, or horseweed, through time on San Juan Mesa from the P4WFEP dataset. Horseweed peaks in 2014, two years after the burn, and decreases over time.



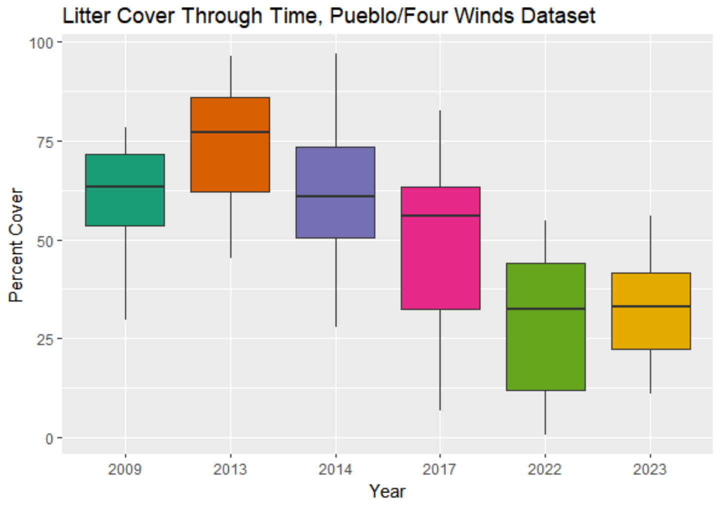


Figure 3.22 Percent cover of litter in San Juan Mesa from both datasets, UA and P4WFEP. Litter decreases in 2023, 11 years after treatment, from pre-treatment levels in 2004 and 2005 (UA). We also see a gradual decrease in litter through time from the P4WFEP dataset, indicating positive treatment effects.

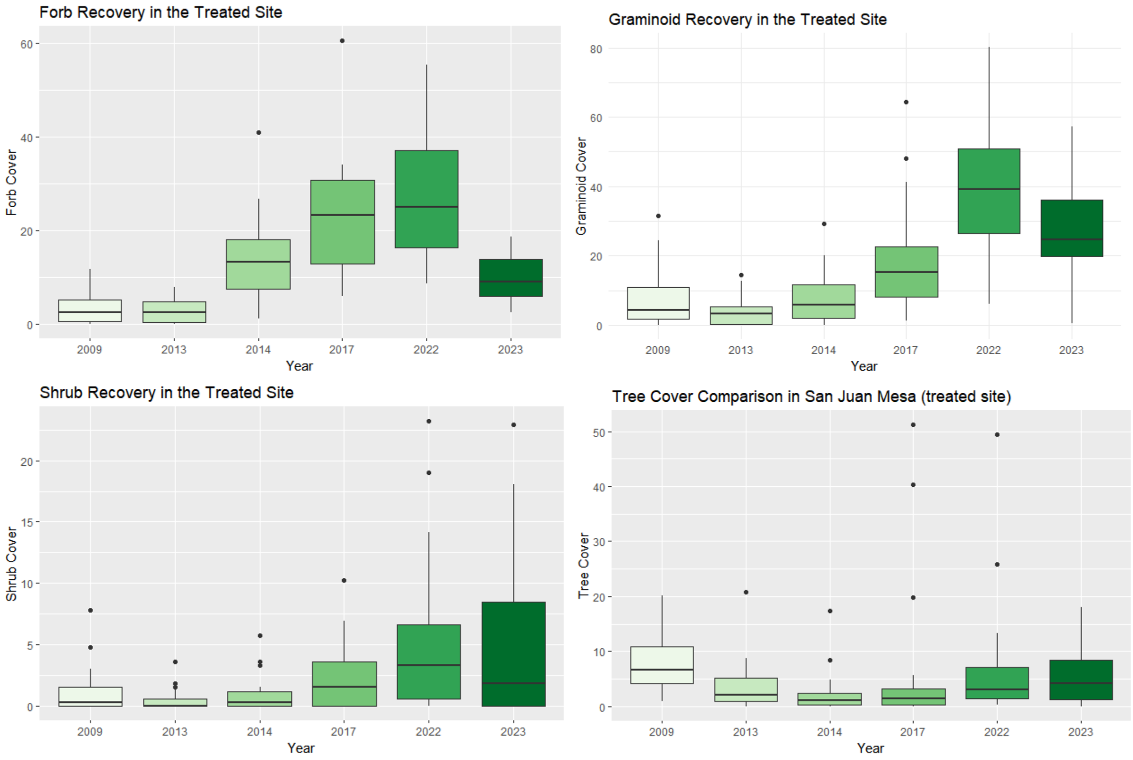


Figure 3.23 Boxplots showing the recovery trajectory of forbs, graminoids, shrubs, and trees through 2009-2023 in the P4WFEP dataset.

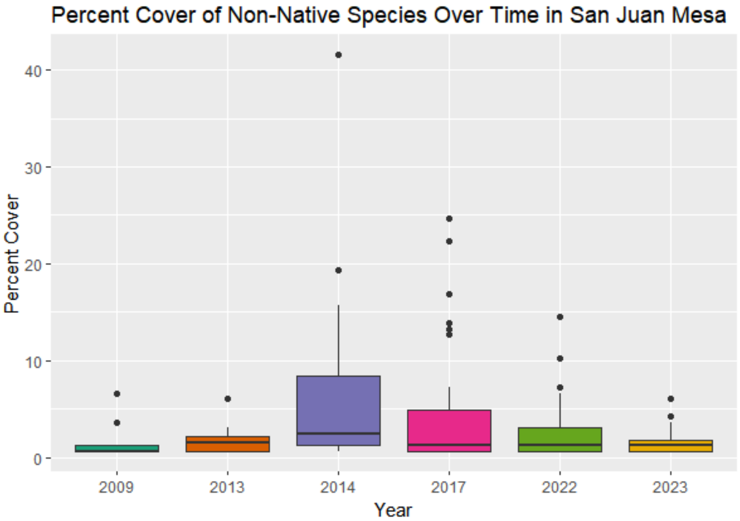


Figure 3.24 Non-native species cover through time in San Juan Mesa, the treated site. There is a peak in non-native cover in 2014 and 2017, 2-5 years after treatment, and a gradual decrease through time. This indicates that treatment affects non-native species in a short-term window after treatment but that long-term effects are low.

Literature Cited

Agee, J. K., & Skinner, C. N. (2005). Basic principles of forest fuel reduction treatments. *Forest Ecology and Management*, *211*(1–2), 83–96. https://doi.org/10.1016/j.foreco.2005.01.034

Anderson, M. J., Ellingsen, K. E., & McArdle, B. H. (2006). Multivariate dispersion as a measure of beta diversity. *Ecology Letters*, *9*(6), 683–693. https://doi.org/10.1111/j.1461-0248.2006.00926.x

Carter, A. J., & Feeney, W. E. (2012). Taking a Comparative Approach: Analysing Personality as a Multivariate Behavioural Response across Species. *PLOS ONE*, *7*(7), e42440. https://doi.org/10.1371/journal.pone.0042440

Collins, B. M., Moghaddas, J. J., & Stephens, S. L. (2007). Initial changes in forest structure and understory plant communities following fuel reduction activities in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, *239*(1), 102–111. https://doi.org/10.1016/j.foreco.2006.11.013

Coop, J. D. (2023). Postfire futures in southwestern forests: Climate and landscape influences on trajectories of recovery and conversion. *Ecological Applications*, *33*(1), e2725. https://doi.org/10.1002/eap.2725

De Cáceres, M., Legendre, P., Wiser, S. K., & Brotons, L. (2012). Using species combinations in indicator value analyses. *Methods in Ecology and Evolution*, *3*(6), 973–982. https://doi.org/10.1111/j.2041-210X.2012.00246.x

Dieterich, J. H. (1983). Fire history of southwestern mixed conifer: A case study. *Forest Ecology and Management*, *6*(1), 13–31. https://doi.org/10.1016/0378-1127(83)90003-8

Dodge, J. M., Strand, E. K., Hudak, A. T., Bright, B. C., Hammond, D. H., & Newingham, B. A. (2019). Short- and long-term effects of ponderosa pine fuel treatments intersected by the Egley Fire Complex, Oregon, USA. *Fire Ecology*, *15*(1), 40. https://doi.org/10.1186/s42408-019-0055-7

Drury, S. (2020). Fuel Continuity. In S. L. Manzello (Ed.), *Encyclopedia of Wildfires and Wildland-Urban Interface (WUI) Fires* (pp. 1–3). Springer International Publishing. https://doi.org/10.1007/978-3-319-51727-8_239-1

Faist, A. M., Stone, H., & Tripp, E. A. (2015). Impacts of Mastication: Soil Seed Bank Responses to a Forest Thinning Treatment in Three Colorado (USA) Conifer Forest Types. *Forests*, *6*(9), 3060–3074. https://doi.org/10.3390/f6093060

Falk, D. A., van Mantgem, P. J., Keeley, J. E., Gregg, R. M., Guiterman, C. H., Tepley, A. J., JN Young, D., & Marshall, L. A. (2022). Mechanisms of forest resilience. *Forest Ecology and Management*, *512*, 120129. https://doi.org/10.1016/j.foreco.2022.120129

Fitzgerald, S. A. (2005). Fire ecology of ponderosa pine and the rebuilding of fire-resilient ponderosa pine ecosystems. *In: Ritchie, Martin W.; Maguire, Douglas A.; Youngblood, Andrew, Tech. Coordinators. Proceedings of the Symposium on Ponderosa Pine: Issues, Trends, and*

Management, 2004 October 18-21, Klamath Falls, OR. Gen. Tech. Rep PSW-GTR-198. Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture: 197-225, 198. https://www.fs.usda.gov/research/treesearch/27269

Fornwalt, P. J., Rocca, M. E., Battaglia, M. A., Rhoades, C. C., & Ryan, M. G. (2017). Mulching fuels treatments promote understory plant communities in three Colorado, USA, coniferous forest types. *Forest Ecology and Management*, *385*, 214–224. https://doi.org/10.1016/j.foreco.2016.11.047

Goff, F., Gardner, J. N., Reneau, S. L., & Goff, C. J. (2005). *Geologic map of the Redondo Peak quadrangle, Sandoval County,New Mexico*. New Mexico Bureau of Geology and Mineral Resources. https://doi.org/10.58799/OF-GM-111

Goodwin, M. J., North, M. P., Zald, H. S. J., & Hurteau, M. D. (2018). The 15-year post-treatment response of a mixed-conifer understory plant community to thinning and burning treatments. *Forest Ecology and Management*, *429*, 617–624. https://doi.org/10.1016/j.foreco.2018.07.058

Grover, H. S., Bowker, M. A., Fulé, P. Z., Doherty, K. D., Sieg, C. H., & Antoninka, A. J. (2020). Post-wildfire moss colonisation and soil functional enhancement in forests of the southwestern USA. *International Journal of Wildland Fire*, *29*(6), 530. https://doi.org/10.1071/WF19106

Guiterman, C. H., Gregg, R. M., Marshall, L. A. E., Beckmann, J. J., van Mantgem, P. J., Falk, D. A., Keeley, J. E., Caprio, A. C., Coop, J. D., Fornwalt, P. J., Haffey, C., Hagmann, R. K., Jackson, S. T., Lynch, A. M., Margolis, E. Q., Marks, C., Meyer, M. D., Safford, H., Syphard, A. D., … Stevens, J. T. (2022). Vegetation type conversion in the US Southwest: Frontline observations and management responses. *Fire Ecology*, *18*(1), 6. https://doi.org/10.1186/s42408-022-00131-w

Hessburg, P. F., Churchill, D. J., Larson, A. J., Haugo, R. D., Miller, C., Spies, T. A., North, M. P., Povak, N. A., Belote, R. T., Singleton, P. H., Gaines, W. L., Keane, R. E., Aplet, G. H., Stephens, S. L., Morgan, P., Bisson, P. A., Rieman, B. E., Salter, R. B., & Reeves, G. H. (2015). Restoring fire-prone Inland Pacific landscapes: Seven core principles. *Landscape Ecology*, *30*(10), 1805–1835. https://doi.org/10.1007/s10980-015-0218-0

Https://arizona-nau.primo.exlibrisgroup.com. (n.d.). Retrieved September 25, 2023, from https://arizona-nau.primo.exlibrisgroup.com

Hurteau, M. D., Bradford, J. B., Fulé, P. Z., Taylor, A. H., & Martin, K. L. (2014). Climate change, fire management, and ecological services in the southwestern US. *Forest Ecology and Management*, *327*, 280–289. https://doi.org/10.1016/j.foreco.2013.08.007

Jang, W., Crotteau, J. S., Ortega, Y. K., Hood, S. M., Keyes, C. R., Pearson, D. E., Lutes, D. C., & Sala, A. (2021). Native and non-native understory vegetation responses to restoration treatments in a dry conifer forest over 23 years. *Forest Ecology and Management*, *481*, 118684. https://doi.org/10.1016/j.foreco.2020.118684

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M. C., Meentemeyer, R. K., Metz, M. R., Perry, G. L., Schoennagel, T., & Turner, M. G. (2016).

Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment*, *14*(7), 369–378. https://doi.org/10.1002/fee.1311

Kelley, S., Kempter, K. A., Goff, F., Rampey, M., Osburn, G. R., & Ferguson, C. A. (2003). *Geologic map of the Jemez Springs 7.5-minute quadrangle, Sandoval County, New Mexico*. New Mexico Bureau of Geology and Mineral Resources. https://doi.org/10.58799/OF-GM-73

Kerns, B., & Day, M. (2018). Prescribed fire regimes subtly alter ponderosa pine forest plant community structure. *ECOSPHERE*, *9*(12). https://doi.org/10.1002/ecs2.2529

Korb, J. E., Daniels, M. L., Laughlin, D. C., & Fulé, P. Z. (2007). UNDERSTORY COMMUNITIES OF WARM-DRY, MIXED-CONIFER FORESTS IN SOUTHWESTERN COLORADO. *The Southwestern Naturalist*, *52*(4), 493–503. https://doi.org/10.1894/0038-4909(2007)52[493:UCOWMF]2.0.CO;2

Korb, J. E., Stoddard, M. T., & Huffman, D. W. (2020). Effectiveness of Restoration Treatments for Reducing Fuels and Increasing Understory Diversity in Shrubby Mixed-Conifer Forests of the Southern Rocky Mountains, USA. *Forests*, *11*(5), 508. https://doi.org/10.3390/f11050508

Marshall, L. A., Falk, D. A., & Mcdowell, N. G. (2019). NITROGEN CAN LIMIT OVERSTORY TREE GROWTH FOLLOWING EXTREME STAND DENSITY INCREASE IN A PONDEROSA PINE FOREST. *Tree-Ring Research*, *75*(1), 49. https://doi.org/10.3959/1536-1098-75.1.49

McCune, B., & Mefford, M. J. (1999). *PC-ORD: Multivariate Analysis of Ecological Data ; Version 4 for Windows ; [user' S Guide]*. MjM Software Design.

Minchin, P. R. (1987). An evaluation of the relative robustness of techniques for ecological ordination. *Vegetatio*, *69*(1), 89–107. https://doi.org/10.1007/BF00038690

Moore, M. M., Casey, C. A., Bakker, J. D., Springer, J. D., Fulé, P. Z., Covington, W. W., & Laughlin, D. C. (2006). Herbaceous Vegetation Responses (1992–2004) to Restoration Treatments in a Ponderosa Pine Forest. *Rangeland Ecology and Management*, *59*(2), 135–144. https://doi.org/10.2111/05-051R2.1

Moore, M. M., Wallace Covington, W., & Fulé, P. Z. (1999). Reference Conditions and Ecological Restoration: A Southwestern Ponderosa Pine Perspective. *Ecological Applications*, *9*(4), 1266–1277. https://doi.org/10.1890/1051-0761(1999)009[1266:RCAERA]2.0.CO;2

Mueller, S. E., Thode, A. E., Margolis, E. Q., Yocom, L. L., Young, J. D., & Iniguez, J. M. (2020). Climate relationships with increasing wildfire in the southwestern US from 1984 to 2015. *Forest Ecology and Management*, *460*, 117861. https://doi.org/10.1016/j.foreco.2019.117861

Multiscale perspectives of fire, climate and humans in western North America and the Jemez Mountains, USA. (n.d.). https://doi.org/10.1098/rstb.2015.0168

Munson, S. M., Muldavin, E. H., Belnap, J., Peters, D. P. C., Anderson, J. P., Reiser, M. H., Gallo, K., Melgoza-Castillo, A., Herrick, J. E., & Christiansen, T. A. (2013). Regional signatures of plant response to drought and elevated temperature across a desert ecosystem. *Ecology*, *94*(9), 2030–2041. https://doi.org/10.1890/12-1586.1

Native American depopulation, reforestation, and fire regimes in the Southwest United States, 1492–1900 CE. (n.d.). https://doi.org/10.1073/pnas.1521744113

Neill, A. R., & Puettmann, K. J. (2013). Managing for adaptive capacity: Thinning improves food availability for wildlife and insect pollinators under climate change conditions. *Canadian Journal of Forest Research*, *43*(5), 428–440. https://doi.org/10.1139/cjfr-2012-0345

Odland, M. C., Goodwin, M. J., Smithers, B., Hurteau, M. D., & North, M. P. (2021). Plant community response to thinning and repeated fire in Sierra Nevada mixed-conifer forest understories. *Forest Ecology and Management*, *495*, 119361. https://doi.org/10.1016/j.foreco.2021.119361

Paper with similar stats methods—Mkd269@nau.edu—NAU Mail. (n.d.). Retrieved October 18, 2023, from https://mail.google.com/mail/u/0/#inbox/FMfcgzGwHLfLqbzrlzmpJDnkvfFCBjvg?projector=1&messagePartId=0.1

Parks, S. A., Dobrowski, S. Z., Shaw, J. D., & Miller, C. (2019). Living on the edge: Trailing edge forests at risk of fire-facilitated conversion to non-forest. *Ecosphere*, *10*(3), e02651. https://doi.org/10.1002/ecs2.2651

Peterson, G., Allen, C. R., & Holling, C. S. (n.d.). *Ecological Resilience, Biodiversity, and Scale*. 13.

Pyron, M. (2010). *Characterizing Communities | Learn Science at Scitable*. Nature Education Knowledge. https://www.nature.com/scitable/knowledge/library/characterizing-communities-13241173/

Relative species abundance. (2022). In *Wikipedia*. https://en.wikipedia.org/w/index.php?title=Relative_species_abundance&oldid=1092830985#cite_note-Hubbell01-1

Reynolds, R. T., Sanchez Meador, A. J., Youtz, J. A., Nicolet, T., Matonis, M. S., Jackson, P. L., DeLorenzo, D. G., & Graves, A. D. (2013). *Restoring composition and structure in Southwestern frequent-fire forests: A science-based framework for improving ecosystem resiliency* (RMRS-GTR-310; p. RMRS-GTR-310). U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. https://doi.org/10.2737/RMRS-GTR-310

Rhoades, C. C., & Fornwalt, P. J. (2015). Pile burning creates a fifty-year legacy of openings in regenerating lodgepole pine forests in Colorado. *Forest Ecology and Management*, *336*, 203–209. https://doi.org/10.1016/j.foreco.2014.10.011

Robinia neomexicana. (n.d.). Retrieved November 14, 2023, from https://www.fs.usda.gov/database/feis/plants/tree/robneo/all.html

Rockweit, J. T., Franklin, A. B., & Carlson, P. C. (2017). Differential impacts of wildfire on the population dynamics of an old-forest species. *Ecology*, *98*(6), 1574–1582. https://doi.org/10.1002/ecy.1805

Roos, C. I., Swetnam, T. W., Ferguson, T. J., Liebmann, M. J., Loehman, R. A., Welch, J. R., Margolis, E. Q., Guiterman, C. H., Hockaday, W. C., Aiuvalasit, M. J., Battillo, J., Farella, J., & Kiahtipes, C. A. (2021). Native American fire management at an ancient wildland–urban interface in the Southwest United States. *Proceedings of the National Academy of Sciences*, *118*(4), e2018733118. https://doi.org/10.1073/pnas.2018733118

Sample, M., Thode, A. E., Peterson, C., Gallagher, M. R., Flatley, W., Friggens, M., Evans, A., Loehman, R., Hedwall, S., Brandt, L., Janowiak, M., & Swanston, C. (2022). Adaptation Strategies and Approaches for Managing Fire in a Changing Climate. *Climate*, *10*(4), Article 4. https://doi.org/10.3390/cli10040058

Singleton, M. P., Thode, A. E., Sánchez Meador, A. J., & Iniguez, J. M. (2019). Increasing trends in high-severity fire in the southwestern USA from 1984 to 2015. *Forest Ecology and Management*, *433*, 709–719. https://doi.org/10.1016/j.foreco.2018.11.039

Springer, J. D., Huffman, D. W., Stoddard, M. T., Sánchez Meador, A. J., & Waltz, A. E. M. (2018). Plant community dynamics following hazardous fuel treatments and mega-wildfire in a warm-dry mixed-conifer forest of the USA. *Forest Ecology and Management*, *429*, 278–286. https://doi.org/10.1016/j.foreco.2018.06.022

Stephens, S. L., Collins, B. M., Fettig, C. J., Finney, M. A., Hoffman, C. M., Knapp, E. E., North, M. P., Safford, H., & Wayman, R. B. (2018). Drought, Tree Mortality, and Wildfire in Forests Adapted to Frequent Fire. *BioScience*, *68*(2), 77–88. https://doi.org/10.1093/biosci/bix146

Strahan, R. T., Stoddard, M. T., Springer, J. D., & Huffman, D. W. (2015). Increasing weight of evidence that thinning and burning treatments help restore understory plant communities in ponderosa pine forests. *Forest Ecology and Management*, *353*, 208–220. https://doi.org/10.1016/j.foreco.2015.05.040

Swetnam, T. W., Farella, J., Roos, C. I., Liebmann, M. J., Falk, D. A., & Allen, C. D. (2016). Multiscale perspectives of fire, climate and humans in western North America and the Jemez Mountains, USA. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *371*(1696), 20150168. https://doi.org/10.1098/rstb.2015.0168

Thomas, Z., & Waring, K. M. (2015). Enhancing resiliency and restoring ecological attributes in second-growth ponderosa pine stands in northern New Mexico, USA. *Forest Science*, *61*(1), 93–104. CAB Abstracts with Full Text.

Verbascum thapsus. (n.d.). Retrieved November 14, 2023, from https://www.fs.usda.gov/database/feis/plants/forb/vertha/all.html#FIRE%20ECOLOGY

Wienk, C. L., Sieg, C. H., & McPherson, G. R. (2004). Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. *Forest Ecology and Management*, *192*(2–3), 375–393. https://doi.org/10.1016/j.foreco.2004.02.004

Williams, A. P., Allen, C. D., Macalady, A. K., Griffin, D., Woodhouse, C., Meko, D. M., Swetnam, T. W., Rauscher, S. A., Seager, R., Grissino-Mayer, H. D., Dean, J. S., Cook, E. R., Gangodagamage, C., Cai, M., & McDowell, N. G. (2012). *Temperature as a potent driver of regional forest drought stress and tree mortality*. *3*, 292–297. https://doi.org/10.7916/d8-b9ec-8z87

Wolk, B., & Rocca, M. E. (2009). Thinning and chipping small-diameter ponderosa pine changes understory plant communities on the Colorado Front Range. *Forest Ecology and Management*, *257*(1), 85–95. https://doi.org/10.1016/j.foreco.2008.08.014

Zald, H. S. J., Kerns, B. K., & Day, M. A. (2020). Limited Effects of Long-Term Repeated Season and Interval of Prescribed Burning on Understory Vegetation Compositional Trajectories and Indicator Species in Ponderosa Pine Forests of Northeastern Oregon, USA. *Forests*, *11*(8), Article 8. https://doi.org/10.3390/f11080834

Chapter 4 Management Implications

In a Nutshell: Fuel reduction treatments positively affect Southwest U.S. ecosystems, enhancing biodiversity and resilience in the face of increasing fire severity and prolonged droughts intensified by climate change. Understory vegetation communities, crucial for biodiversity, provide habitat and forage for wildlife and insects, contribute to soil stabilization and carbon sequestration, and hold cultural significance for indigenous communities all over the world. Because understory vegetation is ecologically important, understanding the long-term effects of prescribed burn and thinning treatments on these communities is paramount. This knowledge equips land managers and policymakers to make informed decisions in addressing the complex challenges posed by uncharacteristically severe wildfires and shifting climate patterns in the Southwest.

Body Text:

Climate Change and Southwestern Ecosystems: Increasing fire severity and prolonged droughts, exacerbated by climate change, threaten Southwestern U.S. ecosystems. Uncharacteristically severe wildfires demand effective management strategies, such as fuels reduction treatments, to mitigate the potential impact on biodiversity and vital ecosystem services. Understanding the dynamics of treatments, particularly their long-term effects on understory vegetation, becomes paramount for informed decision-making by land managers and policymakers in the face of a changing climate.

Fuels Reduction Treatments: Prescribed burn and thinning treatments are widely used to reduce the probability of extreme fire behavior. In this study, we find that such treatments can have additional benefits as promising tools in mitigating climate-induced stressors of forest communities. This study explores their long-term impacts on understory vegetation dynamics. Our analyses reveal that pre-treatment data collection and analysis is crucial for establishing baseline comparisons between treated sites. Following treatment, significant increases in lifeform cover, species richness, and substrate composition indicate potentially positive impacts *of* prescribed burn and thinning treatments on understory communities.

Recovery Trajectories: Over a decade after the mastication and prescribed burn treatments, varying responses in forbs, graminoids, shrubs, and trees depict a nuanced recovery trajectory. The persistence of variations in community composition on treated sites highlights the lasting effects of treatments, as graminoids, shrubs, and forb cover and diversity all increased through time. Native species, initially reduced in abundance after treatment, gradually recovered, emphasizing long term ecological dynamics post-treatment. Non-native species also responded positively to treatment, spiking3-5 years after a prescribed burn, but then decreased after 10 years post-treatment, again highlighting the importance of long-term monitoring. The increased abundance of species such as *Robinia neomexicana*, New Mexico locust, after prescribed fire may hint at a potential mechanism for shrub conversion after higher-severity burns.

Implications for Ecological Management: In addition to their primary objectives to modify fuels and mitigate fire behavior, prescribed burn and thinning treatments can foster increased diversity and plant cover, crucial for enhancing biodiversity and ecosystem resilience. Land managers can consider similar treatments to achieve these positive outcomes as well as the primary objectives of fuel reductions. Long-term comparisons between datasets emphasize the importance of sustained monitoring and collaborative efforts between land managers and scientists. The study advocates for increased communication and shared responsibility in managing ecosystems.

Figures:

Figure 4.1 A native swallowtail butterfly (Papilionidae) pollinating a native thistle, Cirsium undulatum. Photo captured on a plot in the treated site, 11 years after treatment. The butterfly, as well as the copious amounts of native vegetation in the background of the photo, indicates a positive ecosystem recovery after treatment, and showcases how the understory provides important ecosystem services.

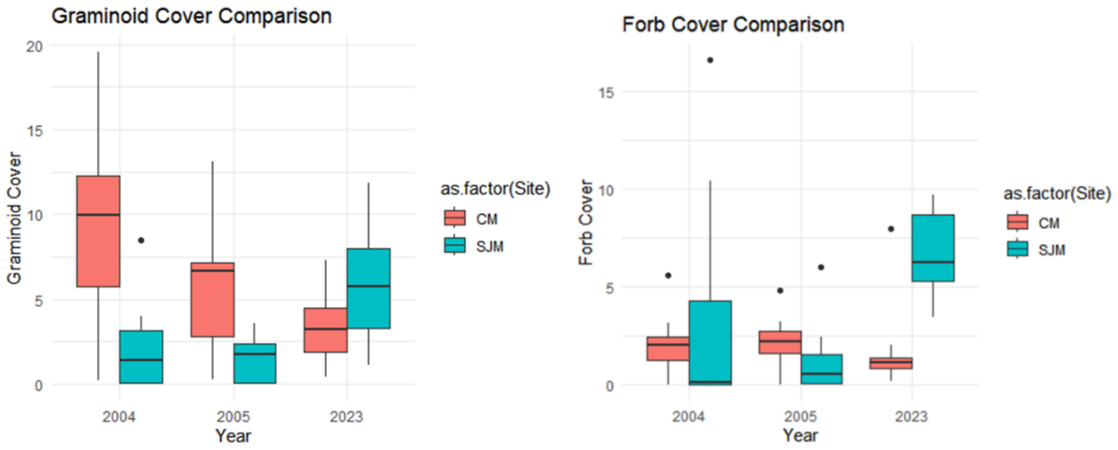


Figure 4.2 Changes in graminoid and forb cover from pre-treatment, 2004 and 2005, to 11 years post-treatment in 2023. The red boxes are the control site (Cat Mesa), and the blue boxes are the treated site (San Juan Mesa). Graminoid cover was formerly higher in the control site, but after treatment was higher on the treated site. This indicates that treatment was beneficial for stimulating graminoid growth. We can also see an increase in forb cover in the treated site in 2023, indicating that fuels reduction treatments are highly positive for forb cover.

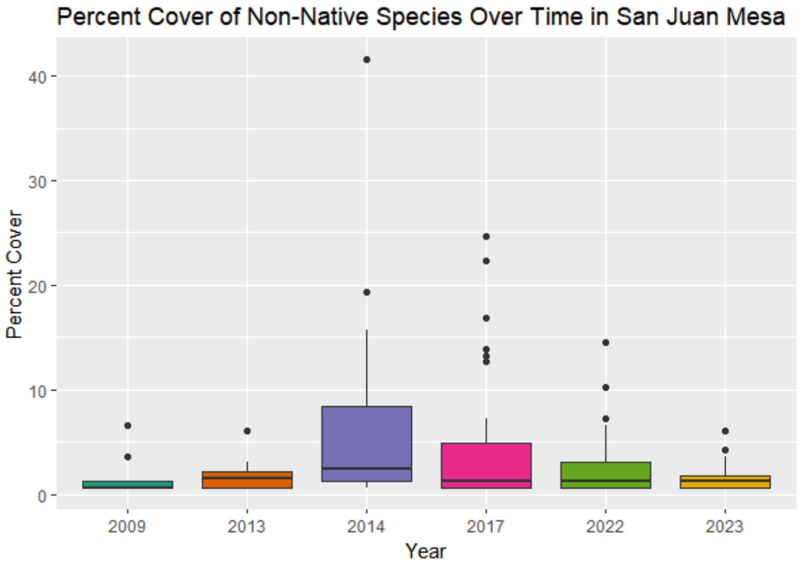


Figure 4.3 The recovery trajectory of non-native species in the treated site. Pre- and immediate-post-burn showed low non-native species cover. In 2014 and 2017, 2-5 years after the prescribed burn, we see that cover spikes. In 2022 and 2023, 10-11 years after the prescribed burn, the cover of non-native species decreases and stabilizes again. With most of the literature regarding understory responses to fuels reduction treatments ending their sampling intervals at 3-5 years post-burn, this figure indicates that long-term management is preferable and imperative to obtaining the full picture. This figure indicates that prescribed burns and thinning treatments do not affect non-native species cover significantly.

Key Findings/Management Implications:

1. **Positive Impact on Understory Vegetation:** Prescribed burn and thinning treatments have a positive impact on Southwest U.S. ecosystems, enhancing biodiversity and ecosystem resilience. The study reveals a notable increase in diversity and plant cover, emphasizing the potential for similar treatments to achieve positive outcomes in managing biodiversity.

2. **Long-Term Effects and Recovery Trajectory:** Understanding the long-term effects of treatments is crucial. The research provides insights into the recovery trajectory of understory vegetation communities over a decade after prescribed burn and thinning treatments. This temporal perspective offers valuable information for land managers,

highlighting the need for sustained, collaborative efforts in monitoring and managing forest ecosystems.

3. **Importance of Collaboration and Monitoring:** The study advocates for continued collaboration between land managers and scientists, emphasizing the significance of long-term monitoring. By synthesizing data from different datasets and encouraging sustained efforts, the research contributes to a comprehensive understanding of ecosystem responses to treatments, laying the groundwork for sustainable forest management in the face of escalating climate change impacts.