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ORIGINAL RESEARCH

Major changes in climate, vegetation, and ecological resilience in recent decades suggest climate smart management strategies for western US dryland shrublands and woodlands

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Abstract

Background Catastrophic wildfire has escalated across the globe in recent decades with devastating consequences for human communities and native ecosystems. Global change processes, including climate warming and land use practices, are altering fuels, fire risk, and ecosystem recovery. Managing ecosystems to reduce fire risk and prevent conversion to undesirable alternative states requires knowledge of the ecological conditions of ecosystems, trajectories of change, and drivers of those changes. We developed an approach for evaluating ongoing changes in climate and vegetation and using that information to determine appropriate fuels and other vegetation management strategies for southwest US dryland shrubland and woodland landscapes. We illustrated the approach at a management appropriate scale—a USDA Forest Service Wildfire Crisis Strategy landscape.

Results We developed an understanding of ecological types, current climatic regimes, ecological resilience to disturbance, and resistance to invasive annual grass (R&R). We then evaluated changes in plant functional type cover, historical fires, and R&R using long-term data. In unburned areas, changes in plant functional type cover included decreases in perennial forbs and grasses but increases in annual forbs and grasses, shrubs, and especially pinyon and juniper trees. In burned areas, tree cover was reduced and both perennial forb and grass and annual forb and grass cover increased. Most ecological types had moderate wildfire risk based on modeled annual burn probabilities and large areas burned since 1998 (16% of study area). These types were likely burning within expected fire return intervals, but areas burned during a single event may have exceeded historical extents and post-fire outcomes had changed. Transitions to warmer temperature regimes occurred between 1980–1999 and 2000–2019 resulting in an 11% decrease in R&R with the greatest impacts in cooler and moister ecological types.

Conclusions We showed that climate warming in southwest drylands has been associated with concurrent changes in vegetation and fuels and decreases in R&R. We provide an approach that allows managers to quantify the ongoing changes at management appropriate scales. We suggest climate smart management strategies to help direct ecosystems into conditions that can decrease fire risk, increase resistance to plant invasions, and reduce vulnerability to climate change.

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Keywords Climate change, Wildfire, Ecological resilience, Resistance to invasion, Drylands, Fuel treatments, Climate smart, Management strategies

Resumen

Antecedentes Los incendios catastróficos de vegetación han escalado en todo el mundo en décadas recientes, con consecuencias devastadoras en comunidades humanas y ecosistemas nativos. Los procesos del Cambio Climático Global, incluyendo el calentamiento global y las prácticas de uso de la tierra, están alterando los combustibles vegetales, el riesgo de incendio, y la recuperación de los ecosistemas. El manejo de los ecosistemas para reducir el riesgo de incendios y prevenir su reconversión a estados alternativos no deseables, requiere del conocimiento sobre las condiciones de los ecosistemas, los cambios de sus trayectorias sucesionales, y los factores conducentes que provocan esos cambios. Desarrollamos una aproximación para evaluar lo cambios proyectados en clima y vegetación, y usamos esa información para determinar las estrategias apropiadas de manejo de los combustibles y la vegetación en los paisajes de arbustales y bosques del sudoeste de los Estados Unidos. Ilustramos la aproximación usada a una escala de manejo apropiada – La estrategia de manejo de la crisis de fuegos de vegetación a nivel de paisaje del Servicio Forestal de los EEUU--.

Resultados Desarrollamos un entendimiento sobre los tipos ecológicos, los regímenes climáticos actuales, la resiliencia ecológica a los disturbios, y la resistencia a los pastos anuales invasores (R&R). Evaluamos luego los cambios en la cobertura de tipos funcionales, los incendios históricos, y las R&R usando datos acumulados durante largo tiempo. En áreas no quemadas, los cambios en los tipos funcionales incluyeron disminuciones en malezas y pastos perennes, e incrementos en malezas y pastos anuales, en arbustos y en especial en pino piñonero (*Pinus edulis*) y enebro (*Juniperus arizonica*). En áreas quemadas, la cobertura arbórea está reducida, y tanto los pastos como malezas perennes incrementan su cobertura. La mayoría de tipos ecológicos tienen un riesgo de incendio moderado basado en el modelado anual de las probabilidades de quema y en las grandes áreas quemadas desde 1998 (16% del área de estudio). Estos tipos podrían quemarse dentro de intervalos previstos de retorno de fuego, aunque áreas quemadas durante un único evento podrían exceder las extensiones históricas y modificar los resultados del post-fuego. La transición hacia regímenes de temperatura más cálidos ocurridos entre 1980 y 1999 y entre 2000 y 2019 resultó en un decrecimiento de R&R con los mayores impactos en tipos ecológicos más fríos y húmedos.

Conclusiones Mostramos que el calentamiento del clima en regiones secas del sudoeste de los EEUU han estado asociadas a cambios concurrentes en vegetación y combustibles y disminuciones de R&R. Proveímos de una aproximación que permite a los gestores cuantificar los cambios previstos a escalas de manejo apropiadas. Sugerimos cambios inteligentes en las estrategias de manejo teniendo en cuenta las variables climáticas, de manera de ayudar a los ecosistemas a disminuir el riesgo de incendio, incrementar la resistencia a la invasiones de plantas y reducir la vulnerabilidad al cambio climático.

Background

Catastrophic wildfire has escalated across the globe in recent decades with devastating consequences for human communities and native ecosystems (Iglesias et al. 2022; United Nations Environment Programme 2022). Global change processes including elevated CO_2 and climate warming have resulted in greater fuel loads, higher temperatures, and more extreme fire weather, while wildland-urban interface development has increased risks to people and property (Ager, 2022); Pan et al. 2022). Large amounts of funding and resources have been allocated to implementing management strategies designed to reduce the exposure of people, communities, and natural resources to the risk of wildfire in fire-prone ecosystems worldwide (e.g., IIJA 2021). These strategies include proactive fuel treatments, supporting post-fire recovery and restoration, and promoting the readiness of human communities. Reducing fire risk through proactive vegetation management, while sustaining the health, diversity, and productivity of native ecosystems, represents a challenge that requires new thinking, approaches, and tools (Ager, 2022).

Resource managers tasked with developing strategies for reducing fire risk while sustaining ecosystems are often faced with moving targets because of ongoing global change processes and their effects on vegetation composition, fuel characteristics, and fire regimes (Hurteau et al. 2014; Westerling et al. (Westerling, and Westerling, 2016); Hessburg et al. 2022). Despite this, current management strategies for addressing fire risk are often static; they target ecological conditions defined at a single point in time. For example, the type of management treatment(s) is often based on the degree of departure from a potential ecological condition in forested ecosystems (Blankenship et al. 2021; Provencher et al. 2024) or the change in ecological state from a reference condition in rangeland ecosystems (USDA NRCS 2022a, b). Strategies that include the effects of climate change typically focus on models of future ecological conditions under different CO_2 relative concentration pathways (RCPs) and base adaptation strategies on modeled projections of an ecosystem's vulnerability to climate change (e.g., Swanston et al. 2016; Sample et al. 2022). Over the past four decades, the dynamic nature of ecosystems and the ongoing effects of climate change and past management on ecological conditions and fire risk have become clear (e.g., Westerling 2016; Hessburg et al. 2022). Furthermore, the importance of understanding the ecological resilience (recovery potential, sensu Holling 1973) of ecosystems for evaluating the likely effects of both wildfire and management treatments has been illustrated in a variety of landscapes (e.g., Hessburg et al. 2015; Chambers et al. 2019). This suggests that strategies for addressing fire risk that couple information on the changes in climate and the vegetation trajectories observed in recent decades with the effects on ecological resilience and fire regimes may be equally or more effective than many current strategies.

Vegetation trajectories can be evaluated with remote sensing data, which provide the capacity to evaluate both abrupt and gradual change in the cover of the plant functional types over time at a variety of spatial scales (Rigge et al. 2021). These data provide a strong surrogate for analyzing historical change in the absence of plot data (Shi et al. 2018). For example, recent analyses using remote sensing data for the western USA showed the magnitude of tree expansion (Filippelli et al. 2020; Morford et al. 2022) and elevational assent and spread of invasive annual grasses in the basin and range ecoregions (Bradley et al.2018; Smith et al. 2022). Similarly, remote sensing products were used to determine the association of climate with both vegetation change (Xian et al. 2012a, b; Homer et al. 2013, 2015) and fire extent across time in semiarid ecosystems (Shi et al. 2018; Holdrege et al. 2024). Determining the amount of change in recent decades in both climate and vegetation cover can help identify "hot spots" with increasing fire risk or climate vulnerability and to prioritize those areas for fuel management and other vegetation treatments where transitions to new or alternative states are most likely.

Concepts related to ecological resilience to disturbance and resistance to plant invasion (R&R) have been widely used to guide natural resources management actions related to vegetation treatments and fire risk (Chambers et al. 2014a, 2019; Hessburg et al. 2015, 2022). Resistance to invasive plants is especially important in drylands because of the potential of these invaders to transform ecosystems into less desirable ecological states (Chambers et al. 2014a). The R&R concepts have been operationalized for dryland shrublands and woodlands through the development and use of R&R indicators based on the environmental characteristics, attributes and processes, and disturbance responses of ecosystems (Chambers et al. 2017b, 2023a, 2024a). Use of climate and water availability variables to develop the R&R indicators allowed evaluation of changes in the indicators in recent decades (Chambers et al. 2024a) and projections of likely future changes (Schlaepfer 2024a). Indicators of R&R can be coupled with assessments of the ongoing changes in climate, vegetation, and wildfire to develop more effective prioritization schemes for vegetation management (Chambers et al. 2017b, 2019, 2023b, 2024f).

More extensive or higher severity wildfires in dryland shrublands and woodlands are causing widespread conversions to novel or alternative ecological states (Fusco et al. 2019; Davies et al. 2021; Smith et al. 2023) as observed in warmer and drier ecosystems with low recovery potential worldwide (Coop et al. 2020; Guiterman et al. 2022). Major anthropogenic drivers of these changes include fire suppression policies (Hai et al. 2023), an increase in the human footprint (Leu et al. 2008; Knick et al. 2011), and greater numbers of human fire starts (Fusco et al. 2016). The first major ecosystem driver is the invasion and expansion of exotic annual grasses and forbs, which increase continuous fine fuels that cure earlier in the growing season and can result in large increases in fire frequency and extent (Bradley et al. 2018). The second is the expansion of native pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.) trees (pinyon-juniper) into shrubland and other dryland ecosystems, which can reduce shrub and herbaceous understory species (surface fuels) and decrease fire spread in the initial phases of expansion (Miller et al. 2019). As stand infilling and tree growth progress, a new strata of crown fuel develops increasing the threat of high severity crown fires (Strand et al. 2013, 2023); Miller et al. 2019). These ongoing changes in fuels and fire risk are being exacerbated by elevated CO_2 and climate warming (Abatzoglou et al. 2016); Balch et al. 2022; Bradford et al. 2020; Pan et al. 2022) and resulting in decreases in R&R to wildfire and management actions (Hurteau et al. 2014; Chambers et al. 2024a; Schlaepfer et al. 2024a).

We developed a new approach for evaluating the ongoing changes in dryland shrublands and woodlands and deriving climate-smart fuels and other vegetation management strategies. We built on prior work that developed R&R indictors for southwest US dryland shrubland and woodland landscapes (Chambers et al. 2024a) and illustrated the approach at a management appropriate scale. We first developed an understanding of the dominant shrubland and woodland ecological types within the landscape, their current climatic regimes, and their relative R&R. We then evaluated the change in cover of the dominant plant functional types, the relative influence of historic fires on the observed changes, and the effects of changes in climate on R&R using long-term remote sensing data. Our key questions were as follows: (1) How has the cover of the plant functional types changed in recent decades? (2) How has historic fire or lack thereof been associated with these changes? (3) How have climatic regimes changed and what were the effects on R&R? We used our results on the magnitude and direction of the observed changes and their likely effects on wildfire risk, R&R, and climate vulnerability to develop climate-smart management strategies for addressing the ongoing changes. Dryland shrublands and woodlands across the region are experiencing similar changes and we believe this approach and the management strategies are broadly applicable.

Study area

The Pine Valley Ranger District (study area) in southwestern Utah, USA, was part of an USDA Forest Service Wildfire Crisis Strategy (WCS) landscape (Fig. 1; USDA Forest Service 2022b). The WCS landscapes were selected to address wildfire risk in locations where it poses the most immediate threats to communities, critical infrastructure, and natural resources (USDA Forest Service 2022a, b). Like many shrublands and woodlands in the western USA, the study area was experiencing increasing burn probabilities due to rapid expansion of the invasive annual, cheatgrass (*Bromus tectorum*), and expansion and infilling of pinyon-juniper (*Pinus monophylla*, *P. edulis*, *Juniperus osteosperma*) trees in shrubland ecosystems (Tuhy et al. 2014; Chambers et al. 2024f). The study area was at the intersection of four different ecoregions, namely, the Central Basin and Range, Wasatch Front, Colorado Plateaus, and Mojave Basin and Range. Consequently, the area was characterized by large elevational gradients with a broad range of shrubland, pinyon-juniper woodland, and forest ecological types. The range in mean annual temperature was 3.9 to 15.7 °C (mean 9.9 °C) and in mean annual precipitation was 318 to 654 mm (mean 473 mm) with cooler and moister conditions at higher elevations and warmer and drier conditions at lower elevations.

The study area was settled by Euro-Americans in the mid-1800s and human communities with a long history of land use were interspersed throughout the area. Native plant understories were often depleted of native perennial grasses and forbs due to more than a century of livestock grazing as well as progressive expansion

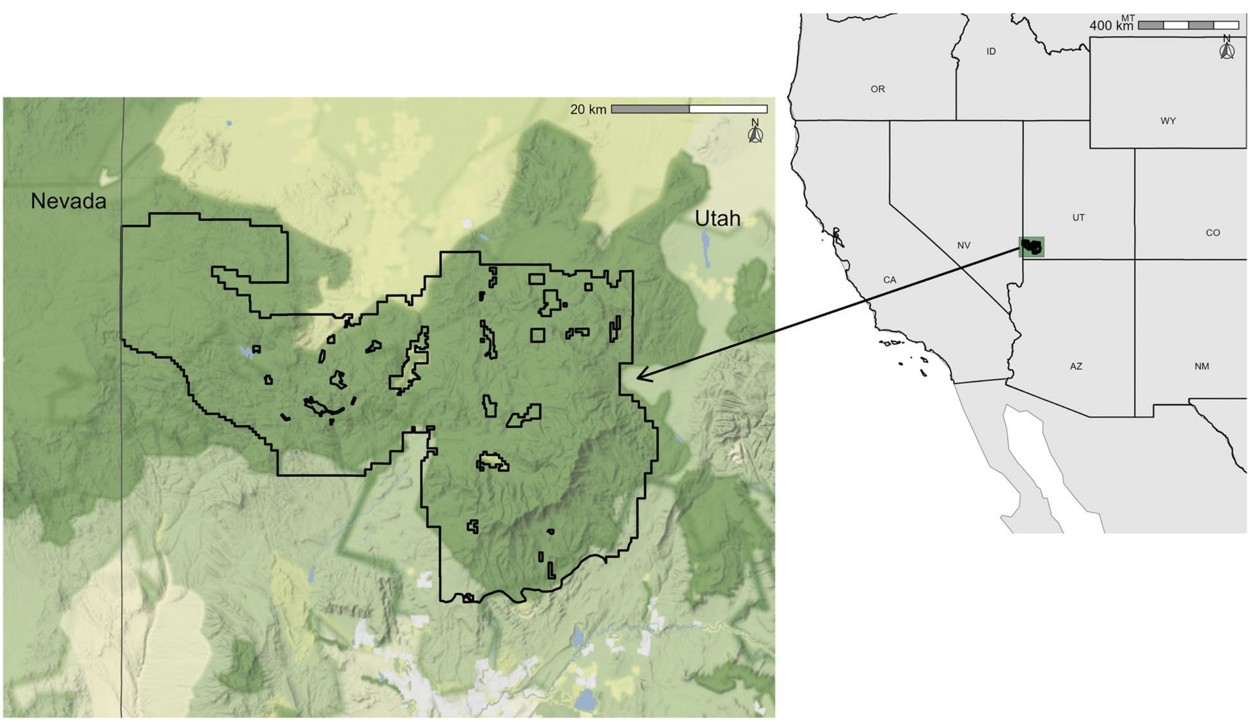

Fig. 1 Map of the Pine Valley Ranger District study area, which is part of the Dixie National Forest and is in the state of Utah in the western USA

and infilling of pinyon-juniper trees into the shrublands (Tuhy et al. 2014; Chambers et al. 2024c). Past vegetation management focused largely on clearing shrubs and trees from the central valleys near the human communities and seeding with introduced forage grasses, such as western wheatgrass (*Pascopyrum smithii*), so large areas had been converted to alternative states dominated by introduced species.

Methods

Climate regimes and resilience and resistance indicators

We developed climate regimes and R&R indicators for a nearly 30 million ha area of the southwest USA that encompassed the study area as described in Chambers et al. (2024a). In brief, we identified proxy soil temperature and moisture regimes (climatic regimes) for the ecological types in the southwest US study area based on mean annual air temperature, mean annual precipitation, and the monsoon index (the ratio of the sum of precipitation during July, August, and September to total annual precipitation) from Daymet annual and monthly climate summaries at a scale of 1 km (Fig. S1; Thornton et al. 2022; https://daymet.ornl.gov). We developed a ruleset for assigning climate regimes to ecological types based on the climate variables. We assigned R&R indicator classes (1 through 10) to the climate regimes and their associated ecological types based on literature review, expert knowledge, and information on the ecological types using an approach similar to Chambers et al. (2023a; Table S1). The climate regime ruleset and R&R indicator classes were used to generate spatial data layers for the climate regimes and R&R classes utilizing the appropriate gridded climate data for southwest drylands (Fig. S2; data are archived in Chambers et al. 2024b, d). We used these data layers to map and analyze the climate regimes and R&R classes for the study area (Fig. S3).

Characteristics of shrubland and woodland ecological types

We identified 11 ecological types for the study area corresponding to categories that were compiled by Tuhy et al. (2014) and that were aligned with the LANDFIRE Biophysical Settings as described in Chambers et al. (2024f). We used the climate regime ruleset and R&R indicator crosswalk to identify the climate regime and R&R class of each ecological type in the study area (Table S2). We used an USDA Forest Service spatial database (Dixie National Forest, unpubl. data) to identify areas with prior fuels and seeding treatments and excluded them from all analyses.

Historical wildfires and wildfire risk

We mapped and calculated the extent and severity of wildfires with areas larger than 405 ha for each ecological type using the Monitoring Trends in Burn Severity (MTBS; mtbs.gov) burned area boundaries dataset for 1986–1997, 1998–2011, and 2012–2023. We also mapped and described the annual burn probabilities for each ecological type using data from Dillon et al. (2023) at the 270 m resolution provided. Burn probability depicts the annual probability of wildfire occurrence at each specific pixel and is derived from simulation modeling of fuelscape data, contemporary weather and ignition patterns, as well as contemporary fire management policies (including fire prevention and suppression efforts; Dillon et al. 2023).

Change in plant functional type cover

We quantified long-term trends in vegetation cover as well as pinyon and juniper expansion and infilling by analyzing Rangeland Analyses Platform (RAP; Allred et al. 2021) data. Estimated mean absolute error (MAE) for RAP cover data was determined from a validation data set of 7500 field plots (Allred et al. 2021). The MAE was ± 7.0% for annual forb and grass pixels, 10% for perennial forbs and grasses, 6.2% for shrubs, and 2.6% for trees. We first analyzed the changes in shrub and tree cover (woody fuels) and perennial forb and grass (PFG) and annual forb and grass (AFG) cover (herbaceous fuels) from 1986 through 2023 with generalized mixed models. We aggregated the original 30 m RAP spatial data to means within 2 ha hexagonal cells. Each hexagonal cell was assigned the ecological type present within the majority (> 50%) of pixels within the cell and whether the majority of the cell had burned in a wildfire since 2000 or received a fuel or seeding treatment from 2009 to 2023. Wildfire occurrence was extracted from the MTBS dataset for burn severities of 2, 3, or 4, which were considered sufficient to assign burn status (e.g., Smith et al. 2023).

We determined the best-fit linear time trends of vegetation cover values for each ecological type and plant functional type (shrubs, trees, PFG, AFG) for areas burned and unburned since 2000 by fitting generalized linear mixed models using package glmmTMB in R (Brooks et al. 2017; R Core Team 2024). We analyzed all years of data from randomly sampled hex cells to provide robust estimates of trends. Sample sizes varied from a minimum of 100 hex cells for relatively rare types, such as burned mountain shrub-Stansbury cliffrose, to 1178 hex cells for unburned Wyoming big sagebrush for a total sample size of 8258 hex cells. Because both the areas of ecological types and areas burned varied, we sampled ecological types in proportion to their occurrence on the landscape. Those ecological types with fewer than 100 hex cells were dropped from the analysis. We modeled vegetation cover as mixed effect beta regression models (logit link), because the data represented time series measured at the same location for 39 years and proportions from 0 to 1 (or percents from 0 to 100%). We dropped any cells that had missing values for any of the variables and added a small constant to those few with values of 0. We modeled each response variable (cover of shrubs, trees, PFG, and AFG) as a function of a three-way interaction between annual time step (continuous scaled and centered), ecological type, and burn history (yes or no), with a random effect of hex cell ID. Final estimates of time trends were reported as marginal trends in the scale of the response value (proportion of cover) with 95% confidence intervals incorporating all fixed effect uncertainty, but not uncertainty due to random effects, using package "emmeans" (Lenth 2024).

We evaluated the relationships between PFG and AFG cover by calculating the median PFG cover that consistently resulted in relatively low median AFG cover in unburned and burned areas for the ecological types using the RAP data. Substantial evidence exists that PFGs are the primary determinants of ecological resilience (Condon et al. 2011; Davies et al. 2012; Chambers et al. 2014b; Prevéy and Seastedt 2014; Larson et al. 2017; Wainwright et al. 2020) and resistance to annual grasses (Chambers et al. 2007, 2014a, 2016, 2017a); Davies et al. 2008; Bansal and Sheley 2016; Prevéy and Seastedt 2014; Larson et al. 2017; Urza et al. 2017).

To analyze the amount of tree expansion in shrublands and infilling in woodlands, we evaluated changes in the proportion of tree cover for each ecological type. We defined three cover classes similar to the phases identified in Miller et al. (2019). In ecological types with low productivity, class 1 tree cover was 5–10%, class 2 tree cover was 10–20%, and class 3 tree cover was > 20%. In ecological types with high productivity, class 1 tree cover was 5–10%, class 2 tree cover was 10–30%, and class 3 tree cover was > 30%. We defined productivity as detailed in Chambers et al. (2024f). Low productivity ecological types were those receiving relatively low mean annual precipitation that had low to moderately low resilience or resistance: blackbrush, mountain shrub-Stansbury cliffrose, and juniper and pinyon. We defined all other ecological types as high productivity. Due to the MAE in RAP tree cover data, 5–10% cover values rather than 1–10% cover were used for class 1. In most of the shrubland and woodland ecological types, tree cover represented pinyon and juniper. Although pinyon-juniper expansion occurs in the Gambel oak and mountain mahogany ecological types, tree species other than pinyon and juniper also occurred in these types.

Once the tree cover classes had been designated, we evaluated the proportions of the tree classes by ecological type and burn history (areas that had or had not burned after 2000) for 1986–2023 using the Theil-Sen estimator (Sen 1968). Tree class in each year was determined by averaging RAP tree cover data over the same 2 ha hex cells analyzed in the continuous cover regressions, and classifying cover into class 1, 2, or 3 depending on the rule set for the majority ecological type within that cell. The proportion of the ecological type not treed or categorized as class 1, 2, or 3 hex cells was calculated and the Theil-Sen slope of proportional cover between 1986 and 2023 determined. Similar to the analyses of continuous cover, areas with known treatment histories and ecological types with insufficient sample sizes were excluded from analyses.

Change in climate regimes and R&R

We evaluated changes in the R&R categories in recent decades by subsetting the 40-year Daymet climate data into two periods: 1980–1999 and 2000–2019. We applied the ruleset for assigning climate regimes based on temperature, the monsoon index, and precipitation to each period. We then assigned R&R indicator classes (1 through 10) to the climate regimes as described for the entire 40-year dataset and mapped and compared the indicators for each period. Data are archived in Chambers et al. (2024c, e). All geospatial and database operations were performed in R 4.4.1 (R Core Team 2024).

Results

Characteristics of shrubland and woodland ecological types

The shrublands and woodlands in the study area had proxy soil temperature regimes that were largely very warm (hypermesic, 11.5–14.5 °C), warm (mesic, 8.5–11.5 °C), or cool (frigid, 6.5–8.5 °C) and proxy soil moisture regimes that were winter moist (xeric, > 305 mm precipitation) or winter moist and relatively dry (aridic < 305 mm precipitation) (Fig. 2, Tables S1 and S2). The monsoon index ranged from 0.22 to 0.26 with a median of 0.24. The study area receives some summer precipitation and supports warm-season plant species, but it is not described as having a summer moist (ustic) soil moisture regime, which is characterized by a monsoonal index of about 0.30 or higher (Chambers et al. 2024a). The R&R of the ecological types paralleled the climate regimes and resilience ranged from moderately low to moderately high while resistance ranged from low to moderately high (Table S2, Fig. S3).

The eleven shrubland and woodland ecological types reflected the climate regimes and the study area's location at the interface of the four different ecoregions (Fig. 3, Table S2). The current cover of the functional types varied among the ecological types. In the shrublands, PFG were generally depleted with covers ranging from about 2 to 8% in most ecological types (Fig. S4). The AFG

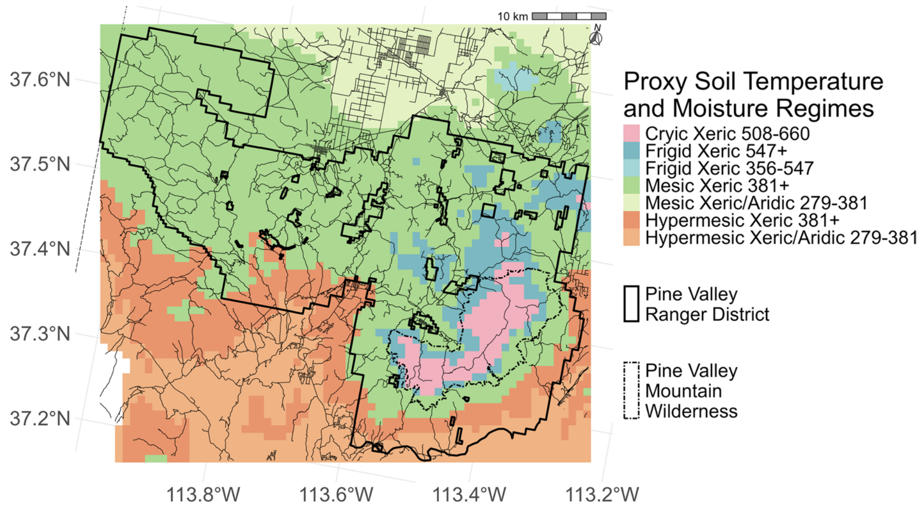

Fig. 2 Proxy soil temperature and moisture regimes (climatic regimes) derived from Chambers et al. (2024a, b) for the study area. The regimes were developed from mean annual temperature and precipitation and the monsoon index using Daymet monthly data at a 1-km scale for 1980–2019 (Thornton et al. 2022). Proxy soil temperature and moisture regime translation: cryic = cold; frigid = cool; mesic = warm; hypermesic = very warm. Udic = moist; ustic = summer moist; xeric = winter moist (> 305 mm precipitation); aridic = winter moist (< 305 mm precipitation)

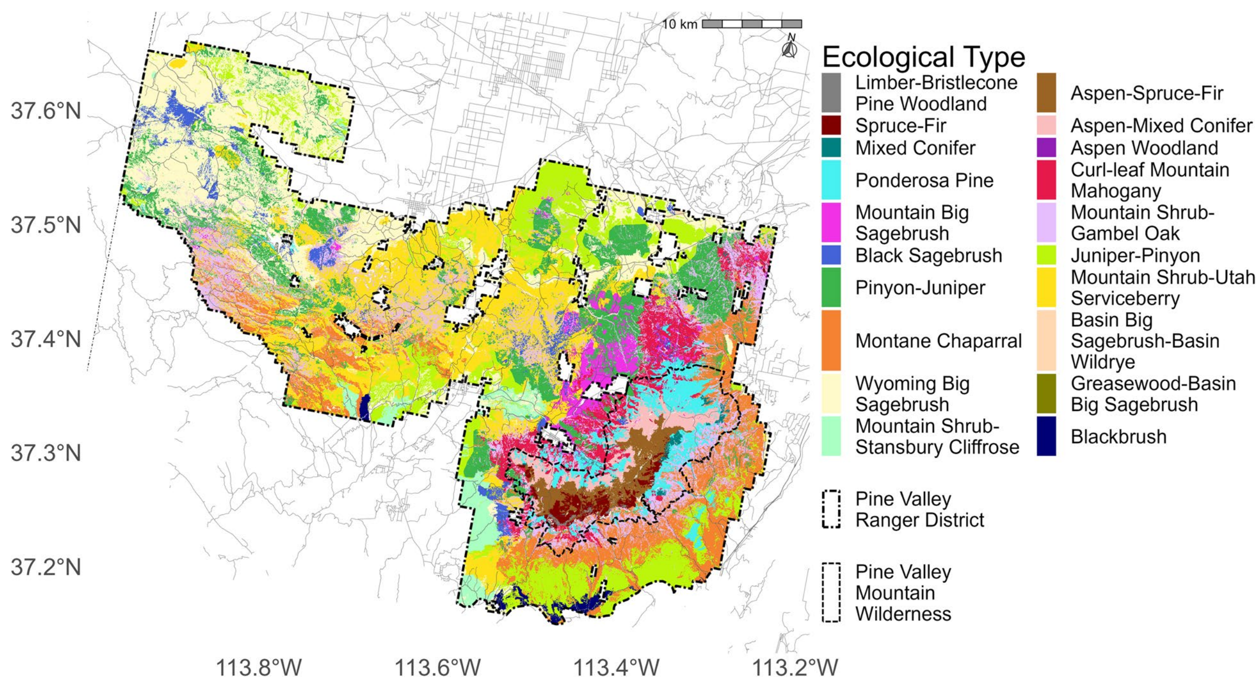

Fig. 3 Map of the ecological types for the Pine Valley Ranger District and Pine Valley Mountain Wilderness area. The ecological types were aligned with the LANDFIRE Biophysical Settings (BpS) developed for the district (Tuhy et al. 2014) and consistent with the ecological site descriptions (ESDs) developed by Stringham et al. (2015) or available in the Ecosystem Dynamics Interpretive Tool (EDIT; USDA and New Mexico State University 2024). Forested areas and aspen woodlands occurred largely in the Pine Valley Mountain Wilderness. Shrublands and woodlands were the focus of this study; curl-leaf mahogany and mountain big sagebrush occurred largely in the northeast surrounding the wilderness area, while the other shrubland types were intermixed across the landscape

were widespread across the study area with cover ranging from about 2 to 12%; the highest covers were in ecological types with very warm to warm and moist climatic regimes. Tree cover in shrubland ecological types ranged from about 6 to 27% with high covers in mountain shrub, chaparral, and big sagebrush. The woodland types had among the lowest covers of all functional types except trees, which had > 30% cover.

Historical wildfires and wildfire risk

Historical wildfires as indicated by MTBS burned the largest areas from 1998 to 2011, intermediate areas from 1986 to 1997, and small areas from 2012 to 2023 (Table 1, Fig. 4). The largest burn extents and most frequent fires were in montane chaparral and mountain shrub-Gambel oak, which are fire-adapted systems (Brooks et al. 2007; Johanson 2011; Simonin 2000; Kitchen and McArthur 2007; Tuhy et al. 2014). Relatively warm and moist mountain shrub and Wyoming big sagebrush types also experienced a series of relatively large burns from 1998 to 2011.

Table 1 The risk of future wildfires and historical wildfires in each ecological type within the study area. Wildfire risk was from modeled annual burn probabilities from Dillon et al. (2023) with 95% CI. *LCI*, lower confidence interval; *UCI*, upper confidence interval. The historical wildfires were from Monitoring Trends in Burn Severity (MTBS) data (mtbs.gov) for 1986–1997, 1998–2011, and 2012–2023, and are shown as square km with the percentage of the area for that ecological type in parentheses

Ecological type	Wildfire risk	Historic wildfires		
	Annual burn probability	MTBS 1986–1997	MTBS 1998–2011	MTBS 2012–2023
	Median (LCI:UCI)	km^2 (%)		
Blackbrush	0.008 (0.006:0.011)	0 (0)	0.3 (2.6)	0.01 (0.1)
Mountain shrub-Stansbury cliffrose	0.014 (0.008:0.019)	0 (0)	14.1 (31.1)	2.6 (5.7)
Montane chaparral	0.011 (0.007:0.016)	25.6 (13.7)	70.2 (37.7)	0.2 (0.1)
Wyoming big sagebrush	0.015 (0.007:0.024)	6.5 (1.9)	39.5 (11.6)	2.3 (0.7)
Mountain shrub-Utah serviceberry	0.012 (0.007:0.020)	12.4 (4.2)	42.8 (14.3)	3.7 (1.2)
Juniper-pinyon	0.013 (0.007:0.018)	1.7 (0.8)	5.0 (2.2)	0.1 (0)
Black sagebrush	0.017 (0.006:0.027)	0.8 (2.0)	4.6 (11.9)	0.2 (0.4)
Mountain big sagebrush	0.019 (0.007:0.024)	0 (0)	1.2 (2.8)	0.2 (0.5)
Pinyon-juniper	0.014 (0.007:0.021)	1.3 (0.7)	13.4 (7.2)	4.0 (2.1)
Mountain shrub-Gambel Oak	0.011 (0.006:0.018)	17.5 (8.4)	61.0 (29.3)	2.4 (1.2)
Curl-leaf mountain mahogany	0.012 (0.005:0.019)	0.2 (0.2)	2.8 (3.8)	5.3 (7.2)

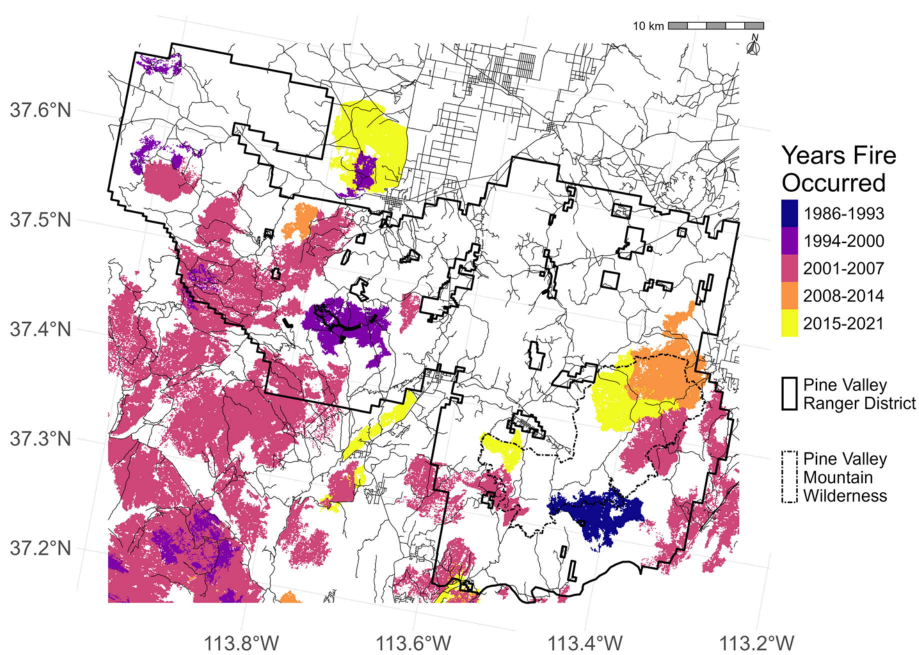

Fig. 4 Historical fires observed in and near the study area from 1986 to 2023 based on Monitoring Trends in Burn Severity (MTBS) data (mtbs.gov). Areas with burn severity values of 2, 3, and 4 are shown indicating burn severities of moderate and greater

Current wildfire risk for the ecological types based on modeled annual burn probability was classified as moderate and ranged from about 1 to 2% (0.01 to 0.02) per year modeled over a 30-year period (Table 1, Fig. S5).

Change in plant functional type cover

Relatively consistent patterns of change were observed from 1986 to 2023 with the amount of change varying among the ecological types. In areas that had not burned since 1986, there were decreases in PFG cover at the 95% CI of about 2 to 5% in all types except mountain shrub-Gambel oak, which declined about 10% (Fig. 5, Table S3, Fig. S6). In contrast, in areas that burned after 2000 PFG often increased. The AFG cover increased over time in unburned areas with only the mountain shrub types and black sagebrush showing relatively large increases. However, after burning AFG increased by 5 to 30% in all ecological types but mountain shrub-Gambel oak. A median PFG cover of 10 to 15% was required to consistently result in relatively low median AFG cover in unburned and burned areas (Figs. S7, S8).

Shrub cover showed increases of about 1 to 5% in unburned areas over time in all types except for those with a large component of big or black sagebrush which declined by 1 to 6% (Fig. 5, Table S3, Fig. S6). In burned areas, shrubs increased about 1 to 4%, except in mountain shrub-Stansbury cliffrose which showed a small decrease and Wyoming big sagebrush which increased by about 11%. Tree cover showed increases of 1 to 8% in the unburned areas for the ecological types with the largest increases in the big sagebrush types and the woodlands. In burned areas, tree cover decreased by about 10 to 20% in all types except mountain shrub-Gambel oak.

Analyses of the change in the proportions of tree cover classes for unburned areas indicated little change in areas without trees (−0.5 to 1.3%), small declines in class 2 (−0.2 to −1%), and increases in class 3 (2–7%) in most types, with especially large increases in class 3 in montane chaparral, Wyoming big sagebrush, and the woodlands (Fig. 6). In burned areas, the decreases in tree cover classes varied among ecological types, ranging from −1.6 to −8.3% for class 2 and from −0.8 to −7.1 for class 3.

Change in climate regimes and R&R

Increasing temperatures caused large changes in climate regimes and thus R&R classes for all ecological types between 1980–1999 and 2000–2019 (Fig. 7, Tables S4, S5, Fig. S9). Cumulatively, 11.4% of the area warmed with the largest losses in cool to cold moist regimes and largest gains in warm moist, very warm moist, and very warm dry regimes. The higher elevation ecological types were most impacted, especially mountain big sagebrush and curl-leaf mountain mahogany.

The changes in climate regimes resulted in a loss of resilience in the high (9, 10) and moderately high classes (7, 8) and gains in the moderate classes (5, 6) (Fig. 8, Table S6, Fig. S9). A mixed result in the moderately low resilience classes was due to a transition of the warm and dry regime (class 4) to the very warm and dry regime (class 3). The loss of resistance was even greater. We observed decreases in the high and moderately high resistance classes (7, 8, 10) and gains in moderately low classes (3, 4).

Discussion

An understanding of the types and magnitudes of change occurring within the different ecological types that characterize landscapes can be used to help reduce the threats of AFG and wildfire and to direct ecosystems to conditions that will be sustainable in the future. Our approach clearly showed the ongoing changes in climate, relative resilience and resistance (R&R), and vegetation trajectories occurring within the ecological types in our study area over the last four decades. In unburned areas, we found decreases in perennial forbs and grasses (PFG) but increases in annual forbs and grasses (AFG), shrubs, and especially pinyon and juniper trees as observed in many dryland shrublands and woodlands. In burned areas, tree cover was reduced and both PFG and AFG cover increased. Transitions to warmer temperature regimes occurred resulting in an 11% decrease in relative R&R with the greatest impacts on cooler and moister ecological types as observed across southwest dryland shrublands and shrublands (Chambers et al. 2024a).

Historical wildfires and wildfire risk

Most ecological types within the study area were classified as having moderate wildfire risk based on modeled annual burn probability. Relatively large areas burned since 1998 (33,700 ha, 16% of study area) similar to areas of the sagebrush biome with winter-dominated precipitation regimes (Holdrege et al. 2024) and of southwest shrublands and woodlands with bimodal precipitation regimes (Mueller et al. 2020). The largest burn extents and most frequent fires were in fire-adapted ecological types, which were characterized largely by root-sprouting shrub species capable of regrowing after fire. In addition, relatively large areas of Wyoming big sagebrush, which is not characterized by fire-adapted species (Kitchen and McArthur 2007), burned from 1998 to 2011.

The shrublands and woodlands in the study area were likely burning within expected fire return intervals (FRI). The estimated range in historic fire return intervals (FRI) in the ecological types varied relative to site productivity, time since wildfire, and relative abundances of the plant functional types. The estimated FRI in early successional

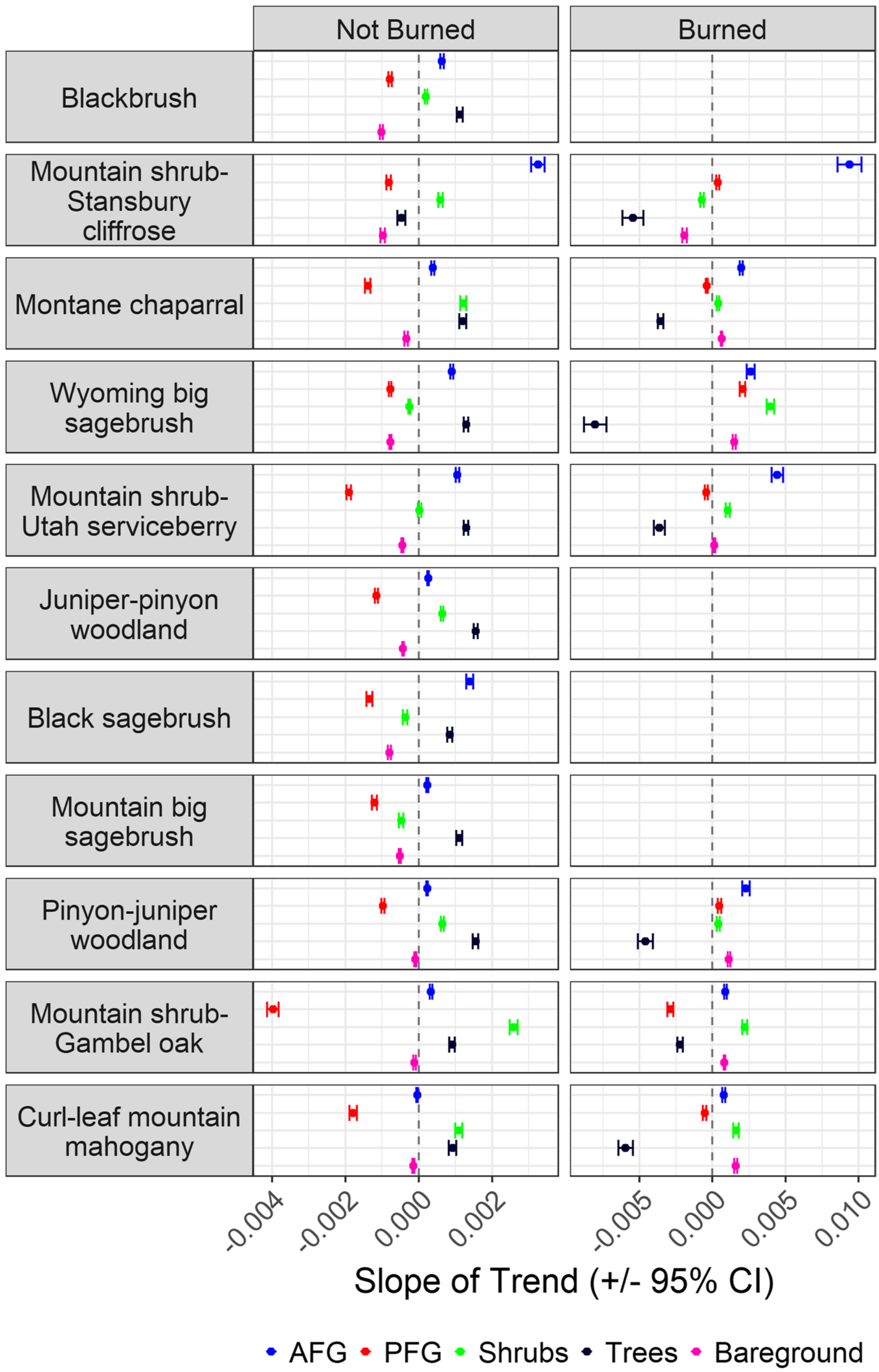

Fig. 5 The trend in vegetation life form cover for each ecological type from 1986 to 2023 in areas that had not burned after 2000 and in areas that had burned after 2000 within the study area. Burned area data were for areas that had a burn severity of 2, 3, or 4 as indicated from Monitoring Trends in Burn Severity (MTBS) data (mtbs.gov). Cover data were from Rangeland Analysis Platform data (RAP; Allred et al. 2021). Time trends are the marginal trends in the scale of the response value (proportion of cover) shown as means (dots) with 95% confidence intervals (whiskers) incorporating all fixed effect uncertainty, but not uncertainty due to random effects. Ecological types that were less than 2% of the area had burned were not included in the analyses (gray panels)

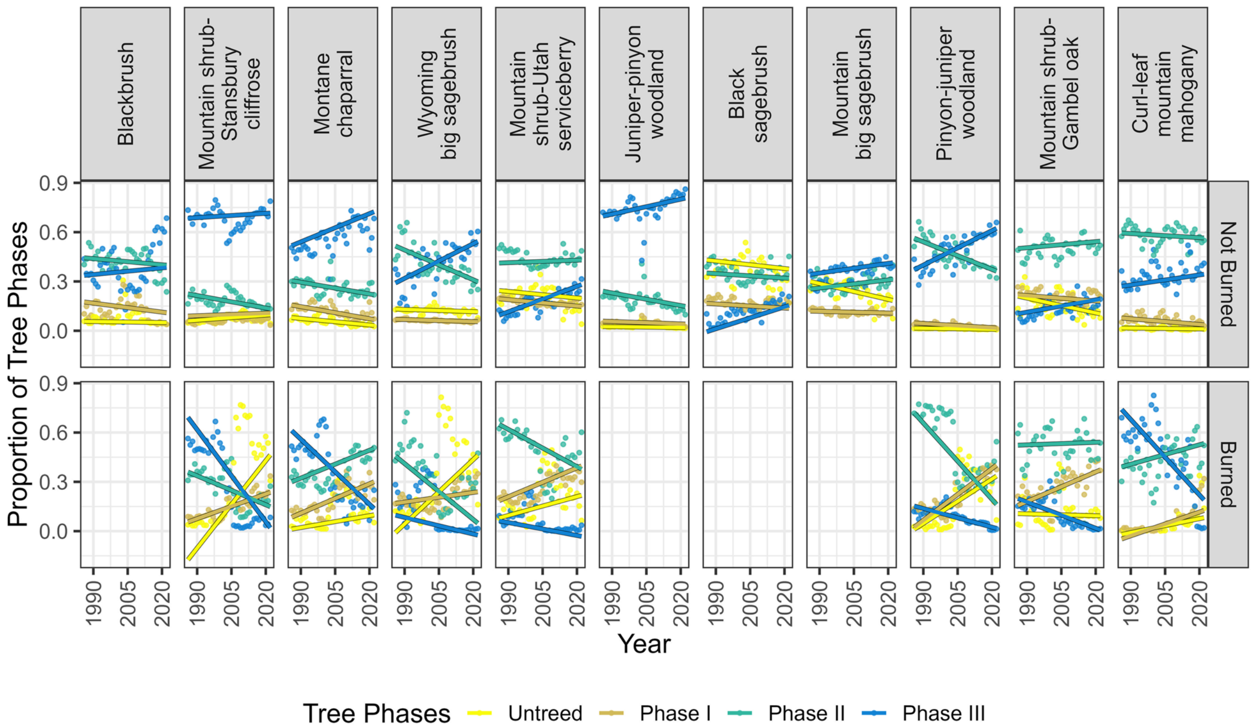

Fig. 6 The proportion of tree cover classes for each ecological type from 1986 to 2023 in areas that had not burned after 2000 and in areas that had burned after 2000 within the study area. Burned area data were for areas that had a burn severity of 2, 3, or 4 as indicated from Monitoring Trends in Burn Severity (MTBS) data (mtbs.gov). Cover data were from the Rangeland Analysis Platform data (RAP; Allred et al. 2021). Tree cover in untreed areas was > 5% and < 10% in cover class 1. In ecological types with low productivity, tree cover was 10 to 20% in cover class 2 and > 20% in cover class 3. In ecological types with high productivity, tree cover was 10 to 30% in cover class 2 and > 30% in cover class 3. Lines indicate the Theil-Sen slope of proportional cover between 1986 and 2022. Ecological types that had burned and were less than 2% of the area were not included in the analyses (gray panels)

areas characterized primarily by grasses and forbs ranged from < 12 to 20 years, in mid successional areas with a mix of shrubs and grasses and forbs from 20 to 80 years, and in later successional areas dominated by shrubs from > 80 to 100 years with slightly longer FRI on warmer and drier sites (Tuhy et al. 2014). Due to interactions among past and present land management practices, annual grass invasion, tree expansion, and a warming climate, areas burned during a single event may have exceeded historical extents and post-fire outcomes had changed.

The relative resilience classes for the ecological types indicated that all but the warmest areas had the capacity to recover following either fuel treatments or wildfire given the appropriate post-treatment or post-fire management (Chambers et al. 2024f). Most ecological types were characterized by warm to cool temperature regimes and relatively high precipitation (> 305 mm), which translates to moderate to moderately high ecological resilience. Exceptions were the blackbrush, mountain shrub-Stansbury cliffrose, and juniper-pinyon types which have areas with very warm temperature regimes. Resistance to cheatgrass was generally moderately low, except for areas of mountain big sagebrush, pinyon-juniper woodland, and curl-leaf mountain mahogany with cool temperature regimes. The relatively low resistance placed the warmer ecological types at high risk of conversion to cheatgrass following treatments or wildfire.

Changes in functional type cover

Large changes in herbaceous and woody cover occurred and thus the expected responses to both wildfire and fuel treatments. In most unburned ecological types, PFG cover showed a strong negative trend (median loss of 4%); declines in C3 grass biomass and perennial herbaceous cover also have been predicted for the sagebrush biome as a whole (Palmquist et al. 2021; Rigge et al. 2021). Depletion of PFG due to livestock grazing coupled with resource competition from other plant functional types, including AFG (Anderson and Inouye 2001; Williamson et al. 2020), shrubs (Pierce et al. 2019; Chambers et al. 2007, 2017a), and pinyon and juniper trees (Johnson and Miller 2006; Roundy et al. 2014; Miller et al. 2019), can result in decreases in PFG cover over time. In turn, decreases in PFG can result in competitive release of

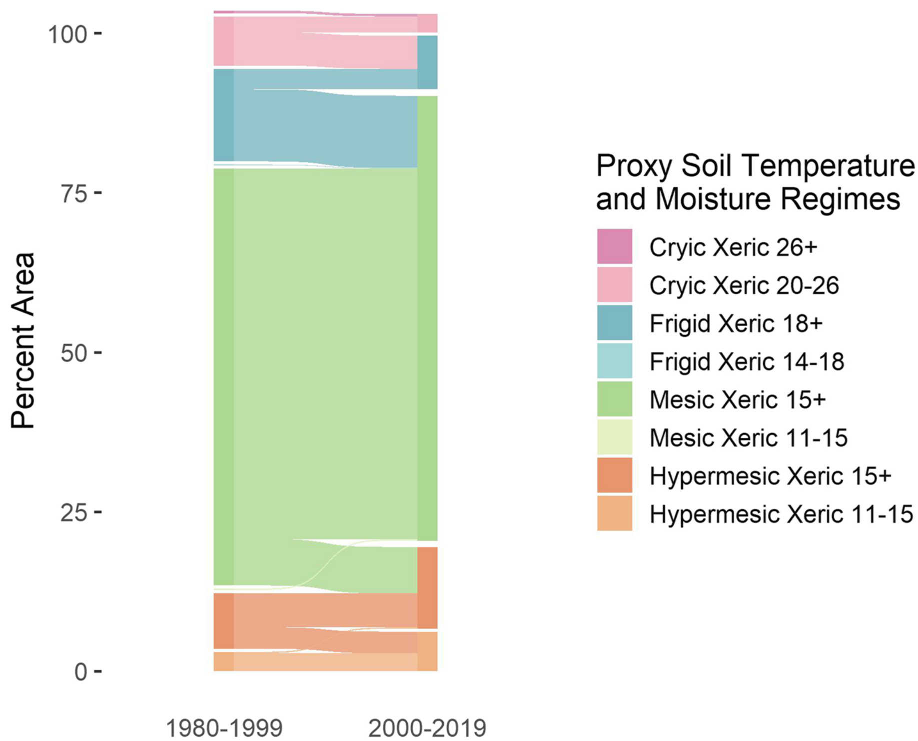

Fig. 7 Alluvial plot showing changes in area of proxy soil temperature and moisture regimes (climate regimes) from 1980 to 1999 and 2000 to 2019 based on Daymet 1-km climate data (Thornton et al. 2022). Soil term definitions: cryic = cold; frigid = cool; mesic = warm; hypermesic = very warm; xeric = winter moist (> 305 mm precipitation); aridic = dry (< 305 mm precipitation)

both AFG (Chambers et al. 2007) and shrubs (Chambers et al. 2017a; Pierce et al. 2019). AFG cover showed positive trends in all relatively warm ecological types with moderately low resistance but was limited in cooler and moister types with low climate suitability and moderate to moderately high resistance in both unburned and burned areas as shown elsewhere (Chambers et al. 2007; Bansal and Sheley 2016). PFG cover trends were less negative in burned than in unburned areas reflecting the moderate to moderately high resilience of much of the study area. To maintain relatively low AFG cover, PFG cover of 10 to 15% was required in both unburned and burned areas for ecological types with low to moderately low resistance. Although decreases in herbaceous species (fine fuels) as observed for PFG can result in decreases in fire risk, increases in these fuels as observed for AFG can increase fuel continuity and flammability elevating fire risk (Ellsworth et al. 2022; Williams et al. 2023).

Trends in shrub cover varied, with a tendency for increases in fire-adapted ecological types with root-sprouting shrubs and no to negative trends in blackbrush and sagebrush types. Although shrub cover can increase due to competitive release in response to depletion of PFG, recruitment can be limited in areas with high levels of either PFG (Chambers et al. 2017a) or AFG. This is particularly true for shrub species dependent on seeds for establishment, like the *Artemisia* species, and on warmer and drier sites (Shriver et al. 2019). In addition, shrub cover can decrease due to direct resource competition from pinyon and juniper trees (Johnson and Miller 2006; Roundy et al. 2014; Miller et al. 2019). Shrubs are an important component of woody fuels but in big sagebrush ecological types shrub cover greater than about 18% results in increased crown fire risk (Ellsworth et al. 2022; Schachtschneider et al. 2024).

Gains in the extent and biomass of trees have been observed across the Intermountain West of the USA over the past 10–20 years (Filippelli et al. 2020; Reinhardt et al. 2020; Morford et al. 2022), and we saw strong positive trends in tree cover for unburned shrublands (median increase 1.5 to 8%) with the greatest increases in big sagebrush types. Positive trends in tree cover were also observed in unburned woodlands (median increase 6 to 7%) likely indicating tree infilling. In shrublands experiencing pinyon-juniper expansion, increased tree cover is often associated with strong resource competition that results in declines in understory shrub and PFG cover and, depending on topographic position and slope, soil redistribution and loss (Pierson and Williams 2016; Miller et al. 2019). Tree covers above 20 to 30%

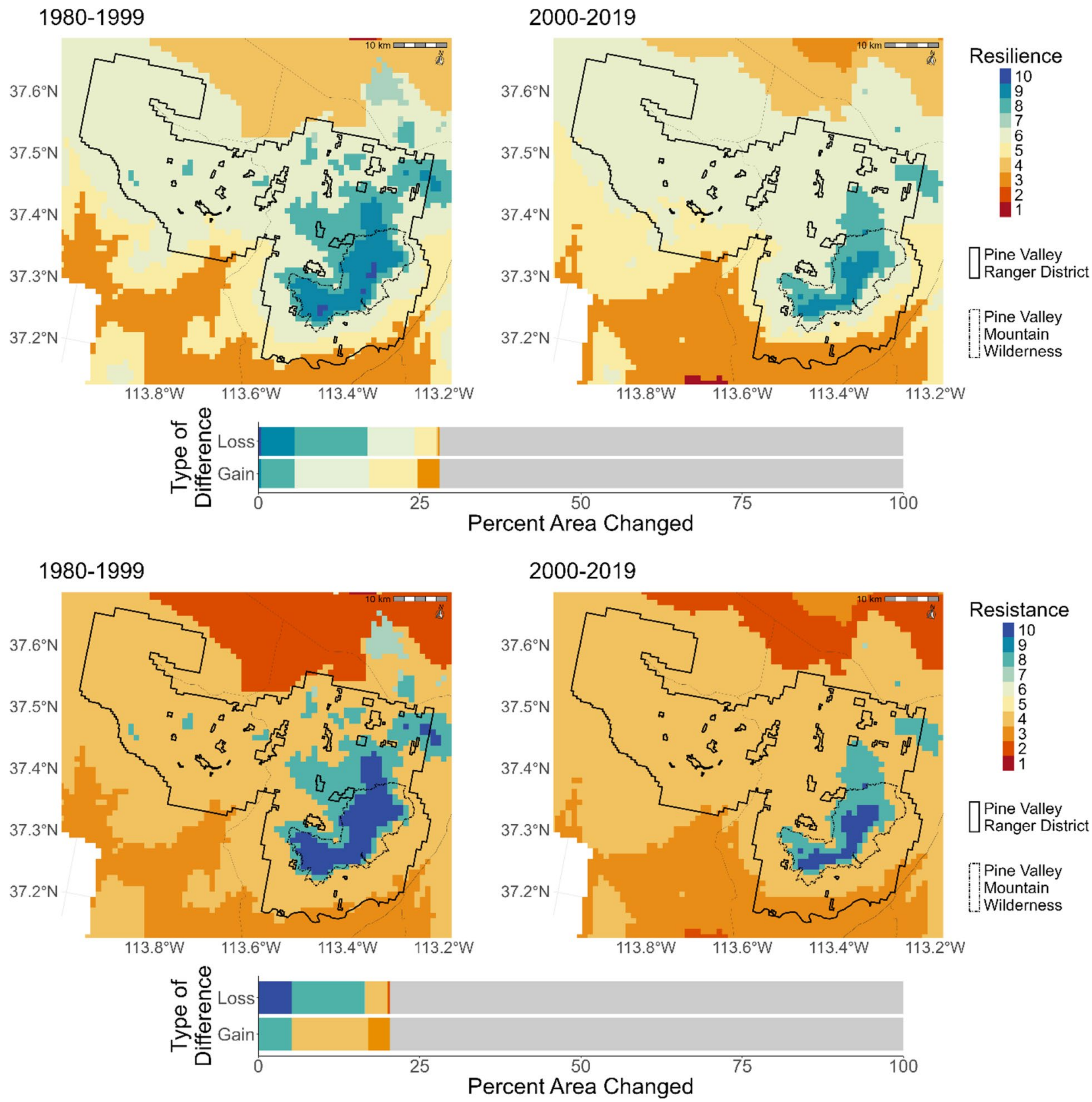

Fig. 8 Maps showing the ecological resilience and resistance to invasion from cheatgrass classes for 1980–1999 and 2000–2019. The amount of change between 1980–1999 and 2000–2019, both gain and loss, are shown in stacked bar plots under each set of maps, with the gray color representing the area with no change. The resilience and resistance classes were based on proxy soil temperature and moisture regimes (climate regimes) calculated from 1-km Daymet climate data means (Thornton et al. 2022) and their associated ecological site descriptions. Higher class numbers indicate greater resilience or resistance

(class 3) often result in longer FRI but increased risk of high severity crown fire, and in the absence of PFG and root-sprouting shrubs, potential conversion to AFG in warmer and drier ecological types (Miller et al. 2019; Williams et al. 2023). In woodlands, tree infilling can result in increased fuel continuity, more extensive and higher severity fires, and loss of old growth trees (Miller et al. 2019). Fewer trees in burned areas were associated with increases in the trends for PFG or shrubs. However, trends in AFG were also positive in climatically suitable areas.

The changes in tree cover classes (phases) we observed were similar to those observed in recent decades across pinyon-juniper ecosystems of the Intermountain West

(Miller et al. 2008, Fillipi et al. 2020). In unburned shrubland and woodland ecological types, we found increases in the proportion of class 3 tree cover and decreases in class 2 tree cover, except in mountain big sagebrush which showed increases in both class 2 and 3 tree cover. Dendrochronological analyses of seven sites in the Great Basin indicated that increases in pinyon and juniper trees were the result of both infill in areas with mixed tree ages and expansion into shrubland ecosystems that had not supported trees over the past several centuries (Miller et al. 2008). The tree age structure indicated that tree dominated areas (class 3) would increase from 20 to nearly 75% of the total woodland by 2035 to 2055 (Johnson and Miller 2006; Miller et al. 2008). In burned areas with sufficient data for analyses, we saw the expected loss of class 2 and 3 tree cover. However, 23 years post-fire the proportion of the area with no trees had decreased in most ecological types indicating reestablishment of trees following wildfire likely aided by residual trees providing seed sources within burned areas.

Change in climate regimes and R&R

The changes in climate we observed since 1986 mirrored the increases in temperatures observed in recent decades for southwest drylands (Zhang et al. 2021). Our prior climate change projections from 1980–2010 to 2070–2100 indicated that higher temperatures would expand the areas of warm and very warm (mesic and hypermesic) and hot (thermic) soil temperature regimes and reduce those with cool (frigid) and cold (cryic) regimes (Bradford et al. 2019). Precipitation was highly variable over time as observed in the climate change projections; however, even in areas where mean annual precipitation increases, higher evapotranspiration is likely to result in drier conditions (Bradford et al. 2020). Cumulatively, the recent climate changes resulted in about an 11% decrease in both R&R with the greatest effects in cooler and moister ecological types.

The influence of climate warming on R&R and plant functional types will have a strong influence on management outcomes. We observed an increase in the monsoon index from 1980 to 2019 in southwest drylands indicating relatively higher summer precipitation (Chambers et al. 2024a). Climate change projections from global change models (GCMs) for plant functional types across the part of the sagebrush biome encompassing the study area indicated declines in perennial C3 grasses and perennial forbs, which are most abundant in winter moist regimes, but widespread increases in perennial C4 grasses, which are most abundant in summer moist regimes (Palmquist et al. 2021). Rhizomatous C4 and other grasses typical of summer moist areas occurred in the study area and increases in these species can elevate resistance to invasion and recovery potential (Bradford and Lauenroth 2006; Lauenroth et al. 2014; Prevéy and Seastedt 2014; Larson et al. 2017). The same GCMs indicated a decrease in sagebrush biomass by mid- to end century for the study area, with the greatest decreases in warmer and drier areas (Palmquist et al. 2021). Warmer temperatures are already resulting in lower resistance to cheatgrass and elevational ascent of the species in winter moist climatic regimes (Smith et. al. 2022). In areas that decline in climatic suitability for cheatgrass due to higher temperatures, other invasive plants, such as red brome (*Bromus rubens*) and redstem stork's bill (*Erodium cicutarium*), may increase (Brooks et al. 2007; Bradley et al. 2016). General increases in populations of *J. osteosperma* and *P. monophylla* across most of their distributions were projected by mid- to end century from GCMs (Noel et al. 2023). However, increasing aridity is likely to affect survival and recruitment of all three species at local scales with the greatest vulnerability in warmer and drier areas and at lower elevations (Shriver et al. 2022).

Prioritizing areas for fuel and other vegetation management treatments

The recent changes in climate and the climate change projections indicate that management strategies should be designed to anticipate the concurrent changes in R&R, fire regimes, and plant functional types. We suggest that management strategies designed to address the ongoing changes in dryland shrublands and woodlands begin with characterizing the ecological types within the landscape and determining their current climatic regimes and relative R&R (Miller et al. 2014, 2015; Chambers et al. 2017b, 2024f). As illustrated here, the next steps are to evaluate recent changes in climatic regimes, fire regimes, R&R and any concurrent changes in vegetation using the available long-term data. Field assessments of potential fuel or other vegetation management treatment areas provide the information on the specific ecological types, plant functional types and relative R&R needed to determine the likely effects of treatments on fuels, fire behavior, and ecological response, and select appropriate treatments (Miller et al. 2014, 2015, 2019; Chambers et al. 2024f).

Coupling information on the types and magnitudes of change with knowledge of the use and effectiveness of different management actions provides the information to develop meaningful management strategies (Table 2). Identifying and monitoring areas experiencing shifts in climate regimes and decreases in R&R provides the basis for determining strategies to facilitate change that limits transitions to undesirable ecological states. Maintaining or increasing PFG is essential for ensuring ecosystem recovery and minimizing increases in AFG (Chambers et al. 2014a, 2016, 2019). Actions to sustain PFG include

Table 2 Management strategies to address the changes in climate, wildfire risk, plant functional types, and R&R in dryland shrublands and woodlands of the southwest USA. The rational for the treatments expand on recent reviews (Miller et al. 2019; Chambers et al. 2024g) and guidelines for selecting treatment areas and types (Miller et al. 2014, 2015; Chambers et al. 2024f). *R&R*, ecological resilience and resistance to invasion; *PFG*, perennial forbs and grasses; *AFG*, annual forbs and grasses

Changes in climate regimes and decreases in R&R

• Monitor changes in climate over time to identify areas where shifts in climate regimes are resulting in decreases in R&R and potential loss of climate suitability for native species

• Monitor areas experiencing shifts in climate to identify changes in plant functional type or species composition and determine appropriate management strategies

• Seed or transplant species adapted to the new climate regimes, such a C4 grasses, following wildfire or treatment to aid transitions

• Reduce livestock use where changes are occurring to prevent loss of vulnerable PFG in transitions

Depletion of PFG and increases in AFG in shrublands and woodlands

• Improve management of livestock grazing (season, duration, intensity) to maintain or increase C3 PFG and facilitate transitions to C4 PFG

• Use herbicides, seeding, and transplanting in areas with depleted PFG (< 10 to 15% cover) and high AFG cover to increase resistance, decrease fire risk, and lower fire transmission to high value resources

• Conduct field assessments prior to fuel and other treatments to determine if PFG are depleted and if herbicides, seeding, or transplants are needed to promote recovery and transitions to new climate regimes

Increases in shrub cover in shrublands

• Use woody fuel treatments to reduce the abundance or alter the distribution of woody fuels, decrease the risk of high severity wildfire, and maintain R&R and biological diversity

o Consider patchy prescribed fire, mowing, or mastication in shrublands with high cover of shrubs (> about 18%)

o Assess the R&R of planning areas to determine if prescribed fire (≥ moderate resilience) or mechanical treatments (≥ moderately low resilience) are appropriate

o Conduct field assessments of understory vegetation prior to treatment to determine if PFG and other high value species are depleted and if herbicides, seeding, or transplants are needed to promote recovery and transitions to new climate regimes

Pinyon and juniper expansion into shrublands and infilling in woodlands

• Use woody fuel treatments in shrublands experiencing pinyon and juniper expansion to reduce the abundance or alter the distribution of woody fuels, decrease the risk of high severity crown fire, and maintain R&R and biological diversity

o Prioritize class 1 and 2 tree cover for prescribed fire or mechanical treatments to address transition into class 3, prevent loss of shrublands, and decrease risk of high severity crown fire

o Assess the R&R of planning area to determine if prescribed fire (≥ moderate resilience) or mechanical treatments (≥ moderately low resilience) are appropriate

o Selectively thin (cut, pile, and burn; mastication) class 3 to address ongoing increases in tree cover

• Selectively thin woodlands to address infilling in areas associated with high value resources to reduce ladder fuels and the potential for high severity crown fires

• Conduct field assessments of understory vegetation prior to treatments to evaluate if PFG and other high value species are depleted and if herbicides, seeding, or transplants are needed to promote recovery and transitions to new climate regimes

• Consider habitat for wildlife, such as mule deer and pinyon jays, in selecting treatment areas

improved livestock management, restoring areas dominated by AFG, and seeding and transplanting following wildfire and vegetation management treatments with C4 and other species adapted to new climatic regimes. Woody fuel treatments can be used to reduce the abundance or alter the distribution of woody fuels, decrease the risk of high severity wildfire, and maintain higher R&R and biological diversity. The severity of the treatments, and thus the use of prescribed fire or mechanical removal, depends on the areas relative R&R (Chambers et al. 2023c, 2024g). Prescribed fire is more appropriate in areas with higher R&R and adequate PFG for recovery, while mechanical treatments are best suited to areas with lower R&R and depleted PFG. Field assessments of potential treatment areas provide the information to determine if PFG are depleted and if herbicides, seeding, or transplants are needed to promote recovery and transitions to new climate regimes (Miller et al. 2014, 2015; Chambers et al. 2024f). Shrublands experiencing rapid pinyon-juniper expansion and infilling with class 1 and 2 tree cover should be among the highest priorities for fuel treatments to address the transition from class 2 to class 3, prevent loss of shrubland ecosystems, and decrease the risk of high severity crown fire in the future (Chambers et al. 2024f, g). To address the large increases in class 3 tree cover in shrublands and infilling in juniper-pinyon and pinyon-juniper ecological types, selective thinning (cut, pile, and burn; mastication) in areas with high value resources, such as infrastructure and communities as well as pinyon jay populations and old growth woodlands, can be used to reduce ladder fuels and the potential for high severity crown fires (Chambers et al. 2024g). Selective thinning in these areas also may improve pinyon-juniper resilience to drought, pathogens, and insects (Redmond et al. 2023).

Conclusions

We showed that climate warming in southwest drylands has been associated with concurrent changes in vegetation and fuels and decreases in R&R. The approach that we used allows managers to identify the ongoing changes at management appropriate scales. We suggest climate smart management strategies designed to anticipate the ongoing changes associated with climate warming and to help direct ecosystems (e.g., Millar et al. 2007; Aplet and McKinley 2017; Schuurman et al. 2022) into conditions that can decrease fire risk, reduce vulnerability to climate change, and increase resistance to plant invasions. Well-conceived management treatments can be used to facilitate change that limits transitions to undesirable ecological states and prevents collapse of ecosystem functions and services.

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1186/s42408-025-00388-x.

Supplementary Material 1.

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Authors' contributions
JCC led the writing of the manuscript and JLB, SK, and MCR assembled the data, conducted the analyses, and executed the figures. JLB, SK, RFM, SB, RB, RWB, MM, MCR, VT, and SGY contributed to writing the manuscript. All authors read and approved the final manuscript.

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Data availability
The datasets on the climate regimes and resilience and resistance indicators for the southwest drylands are available in the following data archives. Contact Jeanne Chambers (Jeanne.chambers@usda.gov) for access to the additional data developed for this manuscript.
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Declarations

Ethics approval and consent to participate
Not applicable.

Consent for publication
Not applicable.

Competing interests
The authors declare that they have no competing interests.

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