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The impacts of wildfires of different burn severities on vegetation structure across the western United States rangelands

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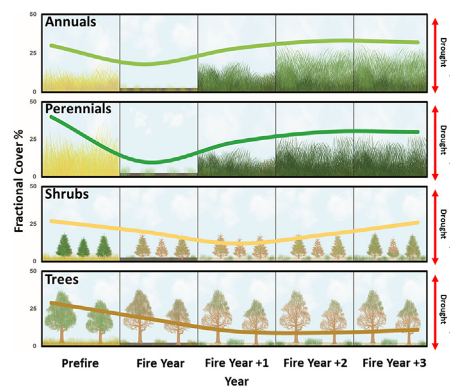
HIGHLIGHTS

- Novel rangeland fractional cover data enabled large-scale assessment of fire impacts.
- Timing in responses to rangeland wildfires differ among plant functional types.
- High severity wildfires led to the largest cover decrease of plant functional types.
- Moist prefire conditions led to greater decreases in herbaceous cover.
- Dry prefire conditions resulted in greater decreases in woody cover.

GRAPHICAL ABSTRACT



Environmental conditions, in combination with amount and type of prefire vegetation cover, influences burn severity and vegetation recovery after wildfire

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ABSTRACT

Large wildfires have increased in western US rangelands over the last three decades. There is limited information on the impacts of wildfires with different severities on the vegetation in these rangelands. This study assessed the impacts of large wildfires on rangeland fractional cover including annual forbs and grasses (AFG), perennial forbs and grasses (PFG), shrubs (SHR) and trees (TREE) across the western US, and explored relationships between changes in fractional cover and prefire soil moisture conditions. The Expectation Maximization (EM) algorithm was used to group wildfires into nine clusters based on the prefire rangeland fractional cover extracted from the Rangeland Analysis Platform. The Standardized Precipitation Evapotranspiration Index (SPEI) with various lag scales from the Gridded Surface Meteorological (GRIDMET) dataset was used to represent antecedent soil moisture conditions. The results showed generally that fractional cover decreased most for AFG and PFG during the fire year, one year postfire for SHR, and two years postfire for TREE. High severity wildfires led to the greatest decrease in cover for all plant functional types, while low severity wildfires caused the least decrease in the functional type cover in most cases, though some variations existed. Furthermore, the impacts of wildfires on vegetation cover were greater in woody (SHR and TREE) types than in herbaceous (AFG and PFG) types. Significant negative correlation existed between percent changes in AFG and PFG cover and SPEI indicating higher prefire soil moisture conditions likely increased fine fuel loads and led to a larger decrease in AFG and PFG cover following wildfires. Significant positive correlation existed between percent changes in SHR and TREE cover and SPEI indicating drier prefire conditions resulted in larger decreases in SHR and TREE cover following wildfires. These findings help better understand the impacts of wildfires on rangelands and provide insights for rangeland management.

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1. Introduction

Wildfires have become more frequent and larger across the western United States (Dennison et al., 2014; Westerling et al., 2006), which affect many ecosystem processes such as vegetation succession trajectories (Barrett et al., 2011), carbon emissions (Engle et al., 2012; Ghimire et al., 2012) and nutrient cycling (McKenzie et al., 2011; Parks et al., 2018). Rangelands, which support grasses, grasslike plants, forbs or shrubs in arid and semiarid regions (Havstad et al., 2009; Mitchell, 2000; Reeves et al., 2015), account for about 30 % of the US land cover (Havstad et al., 2009). During the last 30 years, there has been a significant increasing trend in area burned over western US rangelands (Li et al., 2021; Stasiewicz and Paveglio, 2018). The increasing fire activities could result in changes in land cover such as the loss of native habitats (Arkle et al., 2014) and increases in plant invasion (Ellsworth et al., 2016; Epanchin-Niell et al., 2009). The responses of different rangeland ecosystems to wildfires vary because of the heterogeneous characteristics of rangelands (Fuhlendorf et al., 2017). For example, sagebrush ecosystems, which provide critical habitats for sagebrush-associated wildlife species (e.g., sage grouse), have been reduced due to the increasing land area damaged by fire events (Davies and Bates, 2017; Mahood and Balch, 2019). Moreover, frequent fires in sagebrush ecosystems can promote the dominance of invasive grasses (e.g., *Bromus tectorum*) in sagebrush systems, which may further decrease the cover of native plants (Davies et al., 2011). For mesquite (*Prosopis glandulosa*) ecosystems, plants are generally top-killed by fire and then resprout from underground buds (Heitschmidt et al., 1988), therefore, fire mortality is low (Steinberg, 2001). Given these differential effects of wildfires across ecosystems in the Western US, it is fundamental and critical to evaluate the impacts of wildfires on different rangeland ecosystems to aid land managers and policy makers in evaluating potential impacts, developing appropriate strategies to minimize impacts, and improving capabilities for maintaining and restoring rangelands.

Burn severity, which represents the magnitude of ecological changes caused by fires (Key and Benson, 2006), has been identified as a critical factor in determining ecological effects of fires on ecosystems (Tanase et al., 2011). Burn severity can affect postfire vegetation composition and structure (Van Wagtendonk et al., 2018; Westerling et al., 2006) as well as potentially increase the degradation process through alteration of soil properties (García-Llamas et al., 2019). The majority of burn severity studies have focused on high severity wildfires in forest ecosystems due to numerous negative impacts of fires on human safety and infrastructure within or near these ecosystems (Miller and Ager, 2012), as well as lethal effects of fire on some tree species (Collins et al., 2007; Stevens et al., 2017). Less attention has been given to study of burn severity on rangelands across the western US. Given the increasing trends in rangeland area burned at all levels of severity across the western US (Li et al., 2021), and the heterogenous nature of the rangeland ecosystems across this expansive area, attention is needed as the heterogenous rangeland ecosystems may have diverse and differential responses to the variations in burn severity (Lauvaux et al., 2016). For example, in low elevation sagebrush ecosystems, low severity fires may come with the increased risk of exotic plant invasion, which further increases the subsequent fire risk (Davies et al., 2011). In juniper-invaded ecosystems at middle to high elevations, fires with low or moderate burn severity may be beneficial in reducing juniper and increasing herbaceous cover (Miller et al., 2000). In general, wildfires with low and moderate burn severity can be beneficial in reducing fine fuel loads, removing invasive grasses and releasing nutrients (Fernández-García et al., 2019; Hessburg et al., 2015; Parks and Abatzoglou, 2020; Perry et al., 2011), while wildfires with high burn severity may have long-term consequences on ecosystem structure, function and services (Dillon et al., 2011; Lentile et al., 2007). Evaluating the impacts of wildfires on vegetation structure across the diverse rangeland ecosystems in the western US could aid in identifying the response of these ecosystems to wildfires under different levels of burn severity, thus providing capabilities for assessing prefire vegetation conditions to predict post-fire vegetation outcomes and insights for improved rangeland management.

Patterns of burn severity in heterogeneous rangelands have been found to be controlled by prefire vegetation structure, in addition to weather conditions on a larger scale (Holsinger et al., 2016; Pausas and Ribeiro, 2013). Vegetation structure, considered here as the fractional cover of different plant functional types, is a good indicator that reflects vegetation responses to wildfires. Prefire vegetation cover is a significant factor for assessing wildfire impacts on vegetation and subsequent regrowth after wildfires (Collins et al., 2007; Miller et al., 2013; Stephens, 2001). The recent introduction of high resolution (30-m) rangeland fractional cover products was developed using field data and machine-learning/deep-learning techniques for much of the western US (Allred et al., 2021; Jones et al., 2018; Rigge et al., 2021) enabling assessment and monitoring of rangeland conditions at large geographic scales. These fractional cover products also offer an excellent opportunity to assess wildfires effects on different plant functional types such as annual forbs and grasses, perennial forbs and grasses, shrubs, and trees, and further identify the responses of these plant functional types to wildfires having different levels of burn severity (Allred et al., 2021; Jones et al., 2018).

Weather is another significant driver of wildfire activities (Littell et al., 2009; Liu and Wimberly, 2016; Mueller et al., 2020), and climate variables such as precipitation and temperature, which regulate regional water balance, can influence fuel flammability and the accumulation of the fuel load (Westerling et al., 2006). Warmer and drier conditions in recent decades have increased area burned across the western US (Dillon et al., 2011; Keyser and Westerling, 2017; Morgan et al., 2008; Westerling, 2016), and understanding the influences of prefire weather conditions on wildfire burn severity could be beneficial for predicting future fire behaviors and provide information to develop appropriate management practices (Mueller et al., 2020). Prior precipitation is positively correlated with areas burned as moist conditions increase fuel loads and connectivity, and result in more fuels consumed by fires (Krawchuk and Moritz, 2011; Littell et al., 2009). Climatic variables along with lagged effects (the length of accumulated water deficits) can be used to assess prefire weather conditions (Liu et al., 2017; Vicente-Serrano and López-Moreno, 2005). The Standardized Precipitation Evapotranspiration Index (SPEI) (Vicente-Serrano et al., 2010), which is a good indicator of soil moisture condition (Tian et al., 2018), can be used to examine the relationship between wildfire effects and weather conditions. Unlike the Palmer Drought Severity Index (PDSI) which has a fixed time scale (Guttman, 1998), the multiscalar nature of SPEI provides an opportunity to identify the appropriate time scales at which different plant functional types respond to wildfires. Understanding burn severity, soil moisture conditions, and prefire vegetation structure can help predict with more accuracy how vegetation could respond to wildfires.

The goal of this study was to evaluate the impacts of wildfires of different burn severities on vegetation structure across the western United States rangelands. This is done through examining the changes in fractional cover of plant functional types in response to fires with different levels (low, moderate and high) of burn severity in all large wildfires ($\geq$ 405 ha) across western US rangelands from 1985 to 2017. The correlations between prefire soil moisture conditions using the SPEI drought index as a soil moisture proxy (Barnard et al., 2021; Wang et al., 2015), and percent changes in the fractional covers resulting from wildfires were also evaluated. The objectives of this study were to: 1) identify the timing of largest changes in different rangeland fractional covers (AFG, PFG, SHR and TREE) following wildfires with different severities; 2) assess the impacts of wildfires of different severities on post-fire rangeland fractional covers; and 3) evaluate the relationship between prefire soil moisture conditions and the impacts of wildfires on post-fire vegetation in terms of fractional cover change.

2. Material and methods

2.1. Fire, vegetation and climate datasets

The large wildfire perimeters and burn severity mosaics were obtained from the Monitoring Trends in Burn Severity (MTBS) project, which mapped fires ≥405 ha in the western and ≥202 ha in the eastern halves of the United States since 1984 (Eidenshink et al., 2007). The MTBS product uses differenced Normalized Burn Ratio (dNBR) and the relative differenced Normalized Burn Ratio (RdNBR) derived from prefire and postfire 30-m Landsat imageries to delineate perimeters and area burned with different levels of severity (Eidenshink et al., 2007). MTBS also employs the Composite Burn Index (CBI) (Key and Benson, 2006), which is based on the visible alteration of vegetation and soil, for correlating with NBR and dNBR to establish the regional threshold models for low, moderate and high burn severity classes (Eidenshink et al., 2007; Miller and Thode, 2007). Field data collection for CBI includes collection of information for five strata including substrates; herbs, low shrubs and trees (<1 m); tall shrubs and trees (>1 = and <5 m); intermediate trees (subcanopy, polesized); and big trees (upper canopy) (Key and Benson, 2006). The CBI ranges from 0 to 3 and the values of 1.25 and 2.25 are used as the thresholds between low and moderate burn severity and moderate and high burn severity, respectively (Miller and Thode, 2007). The cross-calibration of burn severity thresholds is applied by MTBS for fires covering different landscapes (forests, shrublands, grasslands) to maintain consistency (Miller and Thode, 2007). The areas burned with low severity are defined as areas where all strata are slightly altered from the prefire state, and prefire plants are generally still viable (Miller and Thode, 2007). The areas burned with high severity are defined as the completely charred or consumption of aboveground biomass for herbaceous plants and shrubs, and trees exhibit >75 % mortality with 100 % crown char and significant branch loss (Eidenshink et al., 2007; Key and Benson, 2006). The areas burned with moderate severity will exhibit traits between areas burned with low and high severity, and numerous potential combinations of distinct low and high indicators may occur to yield a moderate classification overall within the minimum mapping unit (Miller and Thode, 2007). The MTBS product was used in this study to analyze large wildfires (≥405 ha) having low, moderate, and high burn severity across 17 western states including AZ, CA, CO, ID, KS, MT, ND, NE, NM, NV, OK, OR, SD, TX, UT, WA, and WY.

The 30-m annual rangeland fractional plant cover product, which spans the western United States since 1984, was obtained from Rangeland Analysis Platform (RAP; https://rangelands.app/) (Allred et al., 2021; Jones et al., 2018). For the RAP product, around 60,000 vegetation monitoring plots collected by the Bureau of Land Management (BLM) Assessment, Inventory, and Monitoring (AIM), and Natural Resources Conservation Service (NRCS) National Resources Inventory (NRI) were used to train a temporal convolutional network model to predict fractional land cover and to validate estimates of fractional land cover predicted by the model (Allred et al., 2021; Jones et al., 2018). RAP Cover 2.0 used the temporal convolutional network approach because it can better handle satellite time series data and produce a multivariate response for cover (Allred et al., 2021). Moreover, the RAP cover 2.0 included more plot data and Landsat measurements and removed the non-Landsat predictors used in the previous version (Allred et al., 2021). The RAP product includes fractional cover types representing annual forbs and grasses (AFG), perennial forbs and grasses (PFG), shrubs (SHR), trees (TREE), litter (LTR) and bare ground (BG). In this study, only the fractional cover for plant functional types (AFG, PFG, SHR and TREE) from RAP 2.0 were used to assess the impacts of wildfires on vegetation structure (defined as the percent fractional cover of each plant functional type).

To delineate rangeland pixels from other land cover/use types (e.g., forests, crops, developed areas, etc.), the coterminous U.S. rangelands (30 m resolution; circa 2011) product (Reeves and Mitchell, 2011) was used, which was consistent with the rangeland land cover that was used for rangeland delineation in the RAP dataset. One-year prefire and three-year postfire fractional cover values for rangeland pixels within wildfire boundaries were extracted from the RAP product to evaluate the impacts of wildfire on vegetation composition and structure. Wildfires were evaluated for those that occurred from 1985 to 2017. As the coterminous U.S. rangeland map (Reeves and Mitchell, 2011) used by RAP product lacked details on the vegetation types comprising the 30-m pixels, therefore the Existing Vegetation Type (EVT) product from the LANDFIRE program, was used to represent rangeland vegetation types where wildfires occurred. The EVT product provides a consistent characterization of vegetation (Rollins, 2009), and offers a representation of rangeland vegetation types developed by the Society for Range Management (SRM) (Shiflet, 1994). The EVT 2001 product, which was used as the input for the western coterminous US rangeland map (Reeves and Mitchell, 2011) described above, was used in this study.

The Standardized Precipitation Evapotranspiration Index (SPEI), which accounts for both precipitation and temperature effects on soil moisture, was used in this study as a proxy for soil moisture (Vicente-Serrano et al., 2015). SPEI is a multiscale index representing the climatic water balance between precipitation (P) and potential evapotranspiration (PET) (Vicente-Serrano et al., 2010, 2015). The estimation of the SPEI values includes the calculation of accumulated differences between P and PET at different time scales and normalization of the time series based on nonparametric methods in which the probability distributions of the data samples are empirically estimated (Hao et al., 2014; Turco et al., 2019; Vicente-Serrano et al., 2010). 1, 3, 6, 9, 12 and 24-month SPEI data, which represents the accumulated weather conditions for 1, 3, 6, 9, 12 and 24 months, were extracted from the 4-km Gridded Surface Meteorological (GRIDMET) dataset (Abatzoglou, 2013). The SPEI dataset has a 5-day timestep (Abatzoglou, 2013; Smith et al., 2021), and the SPEI date nearest the fire date was used to represent the soil moisture conditions for each wildfire. The SPEI dataset was processed in Google Earth Engine (Gorelick et al., 2017), and SPEI layers were resampled to 30 m to match the spatial resolution of RAP data. The moisture condition categories based on SPEI values are listed in Table 1 (Li et al., 2015; Potopová et al., 2015).

2.2. Methods

All wildfires were masked using the coterminous U.S. rangelands 2011 product (Reeves and Mitchell, 2011) to identify rangeland wildfires within the western US. A total of 9741 wildfires were identified and extracted from the MTBS dataset for further analysis. Google Earth Engine (Gorelick et al., 2017) was used to calculate mean fractional cover within each wildfire perimeter for four plant functional types (AFG, PFG, SHR and TREE) represented in the RAP dataset. The prefire vegetation cover is an important factor influencing wildfires and their severities (Keyser and Westerling, 2017), and clustering wildfires based on the prefire vegetation cover could help better identify different patterns of prefire vegetation structure that can result in wildfires. To accomplish this, the Expectation Maximization (EM) algorithm (Dempster et al., 1977; McLachlan and Krishnan, 2007) was used to cluster rangeland wildfires based on plant functional types in the prefire year (FY-1). The EM algorithm contains an “E step” where the conditional expectation of the complete data log-likelihood and the parameter estimates are calculated given the observed data and an “M step” where the parameters that maximize the expected log-likelihood from the E step are determined (Fraley and Raftery, 2002). The selection of an optimal number of clusters was based on the Bayesian Information Criterion (BIC) (Fraley and Raftery, 2002; Schwarz, 1978). The mean value of fractional cover of each plant functional type (AFG,

Table 1
The soil moisture conditions based on SPEI.

Soil moisture conditions	SPEI
Extremely wet	≥2.00
Severely wet	[1.50,2.00)
Moderately wet	[1.00,1.50)
Normal	(−1.00,1.00)
Moderately drought	(−1.50, −1.00]
Severely drought	(−2.00, −1.50]
Extremely drought	≤−2.00

Table 2
The dominant and subdominant plant functional types and the major existing vegetation types for each cluster. The detailed major existing vegetation types based on areas burned with different levels of burn severity can be found in Table S1.

			Plant Functional Types (PFTs)							
			AFG (%)		PFG (%)		SHR (%)		TREE (%)	
Cluster	Dominant Vegetation Cover	Major Vegetation Types	Mean	SE	Mean	SE	Mean	SE	Mean	**SE**
1	Woody/Perennial Herbaceous	Mountain Big Sagebrush, Western Live Oak, Wyoming Big Sagebrush	2.72	0.46	21.08	0.11	25.32	0.33	19.80	0.19
2	Tree/Annual Herbaceous	Chamise Chaparral, Montane Shrubland, Canyon Live Oak	16.61	0.33	14.95	0.21	10.84	0.22	31.06	0.58
3	Perennial Herbaceous /Tree	Mesquite, Interior Ponderosa Pine, Rough Fescue-Bluebunch Wheatgrass	5.18	0.21	30.65	0.34	10.57	0.07	24.33	0.27
4	Annual Herbaceous /Shrub	Blackbush, Chamise Chaparral, Scrub Oak Mixed Chaparral	24.21	0.15	7.12	0.38	18.70	0.17	7.59	0.07
5	Perennial Herbaceous /Shrub	Wyoming Big Sagebrush, Big Sagebrush-Bluebunch Wheatgrass, Mountain Big Sagebrush	8.78	0.23	39.01	0.29	14.35	0.16	5.30	0.01
6	Mixed Annual & Perennial Herbaceous/Shrub	Wyoming Big Sagebrush, Big Sagebrush-Bluebunch Wheatgrass, Mountain Big Sagebrush	14.03	0.11	28.87	0.40	16.81	0.09	0.84	0.03
7	Mixed Annual & Perennial Herbaceous	Wyoming Big Sagebrush, Introduced Upland Vegetation-Herbaceous, Big Sagebrush-Bluebunch Wheatgrass	33.00	0.06	27.46	0.24	7.59	0.09	0.37	0.26
8	Perennial & Annual Herbaceous/Tree	Blue Grama-Western Wheatgrass, Mesquite, Sand Bluestem-Little Bluestem Dunes	11.95	0.04	60.85	0.32	4.60	0.27	10.44	0.33
9	Perennial Herbaceous	Blue Grama-Buffalograss, Wheatgrass-Bluestem-Needlegrass, Blue Grama-Western Wheatgrass	5.44	0.44	66.43	0.38	5.43	0.09	1.16	0.01

PFG, SHR and TREE) for each wildfire was calculated, and wildfires were clustered using the EM algorithm. Nine clusters were identified, and the mean and standard error (SE) of rangeland fractional cover for plant functional types in each cluster are shown in Table 2. The spatial distribution of each cluster is displayed in Fig. 1.

For each wildfire, years within a five-year window, including prefire year (FY-1), fire year (FY), one year postfire (FY + 1), two years postfire (FY + 2) and three years postfire (FY + 3) were analyzed. The mean values of fractional cover types (AFG, PFG, SHR and TREE) of the five- year window for areas burned with low, moderate, and high severity were calculated, respectively. The percent change in plant functional type cover was calculated for each plant functional type as the differences between prefire and lowest postfire fractional cover for a given plant functional type and then divided by the prefire vegetation fractional cover for the given plant functional type. The calculation of percent change in rangeland fractional cover for wildfires having different severities is shown in Eq. (1).

$$PC_{(i,j,k)} = \frac{\mathrm{Min}\left(Cover_{FY(i,j,k)}, Cover_{FY+1(i,j,k)}, Cover_{FY+2(i,j,k)}, Cover_{FY+3(i,j,k)}\right) - Cover_{FY-1(i,j,k)}}{Cover_{FY-1(i,j,k)}} \times 100\% \tag{1}$$

where i represents cluster (1–9), j represents different fractional covers (AFG, PFG, SHR and TREE), k represents levels of burn severity (Low, Moderate, High).

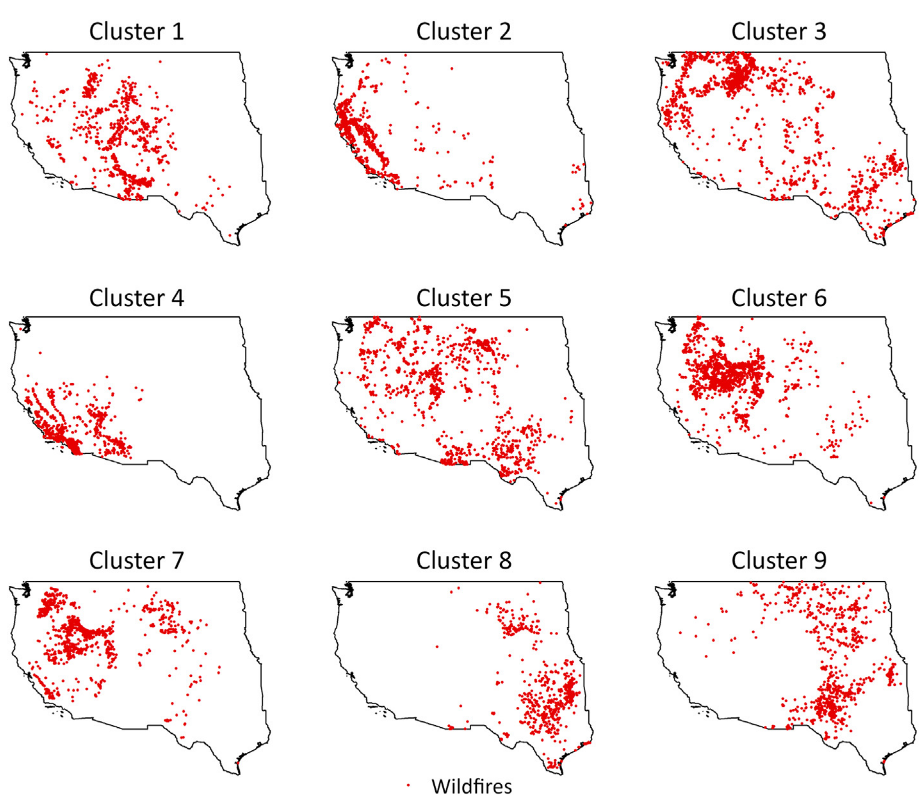


Fig. 1. Nine clusters of rangeland wildfires across western US.

The EVT types for area burned with different levels of burn severity were evaluated if they accounted for >5 % of rangeland area burned within each cluster. The percent of rangeland area burned associated within the EVT type for each cluster was calculated as the percentage of area occupied by the specific vegetation types to the total area burned in a cluster.

The Google Earth Engine platform (Gorelick et al., 2017) was used to extract different time scales SPEI values associated within each wildfire perimeter having low, moderate and high burn severity, respectively. The Pearson correlation matrix was performed between percent change in cover caused by the wildfires with low, moderate, and high burn severity for the plant functional types (AFG, PFG, SHR, TREE) and SPEI associated with different time lags (1, 3, 6, 9, 12 and 24-months). The optimal time scale of SPEI for each plant functional type was determined by the highest number of correlation coefficients that passed the significance test ($p < 0.05$), and only significant correlation coefficients were kept (Li et al., 2015; Russo et al., 2017). All data analysis were performed in R (Team, 2013).

3. Results

3.1. The major vegetation types based on area burned for each cluster

The EM analysis resulted in the formation of nine clusters and their spatial distributions are shown in Fig. 1. Cluster 1 was dominated by woody (SHR + TREE) cover with some perennial herbaceous (PFG) cover (Table 2). For Cluster 2, wildfire sites were characterized as having high Tree/Annual Herbaceous (AFG) cover (Table 2). Perennial Herbaceous/Tree cover were the major fractional cover types that characterized fractional cover in Cluster 3 (Table 2). Wildfire sites in Cluster 4 were dominated by Annual Herbaceous/Shrub types, and sites in Cluster 5 was characterized as being Perennial Herbaceous/Shrub dominated (Table 2). Mixed Annual and Perennial Herbaceous/Shrub characterized the fractional cover for wildfire sites in Cluster 6, and Cluster 7 represented sites dominated by Mixed Annual & Perennial Herbaceous fractional cover (Table 2). Perennial & Annual Herbaceous/Tree represented the dominant cover types across wildfires grouped into Cluster 8, and Cluster 9 was characterized as Perennial Herbaceous cover dominated (Table 2).

3.2. Timing differences of largest changes in rangeland fractional cover following wildfires

Within each cluster's 5-year window (-1 to $+3$ years after wildfires), the lowest AFG and PFG cover occurred during the FY for areas burned with low, moderate and high severities in all clusters except Clusters 1, 2 and 3 which had their lowest AFG and PFG cover during FY + 1 (Fig. 2). For areas burned with low severity in Cluster 9, the lowest value in AFG occurred at FY + 3 (Fig. 2). The SHR cover within the 5-year analysis window had its lowest value in FY +1 in all clusters except in Cluster 1 and 2 for those areas classified as low burn severity where the lowest value occurred during the FY (Fig. 2). The lowest TREE cover occurred at FY + 2 for all clusters with the following exceptions: 1) areas classified as low burn severity in Cluster 1 and 3 which had their lowest TREE cover value during the FY; 2) areas for all burn severity classes in Clusters 4 and 7 and areas burned at low and high severity in Cluster 9 which had their lowest values during FY + 1; and 3) areas classified as moderate severity in Cluster 6 which had its lowest value during FY + 3 (Fig. 2). In general, the cover of all plant functional types decreased in the years after the wildfires with annual and perennial herbaceous species declining most in the fire year and shrubs and trees having their greatest decline in year 1 or year 2 after the wildfires. AFG and PFG cover generally increased quickly after the FY (FY + 1, FY + 2). SHR and TREE cover was slower to recover with cover remaining low or having slight to moderate increases in FY + 2 and FY + 3 (Fig. 2).

3.3. Impacts of wildfires having different severities on rangeland fractional cover

For AFG cover, five out of nine clusters (Cluster 2, 4, 6, 7 and 9) had the largest percent change in AFG cover within areas classified as high severity burns, and the smallest percent change in AFG cover within areas with low severity burns (Fig. 3; Table S2). In general, the clusters with higher AFG cover had the highest percent change in high severity burns. For Cluster 1 and 3, the AFG cover increased most for areas burned with moderate severity. AFG cover increased in areas having high burn severity in Cluster 5, while those areas classified as low and moderate burn severity had decreases in AFG cover. For Cluster 8, the AFC cover decreased most in areas identified as moderate burn severity, and least in areas high severity. PFG cover decreased after wildfires in all clusters except areas burned with low severity in Cluster 4 (Fig. 3; Table S2). Six (Cluster 1, 2, 3, 6, 7 and 9) out of nine clusters had the largest decreases in PFG cover in areas classified as high burn severity, and the smallest percent changes of PFG cover were in areas having low severity burns (Fig. 3; Table S2). For the SHR cover, eight clusters had the largest percent change in cover with areas having high severity burns, and smaller percent change in cover with areas having low severity burns (Fig. 3; Table S2). Among the nine clusters, the SHR cover in Cluster 2 increased after fire compared to declines in cover in the majority of clusters and associated burn severity classes. Across all clusters, TREE cover had the most drastic changes compared to other PFTs in areas having high burn severity. Tree cover in all nine clusters declined after wildfires with the largest decreases in cover within areas classified as high burn severity and smallest declines in areas having low burn severity (Fig. 3; Table S2).

Overall, high burn severity wildfires resulted in larger decreases in fractional cover after wildfires, while low burn severity wildfires had smaller decreases in fractional cover for all plant functional types (AFG, PFG, SHR and TREE). Among all plant functional types, TREE cover had the greatest declines in fractional cover while AFG cover had smaller decreases.

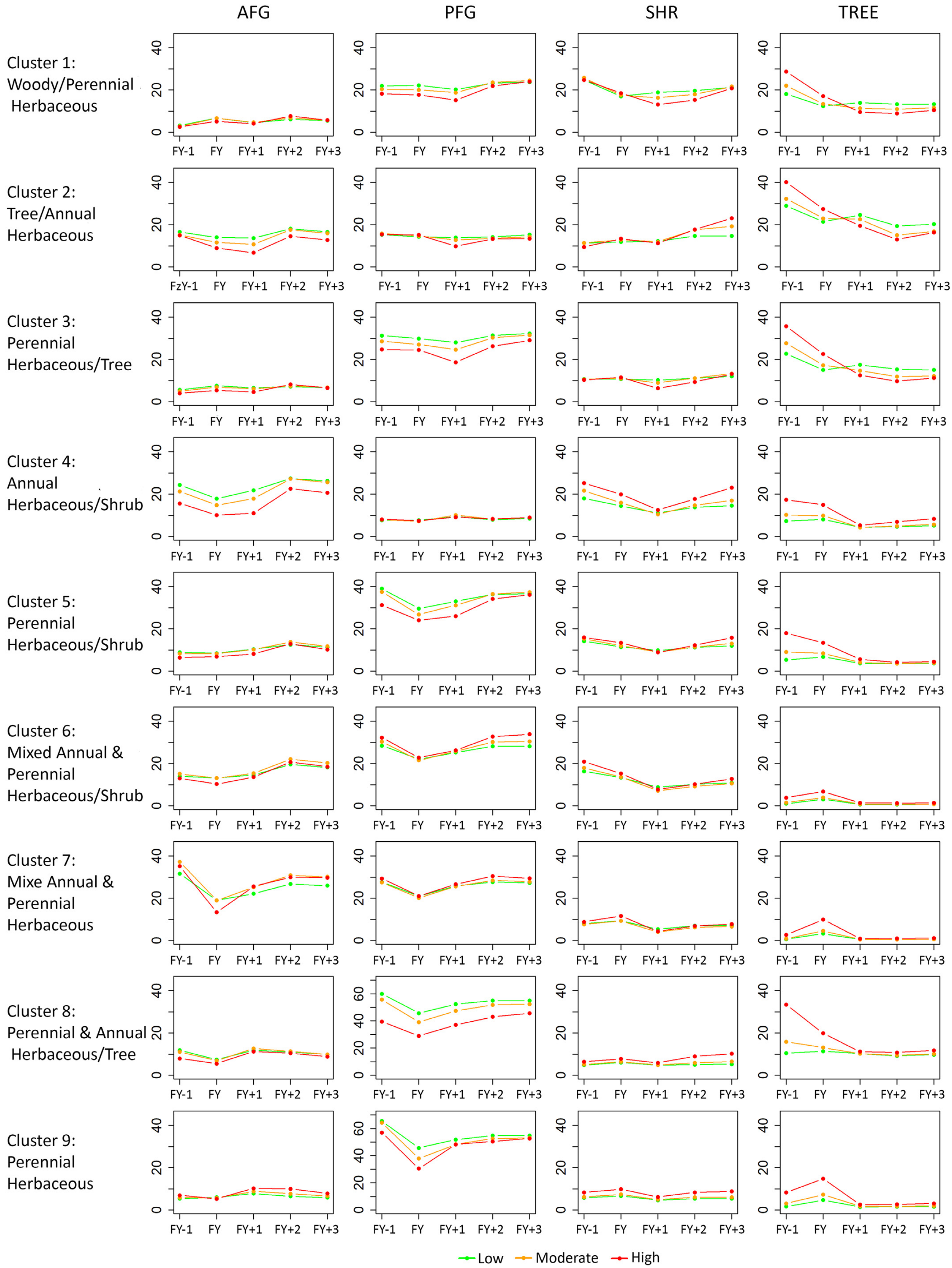
3.4. The relationship between prefire soil moisture conditions and percent change in rangeland fractional cover following wildfires having different severities

Significant correlation existed between percent change in the cover of plant functional types (AFG, PFG, SHR and TREE) and prefire soil moisture conditions as represented by SPEI with different time lags (1, 3, 6, 9, 12 and 24 months) (Fig. S1). In general, the optimal SPEI time scale for AFG was 1 month for areas with moderate and high burn severity, and 12 months for areas with low burn severity (Table 3). For PFG, the optimal time scales were 12, 6, and 1 months for areas having low, moderate and high burn severity, respectively (Table 3). For SHR, the optimal time scale was 6 months for areas having moderate and high burn severity, and 9 months for areas burned with low severity (Table 3). For TREE, the optimal time scale was 12 months for areas having low and high burn severity, and 9 to 12 months for areas having moderate burn severity (Table 3). For areas classified as low burn severity, a majority of the clusters had significant correlation between SPEI with different time lags and percent change in AFG, PFG, SHR and TREE cover, respectively. The majority of significant negative correlations existed between time-lagged SPEI and the percent change in AFG and PFG cover, while the majority of positive correlations existed between time-lagged SPEI and the percent change in SHR and TREE cover (Fig. S1). For areas having moderate and high burn severity classification, a third or more of the clusters had significant correlations between SPEI with different time lags and the percent change in AFG, PFG, SHR and TREE cover, respectively. The correlation patterns for areas with moderate and high burn severity were the similar to the pattern for areas having low burn severity (Table 3; Fig. S1).

4. Discussion

4.1. The response of rangeland fractional cover types to wildfires

The responses of different rangeland fractional cover types to wildfires varied. In general, the AFG and PFG cover tended to decrease immediately following the wildfires likely due to the consumption of herbaceous biomass by the wildfires (Miller et al., 2013; Neary and Leonard, 2020; Wragg et al., 2018). This is consistent with the findings of Donovan et al.


AFG
PFG
SHR
TREE
Cluster 1: Woody/Perennial Herbaceous
Cluster 2: Tree/Annual Herbaceous
Cluster 3: Perennial Herbaceous/Tree
Cluster 4: Annual Herbaceous/Shrub
Cluster 5: Perennial Herbaceous/Shrub
Cluster 6: Mixed Annual & Perennial Herbaceous/Shrub
Cluster 7: Mixe Annual & Perennial Herbaceous
Cluster 8: Perennial & Annual Herbaceous/Tree
Cluster 9: Perennial Herbaceous
FY-1
FY
FY+1
FY+2
FY+3
Low
Moderate
High

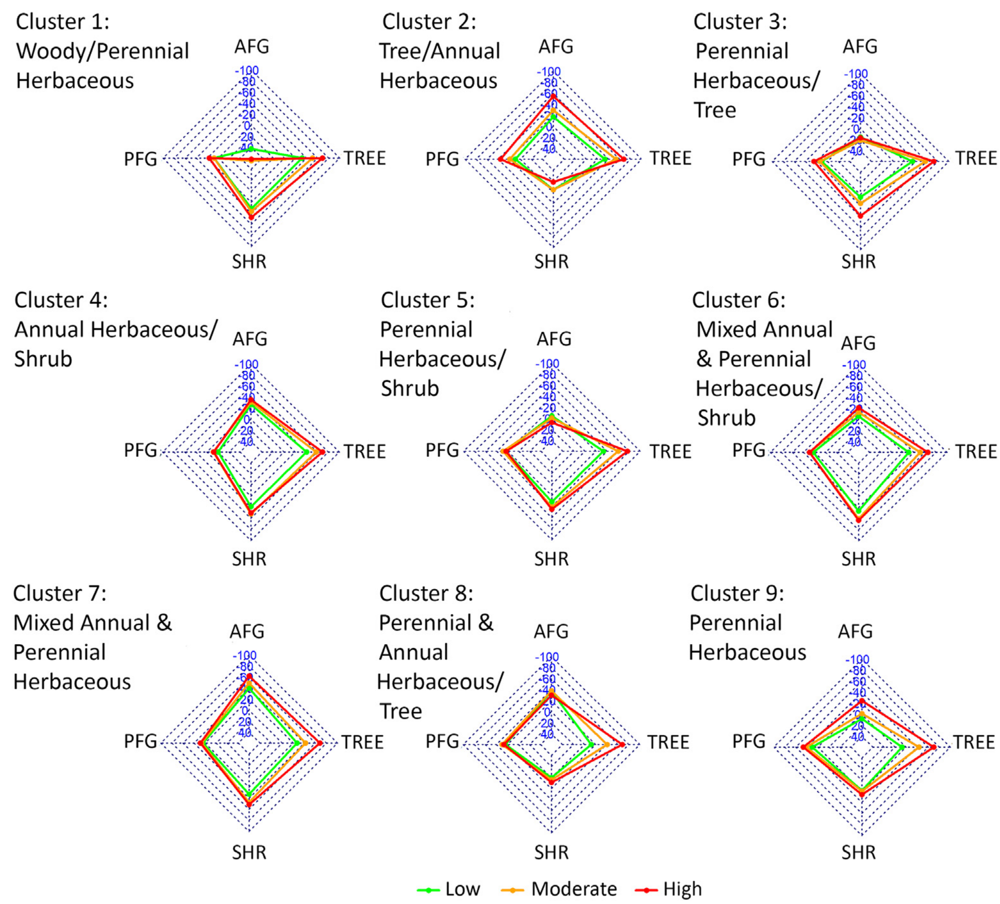


Fig. 3. The percent change in cover of plant functional types including annual forbs and grasses (AFG), perennial forbs and grasses (PFG), shrubs (SHR), and trees (TREE), for the nine clusters. The calculation of percent change in cover was shown in Eq. 1.

(2020) that showed an instant decrease in AFG and PFG cover after wildfires in the Great Plains (Donovan et al., 2020). In this study, higher AFG cover (~33 %) in Cluster 7 (Mixed Annual & Perennial Herbaceous dominated) decreased quickly following the wildfires. Comparable results were found for wildfires in the interior west that had the same vegetation type (Zouhar, 2008). Wyoming Big Sagebrush and Big Sagebrush-Bluebunch Wheatgrass were also dominant EVTs within Cluster 7. Given the moderate to low average shrub cover (7.59 %) across wildfire sites in this cluster (Table 2), these may represent sites where invasive annual grasses have increased at the expense of shrubs. Past research on sagebrush rangelands in the Great Basin has indicated that wildfires can reduce sagebrush (a fire intolerant shrub) cover and invasive annual plants such as cheatgrass (*Bromus tectorum*) can increase after fires, resulting in landscapes being more prone to fire because of the increase in fine fuels (Chambers et al., 2019; D'Antonio and Vitousek, 1992; Uden et al., 2019). Cluster 6 (Mixed Annual & Perennial Herbaceous/Shrub dominated) also had a large number of wildfire sites that occurred in sagebrush rangelands as indicated by the dominant EVTs (Wyoming Big Sagebrush, Big Sagebrush-Bluebunch Wheatgrass, Mountain Big Sagebrush; Table 2). This cluster may represent sagebrush rangelands where fire had not yet occurred, or previous fires may have had low severity given that AFG cover was lower and shrub cover was higher compared to Cluster 7 (Table 2). These sites in Cluster 6 could be vulnerable to *Bromus* invasion as indicated by the increase in AFG cover after wildfire (Fig. 2).

Wildfire sites in Cluster 8 (Perennial & Annual Herbaceous/Trees dominated) and Cluster 9 (Perennial Herbaceous dominated) had high PFG cover (> 60 %). The majority of these sites occur in the Great Plains region of the US (Fig. 1). PFG cover in Clusters 8 and 9 decreased following the wildfires, losing 30 to 50 % of the prefire cover (Fig. 2). However, PFG cover rebounded quickly in the first year after fire and had similar to slightly lower PFG cover 3 years after fire (Fig. 2). Clusters 8 and 9 had EVTs representing grassland vegetation types (Table 2) which are generally top-killed by fires with consumption of aboveground leaves and culms. The cover of these species generally decreases immediately following the wildfires but may recover quickly in the following years (Bailey, 1978; Blaisdell and Holmgren, 1984; Daubenmire, 1968; Hulbert, 1988; Quinnild and Cosby, 1958; Vogl, 1974; Wright and Klemmedson, 1965). Many of these grasses have buds immediately below the soil surface that can quickly resprout after fire (Daubenmire, 1968; Russell et al., 2015), thus allowing these grasses to replace canopy cover after fires if soil moisture and growing conditions are adequate. Wildfire sites in Cluster 8 had higher TREE cover than Cluster 9 (10.44 % vs. 1.16 %, respectively), which may be indicative of rangelands where woody plant encroachment has occurred. In the Great Plains, *Juniperus* species have been encroaching on grasslands and increasing cover at the expense of grasses (Leis et al., 2017; Van Auken, 2007; Van de Water et al., 2003). In the southern Great Plains, mesquite (*Prosopis glandulosa)* has expanded its cover in native grasslands (Ansley et al., 2017; Archer, 1989; Van Auken, 2000) in recent years. Mesquite and

Fig. 2. The rangeland fractional cover from prefire year (FY-1) to three years postfire (FY + 3) of different plant functional types including annual forbs and grasses (AFG), perennial forbs and grasses (PFG), shrubs (SHR), and trees (TREE), for each cluster.

Table 3
The optimal SPEI time lags for percent change in plant fractional cover with different severities and different plant functional types. A "+" in the parenthesis next to the plant functional type/burn severity combination which indicates that the percent change in cover of that plant functional type decreases as lagged prefire soil moisture (SPEI) decreases (i.e., positive correlation). A "- "shows indicates that the percent change in cover of that plant functional type decreases as lagged prefire soil moisture (SPEI) increases (i.e., negative correlation). See Fig. S1 for all combinations of time-lagged SPEI and percent change in area burned for each burn severity level and each plant cover type.

Plant functional types	Burn severity		
	Low	Moderate	High
AFG	SPEI-12 (−)	SPEI-1 (−)	SPEI-1 (−)
PFG	SPEI-12 (−)	SPEI-6 (−)	SPEI-1 (−)
SHR	SPEI-9 (+)	SPEI-6 (+)	SPEI-6 (+)
TREE	SPEI-12 (+)	SPEI-9,12 (+)	SPEI-12 (+)

certain Juniperus species (redberry juniper, *Juniperus pinchotii*) have buds that are below the soil surface that allow these species to resprout after fire, thus allowing these species to recover over time after fires (Starns et al., 2022; Steuter and Britton, 1983; Wright et al., 1976).

The SHR cover decreased most at one year, postfire. Cluster 1 and 4 had higher shrub cover compared with other clusters, and the major shrub vegetation types were Mountain Big Sagebrush, Wyoming Big Sagebrush, Blackbush and Chamise Chaparral (Table 2). Mountain Big Sagebrush and Wyoming Big Sagebrush which have highly flammable wood, bark, and foliage (Brown, 1982; Mueggler, 1976), are typically killed by wildfires resulting in the decrease in cover of these species (Baker, 2006). Stands of blackbush are subject to wildfires which can spread easily due to the dense and resinous foliage on these shrubs (Bowns and West, 1976), and cover of blackbush has been found to decrease more one year after the wildfire (Paysen et al., 2000). In examining burn severity within wildfire perimeters that included blackbush EVTs in the Mojave Desert, Klinger et al. (2019) found reductions in woody plant cover with increasing burn severity and noted that this could lead to state changes in vegetation composition and structure. Chamise Chaparral, a fire-sensitive species, has physical adaptations enhancing its flammability (Hanes, 1971; Mutch, 1970) and its cover decreased most at one year, postfire in California (Storey et al., 2016) similar to that seen in this study. In general, TREE cover remained low at three years postfire. The TREE cover was higher in Clusters 2 and 3 compared with other clusters and tree species within the Clusters (Table 2) have characteristics that can increase chances of crown damage or top kill that would reduce cover after wildfires. For example, the reduced tree cover in Interior Ponderosa Pine could be the result of intense radiant heat which can kill this species of tree (Alexander, 1986). Canyon Live Oak is easily ignited and top-killed by fires leading to decrease in its cover (Plumb, 1981).

4.2. Differential impacts of wildfires with different burn severities on rangeland fractional cover

Overall, high severity wildfires led to greater decreases in the cover of the different plant functional types, and low severity wildfires resulted in smaller decreases. Wildfires having high burn severity consume substantial amounts of aboveground fuels, especially fine fuels from herbaceous plants and litter, leading to larger decreases in rangeland cover. In western US rangelands, the various physical and chemical characteristics of the shrubs and trees (Table S3) can induce crown fires resulting in a greater decrease in woody plant cover. For example, the sagebrush species such as Mountain Big Sagebrush and Wyoming Big Sagebrush are very flammable and have slow recovery times following fires (Baker, 2006, 2011; Daubenmire, 1975). The thin and flakey bark of Canyon Live Oak can ignite and carry fires up the trunk and into the crown resulting in high severity fires (Griffin, 1978; Plumb, 1981). The Scrub Oak Mixed Chaparral, whose branches are close to the burning surface fuels, provides good sources of woody fuels, and are conductive to crown fires leading to large decreases in woody cover of these species after wildfires (Harper, 1985). The Chamise Chaparral, which has structural characteristics that include large amounts of small-stemmed materials, along with the chemical composition of the foliage having high energy ether extracts, can induce high burn severity fires that can reduce cover of woody species in this vegetation type (Countryman, 1970; Philpot, 1977).

Low and moderate severity wildfires led to a decrease in the covers of plant functional types, and the extent of the decrease in herbaceous (i.e., AFG and PFG) cover was lower than that exhibited by woody species (i.e., SHR and TREE) cover. Low severity wildfires may not consume all of the plant biomass (especially in trees and shrubs), and some plants (or portions of plants) can still be standing which can lead to smaller decreases in plant cover, whereas moderate severity wildfires consume more aboveground fuels resulting in larger decreases in plant cover (Neary and Leonard, 2020; Switky, 2003). In western US rangelands, Bluebunch Wheatgrass and Introduced Upland Vegetation-Herbaceous were the major herbaceous vegetation types that had areas classified as having low and moderate burn severities (Table S3). Bluebunch Wheatgrass has dense tufted culms, and is easily consumed by fires (Agee and Maruoka, 1994). Fires can consume aboveground biomass of herbaceous species in the Introduced Upland Vegetation-Herbaceous vegetation type resulting in a decrease in herbaceous cover (Akinsoji, 1988). Due to these species rapid recovery following wildfires, changes in herbaceous vegetation types may occur on temporal scales shorter than that of map production which may lead to the underestimation of decrease in herbaceous cover following the wildfires.

4.3. The relationship between lagged moisture conditions and the percent cover change for different plant functional types

For herbaceous vegetation types (i.e., AFG and PFG), negative correlations between percent change in these cover types and SPEI with various lag scales indicated that wetter prefire conditions, as indicated by more a positive lagged SPEI, led to larger decreases in these cover types following wildfires. Wetter prefire moisture conditions would suggest that soil moisture was at levels that facilitated greater herbaceous fuel accumulation and fuel connectivity, thus allowing wildfires to consume more fuels, resulting in large decreases in cover of annual and perennial forbs and grasses (Fusco et al., 2019; Harvey et al., 2017; Huang et al., 2020; Keyser and Westerling, 2017). Some herbaceous vegetation types, such as the Introduced Upland Vegetation-Herbaceous type, could be sensitive to the changes in moisture conditions due to the annual plant species in this type having relatively shallow roots and low water storage capacity (Wu et al., 2018; Xu et al., 2018).

For woody vegetation types (i.e., SHR and TREE), drier soil moisture conditions (more negative SPEI) resulted in larger decreases in these cover types following wildfires. Fuels are relatively abundant in woody ecosystems, and fuel flammability is a primary constraint on the impacts of wildfires on vegetation cover (Flannigan et al., 2013; Jolly et al., 2015). Warm and dry conditions can dry out fuels, and increase the potential for ignitions (Pausas, 2004; Williams et al., 2001). The relatively longer lagged SPEI for shrubs and trees (i.e., the moisture/drought conditions 6, 9 and 12 months prior to the fire) indicated that longer term drought conditions led to greater decreases in woody cover following wildfires, and comparable results were found in other studies (Littell et al., 2009; McKenzie and Littell, 2017).

4.4. Limitations

There are several limitations in this study. First, the prefire rangeland composition from RAP dataset is an annual estimate of fractional cover, and higher temporal resolution may provide better estimation. RAP accuracy may vary across the landscape and across different plant functional groups, which could increase uncertainty about fire effects in more heterogeneous areas and for woody functional groups, especially if training and verification data for the fractional cover products lack data from wildfire areas (Germino et al., 2022). Second, although the use of the Existing Vegetation Type (EVT 2001) in this study is consistent with the rangeland map used in developing the RAP dataset (Jones et al., 2018), the EVT map may not reflect changes in vegetation types that could have occurred since the development of the 2001 EVT product. Third, the burn severity mosaics from MTBS dataset, which are the only mosaics available for the scale of western US, has limitations. Changes in vegetation cover between prefire and postfire images in some herbaceous types may be underestimated as some vegetation types may recover quickly following fires, although MTBS program has worked to address this issue by using images immediately after a fire to characterize burn severity (Eidenshink et al., 2007; Picotte et al., 2020; Stambaugh et al., 2015; Zhu et al., 2006). Stambaugh et al. (2015) noted that performance of remote sensing derived burn severity was lower for grasslands compared to woodlands, and accuracy in predicting the correct severity class in grassland was highest for moderate severity classification and lowest for low and high severity classification. In this study, the MTBS severity estimations were used across all EVTs to maintain consistency. However, future work is needed to determine if new protocols need to be developed for burn severity classification in grasslands. Fourth, the spatial resolution of climate data (SPEI) does not match that of the RAP dataset (4 km vs. 30 m, respectively), and higher spatial resolution climate data could enhance the exploration of relationships between soil moisture conditions and vegetation responses to wildfires. Lastly, this study did not examine other factors such as topography and elevation that could influence burn severity. This could be explored in future studies at appropriate spatial scales and if data are available.

Evaluating the impacts of wildfires on vegetation structure are important for understanding rangeland fire regimes and identifying the responses of diverse rangeland ecosystems to wildfires with different severities. This information could aid in evaluation of vegetation and soil moisture conditions that could provide early warning for areas vulnerable to state changes resulting from high fuel loads and dry conditions (e.g., sagebrush ecosystems), and help predict future vegetation patterns. For example, the fires in woody-dominated ecosystems could lead to larger decrease in vegetation cover, and need relatively longer time to recover to prefire conditions. The fires in PFG-dominated ecosystems would have lower decreases in vegetation cover which may take shorter time to recover, and some plant functional types may have cover greater than prefire conditions. The SPEI with different time lags can assist in monitoring accumulated dry or moist soil conditions and further help develop trajectories of potential fractional cover changes in the event of a future wildfire.

5. Conclusions

Rangeland wildfires in the western US have become more frequent and larger over the last few decades with significant ecological and societal implications, but our understanding of the impacts of wildfires on vegetation, and how they vary with different levels of burn severity, is limited. This study examined the impact of wildfires of different burn severity levels on rangeland vegetation, in terms of fractional covers of plant functional types including annual forbs and grasses (AFG), perennial forbs and grasses (PFG), shrubs (SHR), and trees (TREE), as well as the influence of prefire soil moisture conditions. Overall, the cover of all plant functional types decreased following the wildfires, but their timing of response differed. The AFG and PFG cover decreased almost immediately following the wildfires as aboveground herbaceous biomass was consumed by wildfires. Generally, the SHR cover had the greatest decrease at one year postfire, and the TREE cover had the greatest decrease at two years postfire. High burn severity wildfires led to larger decreases in the cover of plant functional types, while low burn severity wildfires resulted in smaller decreases, in general. Furthermore, the extent of the decrease in herbaceous (i.e., AFG and PFG) cover was lower than that in woody (i.e., SHR and TREE) cover following wildfires of low and moderate severity, likely due to the ability of herbaceous vegetation to recover rapidly after fire. The patterns of correlations between SPEI with different lag scales and percent change in cover of plant functional types suggested that wetter prefire conditions led to larger decreases in AFG and PFG fractional cover, while drier prefire conditions resulted in larger decreases in SHR and TREE cover following wildfires. These findings improve our knowledge of the effects of wildfires of different burn severities on rangeland vegetation, as well as the relationships between soil moisture conditions and vegetation responses to wildfires.

CRediT authorship contribution statement

Zheng Li: Conceptualization, Methodology, Formal Analysis, Writing-Original Draft, Writing-Reviewing & Editing. **Jay P. Angerer:** Conceptualization, Methodology, Writing-Reviewing & Editing. **X. Ben Wu:** Conceptualization, Methodology, Writing-Reviewing & Editing, Funding Acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.157214.

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