Forest Ecology and Management 429 (2018) 617–624

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

The 15-year post-treatment response of a mixed-conifer understory plant community to thinning and burning treatments

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ARTICLE INFO

Keywords:
Disturbance
Plant diversity
Fire Exclusion
Prescribed fire
Mechanical thinning
Shrub cover
Sierra Nevada

ABSTRACT

Disturbance is central to maintaining diversity in forest ecosystems. In the dry forests of the western United States, over a century of fire exclusion has altered the fire regimes of these forests, resulting in high fuel loads and a loss of plant diversity. Mechanical thinning and prescribed fire are widely used to restore structural complexity and species diversity in many western U.S. forests. While studies have shown that the reintroduction of fire into these forests initially promotes plant diversity, there is limited information on the persistence of this effect. We evaluated the effects of thinning and burning treatments on the understory plant community fifteen years after treatment in an old-growth, Sierran mixed-conifer forest. Using a full-factorial design, including three levels of thinning and two levels of burning, we found mechanical thinning and prescribed fire reduced litter depths and increased the availability of bare ground, resulting in an initial increase in herb cover. However, fifteen years after treatment, litter depths and shrub cover increased, resulting in a more homogenous understory community and a loss of herb cover. Overall, our results suggest that while thinning and burning treatments initially promote herbaceous plant cover, these effects are short lived in the absence of a second disturbance.

1. Introduction

Disturbance plays an important role in maintaining species diversity in ecosystems across the globe. Humans have intervened in many cases by altering disturbance frequency and severity. While diversity tends to be highest at intermediate levels of disturbance, intermediate is a function of the productivity of the system (Connell, 1978; Kondoh, 2001). Further, the spatial variability of disturbance can interact with microsite variability, creating fine-scale habitat heterogeneity that is more likely to sustain higher species diversity (Denslow, 1980; Fraterrigo and Rusak, 2008; Roberts and Gilliam, 1995; White and Jentsch, 2001).

Wildfire has played a central role in shaping forest ecosystems across the United States (Agee, 1993; Allen et al., 2002; Baisan and Swetnam, 1990; Bowman et al., 2009). Wildfires burn at different intensities and frequencies, interacting with topographic position, fuel type, edaphic conditions, and weather to produce varying effects across the landscape (Fites-Kauffman, 1997; van Wagtendonk and Fites-Kauffman, 2006). The resulting spatial and structural complexity promotes diversity at the site, stand, and landscape scales. In fire-adapted conifer forests, the majority of this biodiversity is found in the understory plant community (Palik and Engstrom, 1999; Shevock, 1996).

Over a century of fire exclusion has altered the structure and function of frequent-fire forests. High tree densities and surface fuel loads now characterize historically frequent fire forests and with these changes, forest conditions have become more homogenous (Agee and Skinner, 2005; Covington et al., 1997; Gilliam and Platt, 1999). In the absence of disturbance, depauperate plant communities dominated by trees and shade-tolerant species have replaced the once diverse assemblage of herbaceous and woody plants in frequent-fire forest ecosystems (Griffis et al., 2001; Kirkman et al., 2004).

Homogenous forest structure coupled with increasing temperature and longer, drier fire seasons has increased the occurrence of fire and the proportion of wildfires that burn under high-severity in dry forest types (Miller et al., 2009; Westerling, 2016). Large high-severity burn patches trade one homogenous condition for another and the post-fire vegetation community can increase the probability of subsequent high-severity fire (Coppoletta et al., 2016, Guiterman et al., 2017).

Reintroducing fire to drier forests in a manner characteristic of natural fire regimes can promote plant diversity if structural heterogeneity and the accompanying microsite variability are restored (Knapp

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https://doi.org/10.1016/j.foreco.2018.07.058
Received 18 May 2018; Received in revised form 24 July 2018; Accepted 30 July 2018

et al., 2007; North et al., 2005a). Prior work in longleaf pine forests of the southeastern US has demonstrated that regular fire events are required to restore forest structural heterogeneity, which supports substantial understory plant diversity (Franklin et al., 2007; Kirkman et al., 2016). In the western US, the area treated by repeat burning has been limited because of a shorter history of prescribed fire use and the scale of the fire hazard problem. However, one study of repeated fire in Sierran mixed-conifer forest found that initial entry burns increased understory plant richness and cover, and that repeated burning could enhance the distributions of species impacted by fire suppression (Webster and Halpern, 2010).

The Sierra Nevada contains over 50% of California's vascular plant species (Davis and Stoms, 1996; Potter, 1998) and the southern Sierra contains the highest species richness and the most endemic and rare species across the range (Shevock, 1996). Historically, shrubs such as ceanothus (*Ceanothus* sp.) and manzanita (*Arctostaphylos* sp.) covered about 22% of the forest understory in mixed-conifer forests of the central Sierra Nevada (Hasel et al., 1934). The patchy distribution of shrubs in the understory was likely associated with a heterogeneous canopy with gaps that allowed for increased light availability on the forest floor (Conard et al., 1985; Cronemiller, 1959; Show and Kotok, 1924). Historically, frequent, low- and mixed-severity fires shaped the structure, function and composition of Sierran mixed-conifer forests (North et al., 2009). The fire season was constrained primarily to the later summer months following snowmelt and a drying of fuels and the typical mean fire return interval for Sierra mixed-conifer was 12–20 years (North et al., 2005b, Van de Water and Safford, 2011). Generally high severity made up 5–10% of total fire area and consisted of many small (< 4 ha), and few large patches (< 100 ha, Collins and Skinner, 2014). The resulting heterogeneity was characterized by a distribution of patch types, including closed-canopy patches, shrub patches and open gaps that promoted substantial understory plant diversity (e.g., Hutchings et al., 2003). Increases in canopy cover as a legacy of fire exclusion have resulted in a less diverse and homogenous understory community (North et al., 2005a). Forest structural and compositional heterogeneity across scales, from the microsite to the landscape, are important for maintaining a range of abiotic conditions that promote a diverse understory plant community (Beatty, 2014).

Management techniques that utilize fire and fire-surrogate (i.e. thinning) treatments can recreate within-stand heterogeneity and promote a diverse herbaceous and shrub understory (Knapp et al., 2013). While fire-surrogate treatments can alter forest structure and light availability, the understory plant community has been found to have a muted response to mechanical-only treatments because fire is required to reduce surface fuels and create establishment opportunities (Collins et al., 2007; Dodson et al., 2008; Wayman and North, 2007; Webster and Halpern, 2010). However, what remains unanswered is if the effects of both mechanical and burning treatments on the understory plant community persist over time.

We used data from the Teakettle Experimental Forest in the Southern Sierra Nevada to answer the question: How do burning and thinning treatments alter plant community composition and structure 15-years after treatment? We hypothesized that: (1) while burning and thinning treatments initially increased understory diversity, 15-years after treatment understory plant diversity would decline as shrub cover and fuel loads increased; (2) herb cover and richness would decrease as shrub cover and fuel loads increased; and (3) shrub cover would increase as live tree density decreased.

2. Material and methods

2.1. Study site

This study was conducted within the 1300 ha reserve of old-growth forest that was established as the Teakettle Experimental Forest in 1938. The Experimental Forest has no prior history of logging or known history of stand replacing disturbance. Teakettle is located 80 km east of Fresno, CA on the north fork of the Kings River with an elevation ranging from 1900 to 2600 m. The climate is Mediterranean, typical of the west side of the Sierra Nevada, with an average annual precipitation of 125 cm that falls predominately as snow between November and April (North et al., 2002). Over the period of this study an extreme drought occurred from 2012 to 2015, with California's driest 12-month period recorded during this event (Swain et al., 2014).

The mixed-conifer forest type at Teakettle is comprised of white fir (*Abies concolor*), incense-cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*) and Jeffrey pine (*Pinus jeffreyi*, Rundel et al., 1988). Red fir (*Abies magnifica*) and black oak (*Quercus kelloggii*) are also present in the overstory, but at low densities (North et al., 2007). Other hardwood species include willow (*Salix spp.*), bitter cherry (*Prunus emarginata*) and canyon live oak (*Quercus chrysolepis*). Prior to the last known wildfire in 1865, the mean fire return interval at Teakettle was 17.3 years (North et al., 2005b). The 1865 reconstructed forest structure of this mixed-conifer forest was characterized by a low density (67 trees ha^{-1}) of larger trees (quadratic mean diameter 49.5 cm), with Jeffrey pine and sugar pine accounting for 48.9% of the stems (North et al., 2007). Following fire exclusion, a substantial number of establishment events for white fir and incense-cedar occurred, coincident with years of high precipitation (North et al., 2005b). This resulted in increased tree density (469 stems ha^{-1}), which was dominated by white fir (67.6%, North et al., 2007). Prior to treatment, white fir and red fir comprised approximately 86 percent of the basal area at Teakettle, with sugar pine, Jeffrey pine and incense cedar comprising the remaining 13 percent.

The majority of the plant species diversity in this mixed-conifer forest is in the understory. Prior to treatment, 123 herbaceous species and 14 shrub species were identified within the Experimental Forest. During this period, total shrub cover was 27.2% with the most common species being mountain whitethorn (*Ceanothus cordulatus*), which accounted for almost 30 percent of the total shrub cover (North et al., 2002). Other common shrub species include bush chinquapin (*Chrysolepis sempervirens*), pinemat manzanita (*Arctostaphylos nevadensis*), green leaf manzanita (*A. patula*), snowberry (*Symphoricarpos mollis*), sticky currant (*Ribes viscosissimum*), Sierra gooseberry (*R. roezlii*) and hazelnut (*Corylus cornuta*). The two most abundant shrub species, mountain whitethorn and bush chinquapin, are found throughout the entire forest. The most common herbaceous species prior to treatment was *Monardella odoratissima*. For a complete site description, see North et al. (2002).

2.2. Treatments and data collection

Within the mixed-conifer zone of Teakettle, 18 permanent 4 ha treatment units were established in 1998. Using a full-factorial design, three replicates of each treatment unit were randomly assigned one of two levels of prescribed burning (burn and no burn) and one of three levels of thinning (no thin, understory thin, and overstory thin) for a total of six treatments. For the thin and burn treatments, thinning was implemented in 2000, followed by prescribed burning in 2001. The thin-only treatment units were thinned in 2001. Prescribed burning was applied in late October 2001 after the first major fall rain, resulting in a slow creeping ground fire intended to consume surface fuels while minimizing overstory ignition. Understory thinning removed trees between 25 and 76 cm in diameter while retaining at least 40% canopy cover, following prescription guidelines in Verner et al. (1992). Overstory thinning removed trees greater than 25 cm in diameter, while retaining approximately 22 regularly spaced large diameter (> 100 cm) trees per hectare. The understory thinning treatment reduced stem density from a pre-treatment mean of 469 trees per hectare (TPH) to a post-treatment mean of 239.5 TPH, reducing mean basal area by 15.2 $m^2\,ha^{-1}$. The overstory thinning post-treatment mean was 150.3 TPH, with a mean basal area that was reduced by 33.7 $m^2\,ha^{-1}$ (North