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Nonnative species influence vegetative response to ecological restoration: Two forests with divergent restoration outcomes

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ABSTRACT

Changes in the vegetative structure and diversity of ponderosa pine forests have generated interest in conducting ecological restoration projects to improve the overall forest health of these ecosystems. Ecological restoration prescriptions often consist of thinning trees to emulate pre-1870s forest structure followed by prescribed burning. Disturbances associated with ecological restoration can, however, promote invasion by nonnative species. We compared two northern Arizona ponderosa pine forests treated for ecological restoration, one at the Fort Valley Experimental Forest and one at Mt. Trumbull on the Grand Canyon-Parashant National Monument. We examined the response of native and nonnative plant species, as well as all species combined, to treatments at the two forests. Both study sites showed a significant increase in native and nonnative species cover and richness by the fifth year post-treatment that remained significant by the tenth year post-treatment. Despite these general trends in native and nonnative community development, the understory vegetation at the two sites followed diverging successional patterns after treatment. By the tenth year post-treatment Fort Valley was dominated by native species and Mt. Trumbull was dominated by a single nonnative species, cheatgrass. The differences in post-treatment understory recovery are likely due to pretreatment forest conditions. At Fort Valley, nonnatives were present, but accounted for only 0.11% of the pretreatment cover. At Mt. Trumbull, nonnatives accounted for 5.26% of the pretreatment understory cover, with cheatgrass accounting for approximately 4% of the understory cover. Additionally, the soil seedbank at Fort Valley had greater overall species richness and greater native perennial grass richness than Mt. Trumbull. We propose that the application of ecological restoration treatments should be targeted to sites with low abundance of nonnatives prior to treatment. Sites containing high abundance of nonnatives prior to treatment should be managed for nonnative species mitigation before initiating any ecological restoration projects.

1. Introduction

A century ago, the plant community of southwestern ponderosa pine forests consisted of fewer trees and greater understory diversity (Weaver, 1951; Cooper, 1960; Covington and Moore, 1994; Bakker and Moore, 2007; Laughlin et al., 2011). With the advent of Euro-American settlement in the region, fire suppression, extensive livestock grazing, and commercial logging practices altered the successional trajectory in favor of more trees and reduced understory. Contemporary southwestern ponderosa pine forests often contain dense, interconnected canopies with thick forest floor litter and duff layers. The commensurate shading and lack of bare mineral soil for seed germination has resulted in forests generally depauperate of understory vegetation (Covington and Moore, 1994; Laughlin et al., 2011). The disjunction between current and historical conditions in southwestern ponderosa pine forests has generated interest in using ecological restoration techniques to improve overall forest health and increase biodiversity (Covington et al., 1997; Moore et al., 1999). A common ecological restoration prescription in southwestern ponderosa pine forests is to thin trees to emulate historical forest structure from a time period pre-dating the disruption of the past disturbance regime, then reintroduce fire to the system (Covington and Moore, 1994; Moore et al., 1999).

Ecological restoration treatments consisting of tree removal and prescribed fire intentionally generate ecological disturbances. Tree thinning perturbs the soil and increases light infiltration to the understory, while prescribed fire removes much of the understory aboveground biomass, alters short-term nutrient cycling, and

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consumes accumulated litter and duff layers from the forest floor (Covington et al., 1997; DeLuca and Zouhar, 2000; Johnson and Curtis, 2001; Selmants et al., 2008). The ecosystem changes generated by restoration-induced disturbances promote increased growth and diversity in the native vegetative community (Moore et al., 2006). Disturbances associated with ecological restoration treatments can, however, have the unintended consequence of exposing the ecosystem to greater risk of invasion by undesired and detrimental nonnatives (Hobbs and Huenneke, 1992; Crawford et al., 2001; Griffis et al., 2001; Allen et al., 2002). While the presence of nonnatives is contrary to the objectives of ecological restoration (SER, 2004), an elevated presence of nonnatives in response to thinning and burning treatments is common (Wienk et al., 2004; Dodson and Fiedler, 2006; Moore et al., 2006; Nelson et al., 2008; Sabo et al., 2009). The risk of nonnative invasion has raised concerns regarding the utility of historical reference conditions as a standard protocol for designing ecological restoration treatments (Allen et al., 2002; D'Antonio and Meyerson, 2002; Noss et al., 2006; Hobbs et al., 2009; Jackson and Hobbs, 2009).

In this study we examined ecological restoration treatments in two ponderosa pine forests in northern Arizona, USA. The first study site is in the Uinkaret Mountains north of the Grand Canyon and the second is near the city of Flagstaff. Ecological restoration treatments were based on historical reconstruction of the overstory density and distribution combined with the application of prescribed fire. For both studies we have at least 10 years of post-treatment data collected with the same experimental design and methodology. We hypothesize that (1) the native understory will increase in cover and species richness in response to ecological restoration, (2) nonnatives will also increase in response to ecological restoration, and (3) the understory response to ecological restoration will be consistent in both study sites.

2. Methods

2.1. Study sites

The Fort Valley site is located in and adjacent to the Fort Valley Experimental Forest, approximately 15 km northwest of Flagstaff, AZ in the Coconino National Forest (35°16′19″N, 111°41′22″W). Elevation of the study area is approximately 2250 m. Mean annual precipitation during the duration of the sampling period (1997–2011) was 45 cm (Western Regional Climate Center, http://www.wrcc.dri.edu). Soils consist of fine, smectitic, frigid Mollisols and Alfisols derived from basaltic parent materials (Stoddard et al., 2011). Vegetation at this site is dominated by ponderosa pine, consisting of groups of mature trees intermixed with numerous dense thickets of smaller diameter trees. Prior to treatment, total trees per hectare averaged approximately 1188 and 901, respectively, in the control and treated units. In 1999, after both thinning and burning, tree density decreased by 0.1% and 84%, respectively, in the control and treated units (Korb et al., 2007). The site has not been actively grazed by cattle since the establishment of the Fort Valley Experimental Forest in 1909. The treated units were not seeded after burning except for a portion of each slash pile that was used in a slash pile amelioration study (Korb et al., 2004).

The Mt. Trumbull site is in the Grand Canyon-Parashant National Monument (GCPNM), located in the Uinkaret Mountains (36°22′N, 113°7′W). GCPNM is co-managed by the Bureau of Land Management and National Park Service. Elevation of the study area is approximately 2150 m. Mean annual precipitation during the duration of the sampling period (1997–2010) was 36 cm (Western Regional Climate Center, http://www.wrcc.dri.edu). Soils consist of shallow, cindery Inceptisols derived from basaltic parent materials (McGlone et al., 2009b). Vegetation at this site is dominated by ponderosa pine and Gambel oak (*Quercus gambelii*). Prior to treatments, total trees per hectare averaged approximately 1372 and 1375 respectively in the control and treated units. In 2000, after both thinning and burning, tree density decreased by 7% and 75% in the control and treated units (Fulé et al., 2007). The site is grazed from July 1 to October 31 by a maximum of 88 head for a total of approximately 300 AUMs for an average of 0.18 AUMs ha^{-1}. Cattle were excluded from the site between 1998 and 2002. The treated units were not seeded after burning.

2.2. Experimental design and treatments

The same experimental design was implemented at both Fort Valley and Mt. Trumbull. Treated units received a historical evidence-based ecological restoration treatment that included both thinning and prescribed fire. Tree thinning was based on site-specific reconstructions of pre-1870s forest structure using principles described by Moore et al. (1999). For each site, evidence of old-growth individuals (trees established prior to Euro-American settlement of the region in 1870) was used as a basis for live tree retention. In addition to retaining all old-growth trees, 1.5–3 additional trees were retained as replacement per old-growth individual evidence (e.g. logs, snags, and stumps from trees established prior to the 1870s). This procedure was designed to leave 150–300% of the trees per hectare we estimated existing in our pre-1870s forest structure reconstruction. The retention of more trees per hectare than levels determined by forest reconstruction offset potential errors in reconstructions and unpredicted mortality after treatment. The number of younger replacement trees depended on tree size. If the replacement trees were <40.6 cm diameter at breast height, the higher retention rate was used. In addition, trees in close proximity to each piece of evidence were preferentially retained to more closely approximate the pre-1870s spatial pattern. Prescribed fire was applied by drip torch using strip head fires. Additionally, each treated unit was paired with a control unit that was neither thinned nor burned.

Fort Valley contains three replicated experimental blocks and Mt. Trumbull contains four experimental blocks. Each experimental block contained two treatment units that were randomly assigned as either a treated or a control unit. Treatment units ranged in size from 14 to 16 ha, and each contained twenty 400 m^2 (11.28 m radius) fixed area monitoring plots established on a 60 m × 60 m grid. Plot centers were permanently marked with iron stakes to ensure exact relocation for sampling in subsequent years. Sampling protocol for herbaceous vegetation was modified from the National Park Service fire monitoring protocol (USDI NPS, 1992). Each plot included one 50 m point-line intercept transect oriented parallel to the prevailing slope and centered on plot center. Sampling points were located every 30 cm along each transect for a total of 166 points per plot. Species information was recorded any time a portion of a plant's live aboveground biomass intersected a sampling point. Species presence/absence was also record within a 10 m × 50 m (500 m^2) belt transect centered on the point-line intercept transect. Taxonomic nomenclature and species nativity follow the USDA Plants Database (USDA NRCS, 2012).

Tree thinning was conducted in 1999 at both sites except for one experimental block at Mt. Trumbull that was thinned in 1998. All blocks at both sites were initially burned in spring 2001 except for one experimental block at Fort Valley that was burned in spring 2000. All blocks at both sites were reburned in fall 2007. All pretreatment data were collected in 1998. We collected data in the first growing season post-treatment in summer 2001 at both sites except for the experimental block at Fort Valley that was burned in spring 2000 where we collected data in summer