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Original Research

Understory Responses to Tree Thinning and Seeding Indicate Stability of Degraded Pinyon-Juniper Woodlands☆

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Depauperate understory plant communities resulting from intensive livestock grazing in pinyon-juniper woodlands of the western United States may represent degraded stable states, resistant to ecological restoration treatments. In this study, we analyzed 10-yr understory plant community responses to restoration treatments that included tree thinning to approximate historical densities of pinyon pine *(Pinus edulis)* and juniper *(Juniperus osteosperma)*, scattering of thinning slash to improve soil conditions, and seeding at two woodland sites (Craig Ranch and Goose Pond) in northwestern Arizona. Results showed that thinning resulted in significant reductions in tree density at both sites, as well as reductions in tree basal area at the Goose Pond site. Boles, branches, and tops of the thinned trees scattered across the study sites resulted in few changes to woody surface fuel loading. Thinning and addition of woody material, along with seeding, resulted in only minor changes in understory cover and species richness at both sites. However, plant cover and species richness were both negatively correlated with tree density. Degraded conditions at the sites appeared to be stable, and we suggest that treatments implemented in our studies may have not been intensive enough to produce significant understory responses and meet restoration objectives. Managers aiming to restore understory diversity at similar sites may be required to use heavier thinning prescriptions and repeated seeding. More work is needed to test new restoration approaches that are designed to drive degraded pinyon-juniper woodlands over resilience thresholds toward more diverse understory communities.

Introduction

Pinyon-juniper ecosystems are highly variable in composition and structure and are distributed broadly on semiarid sites across western North America (Romme et al., 2009). Degraded ecological conditions stemming from intensive land use are common in pinyon-juniper woodlands and historical savannas. For example, intensive fuelwood harvesting in the mid- to late 1800s left previously wooded landscapes of the Great Basin and Southwest denuded of tree cover (Young and Budy, 1979; Bahre and Hutchinson, 1985; Creque et al., 1999). Similarly, extensive areas of woodland were converted to grassland for livestock production by chaining, cabling, burning, and other methods in the mid-1900s (Young and Budy, 1979; Romme et al., 2009). In contrast, degradation in the form of increases in tree density, loss of understory plant community abundance, and accelerated soil erosion have been widely reported in woodlands and savannas across the range of the pinyon-juniper type (Campbell, 1999; Jacobs and Gatewood, 1999; Brockway et al., 2002; Romme et al., 2009). Increased tree cover in these systems is thought to be due to a combination of natural and anthropogenic factors including intensive livestock grazing, climatic variability, increases in atmospheric CO_2, and interruption of natural fire regimes (Altschul and Fairley, 1989; Shinneman et al., 2008; Poulos et al., 2009; Romme et al., 2009; Margolis, 2014). Even in persistent woodlands where tree cover may have been minimally affected by fire exclusion, intensive livestock grazing has resulted in decreases in understory diversity and increased soil erosion (Beymer and Klopatek, 1992; Shinneman et al., 2008). On severely degraded sites, ecological conditions may represent alternative stable states that are resistant to restoration treatments.

Since Holling's (1973) seminal discourse on the topic of ecosystem dynamics, the term "resilience" has received much attention in scientific literature and fields of natural resource management. Holling (1973) defined ecological resilience as "a measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables." A central aspect of ecological resiliency is "resistance," which is defined as the "ease or difficulty of changing the system" (Walker et al., 2004). Although recently resilience has been viewed normatively as a desirable property (Brand and Jax, 2007), in Holling's (1973) original definition, resilience is interpreted as neither desirable nor undesirable (Seidl

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et al., 2015). Ecological systems functioning within natural ranges of variability, as well as those that have been simplified or degraded, may exhibit stable state conditions that are resistant to management actions. For example, Carpenter and Cottingham (1997) described processes leading to degradation and the development of new resilience mechanisms that must be overcome to restore lake ecosystems. Suding et al. (2004) explained that ecological resilience to restoration may indicate a shift of a plant community to an alternative degraded state. Similarly, Briske et al. (2008) explain that strong negative feedbacks may increase resilience of alternative stable states and challenge restoration of degraded rangelands.

In cases of ecosystem degradation, a goal of ecological restoration is to drive systems from stable, degraded states, across resilience thresholds, toward more desirable basins of attraction (Walker et al., 2004). To facilitate such transition, degraded woodland systems may require active restoration treatments that include manipulation of vegetation structure, alteration of microclimate and soil conditions, and addition of seeds or propagules (Rey Benayas et al., 2009). Restoration treatments developed for degraded pinyon-juniper ecosystems commonly include tree thinning, amending soil conditions by adding organic matter and thinning slash, and seeding with native understory plant species (Jacobs and Gatewood, 1999; Huffman et al., 2008a; Jacobs, 2015). Tree thinning can reduce interference and competitive effects and provide more light and soil resources for understory plants. Some restoration thinning prescriptions call for close adherence to site-specific, historical reference conditions. This approach centers on retaining all pre-Euro-American settlement trees (i.e., those that predate onset of industrial land uses), plus keeping some number of postsettlement trees to account for recent mortality and removing the remainder (e.g., Huffman et al., 2008a). In some persistent woodlands, thinning intensity following this approach may be minor due to relatively high numbers of presettlement-aged trees (Romme et al., 2009). Addition of organic matter, commonly done by scattering woody material or "slash" remaining from thinning activities, can ameliorate microclimatic conditions, provide safe sites for seedling establishment, and increase microbiotic activity in soils (Tongway and Ludwig, 1996; Breshears and Barnes, 1999; Stoddard et al., 2008). Although thinning, slash additions, and seeding have been shown to be generally effective for increasing understory production and abundance, some studies have failed to find important effects of these treatments (Lavin et al., 1981; Brockway et al., 2002; Huffman et al., 2013). Seeding with native species can increase understory cover (Redmond et al., 2014), and appropriate seed mixes can be developed using information from local observations, as well as reports describing species composition and relative abundance at minimally impacted reference sites.

In this study, we examined understory plant community responses to ecological restoration treatments at two pinyon-juniper woodland sites in northwestern Arizona. Both sites were described by local land managers as showing undesirable conditions in terms of high tree density, low understory cover, and low plant species richness. These conditions were considered to be due primarily to livestock grazing history and changes to the natural fire regimes. At these sites we established two small, identical studies to experimentally address the following research questions: 1) Do restoration treatments including tree-thinning prescriptions guided by reference conditions, scattering thinning slash, and seeding lead to increases in plant cover and species richness? 2) How do understory responses differ across sites with contrasting soils characteristics; and 3) Can such treatments move understory conditions to more diverse, stable states?

Methods

Study Sites

We initiated repeated studies in 2002 at two sites ("Craig Ranch" [CR] and "Goose Pond" [GP]) on Grand Canyon — Parashant National Monument near Mount Trumbull, Arizona (Fig. 1). The two sites were within 5 km of one another and were generally similar in terms of vegetation and woodland structure; however, the sites differed in soil characteristics. Soils at the CR site (lat. 36°26′1″N, long. 113°9′40″W) are shallow to deep gravelly sandy loams to very cobbly clays, derived from limestone, basalt, and sandstone alluvium and colluvium. Soils at the GP site (lat. 36°24′46″N, long. 113°12′15″W) are shallow to very deep, very cindery loams derived from alluvial and colluvial, scoriaceous basalt, and pyroclastics (USDA Soil Conservation Service, unpublished). Although the sites differed in terms of soil texture and parent material, in 2002 soils at both sites were characterized by erosion pavement, presumably due to livestock grazing. Elevation of the sites ranges approximately 1900 — 1950 m. Average annual precipitation in the area near the sites is ≈50 cm and is distributed in a bimodal seasonal pattern with notable peaks during the months of July — August and December — January (WRCC, 2015).

Vegetation at the sites is classified as Great Basin Cold Temperature Woodland (Brown, 1994). Overstories were composed exclusively of pinyon pine (*Pinus edulis* Engelm.) and Utah juniper (*Juniperus osteosperma* Torr.). Trees were generally arranged in all-aged groups with pronounced interstitial openings ("interspaces"). Understory vegetation composition was typical of pinyon-juniper woodlands in northwestern Arizona. Understory vegetation was generally sparse in spatial distribution and depauperate in species (Stoddard et al., 2008). Livestock grazing was halted at both sites in the early 2000s, just before initiation of this study. No livestock grazing was permitted at either site during the course of our research.

Study Design and Field Sampling

At each of the two sites, we delineated 9 ha for study. Each 9-ha area was divided into two 4.5-ha units, and one of each pair was randomly selected for treatment while the other remained as an untreated control. Thus, at each site, separate but identical studies were designed. Sample plots to characterize pretreatment conditions and posttreatment responses of overstory structure, forest floor, and understory vegetation were arrayed on a 60-m systematic grid, established 60 m from the treatment boundaries within the units (i.e., 60-m treatment buffer). Six sample plots per unit were established ($N = 12$ per site) (see Fig. 1).

Sample plots established in the study units were circular and 0.04 ha in size. For long-term monitoring purposes, we used steel rebar driven into the soil to mark plot centers, and these points were georeferenced. In 2002, before treatments were implemented, we measured tree density, size, and species composition; woody surface fuel loading; and understory community characteristics on each plot. All live and dead trees on plots were numbered using aluminum tags nailed to tree bases. Species and diameter at root collar (drc) of each tree (live and dead) were recorded. To determine tree ages, we collected increment cores from all live trees ≥ 20 cm drc and from a 20% random subsample of smaller trees (< 20 cm drc). Fuel loading was estimated using methods described in Brown (1974). One 15-m planar fuels transect was established on each plot with the proximal end anchored at plot center, and the transect direction was determined randomly. A piece of steel rebar was driven into the soil at the distal end of each fuels transect for subsequent relocation and remeasurement. On each transect, forest floor depth was measured at points every 1.5 m, and layers were classified as "litter" layer (recent, undecomposed) and "duff" (lower, decomposing). Woody surface fuels intersecting transects were measured for diameter and tallied by moisture timelag classes according to Brown (1974). Timelag classes represent the length of time required for wetting or drying of fuels of different sizes, relative to the equilibrium moisture content. Timelag classes were "1-hr" (< 0.63 cm), "10-hr" (0.63 — 2.5 cm), "100-hr" (2.5 — 7.6 cm), and "1 000-hr" (> 7.6 cm). The largest timelag class (1 000-hr) was further subdivided into sound ("1 000-hr-s") and rotten ("1000-hr-r") categories. To sample understory

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