

Restoration of degraded transitional (piñon–juniper) woodland sites improves ecohydrologic condition and primes understory resilience to subsequent disturbance

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ABSTRACT

An overstory thinning and slash mulch treatment designed to improve ecohydrologic condition of degraded transitional (piñon–juniper) woodland sites was evaluated at four sites over a cumulative 16-year period beginning in 1994. Study sites were located within Bandelier National Monument and Santa Fe National Forest on the east-facing Pajarito Plateau of the Jemez Mountains in North-Central New Mexico. Across all sites, total understory cover increased several-fold at 3 to 5 years post-treatment relative to both pre-treatment condition and control, whereas measures of runoff and sediment production were reduced by an order of magnitude. During the course of post-treatment monitoring, several coincident disturbances (multi-year drought beginning in fall 1999, wildfire in spring 2000 and piñon *ips* beetle outbreak in 2002) caused widespread vegetation mortality and allowed differential recovery patterns to be documented across treatment and control areas. Response to these unplanned disturbances suggested prior restoration to improve ecohydrologic function of denuded intercanopy locations also primed understory resilience as measured by the relative capacity of restored areas to regain levels of effective cover. Treatment areas affected by drought-beetle tree mortality exceeded pre-drought levels of understory cover within two growing seasons while retaining improved hydrologic function; by contrast, control areas continued to exhibit accelerated runoff and erosion despite temporary improvements in understory cover post-disturbance. Post-fire outcomes suggest that prior restoration also enhanced levels of understory cover post-burn relative to control, despite negative-soil-heating effects from consumed slash. Copyright © 2014 John Wiley & Sons, Ltd.

KEY WORDS restoration; woodland; resilience; ecohydrology

Received 14 April 2014; Revised 7 September 2014; Accepted 2 December 2014

INTRODUCTION

Ongoing and projected changes in regional climates have prompted ecological risk assessment of species and community level response. These efforts range from predictive modelling to case studies and most often utilize extant species range and community attributes to project future potential in relation to shifting climate envelopes. However, current and future species distributions and community assemblages are controlled by a myriad of factors, not least of which are anthropogenic effects, all of which can limit real-world applicability of simple bioclimatic models. Moreover, many ecological communities and the physical habitats they currently occupy are degraded, which may constrain future site potentials. Enhancing elements of system resilience (i.e. capacity to recover to a desired state after disturbance) should be a

common adjunct of functional restoration efforts to repair degraded systems (Lake, 2012). Yet, presumed benefits of restoration in bolstering resilience and other favourable system responses to changing conditions have received surprisingly little attention (Floyd and Romme, 2012). This may reflect a paucity of cases where treatment monitoring was sustained long enough to assess restoration legacies or to characterize episodes of resilience post-disturbance. Where monitoring data are available, resilience can be quantified using relevant measures of system recovery to a desired state and the relative strength of the response should reflect the underlying biological capital (e.g. soil, nutrients and plant materials) available to support that recovery.

The Jemez Mountains in North-Central New Mexico were intensively grazed during the historic settlement period beginning ca 1850, including areas within Bandelier National Monument and Santa Fe National Forest (Allen, 2007). Consumption of grassy fuels interrupted patterns of frequent, low intensity surface fire associated with ecotonal Ponderosa pine (*Pinus ponderosa* Dougl. ex Laws. *scopulorum* Engelm.) savanna communities

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(Allen, 2007). Prior to historic grazing, Colorado piñon pine (*Pinus edulis* Engelm.) – one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg.) – woodlands were largely restricted to unproductive (e.g. steeper and rocky) settings adjoining and interspersed with the (swales, pumice patches and gentle mesa) locations dominated by Ponderosa savanna (Jacobs *et al.*, 2008). Age-class and observational data suggest that the historic woodland component was relatively open canopy, but mostly persistent in nature (*sensu* Romme *et al.*, 2009) with sparse understories and only supporting patchy crown fire (Jacobs, 2008; Jacobs *et al.*, 2008). Cessation of fire disturbance within Ponderosa savanna allowed fire intolerant woodlands to encroach and eventually dominate after beetle mortality dramatically reduced distribution of Ponderosa below 2000 m during the 1950s multi-year drought (Allen and Breshears, 1998).

Extensive tree die-off during the 1950s was followed by moist conditions favourable to woodland establishment during subsequent decades, and these events facilitated a rapid transition from topographically patterned mosaics of Ponderosa savanna and persistent woodland to increasingly homogeneous *transitional* woodland cover (Allen, 2007). The pace and magnitude of large disturbances in this landscape have escalated dramatically since 2000 and illustrate how climatic extremes acting on vulnerable ecohydrologic systems can promote rapid state transitions well ahead of mean thresholds. Recent multi-year droughts have been accompanied by warmer temperatures and increasing evaporative demand, drought stress, beetle populations and plant mortality (Breshears *et al.*, 2005; Rich *et al.*, 2008; Williams *et al.*, 2012).

In these transitional woodlands, a common outcome of encroachment has been the desertification of intercanopy locations, with an increase in bare soil exposure as herbaceous understories are suppressed by expanding tree canopies (Jacobs, 2008; Jacobs *et al.*, 2008). This trajectory becomes self-reinforcing when size and connectivity of bare soil patches reach critical thresholds, enhancing runoff and soil erosion and further degrading site conditions (Davenport *et al.*, 1998; Wilcox *et al.*, 2003; Williams *et al.*, 2013). Transitional woodlands can be considered ‘functionally degraded’ when intercanopy patches are dominated by exposed soils and vulnerable to high rates of runoff and erosion. Unequal distribution of resources between canopy and intercanopy locations reflects cumulative accumulation patterns associated with long-lived tree patches and that contributes directly to desertification of adjacent intercanopy locations (Schlesinger *et al.*, 1990). As limited resources are concentrated within canopy patches, runoff connectivity between intercanopy locations can cross thresholds that then support soil loss across the entire site (Davenport *et al.*, 1996).

Efforts to stabilizing rapidly eroding soils in these degraded transitional woodlands and mitigate impacts on

embedded cultural resources became a major focus of land managers at Bandelier beginning in the early 1990s (Chong, 1993; Davenport *et al.*, 1998; Jacobs and Gatewood, 1999; Jacobs *et al.*, 2008; Jacobs, 2011; Loftin, 1999). Given the rapid system changes since the 1950s (e.g. type conversion of Ponderosa savanna to degraded transitional woodland), structural restoration of historical conditions was not considered feasible. Restoration of ecohydrologic function instead became the primary focus, with intent to retain remaining biological capital (e.g. soil and plant materials) in support of a full range of native vegetative assemblages possible under current and future conditions. Restoration treatments consisting of a variable mechanical overstory thinning and dispersal of cut trees as slash mulch into bare soil intercanopy locations were designed to reduce accelerated rates of runoff and erosion and promote establishment of an effective understory cover. Study sites were established beginning in 1994 and monitored over a cumulative 16-year period through 2010. During the course of post-treatment monitoring, several unplanned and coincident disturbances (i.e. multi-year drought beginning in fall 1999, wildfire in spring 2000 and piñon *ips* beetle (*Ips confusus* Leconte) outbreak in 2002) provided opportunities to evaluate whether functionally restored systems were also more resilient and better able to respond favourably to anticipated effects of changing climate.

METHODS

Study site

Three piñon–juniper woodland sites within Bandelier National Monument and a fourth on the adjacent Santa Fe National Forest were established on the east-facing Pajarito Plateau in the Jemez Mountains of North-Central New Mexico (Figure 1) and monitored over a cumulative 16-year period (1994 to 2010). The four study sites were locally representative of mesa-top woodlands ranging from 1900 to 2100 m: Garcia Mesa, site 3 (2100 m), Alamo Mesa, site 2 (1900 m), South Mesa, site 4 and Frijoles Mesa, site 1 (2000 m). Prior to treatment, woodland canopy within the four study areas was a mixed piñon–juniper overstory ranging from ~25% to 60% cover and split approximately between the two co-dominants. Exposed soils dominated intercanopy locations, whereas dense litter mats were often present under tree canopy; the latter often forming slightly elevated mounds relative to the adjacent intercanopy. Related study sites provided supporting information on area woodland soils and hydrology (Supporting information).

Climate and recent disturbance patterns

Representative monthly precipitation compiled by water year for the October 1991 to September 2012 period

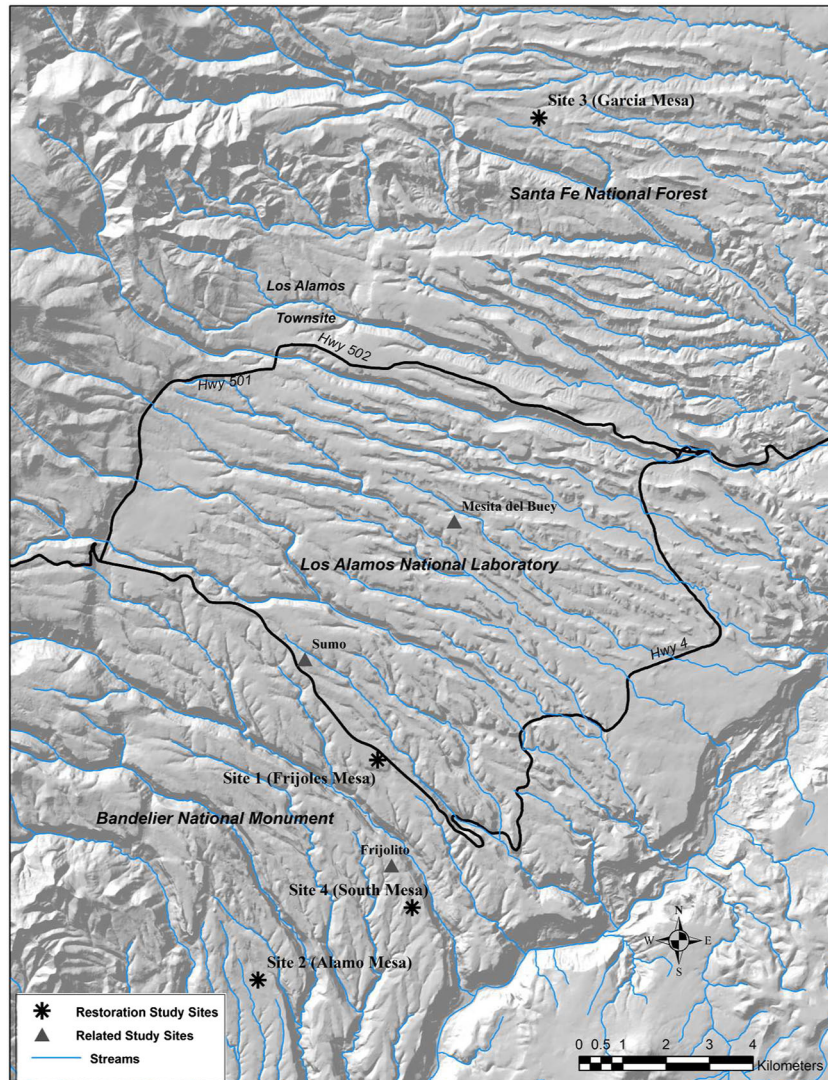


Figure 1. Location of woodland restoration sites (Sites 1, 2, 3 and 4) within North-Central New Mexico including portions of Bandelier National Monument and the adjacent Santa Fe National Forest. Also shown are nearby reference study sites Frijolito Experimental Watershed, Mesita del Buey and Sumo.

highlights moisture patterns most relevant to end-of-growing-season measures of vegetation cover (Figure 2). Spring is typically warm and dry followed by a wet summer period. Monsoonal (late June to mid-September) rains generally contribute 50% or more of mean annual precipitation, but large temporal and spatial variability in summer moisture is an overriding pattern (Hastings *et al.*, 2005). Winter snow cover is also variable in amount and persistence; from late fall to early spring, repeated frost heave and solar drying of exposed soils can reduce bulk density and structural integrity making them vulnerable to erosion by summer precipitation events. A severe winter–spring drought from October 1996 through May 1997 muted understory response in the subsequent growing season. A multi-year drought and associated bark beetle outbreak (2000–2003) resulted in >95% mortality of

mature overstory piñon, and two of four study sites burned in spring 2000.

Data collection and analysis

Vegetation. Prior to restoration, sites were subdivided into treatment and control areas, and pre-treatment data collected in the year of establishment. Sites 1–3 used an unbalanced, replicated block design; treatment was assigned non-randomly to contiguous (15 m²) plots ($n=5$) in each of three replicates; control plots ($n=1$) were offset 15 m in adjacent untreated areas (Jacobs and Gatewood, 1999). Vegetation transects (2 × 21-m long) were established diagonally across each plot.

Site 4 was established as a paired-watershed case study to evaluate restoration response across multiple parameters

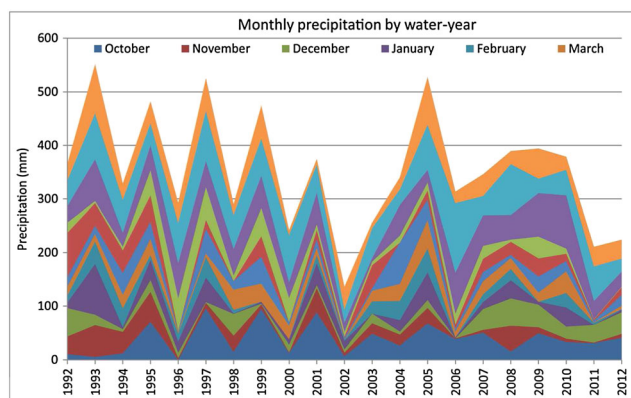


Figure 2. Monthly precipitation compiled by water year (1 October to 30 September) for the October 1991 to September 2012 period; water year provides a good basis for relating precipitation patterns to seasonal vegetation growth. Rainfall data are from the Banderier National Monument weather station located ~1 km northeast of site 4 at 1980-m elevation.

including vegetation (cover and biomass), hydrology (surface runoff and sediment transport), soil (soil micro-topography, moisture and nutrients) and biota (ground-dwelling arthropods, butterflies, birds and ungulate grazing) Jacobs *et al.*, 2002. Vegetation transects representing replicate 1 ($n=20$, in two 50-m segments) were established at regular intervals around the perimeter of each 100-acre watershed and run perpendicular downslope. Additional vegetation transects were sampled within ungulate exclosures ($n=5$, 50 m) and across archaeological sites ($n=5$, 2×25 m) providing two additional estimates of understory cover within site 4; the data from these two sub-locations were designated as replicates after preliminary analysis indicated that they were not significantly different from each other or from replicate one (Jacobs and Gatewood, 1999; Jacobs *et al.*, 2002).

Vegetation and ground cover were monitored across all four study sites over a cumulative 16-year period (1994–2010), but not all sites or vegetation transects within a site were sampled in all years. Treated areas at each site were mechanically thinned in the year following establishment. Piñon and juniper overstory was reduced by 75–100% (from 35–45% to 10–0%). Thinning intensity varied by site from 100% (sites 1 and 2) to 75% (sites 3 and 4); trees were lopped, and the green slash (i.e. retaining fresh needles) was scattered onto bare soils of adjacent intercanopy locations. Sites 1 and 3 were established in 1994 and treated in 1995, site 2 was established in 1995 and treated in 1996 and site 4 was established in 1996 and treated in 1997.

At all sites, vegetation cover data were collected using a standardized line-transect method; basal and overstory cover components were sampled separately, by species or cover type, and at centimetre resolution. Sampling was conducted in late fall (but prior to first frost) to capture cumulative understory response at end-of-growing season. Raw transect

data were compiled using PERMPLOT v.8.0 (Bracker, 2010). Changes in mean patch size of effective cover types (i.e. understory basal cover and ground litter versus bare soil) were derived from vegetation transect data ($n=1$), sampling the paired 0.3-ha runoff micro-watersheds. Species richness was derived from cover data and represents a simple count of species present in the sample.

Basal values represent ground intercept and cumulatively total 100%, whereas canopy values record overstory per species and because of overlap can potentially exceed 100%; total understory cover is the sum of basal and canopy by forb, grass, sub-shrub and shrub growth forms. Pre-burn and post-burn fuels data were collected along vegetation transects within the two burned locations (sites 1 and 3, $n=5$ per replicate) using a modified brown-line method that compiled frequency of woody intercepts within four discrete size classes (Jacobs and Gatewood, 2002). Understory biomass (forbs and grasses) was collected from site 4 through 2 years post-treatment (1996–1998). Vegetation was clipped from four 0.5-m² quadrats offset at defined intervals along each of twenty transects per treatment ($n=80$); biomass was sorted by species, oven dried and weighed (Jacobs *et al.*, 2002; Oertel, 2004).

Vegetation cover data were organized to support analysis of treatment-control response relative to (1) pre-drought period (1994–1999), (2) drought and post-drought period (1999–2010) and (3) fire effects (1999–2001). Additional analysis quantified: (1) vegetated and litter versus bare soil mean patch size across paired runoff plot locations; (2) relative abundance of an invasive annual, cheat grass (*Bromus tectorum* L.), (3) drought-beetle-induced mortality of the piñon overstory and (4) effects of protection from grazing on treatment response of two cool-season grasses, mutton (*Poa fendleriana* (Steud.) Vasey) and June (*Koeleria macrantha* (Ledeb.) Schult.), which green up in early spring and are preferred by native ungulates.

Vegetation response to restoration and subsequent drought-beetle-fire disturbances was assessed at the individual site level using the repeated measures statement in the MIXED procedure in SAS 9.2 (SAS, 2008) where time was the repeated measures factor. Least-squares means (Lsmeans) were calculated by growth form for each treatment*year combination, and Tukey adjusted p -values provide measures of significance for multiple treatment*year comparisons within sites. Summary responses (Lsmeans averaged across sites by growth form by treatment*year) were calculated to illustrate representative patterns for the larger study area and provide the primary reference for discussion of restoration outcomes. Summary responses for the pre-drought period (1994–1999) average site Lsmeans by treatment year (0 to 5) to minimize influence of annual rainfall variability, whereas summary responses for the post-drought period (1999–2010) average site Lsmeans by

data year to minimize influence of post-treatment year. Summary measures of understory cover recovery post-drought-beetle-fire disturbance in treated versus control areas were used to infer relative system resilience attributable to prior restoration.

Hydrology. Hydrologic parameters (surface elevation, sediment production and runoff) were only collected from site 4 and for variable time frames. Changes in soil surface elevation were assessed using a soil erosion bridge method (1996–2000, 2010), sediment production was estimated per event from 0.1-ha catchment areas (2000–2002) and surface runoff was calculated per event from instrumented 0.38-ha micro-watersheds (2003–2009). Each of the sediment catchments and runoff micro-watersheds was instrumented with its own rain gauge given the extreme spatial variability of summer monsoonal events (Hastings *et al.*, 2005).

Estimated sediment loss/gain (kg/ha) by treatment year was derived from erosion bridge measured changes in soil surface elevation (mm). Bridges were stratified across four soil–slope settings: Upper slope intercanopy soils (pumice and non-pumice) and channel (upper and lower) and bridges were oriented perpendicular to the slope. Bridges were grouped into 18 sets of five located between each of the 20 vegetation transects ($n=90$) per treatment. Non-pumice sites received two of the five total bridges per set. Thirty soil elevation points plus one reference point are sampled across each bridge per reading; bridge averages are subtracted from the previous reading to determine net loss or gain.

Sediment production data were collected from $n=6$, ~0.1-ha catchments, stratified across two soil settings (pumice, $n=4$ and non-pumice, $n=2$) within adjacent treatment and control areas. Accumulated sediment was harvested after each event and measured volume converted to equivalent weight using catchment-specific estimates of soil bulk density (Oertel, 2004). Sediment yields (kg/ha/storm) were calculated for each catchment event by adjusting sediment volume to catchment contributing area. Each observation represents catchment level sediment yield (kg/ha) and the associated maximum 30-min rainfall intensity (R_I -30 mm/h). Sediment yield regressions were developed by treatment and soil setting using the compiled catchment data ($n=108$ per treatment) representing 18 rainfall-sediment transport events sampled over a 3-year period from 2000 to 2002 (Hastings, 2002; Hastings *et al.*, 2003; Oertel, 2004).

Runoff data were collected from paired ($n=1$) micro-watersheds (~0.38 ha) that were established in comparable settings within adjacent treatment and control areas. Micro-watersheds were matched relative to size, aspect, slope, and soils and located within ~500 m of each other. Prefab flume

structures instrumented with pressure transducers recorded stage height; nonevent data were collected hourly until stage height reached a minimum trigger value (0.05 ft) that activated a high frequency (5-s interval) data collection mode. Stage (feet) was converted to flow (cfs) using manufacture rating curves. Each rainfall-runoff observation represents the value of the maximum cumulative runoff within a 10-min period (R_I -10 cfs) and associated maximum 10-min rainfall intensity (P_I -10 mm/h) for the same location and period; maximum rainfall-runoff values were temporally synchronous with maximum runoff lagging maximum rainfall by one to several minutes in all cases. Runoff yield regressions were developed using data from $n=67$ rainfall-runoff events per treatment measured over a 7-year period from 2003 to 2009.

Soil Nutrients. Plant available soil nitrogen was assessed at 3 years post-treatment (1999) in control versus treatment areas of site 4 using a resin bag method (Lajtha, 1988); *in situ* resin bags ($n=50$) were deployed using a replicated design and stratified by canopy-intercanopy soil location. Bags were retrieved after 30 days, and total available nitrogen ($\text{NH}_4 + \text{NO}_3$) was determined by laboratory analysis.

RESULTS

Vegetation

At the individual site level, treatment-control areas were very similar prior to treatment with no significant within-site differences for measured cover parameters; when averaged across sites, vegetation and ground cover components were also approximately equal (Figure 3a and b). Site level lsmeans (treatment*cover type), and p -values for pre-treatment differences (i.e. year 0) between control and treatment, are provided in the Table S1.

Representative data from site 4 are used to illustrate effects of a ~75% overstory reduction on generated slash cover and exposed soils (Figure 4). Treatment response was characterized by an initial pulse of annual and biennial forbs during the first two growing seasons, with perennial grasses and sub-shrubs increasingly more linearly through year 5. Some ephemeral early season species were not well represented in the data set and annual and biennial forbs that respond opportunistically to moisture contributed to high variability in annual cover estimates. By 3 to 5 years post-treatment, basal and total understory cover components increased on average several-fold in restored areas relative to both pre-treatment status and control, with corresponding increases in understory species richness and decreases in bare soil exposure (Figure 3a and b).

Understory (forb and grass) biomass was two-fold higher in treated (463 kg/ha) versus control (154 kg/ha) areas of

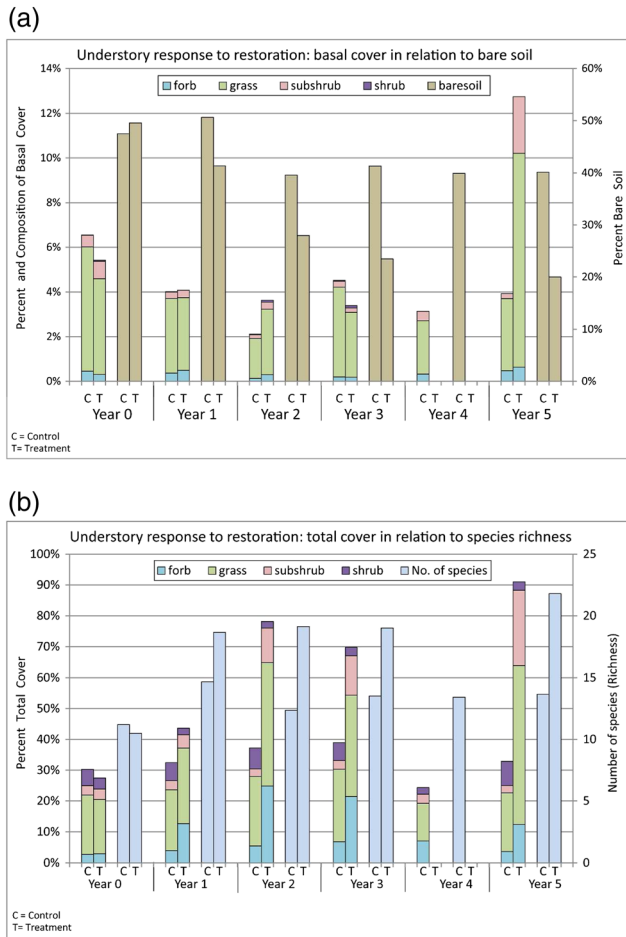


Figure 3. Summary changes in (a) basal and (b) total understory vegetation cover through treatment year 5 across sites; stacked bar graphs of component growth form the following: forb, grass, sub-shrub and shrub in relation to changes in (a) bare soil exposure and (b) changes in understory species richness. Site 1 contributed data to averages for all six data years, site 3 to four data years, and sites 2 and 4 to three data years (Table S1).

site 4. Shrub cover was generally unresponsive to treatment and remained unchanged through year 5. Exposed soils were lower, and species richness higher in treated versus control areas by year 5. During the pre-drought period, a supporting study at site 4 documented an increase in butterfly abundance and diversity to the early forb-dominated understory response (Kleintjes *et al.*, 2004). In contrast to butterflies, the response of ground-dwelling arthropods to treatment was equivocal (data not shown) with species level abundances primarily tracking seasonal to annual patterns of temperature and precipitation (Oertel, 2004). Site level lsmeans (treatment*cover type*year) and *p*-values for differences from the prior data year (e.g. year 1 post-treatment vs year 0) are provided in Table S1.

Prior to drought mortality (in 2000), the control area at site 4 supported a mixed piñon (25%) and juniper (19%) overstory. The majority of piñon trees died and dropped needles by 2003; by 2005, only a 21% juniper overstory

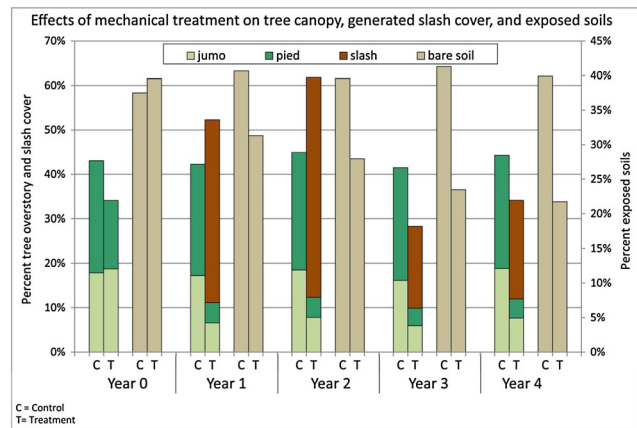


Figure 4. Representative effects of mechanical treatment on tree canopy, generated slash cover and exposed soils at site 4. Mechanical treatments reduced live tree cover by two-thirds (from 34% to 11%), and slash was distributed as a mulch on exposed soil intercanopy locations, increasing effective cover and reducing exposed bare soils by nearly 20% relative to control.

remained, and dead standing piñon began falling (Figure 5). Summary responses of basal and total understory vegetation cover (Figure 6a and b) are shown in relation to changes in bare soil exposure and understory species richness across treatment and control areas from 1999 (pre-drought at treatment years 3 to 5) through 2010 (first year of the second multi-year drought).

Basal and total cover in treated areas, that were reduced during the initial year of drought in 2000, rebounded to pre-drought levels by 2008 before falling again during the second drought beginning in 2010 (Figure 6a and b). Although the basal cover rebounded, associated increases in bare soil were sustained throughout, with control and treatment areas having comparable levels by 2010. Basal components of cover in control areas remained relatively stable throughout with minor increases following widespread piñon mortality and increased moisture from 2004 to 2008. Treatment areas consistently maintained total herbaceous cover at levels twofold higher than control throughout the post-drought period. Within control areas, total understory cover experienced a temporal pulse from 2004 to 2008. Shrub cover remained relatively unresponsive throughout the post-drought period across both treatment and control areas. Species richness was consistently higher in treated areas except for a few years post-drought (2005–2008) when control area richness was temporarily boosted by the post-piñon mortality release of annual and biennial forbs; site level lsmeans (treatment*cover type*year) and *p*-values for differences from the prior data year (e.g. 2000 vs 1999) are provided in Table S2.

Total understory response to restoration within the enclosure at site 4 was not notably different from areas exposed to grazing when analysed at the growth form

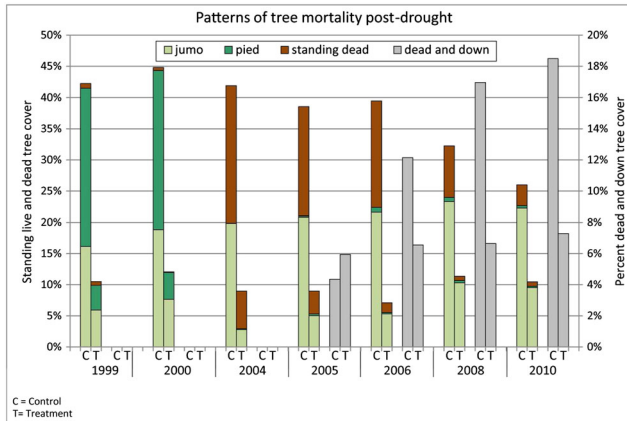


Figure 5. Severe multi-year drought and associated bark beetle outbreak (2000–2003) resulted in >95% mortality of mature overstory piñon across all study locations. The majority of piñon trees died and dropped needles within a 2-year (2002–2003) time frame, and dead standing individuals began falling over in 2005.

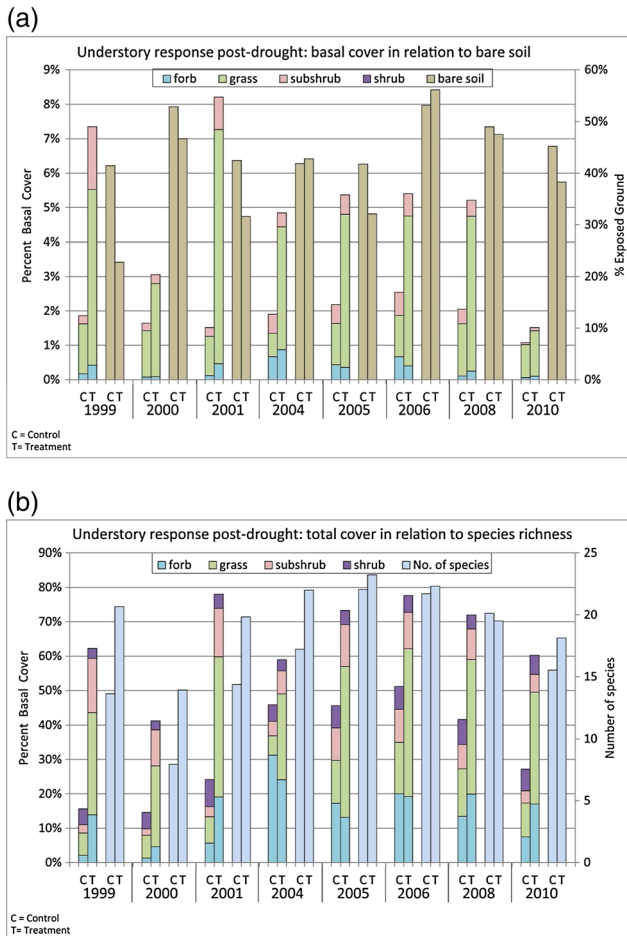


Figure 6. Summary post-drought response of (a) basal and (b) total understory vegetation cover across sites; stacked bar graphs of component growth form from the following: forb, grass, sub-shrub and shrub in relation to changes in (a) bare soil exposure and (b) changes in understory species richness. Site 4 contributed to averages in all data years and site 1 to 1999 and 2000 (Table S2).

level. However, two cool-season grasses (mutton and June) had nearly twice as much cover within the exclosure relative to treated areas unprotected from grazing (2.2% vs 1.1%) and sustained this 2:1 cover advantage into the first year of the drought in 2000 (1.1% vs 0.5%). A temporary pulse in cool-season grasses was also documented within the control area of site 4 in 2008 and casually correlated with the period when nearly half of the standing dead piñon had fallen over; however, cool-season grass cover returned to pre-pulse (2006) levels by 2010.

Burned treatment and control areas showed large reductions in understory cover at the end of the first growing season post-fire, but rebounded strongly by the second year particularly at the upper elevation site 3 where pre-disturbance grass cover was relatively high (Figure 7a and b). Site 3 treatment areas experienced ~50% fire mortality of live tree canopy (from 6% to 3%) with total understory recovering about two-thirds of pre-burn levels at two years post-burn. Site 3 control locations experienced nearly 100% overstory tree mortality, except where buffered by adjacent thinned areas, with total understory cover exceeding pre-burn levels by one-third at year 2 post-burn. Fuel loading (primarily slash from treatment) was reduced by burning from 9.08 to 1.65 t/acre at site 1 and from 4.84 to 2.44 t/acre at site 3 (Jacobs and Gatewood, 2002). Bare soil exposure nearly doubled in control areas and more than quadrupled in treatment locations with only minor reductions by the second season post-fire. Site level lsmeans (treatment*cover type*year) and *p*-values for differences from the prior data year (e.g. 2000 vs 1999) are provided in the Table S3.

Trace levels of cheat grass were detected within control areas of site 4 in 1997, but not reported from treated areas until after 2000 more than 4 years post-treatment; and only expanded across the site after 2004 with return of more favourable winter–spring moisture (Figure 8). The spatial distribution of cheat grass was highly clumped and associated with pumice soils on east-facing slopes. In 2008, the treated areas averaged >3% cheat grass cover compared with ~1% in control. By 2010, cheat grass was found on 35% of control and 80% of treatment transects; the treatment area averaged >7% cover with control just over 3%. Cover estimates of cheat grass (per transect per year per treatment) are provided in Table S4.

Hydrology

Estimated sediment loss/gain (kg/ha) by treatment year derived from measured changes in soil surface elevation (mm) are grouped by four soil–slope settings: upper slope intercanopy soils (pumice and non-pumice) and channel (upper and lower). From an arbitrary baseline in year 0

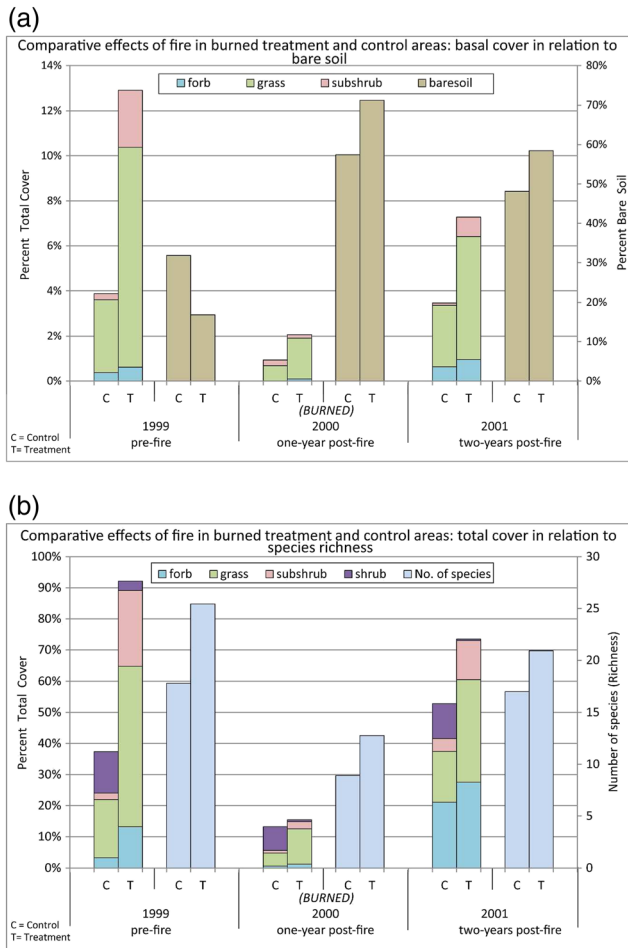


Figure 7. Summary understory response to burning at 5 years post-treatment; stacked bar graphs of component growth form from the following: forb, grass, sub-shrub and shrub in relation to changes in (a) bare soil exposure and (b) changes in understory species richness. Sites 1 and 3 contribute data for all three data years (Table S3).

(pre-treatment), both treatment and control areas showed sizable sediment losses in year 1 (Figure 9). Losses moderated across both areas in 1998, probably due to reduced precipitation intensity. Treatment areas diverged from control in 1999 and continued to show net sediment gains through 2010. Although control areas also had net gains by 2010, this was primarily due to lower channel storage; treatment areas in contrast aggraded strongly in upper slope settings reflecting diminished runoff from intercanopy non-pumice soil locations.

Sediment yield regressions suggest that treatment reduced sediment production by an order of magnitude relative to control over a range of precipitation intensities (Figure 10); an analysis of variance indicated the treatment effect was highly significant ($p < 0.0001$) in reducing sediment yield. Hastings *et al.* (2003) previously reported a two order of magnitude reduction in sediment production for the same study site over a 2-year period (2000–2001). Cumulative sediment production in 2002 was one-third to

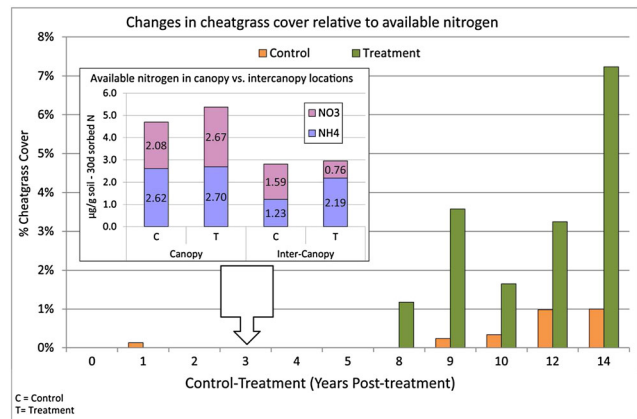


Figure 8. Changes in cheat grass cover (*Bromus tectorum* L.) across treatment and control areas at site 4 in relation to available N₂ measured at 3 years post-treatment (Table S4).

one-tenth of the previous 2-year (2000–2001) average; Whereas severe drought in 2002 reduced understory cover, it also minimized opportunities for runoff and sediment production. When sediment yield analysis is restricted to non-pumice soil catchments ($n=4$), an even stronger treatment effect is found (Figure 10) highlighting the relative stability of pumice patches regardless of treatment.

Runoff yield regressions suggest treatment reduced runoff by an order of magnitude relative to control across a range of precipitation intensities (Figure 11a). Initiation of runoff within the treated area 0.3 ha micro-watershed appeared to have a much higher rainfall intensity threshold (I_{10} of ~80 mm/h) relative to the control (I_{10} of ~20 mm/h). Hydrographs of the 13 August 2006 precipitation event highlight differential runoff across the paired (0.3 ha) treatment-control micro-watersheds (Figure 11b); this was the most intense rain event documented during the 7-year measurement period and illustrates high spatial variability typical of convective cells (Hastings *et al.*, 2005). Maximum 10-min rainfall intensity (P_{I-10}) was one-third higher on the control (121.92 mm/h) versus treatment (83.82 mm/h) micro-watershed for this event, whereas the highest cumulative 10-min runoff (R_{I-10} cfs) was about five times higher for the control (14.11 cfs) versus treatment (2.74 cfs). Runoff yield regressions recalculated without the 13 Aug 06 event ($n=66$) also documented an order of magnitude difference between treatment ($FLOW = 0.001 * PRECIP$; $R^2 = 0.30$) and control ($FLOW = 0.019 * PRECIP$; $R^2 = 0.63$) with improved R^2 values.

The treatment micro-watershed had consistently lower means for exposed soil patch size after year 2, and by 2008, exposed soil patch size was twice as high in control versus treatment (Figure 12). Treatment generally had higher means for effective (vegetated and litter) cover patch size, with the exception of several years post-drought (i.e. post-treatment years 9 and 12, in 2005 and 2008).

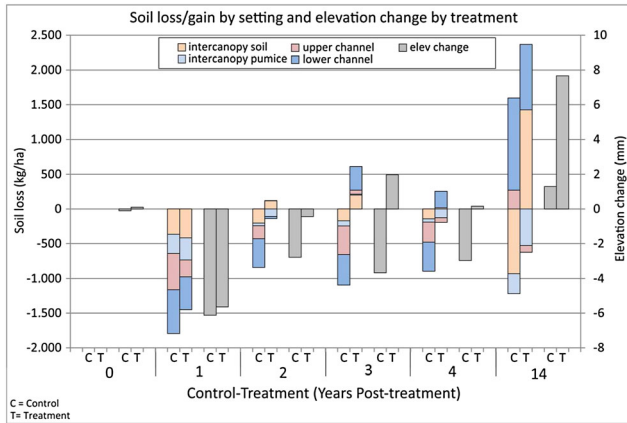


Figure 9. Estimated sediment loss/gain (kg/ha) by treatment year derived from measured changes ($n = 90$) in soil surface elevation (mm). Annual changes are grouped by four soil-slope settings: upper slope (uc) intercanopy soils [pumice (p) and non-pumice (np)] and channel [upper (uc) and lower (lc)].

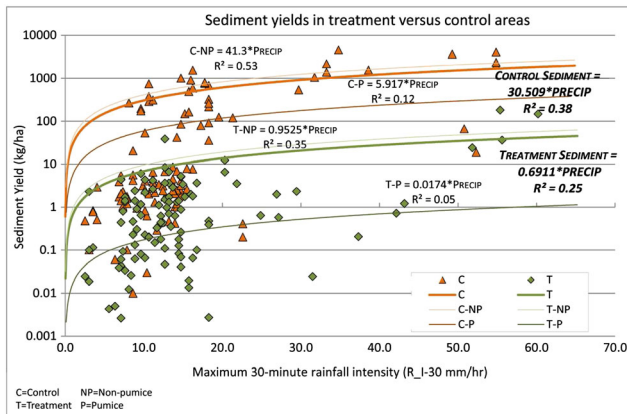


Figure 10. Modelled sediment yields (by treatment and soil setting) using the compiled (0.1 ha) catchment data ($n = 108$ per treatment) representing 18 rainfall-sediment transport events over a 3-year period (2000–2002) suggest that treatment reduced sediment production by an order of magnitude relative to control over a range of precipitation intensities. An even stronger treatment effect is found when analysis is limited to non-pumice (NP) soils (dashed trend line), whereas analysis limited to pumice (P) soils shows a weaker effect. Rainfall-sediment data events represent catchment level sediment yield (kg/ha) and associated maximum 30-min rainfall intensity (R_{I-30} mm/h).

Soil Nutrients

Total plant available nitrogen ($\text{NH}_4 + \text{NO}_3$) in canopy soils was comparable across treatment ($5.37 \mu\text{g/g}$) and control ($4.70 \mu\text{g/g}$) locations at 3 years post-treatment (1999) and evenly split between nitrate and ammonium (Figure 8). Intercanopy locations however appeared to show some differences in available nitrogen, with ammonium (NH_4) in treatment intercanopy soils nearly twice levels in control ($2.19 \mu\text{g/g}$ vs $1.23 \mu\text{g/g}$), whereas available nitrate (NO_3) was just half of control ($0.76 \mu\text{g/g}$ vs $1.59 \mu\text{g/g}$). Intercanopy soil locations were relatively nutrient poor with only two-thirds the available nitrogen of canopy soils (~ 3 vs $5 \mu\text{g/g}$) regardless of treatment.

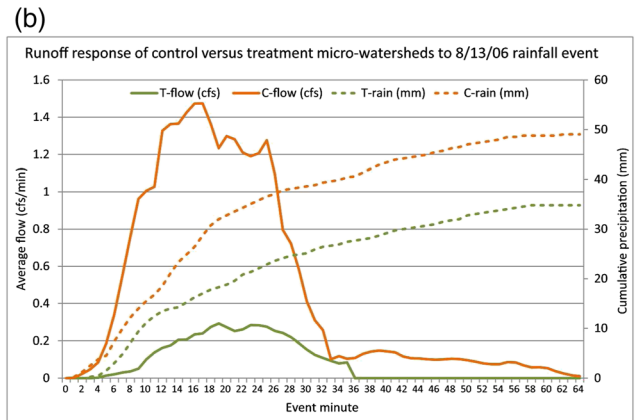
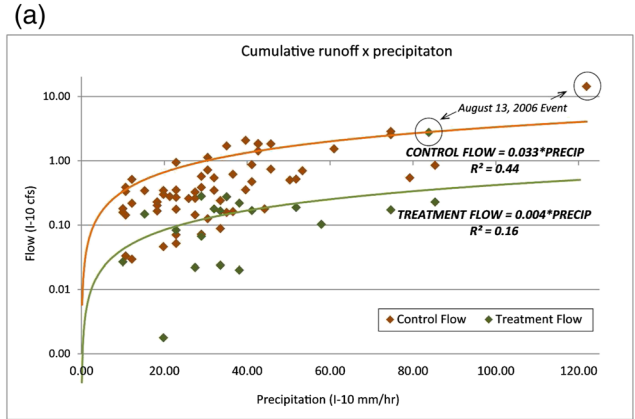


Figure 11. (a) Modelled runoff yields for ($n = 67$) rainfall-runoff events over a 7-year period (2003–2009) across paired treatment-control (0.3 ha) micro-watersheds ($n = 1$) suggest that treatment reduced runoff by an order of magnitude relative to control across a range of precipitation intensities. Rainfall-runoff data events represent the highest cumulative runoff total within a 10-min period (R_{I-10} cfs) and the associated maximum 10-min rainfall intensity (P_{I-10} mm/h). Runoff yields modelled without the extreme 13 Aug 06 event ($n = 66$) also documented an order of magnitude difference. (b) Hydrographs of an intense convective thunderstorm on 13 Aug 06 highlights the differential runoff response (average cfs/min) to measured precipitation (cumulative mm/min) across paired (0.3 ha) treatment-control micro-watersheds. Maximum 10-min rainfall intensity (P_{I-10}) was about one-third higher on the control (121.92 mm/h) versus treatment (83.82 mm/h) micro-watershed, whereas the highest cumulative 10-min runoff (R_{I-10} cfs) was about five times higher for the control (14.11 cfs) versus the treatment (2.74 cfs).

DISCUSSION

Functional restoration of woodland transitional sites enhanced the capacity of degraded intercanopy locations to support herbaceous plant growth and mitigated accelerated runoff and erosion. Event level measures of sediment production and surface runoff documented dramatically improved hydrologic function in treated areas where increases in effective cover acted to reduce bare soil patch size and connectivity. Across all locations, understory cover increased several-fold at 3 to 5 years post-treatment relative to both pre-treatment condition and control (Figure 4a and b), whereas measures of runoff and

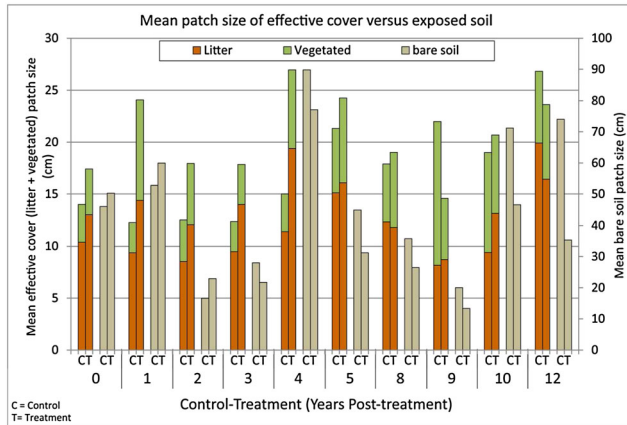


Figure 12. Annual (1996–2008) changes in mean patch size for hydrologically effective basal cover types (vegetated and litter) versus bare soil for the paired (0.3 ha) runoff micro-watersheds. The treatment micro-watershed generally had higher means for effective cover patch size except for several years post-drought (post-treatment years 9 and 12, in 2005 and 2008) and lower means for exposed soil (bar graph and negative values) after year 2 post-treatment (1998). Control area vegetated and litter patch size increased after 2004 in response to drought mortality effects. By 2008, exposed soil patch size was twice as high in control versus treatment.

sediment production were reduced by an order of magnitude (Figures 10 and 11a and b). Thinning and slash treatments can redistribute limited (water, light and nutrient) resources from canopy to intercanopy locations, enhance capture and infiltration of precipitation (through increases in surface roughness that intercept rain drop impact and runoff), and mitigate loss of soil moisture (by shading soil surfaces to reduce evaporative loss and diurnal temperature flux). Recovery processes initiated by restoration may become self-reinforcing in desertified systems where understory cover reduces size and connectivity of intercanopy bare soil patches (Figure 12), increasing retention of water and soil and enhancing site productivity.

Measures of shallow (<15 cm) soil moisture and temperature collected at site 4 as part of the current study were inadequate to model treatment effects given small sample sizes, temporal data gaps and large spatial-temporal variability. However, in a comparable thinning study conducted in western juniper woodlands in Eastern Oregon, Bates *et al.* (2000) reported increased soil water content and plant water availability at two growing seasons post-treatment and suggested slash reduced evaporative loss of soil water. A second study in Arizona, which evaluated effects of slash mulching separately from overstory thinning, also reported improved intercanopy site conditions (i.e. soil moisture) for seeded native grasses as evidenced by enhanced establishment and reproductive status (Stoddard *et al.*, 2008).

In the current study, plant available nitrate (NO_3) in treated intercanopy areas was reduced by half relative to control suggesting possible effects of slash additions on microbial populations, whereas plant available ammonium (NH_4) in treated intercanopy locations almost doubled

perhaps from increased herbaceous biomass inputs. Stoddard *et al.* (2008) also reported a decrease in plant available nitrate (NO_3) along with increases in both arbuscular mycorrhizal fungi and microbial carbon biomass, but no change in plant available ammonium (NH_4) 1 year after slash addition. Carbon additions can temporarily reduce available nitrate through microbial immobilization and thus might have potential to mitigate invasion of exotic plants with high nutrient demands (Blumenthal *et al.*, 2003).

The near absence of cheat grass invasion within treated areas early on, despite notable improvements in intercanopy site conditions, could be attributable in part to this slash-mediated nitrogen-immobilization effect. Moreover, the relative absence of cheat grass from both treatment and control areas until the early post-drought period suggests that disturbance from mechanical treatment was not the primary driver of invasion at this location. However, once established, improved site conditions within treated areas promoted growth of cheat grass relative to control. With onset of favourable moisture in fall 2013 following a second multi-year drought (2010–2013), cheat grass cover continued to expand across both treated and control areas (although primarily on pumice soils), potentially threatening long-term successional outcomes and ecohydrologic stability of these woodland systems.

Live tree canopy interception of rainfall can exasperate runoff erosion from bare soil intercanopy locations by focusing event intensity at the drip line; thinning overstory can mitigate this effect by exposing absorbent litter mounds (Breshears *et al.*, 1997). For example, a typical woodland stand with 35% live tree canopy prior to treatment was reduced to 10% cover, and the slash was used to create a mulch cover of 40% on bare soil intercanopy locations (Figure 4). Slash interception of rain drops can reduce mobilization of soil for subsequent transport by surface runoff. Slash can also reduce mean patch size of bare soil while promoting an increase in effective cover patch size (Figure 12) and thus dampen the magnitude of runoff and sediment transport relative to control (Figures 9, 10, 11a and b). Cumulatively, these effects can rebalance the spatial extent and pattern of sink-source areas shifting hydrologic response above critical runoff thresholds and mitigating accelerated erosion (Davenport *et al.*, 1998). However, the physical benefits of treatment can be ephemeral as green needles fall off the slash within <2 years and fine litter is rapidly degraded by solar radiation, providing the degraded site a relatively short window of opportunity for establishment and growth of understory vegetation.

Long-term monitoring data also suggest that restoration enhanced understory resilience (i.e. capacity to recover) to subsequent fire and drought disturbance, by boosting perennial herbaceous cover and recharging annual and biennial soil seed banks (Figure 6a and b). Restoration is

thought to improve retention of nutrients, organic material (e.g. carbon) and moisture in soils, enhance species diversity and bolster available plant materials (vegetative and seed), thus enabling more robust post-disturbance understory responses relative to untreated controls. However, realization of these benefits requires appropriately matched sites and methodologies, as well as sufficient lead time between initial restoration treatment and subsequent disturbances.

Fire effects were highly variable depending on fire behaviour and fuel loading, burn severity, pre-burn vegetative condition and post-fire precipitation (Figure 7a and b). The favourable outcome of the burned control area at site 3 reflects relatively high pre-burn understory cover combined with nearly complete overstory mortality. Whereas mechanically thinned areas can sometimes mitigate running crown fire behavior, residual slash enhances fuel loads and surface heating that can exasperate mortality of perennial understory if these sites are burned too soon post-treatment.

While drought-beetle (and fire) induced tree mortality is akin to mechanical thinning in reducing live canopy, the potential benefits of woody debris to system response are not fully realized because needle litter falls directly onto existing canopy mounds (or consumed by burning) and fallen tree coverage of intercanopy locations is uneven (Figure 5). By comparison, mechanical treatments uniformly redistribute branches with needles as mulch onto intercanopy soils where it physically mitigates rain drop splash, runoff and sediment transport while enhancing infiltration and reducing evaporation; when needles fall off of branches after several seasons, they blanket the intercanopy soil with litter and eventually contribute organics and nutrients (Figure 4). However, prior thinning was insufficient to mitigate piñon die-off at this location, where insect populations reached epidemic levels and simply overwhelmed trees within relatively small islands of treatment. Drought-beetle disturbance likely enhanced understory response in treated areas by further reducing overstory competition, although drought initially acted to reduce understory cover. Control areas thinned by drought-beetle mortality recorded temporary increases in understory cover, but these were insufficient to mitigate accelerated levels of runoff and erosion.

CONCLUSIONS

Hydrologic and vegetative response to restoration in degraded transitional (piñon–juniper) woodlands was documented over a cumulative 16-year period and provided opportunities to assess the relative resilience of treated versus control areas to subsequent drought-beetle and fire disturbance. Restoration enhanced the capacity of

degraded intercanopy locations to support herbaceous plant growth and mitigated accelerated runoff and erosion. Treatment increased understory cover several-fold at 3 to 5 years post-mechanical treatment and reduced runoff and sediment production by an order of magnitude. Restored areas also showed greater resilience to subsequent drought-beetle and fire effects, recovering to higher levels of effective understory cover post-disturbance. With sufficient lead time appropriately designed functional restoration treatments can sometimes remedy legacy degradation, and where successful these restored systems should have inherently greater resilience to predicted climate perturbation.

ACKNOWLEDGEMENTS

Craig Allen, Kay Beeley, Richard Gatewood, Brian Hastings, Esteban Muldavin, Nathan Myers, Rebecca Oertel and David Snyderman along with numerous Student Conservation Association interns provided critical support for this project. Sam Loftin and two anonymous reviewers provided comments on an earlier draft of this manuscript. Elaine Jacobs assisted with preparation of the final graphics. Portions of this work were funded by grants from the National Park Service and US Geological Survey.

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