

Watershed Restoration
in Degraded Piñon-Juniper Woodlands:
A Paired Watershed Study
1996-98(9)

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Abstract

Restoration studies, conducted in degraded woodland areas of Bandelier National Monument, New Mexico assessed the utility of mechanical treatments to stabilize soils and cultural resources. Piñon-juniper woodland systems have colonized millions of acres of western grazing lands since ca. 1850, transforming former grassland and savanna communities into woodlands, and encroaching into the understories of upland forests. A poor understanding of this dynamic vegetation type has limited the success of management initiatives to improve watershed, wildlife, and range conditions, protect cultural resources, and mitigate impending wildfire hazards. In local degraded woodlands, exposed soils predominate and summer monsoons generate unsustainable sediment loss, which threatens embedded cultural resources. In early 1997, mechanical thinning and slash mulching treatments were applied to one of two, paired, 40 ha watersheds. Response to treatment was quantified using both biotic (plant cover and biomass, abundance of arthropods, and ungulate utilization) and abiotic measures (ground cover and changes in soil surface topography). In late 1999, at three years post-treatment, total herbaceous plant cover had increased three-fold over pre-treatment and control levels, while soil erosion rates, as inferred from changes in surface topography, were significantly reduced relative to control.

Introduction

In north-central New Mexico, piñon-juniper woodlands, comprised of Colorado piñon (*Pinus edulis* Engelm.) and one-seeded juniper (*Juniperus monosperma* (Engelm.) Sarg.), have become a dominant cover type occupying millions of acres. Historic grazing pressures, and the concomitant loss of fine fuels during the last 150 years, disrupted natural fire regimes in many western grassland, savanna, and forest. Loss of periodic fire disturbance in these plant communities in the latter half of the 1800's, provided fire intolerant woodland species opportunities to both expand in range and increase in density (Miller and Wigand, 1994; Gottfried et al., 1995; Gruell, 1999; Tausch, 1999). Expansion of woodlands from 'traditional' fire proof habitats (i.e. rocky, shallow soil sites) onto deeper soil sites (i.e. former grass dominated communities), as well as increasing crown closure within older growth stands, imposes competitive effects on herbaceous ground cover and extends the physical impacts initiated by historic grazing. Loss of herbaceous cover, and conversely exposure of bare soils, can trigger accelerated runoff and erosion resulting in degradation of both the upland watershed and downstream riparian zone. Within woodlands areas of Bandelier National Monument, accelerated runoff and losses of soil are progressively degrading biological communities and embedded cultural resources.

With the structural homogeneity imposed by a woodland overstory, increasing canopy closure can make these communities vulnerable to large-scale, episodic mortality resulting from drought, insect, disease, and fire. Catastrophic loss of the tree overstory, through any of these episodic disturbances, can leave the system vulnerable to extreme degradation from erosion, loss of biological diversity and productivity, and establishment

of invasive plants. In woodland areas of Bandelier with productive soils, canopy closure was recently approaching levels sufficient to sustain crown fire until extreme drought in 2002 precipitated widespread mortality of piñon pine.

While an appropriate management response to woodland issues is desirable, an incomplete understanding of the historical dynamics of western U.S. woodland systems has fueled controversy over best management practices (Swetnam et. al, 2000). Central are questions about the relative roles of climate and episodic disturbance patterns on long-term vegetation dynamics versus the effects of more recent land-use history including harvest, ungulate herbivory, and loss of fire regime. This unresolved debate constrains efforts ideally designed to improve watershed, wildlife, and range conditions, restore system health and productivity, and mitigate impending wildfire hazards.

Results from small scale experimental restoration work initiated at Bandelier in 1994 (Jacobs and Gatewood, 1999) were encouraging and suggested recovery of degraded woodland areas was possible through the application of a mechanical overstory thinning and slash mulching treatment. Results after three years post-treatment were highly significant with two to seven-fold increases in total herbaceous cover relative to both control and pre-treatment condition. Tree thinning and the increase in needle and branch litter and associated decrease in exposed soil, resulting from distribution of the slash mulch, apparently improved site conditions for herbaceous plant growth and were casually observed to mitigate the erosional effects of summer monsoonal events on previously exposed intercanopy soils.

Building on these preliminary results, a large scale, paired watershed study was initiated in 1996 (Jacobs et. al, 2000) to validate findings across a greater range of site conditions, and measure response to treatment across multiple biotic and abiotic parameters at functional ecosystem scales. This report presents results for selected vegetation, soil, ungulate, arthropod, and cultural site components, from the paired watershed restoration study for the 1996-1998(9) timeframe. Response to treatment was assessed by documenting changes in surface cover (i.e. vegetation, litter, bare soil), plant biomass, and rates of soil erosion as inferred from changes in surface micro-topography (i.e. soil erosion bridges). In addition, a community level response was inferred by documenting changes in ground dwelling arthropod populations and site utilization by ungulate species. Cultural site stability was also assessed in order to document the effects of restoration treatment on cultural resources. Soil moisture, temperature and nutrient data, were also collected; methods are presented, but results for these parameters are not currently available.

The functional watershed framework of this study provided opportunities to conduct numerous supporting studies beyond the original scope of work; these are briefly summarized below:

- A first order, soil survey, conducted in 1997 delineated 12 soil types units within the two watersheds and immediately adjacent area (Davenport, 1997); this classification was simplified to four soil types using presence/ absence of surficial pumice and an underlying argillic horizon (Julius, 1999).
- An analysis of woodland age structure and understory composition across the three major soil types was conducted within the treated portion of the watershed study area (Julius, 1999).
- Butterfly response to watershed restoration treatment, recorded as species abundance and diversity along established vegetation transects in each of two watersheds over three separate years, was recently published as a separate report (Kleintjes et al, in press).
- Avian response to watershed restoration, based on data from representative point counts in each of two watersheds across four sample years, is in preparation (Fettig, personal communication).
- Sediment production data in response to watershed restoration treatment, measured in six sediment dams per watershed and analyzed as a function of soil type and rain event intensity, has also been recently published in a separate report (Hastings et. al., 2002).
- Surface runoff and suspended sediment production, in response to restoration treatment and across a range of event intensities, are being documented within each of the two watersheds in partnership with the USGS-WRD (Jacobs, personal communication).
- The use of fire as a tool for long-term maintenance of mechanically restored woodland savanna systems was also recently evaluated (Jacobs and Gatewood, 2002).
- Historical perspective on woodland systems and the central role of periodic disturbance (i.e. fire and drought) in shaping these communities has been documented in several USGS-BRD sponsored studies (Allen and Breshears, 1998; Allen, personal communication).
- Collaborative work with the USFS Rocky Mountain Experiment Station and Los Alamos National Laboratory also yielded several publications in support of woodland restoration efforts (Brockway et. al., 2002; Loftin, 1999).
- Finally, a project recently funded by NPS and USGS-BRD will characterize woodland systems and the effects of fire across a multi-state area where comparable issues (i.e. erosion, fire, and invasive plants) are limiting productive management of degraded lands (Jacobs, personal communication).

Study Area

GEOLOGICAL HISTORY AND TOPOGRAPHY

Bandelier National Monument is located on the southeast flank of the Jemez Mountains in north central New Mexico. The volcanic Jemez Mountains was formed from a series of events dating from 13 million years B.P. to as recently as 50,000 years ago. The Valles Grande, the central feature of this landscape, was created by two major eruptions at 1.6 and 1.2 million years B.P.; cumulatively these deposited the 300-meter thick Bandelier tuff in two distinct layers. The most recent eruption (El Cajete at 50-60,000 B.P.) covered the local landscape with many meters of pumice, much of which has been subsequently eroded and reworked, leaving pumice patches predominately on east facing slopes and deep alluvial deposits on lower slopes (Goff et. al., 1996). The eastern facing flank of the Jemez Mountains, called the Pajarito Plateau, has been incised into alternating landscape patterns of mesas separated by steep walled canyons. The study sites are located within two adjacent watersheds on South Mesa, within Bandelier National Monument, at an elevation ranging from 1900 to 2000m (figure 1). The mesa generally slopes from 2 to 3 percent to the southeast. The two watersheds drain from north to south into Lummis Canyon at a gradient of 2.5 percent with side slopes (facing east and west) ranging from 11 to 15 percent.

CLIMATE

Cool, dry winters and warm, wet summers characterize the climate across the region. The mean daily temperature extremes range between -10°C and 6°C in January and 13°C and 30°C in June, considered the coldest and warmest months of the year, respectively. Mean annual precipitation is about 427mm as recorded at the fire lookout tower near park headquarters on the mesa immediately north of the study site (unpublished weather records for 1925-2002, on file at Bandelier National Monument). Winter precipitation is delivered principally as snow by low-pressure systems that sweep from west to east across the Southwest, coalescing with moisture from the Pacific Ocean or the Gulf of Mexico. Winter precipitation is generally followed by a seasonal dry period during the months of May and June. This dry period is defined as much by increased potential evapo-transpiration that accompanies increased day length, solar radiation, and temperatures, as by decreased precipitation. The spring dry period is usually relieved by the onset of the Mexican monsoon; this weather pattern typically delivers at least 40 percent of the annual precipitation during the months July through September, and is associated mostly with short duration, high intensity, thunderstorms. Each summer as high pressure becomes entrenched off the coast of Baja California, low pressure in the Southwest feeds pacific moisture across the region fueling the development of afternoon thunderstorms. The magnitude, frequency, and tracking of individual large, intense thunderstorm cells during this period can account for large year-to-year variability in annual rainfall and between local areas. From 1996 through 1999, mean annual precipitation was normal to above normal, except for 1998, which was below normal (figure 2). Although a minor drought was experienced from October 1995 until June 1996, rainfall was normal to above normal for the July-September growing season period

from 1996 through 1999. Since 2000, annual precipitation has been below mean values, with several dry winters and relatively weak monsoons, although growing season moisture was only slightly below normal until 2002, when relatively severe drought conditions prevailed.

SOILS

A first order soil survey was conducted in 1997 and identified 12 mapping units within the two watersheds and the adjacent surrounding area (Davenport, 1997). Soils encountered during the mapping process are in various stages of development ranging from entisols to alfisols. The presence of buried and exposed buried soils suggests alternating periods of stability and instability throughout the geomorphic history of the landscape (Davenport, 1997). Surface horizons range in texture from loam to sandy loam, with deeper horizons ranging from clay loam and silty-clay loam to sand. Soil depth can be highly variable across relatively short distances, although the deeper, more productive soils are generally alluvial deposits on lower slopes. For the purposes of this study, we use a generalized version of the Davenport soil types provided by Julius (1999). This generalized approach recognizes four soil groups (figures 3 and 4) depending on presence or absence of surficial pumice and a buried argillic horizon (<120cm deep): pumice-argillic (PA), pumice- non-argillic (PNA), non-pumice-argillic (NPA), and non-pumice-non-argillic (NPNA). Surficial pumice soils provide greater infiltration capacity while mitigating evaporative losses and physical heaving, however it can be relatively lacking in water storage capacity and organics. Non-pumice soils generally are more vulnerable to surface erosion given low infiltration capacity (i.e. high runoff in absence of vegetation) and susceptibility to frost/drought heaving. Presence of an argillic horizon can act as an impermeable layer which perches water at depth; the relative depth of the argillic horizon determines whether perched water is available to plants and whether more available to shallow rooted herbaceous plants (argillic horizon <120cm) or more deeply rooted woody plants. A soil survey of the park by the NRCS has also recently been completed.

VEGETATION

The vegetation in the control and treatment watersheds is broadly characterized as piñon-juniper woodland with 35% to 40% mean tree canopy cover (figure 5). At a finer scale, and relative to soil type and topographic position, tree cover can vary from relatively open stands with established herbaceous, sub-shrub, and shrub understory, to nearly closed stands with needle litter beneath canopies and bare soil dominating intercanopy spaces. Considerable numbers of Ponderosa pine snags and logs are scattered across portions of the watersheds; in these areas Ponderosa was the formerly the dominant tree species until severe drought and subsequent bark beetle outbreak in the mid-1950's effectively converted these areas to woodland (Allen and Breshears, 1998). Establishment of piñon pine and one-seed juniper into the understory of the former ponderosa pine savanna, after cessation of fire in 1880, likely exacerbated drought conditions, and set the stage for type conversion to woodland. Remnant stands of ponderosa pine are now largely restricted to stringers along the primary channel in the bottom of the two watersheds; this drainage area is not a part of the sampled study area. More recently, drought conditions

beginning in 2000 and intensifying in 2002, with associated beetle infestation of moisture stressed trees, have resulted in near complete mortality of Ponderosa Pine (90%+) and very high mortality of the piñon pine (80%+). The severity of the current episode of mortality is likely exacerbated by relatively high density of trees both competing for limited moisture and facilitating beetle transmission. Casual observations suggest mortality levels of piñon pine and ponderosa within the treatment watershed are nearly as high as within the control area, despite removal of nearly three-quarters of tree density in 1997. However, beetle levels may have overwhelmed any moisture benefits within a treatment surrounded by landscape of untreated woodland. Additional field work is necessary to document the effects of severe drought (i.e. piñon pine mortality and herbaceous response across treatment and control areas) and implications for degraded woodlands and restoration.

The generally sparse herbaceous understory (<10% cover) is comprised principally of native, warm season grasses, including little bluestem, blue grama, and mountain muhly; these species are typically found in intercanopy spaces. Cool season grasses, including muttongrass, June grass, and littleseed ricegrass, are often found beneath the protective canopy of trees. A distinctive cool season grass of the intercanopy, Galleta, was probably more common before grazing and woodland expansion. A great variety of perennial forbs, as well as annual and biennial forbs can be found depending on local site conditions and weather patterns. Common shrubs include oak, rabbitbush, and sumac, with sub-shrubs such as wormwood, snakeweed, and pinque. Several genera of cacti are also present, with species of prickly pear dominating. Identification of specimens and nomenclature follow the Flora of Bandelier (Jacobs, 1989) and are based on voucher specimens housed at the Bandelier National Monument Herbarium. A recent Vegetation Management Plan (Jacobs, 2003) provides an overview of plant community classification and vegetation issues at Bandelier. A listing of common vascular plant species encountered in the study area, with scientific and common names, is provided in Table I.

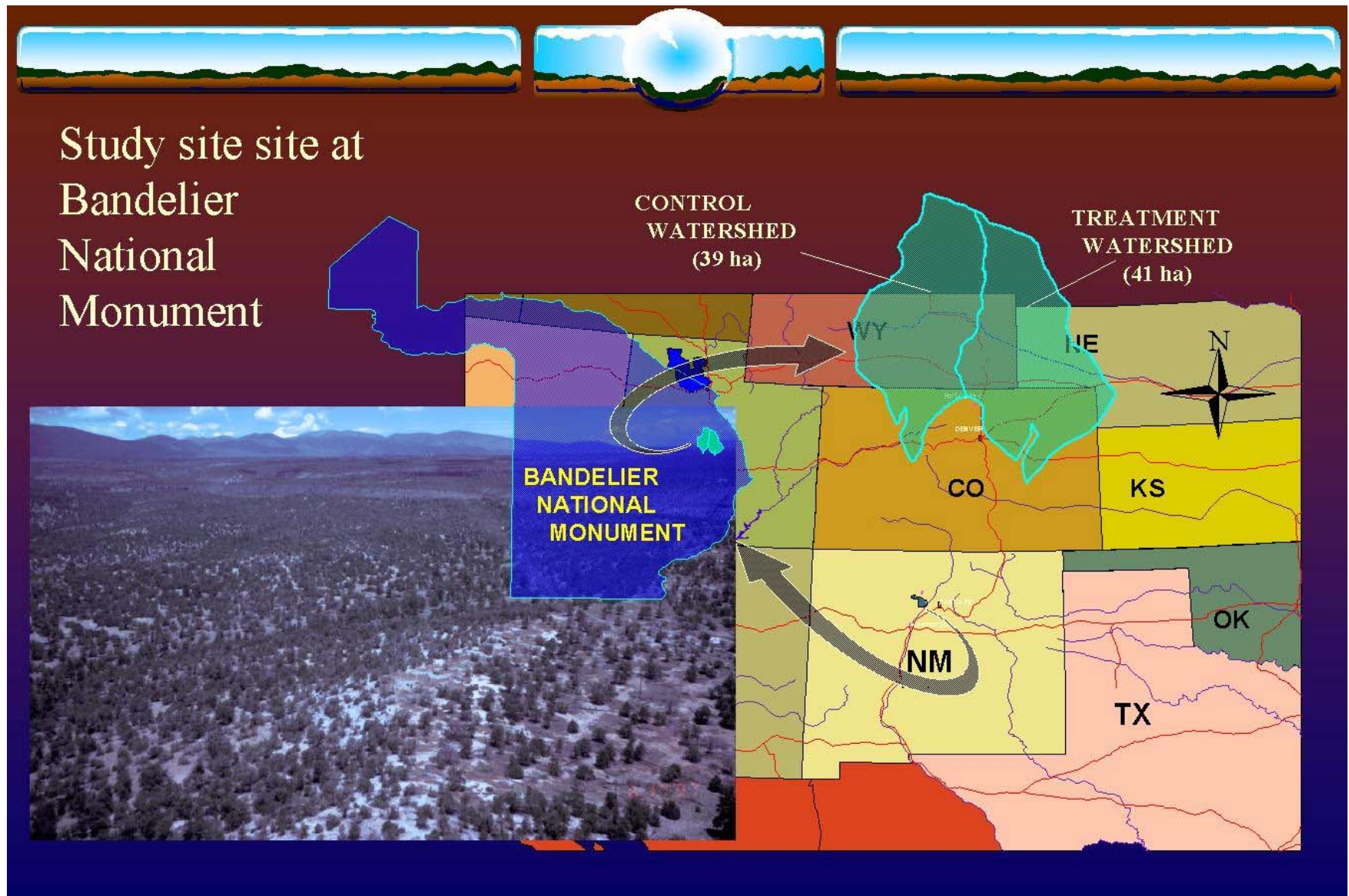


Figure 1. Location of the paired watershed restoration site, Bandelier National Monument, Los Alamos, New Mexico

Monthly Rainfall Patterns (1996-1999)

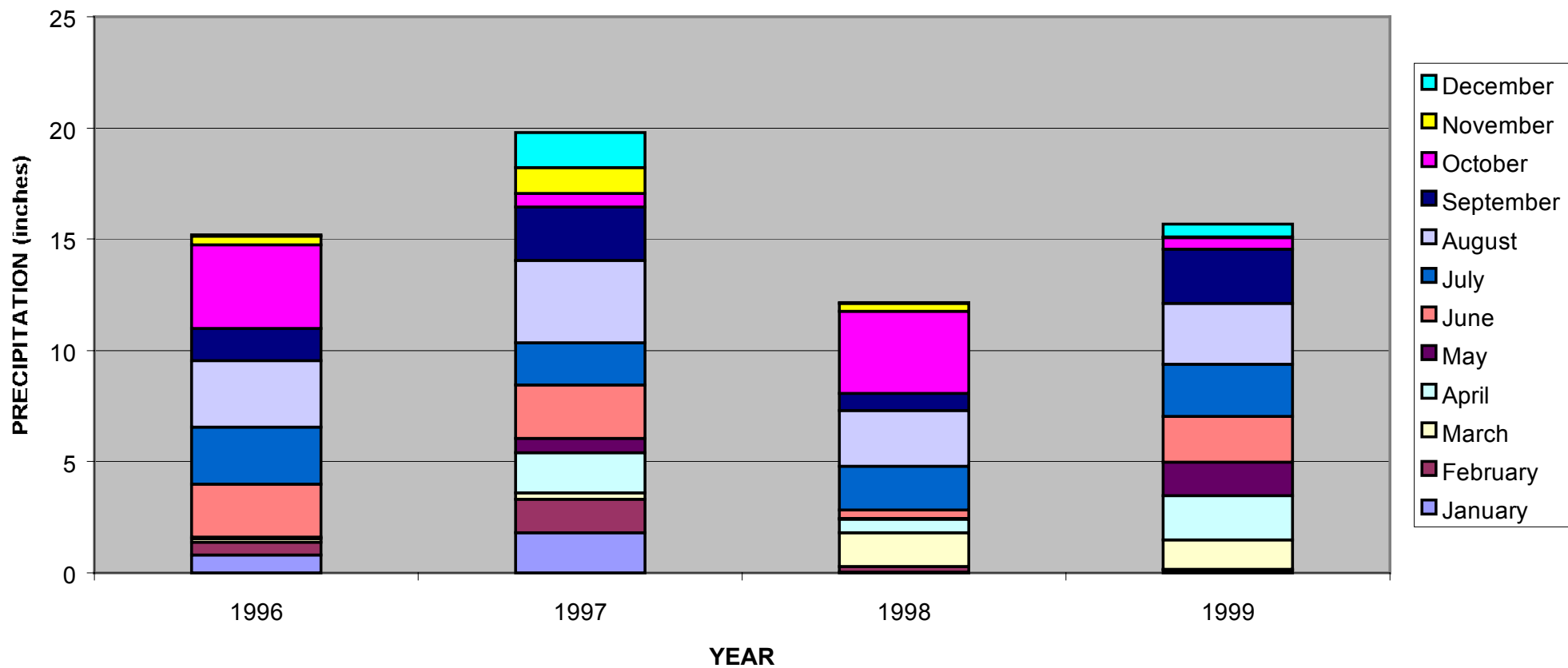


Figure 2. Monthly rainfall patterns for each of four years (1996-1999); from unpublished weather data collected at Bandelier National Monument, Los Alamos, NM

Generalized soil map

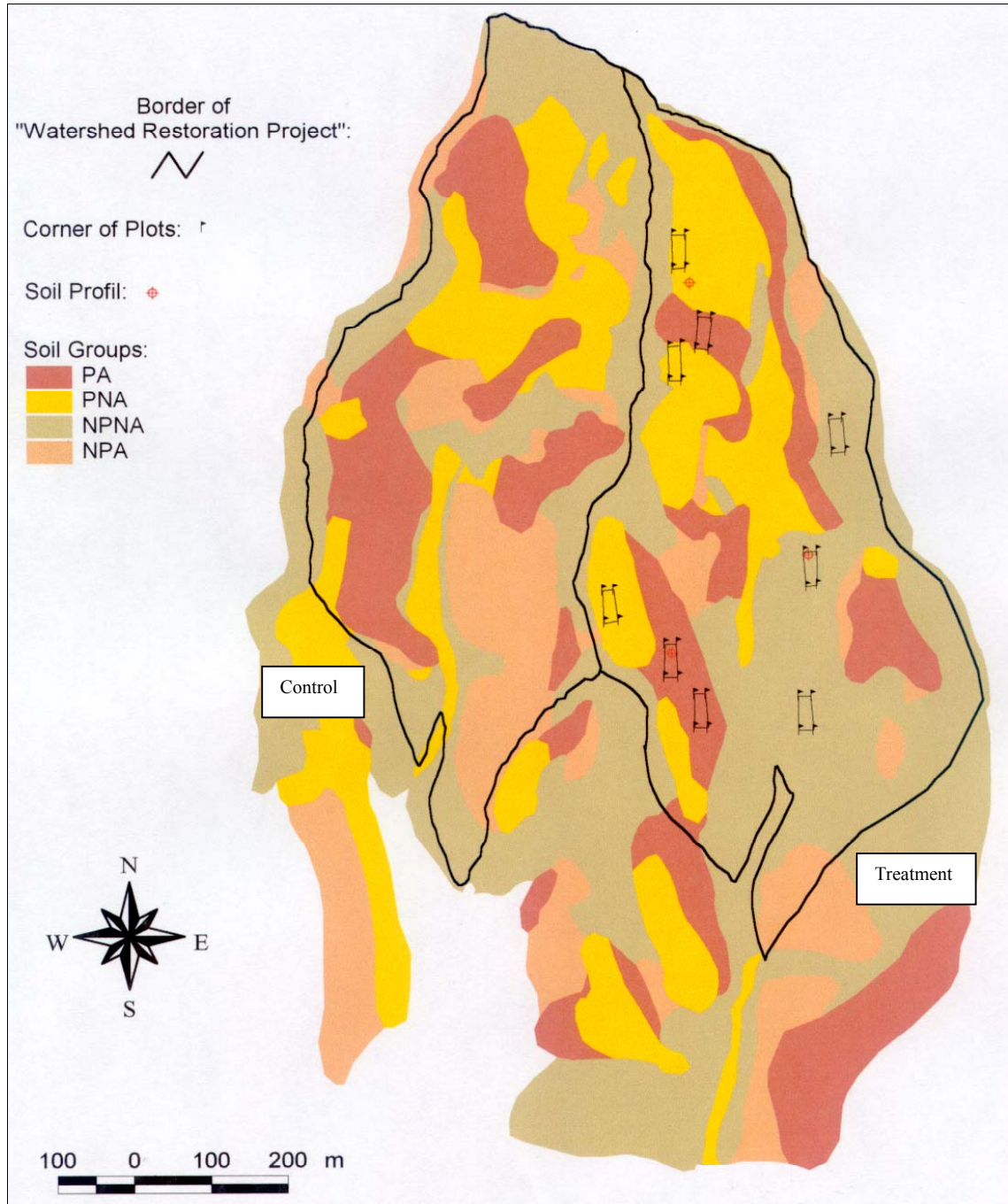


Figure 3. Generalized soil map of the paired watershed restoration study area (Julius, 1999).

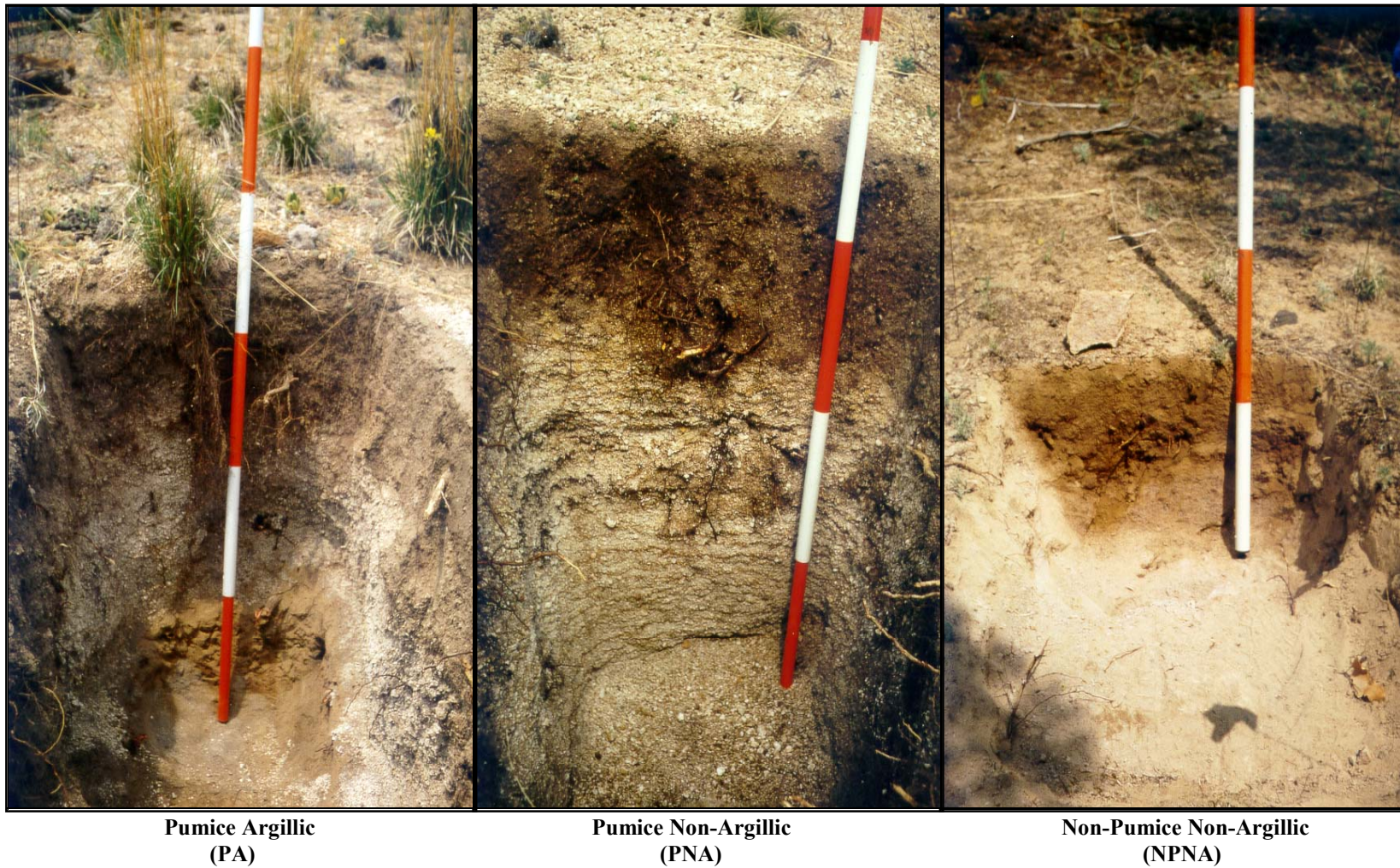


Figure 4. Major soil types within watershed project area; see text (Study Site: Soils) for detailed soil descriptions

Aerial view of the paired watershed restoration study area

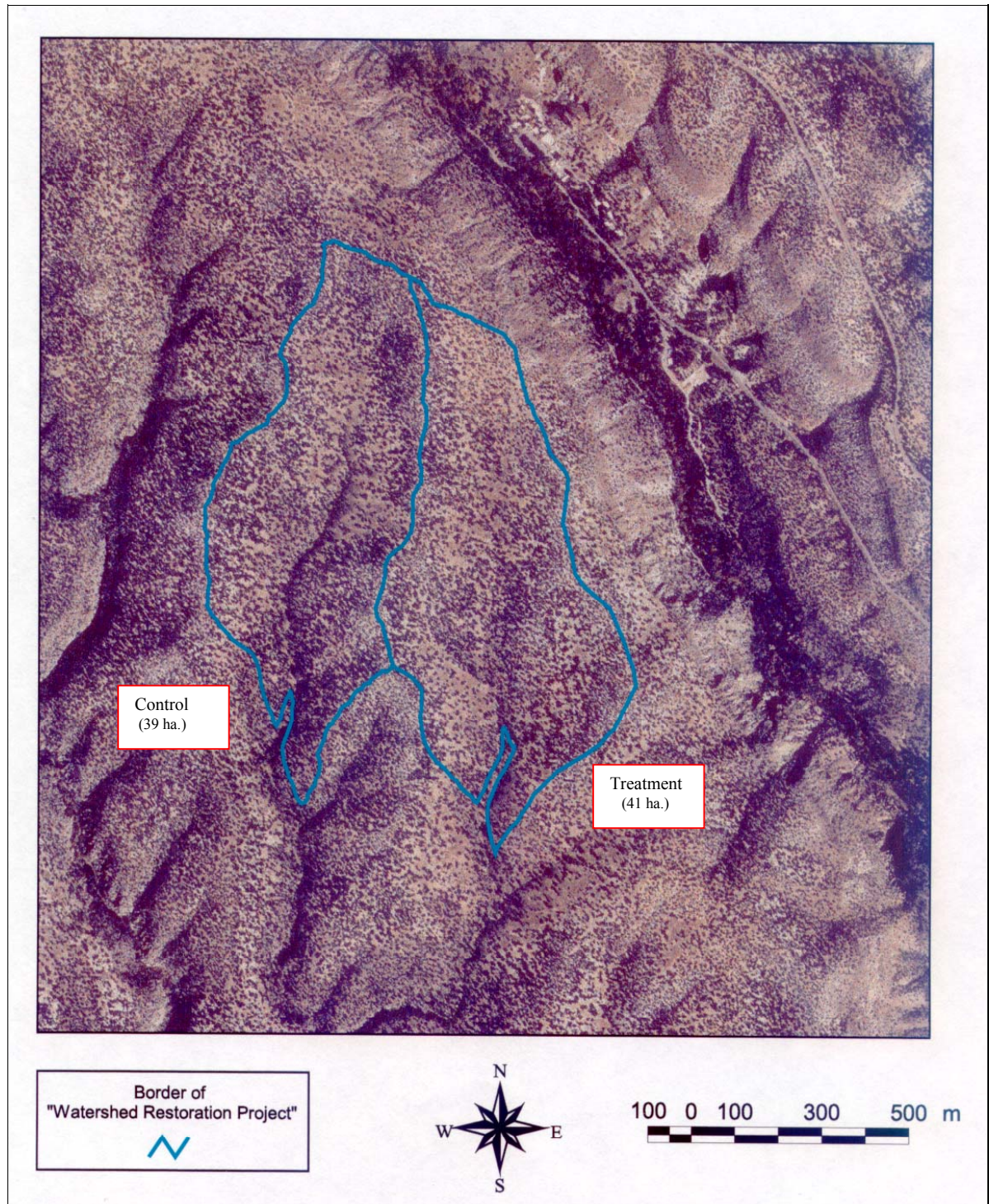


Figure 5. Aerial view of the paired watershed restoration study area (ca. 1991) showing general topography and vegetation within woodland zone of Bandelier National Monument, Los Alamos, NM

TABLE I: Listing of common vascular plant species in the study area

TREES	One-Seed Juniper	<i>Juniperus monosperma</i>
	Piñon Pine	<i>Pinus edulis</i>
	Ponderosa Pine	<i>Pinus ponderosa</i>
SHRUBS	Wavyleaf Oak	<i>Quercus unguolata</i>
	Mountain Mahogany	<i>Cercocarpus montanus</i>
	Rabbitbrush	<i>Chrysothamnus nauseosus</i>
	Apache Plume	<i>Fallugia paradoxa</i>
	Skunkbush	<i>Rhus trilobata</i>
SUB-SHRUBS	Snakeweed	<i>Gutierrezia sarothrae</i>
	Pinque	<i>Hymenoxys richardsonii</i>
	Wormwood Sagebrush	<i>Artemisia species</i>
GRASSES	Little Bluestem	<i>Andropogon scoparius</i>
	Mountain Muhly	<i>Muhlenbergia montana</i>
	Bluegrama	<i>Bouteloua gracilis</i>
	June Grass	<i>Koeleria cristata</i>
	Mutton Grass	<i>Poa fendleriana</i>
	Little-seed Ricegrass	<i>Oryzopsis micrantha</i>
	Three-awn	<i>Aristida species</i>
	Bottlebrush Squirrel Tail	<i>Sitanion hystrix</i>
	Galleta	<i>Hilaria jamesii</i>
PERENNIAL FORBS	Golden Aster	<i>Chrysopsis villosa</i>
	Deervetch	<i>Lotus wrightii</i>
	James Hiddenflower	<i>Cryptantha jamesii</i>
	Spreading Fleabane	<i>Erigeron divergens</i>
	Trailing Fleabane	<i>Erigeron flagellaris</i>
	Sunflower	<i>Helianthus petiolaris</i>
	Plumed Boneset	<i>Kuhnia chlorolepis</i>
	Many-headed Groundsel	<i>Senecio multicapitata</i>
	Groundsel	<i>Senecio species</i>
	Indian Paintbrush	<i>Castilleja integra</i>
	White Prairieclover	<i>Petalostemum candidum</i>
	Purple Wallflower	<i>Sisymbrium linearifolium</i>
	Golden Blazing Star	<i>Mentzelia pumila</i>
	Long Trumpet Gilia	<i>Gilia longiflora</i>
	Wormwood Sagebrush	<i>Artemisia species</i>
ANNUAL OR BIENNIAL FORBS	Golden Ragweed	<i>Bahia dissecta</i>
	Mullein	<i>Verbascum thapsus</i>
	Lamps Quarters	<i>Chenopodium fremontii</i>
	Ragleaf Goosefoot	<i>Chenopodium graveolens</i>
	Kings Lupine	<i>Lupinus kingii</i>
	Wrights Clubflower	<i>Cordylanthus wrightii</i>
	Doveweed	<i>Croton texansis</i>
SUCCULENTS	Prickly Pear	<i>Opuntia species</i>
	Banana-leaf Yucca	<i>Yucca baccata</i>
	Narrow-leaf Yucca	<i>Yucca glauca</i>

Problem Statement

Piñon (*Pinus edulis* Engelm.) and one-seed juniper (*Juniperus monosperma* (Engelm.) Sarg.) woodlands occupy nearly 10,000 acres within Bandelier National Monument and millions of acres in north-central New Mexico. Domestic livestock grazing, beginning in the late 1600's and intensifying with the arrival of the railroad in 1880, transformed fire dependent grassland and savanna communities into woodlands choked with young tree growth. Overgrazing reduced standing herbaceous biomass and continuity of fine grass fuels, effectively preventing fire from propagating in these systems (Gottfried et. al., 1995). As the soil-stabilizing, grassy ground cover was supplanted by increasing densities of trees, highly erodible soils were exposed to physical processes (see Conceptual Model). Bandelier National Monument, a mountain wilderness and cultural landscape of abandoned pre-historic settlements, was established to preserve the unique ancestral remains of Puebloan Indian peoples. The majority of the archeological sites within Bandelier National Monument occur within piñon-juniper communities; of these over 90% have sustained erosional damage (Mozzillo, 1999). Rapid soil loss in degraded piñon-juniper communities appears unsustainable and threatens the integrity of the thousands of pre-historic cultural sites. Without prompt management action to stabilize soils, we predict wholesale loss of soils and cultural resources within the next 100 years. Approximately 8,000 acres of woodland at Bandelier are in critical condition, relative to soil loss and threats to cultural resources, and could benefit from proposed management restoration actions.

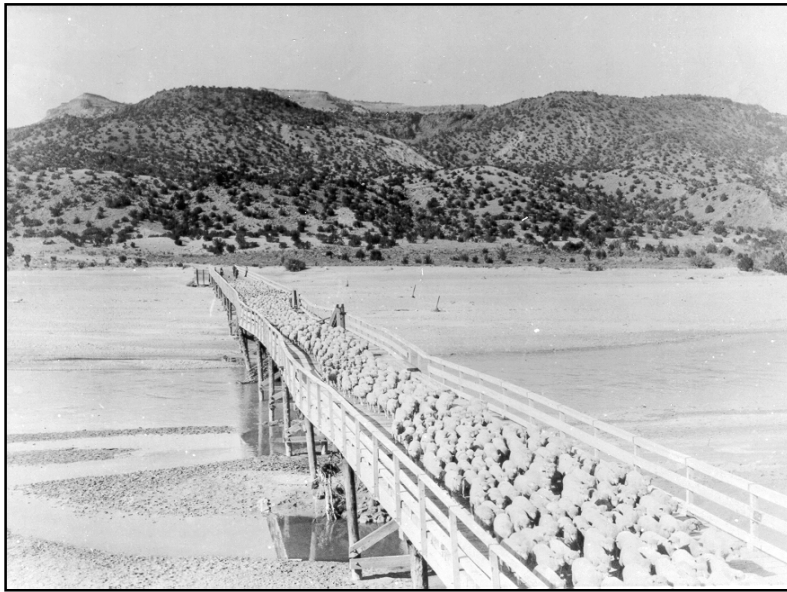
The loss of fire disturbance, initially as a result of overgrazing and subsequently through active suppression (Allen, 1989; Touchan et. al., 1994; Gottfried et. al., 1995; Morino et. al., 1998) had a profound influence on the dynamics of piñon-juniper systems. Age class information from piñon-juniper woodland sites in our area (see Historical Evidence) suggests an exponential increase in piñon-juniper stem densities in former grassland and savanna areas beginning around 1800 (Julius, 1999; Davenport et. al., 1996; Gottfried et. al., 1995;). Overstocked with young trees and lacking an effective ground cover, degraded piñon-juniper systems are poorly equipped to manage limited soil and water resources. Water competition is limiting for herbaceous plants in dense woodland situations, where surface runoff is high, infiltration low, and trees harvest most of the available soil moisture (Breshears et. al., 1997). Intensive characterization of erosional processes in a one hectare portion of a degraded piñon-juniper woodland at Bandelier (Wilcox et. al., 1996) suggest average annual soil losses of 10,000 to 20,000 kg/ha, most of it occurring during intense thunderstorm events typical of the summer monsoons. On the basis of both soil erosion bridge and sediment catchment data, soil loss within degraded piñon-juniper communities at Bandelier can be conservatively estimated at 1.0 cm per decade (Davenport et. al., 1996; Davenport, 1997; Davenport et. al., 1998; Wilcox and Breshears, 1995); an unsustainable rate given shallow soils with average depths of 1.0 to 12.0 dm. Despite the cessation of intensive livestock grazing pressure around 1940 (within Bandelier National Monument), degraded piñon-juniper communities appear to have little inherent capacity to recover on their own. Repeated measures of herbaceous cover in ungulate exclosures, established in 1975 on degraded piñon-juniper sites at Bandelier, provide compelling evidence that exclusion of grazing alone is

insufficient to promote recovery of these systems. (Chong, 1993).

Woodland areas above ~1900 meters elevation were selected as the initial focus for restoration studies, because of more favorable site conditions for re-establishment of an effective herbaceous ground cover. Rough terrain, high cultural site density, and the presence of designated wilderness precluded consideration of mechanized ground disturbing activities (such as chaining, drilling and other agronomic techniques utilizing heavy equipment) typically associated with large-scale restoration efforts. Small-scale efforts to develop and test restoration methodologies compatible with natural, cultural and wilderness values were initiated in 1991 (Chong, 1993, 1994) and 1994 (Jacobs and Gatewood, 1999). Based on promising results from small-scale studies, paired watershed level studies were initiated in 1996, both to validate treatment response at functional ecosystem scales as well as to educate the public about degraded systems and restoration options.

Results from experimental work, conducted by NPS, USGS-BRD and USFS staff at Bandelier and on the adjacent Santa Fe National Forest, suggest restoration of degraded, upper elevation, piñon-juniper woodlands using an overstory thinning and slash mulching restoration treatment, is feasible (Jacobs and Gatewood, 1999; Loftin, 1999). Thinning of the tree overstory reduces competition for limited water and nutrient resources, while the scattered slash provides benefits to exposed soils: reducing runoff and sediment transport, increasing infiltration and soil moisture, providing shade and moderating soil temperature fluctuations, freeze-thaw action and evaporation, redistributing nutrients, and mitigating grazing impacts. Combined, these effects appear to create favorable micro-sites for increased productivity of remnant herbaceous plants as well as for germination, establishment and growth of new herbaceous individuals (see Conceptual Model and Pictorial Overview). Testing of this restoration treatment, at multiple scales has produced compelling results relative to both pre-treatment condition and controls. Increases in total herbaceous cover and biomass, and decreases in exposed soils, in response to treatment are highly significant; in addition, the magnitude of the response appears to be sufficient to effectively mitigate accelerated erosional processes.

The current report documents selected results from the paired watershed study for the 1996-1998(9) time period. However, assessments of ecological restoration efforts require assessment over longer periods of record to increase confidence across multiple parameters. Additional data were collected for an additional two years (1999-2000) and will be reported elsewhere. These data may provide more compelling support for mechanistic theory relative to system response and allow year to year climatic variability to be factored out. Analysis of the separate datasets, which correlates changes in herbaceous cover and biomass with erosion rates, soil moisture and temperature, and relates abundance and diversity of arthropods, and utilization of the study site by avian and ungulate species, is critical to a more complete understanding of ecosystem response to treatment. An integrated analysis of the complete, five-year dataset is planned.

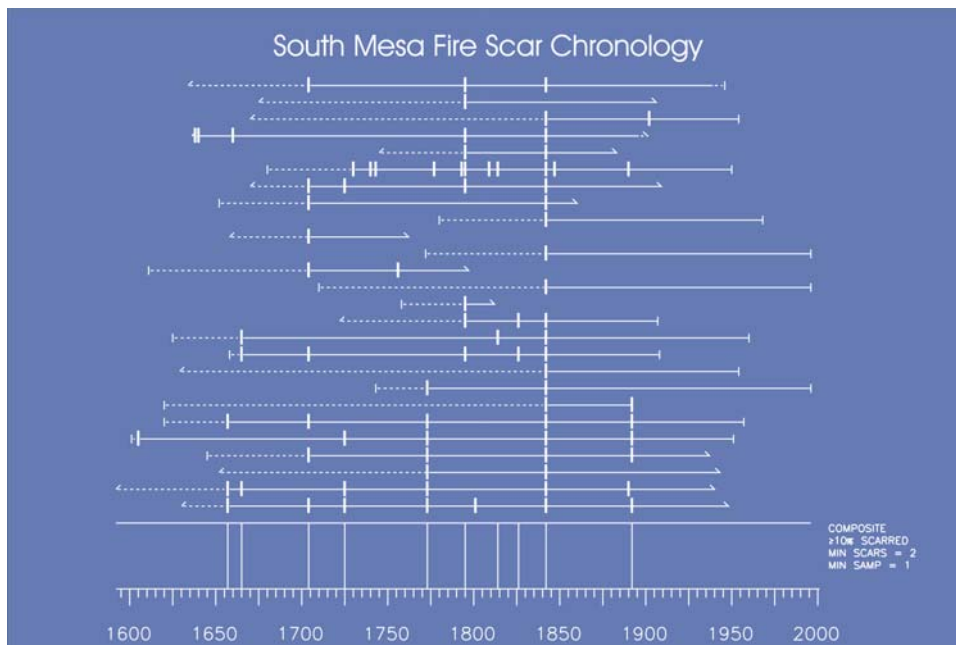


Historical

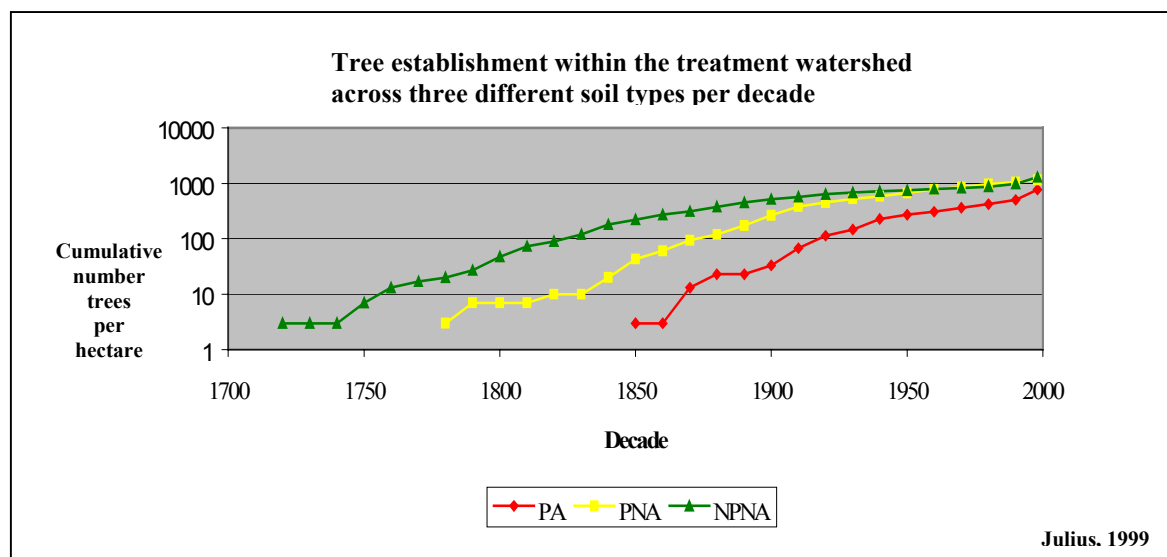
Evidence

Sheep crossing the Rio Grande
in early 1900's (left)

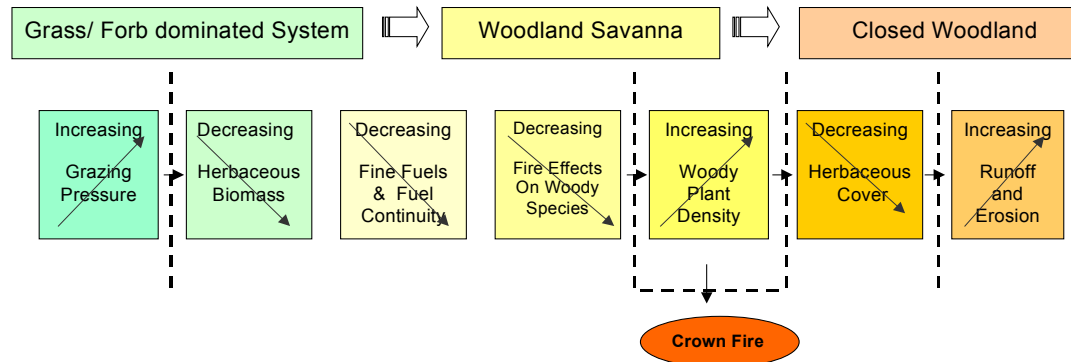
Local fire history
and woodland age class (below)



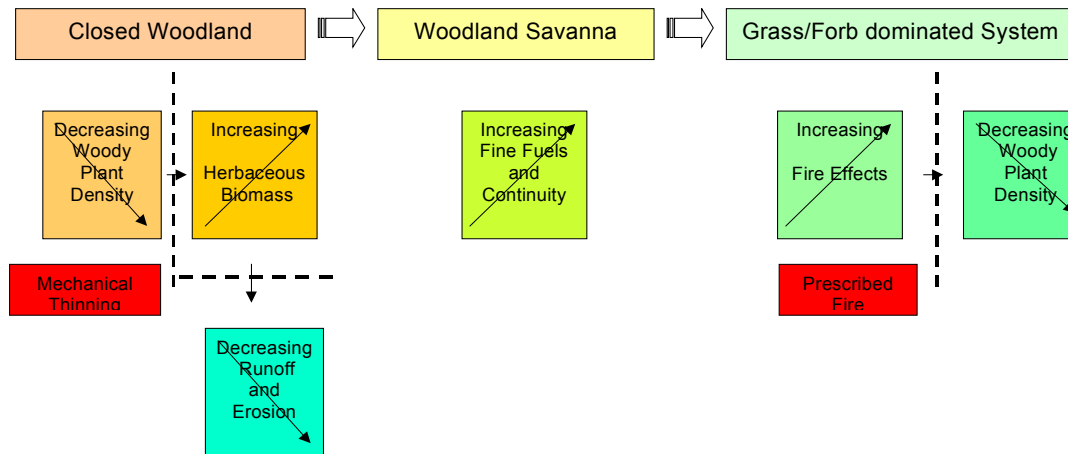
Morino et. al. 1998



Julius. 1999



Conceptual model of grassland-savanna-woodland continuum and relation to degradation/ restoration processes



Conceptual model of degradation and restoration processes and effects with key disturbance and system response variables noted.

Pictorial Overview: Watershed Restoration Project



Typical view of degraded woodland



Mechanical treatments allow selective removal of smaller (<6-8" DBH) trees and avoid damage to remnant herbaceous cover and cultural resources



Slash mulch improves micro-site conditions by mitigating surface runoff and evaporation



Aerial view looking across divide between treatment and control watersheds



Initial response to restoration is forb dominated



Generalized response to restoration

Methods

SITE SELECTION AND EXPERIMENTAL DESIGN

The general study site was selected using a variety of imagery to identify degraded woodland areas within Bandelier National Monument of sufficient size and with topographically comparable adjacent watersheds. Field reconnaissance was subsequently used to confirm similar soil and vegetation attributes across the two paired watersheds. An experimental design was developed to sample multiple biotic and abiotic parameters, provide a measure of system response to treatment, and potentially allow spatial and temporal correlation among the various indicators (i.e. integrated system response). Rationale for selection of various indicators, and specifics on sampling design, layout, and data collection methods are discussed below under the individual parameters. All sample points were permanently marked with labeled (aluminum and rebar) stakes driven several feet into the ground and geo-referenced using GPS to map into a GIS database. Using GPS and GIS generated maps, permanent points can be easily relocated with minimum effort, allowing repeated measures to be taken from the same locations. Baseline data for most parameters was collected during the summer and fall of 1996.

The restoration treatment was randomly assigned to one of the two 40 ha (100 acre) watersheds. The treatment consisted of a targeted reduction (i.e. two-thirds) in woodland tree density (i.e. one seed juniper and Colorado piñon pine), based in part on results of earlier work (Jacobs and Gatewood, 1999; Loftin, 1999; Bledsoe and Fowler, 1992), with a focus on removal of smaller diameter (presumed younger aged) individuals less than 6-8 inches just above base (except as noted for cultural sites). Individuals of Ponderosa Pine as well as woody shrubs were retained in all cases. The slash generated from the thinning operation was scattered onto adjacent exposed soil intercanopy areas. This prescription was selected to balance multiple objectives including: reducing tree density and competitive effects while maintaining older growth within the stand structure, generating adequate slash for mulching exposed soil surfaces, and releasing an herbaceous response sufficient to stabilize soils. The prescription was more conservative than those applied to earlier small-scale research plots (Jacobs and Gatewood, 1999), where tree canopy reductions of 95-100 percent yielded three to seven fold increases in herbaceous cover. Conceptually, leaving the larger trees was expected re-establish a stand structure similar to pre-grazing conditions. However this overly simplistic approach will be refined for future treatments on the basis of work by Julius (1999), who showed that diameter-age relationships are strongly correlated with soil type. The relative productivity of different sites therefore, can markedly affect annual growth increments, with smaller, older trees on shallow soils and larger, younger trees on deeper soils. Removal of two-thirds of total tree density was expected to reduce competition for soil moisture (and other resources) that limited herbaceous plant growth. In addition, lopping and scattering the slash into intercanopy areas was anticipated to serve as a physical barrier to reduce runoff and sediment transport, and conversely increase infiltration, while shading effects mitigated evaporation and buffered soil temperature.

Initial mechanical treatment was implemented in February 1997; a six person sawyer crew used chainsaws to fell trees, lop branches and trunk sections, flush cut stumps, and scatter slash preferentially onto bare soil intercanopy sites (see Pictorial Overview). This treatment was applied beginning at the watershed divide and going down slope 100 to 150 meters, leaving some untreated areas in the drainage bottom. Due to endangered species restrictions, restoration activities were suspended on March 1st with ~60% of the total area treated; mitigating measures allowed resumption of treatment on April 1st and the remaining portion of the treatment watershed was completed by the end of May.

VEGETATION AND SOIL COVER

One of the most robust measures of restoration treatment response is vegetative cover, although precipitation patterns can introduce considerable variation in the response of semi-arid plant communities. Vegetative response to restoration treatment was monitored using a line-intercept method to quantify basal and foliar cover of plants and associated ground cover (litter, rocks, wood, aerial slash, etc). Vegetation transects, 100 meters in length, were established on the upper one-half to two-thirds of the hillslope, perpendicular to the topographic contour (figure 6, Methods Overview). These transects were composed of two 50m segments (51.5m including the 1.5m unread leader) designated as either upper or lower segments, which were treated as separate observations (n=40 for each watershed). Beginning with a random start, the watershed transects were systematically placed, at 75 to 150 meter intervals around the periphery of each watershed. Each transect began at the ridge top and extended down slope, sometimes reorienting transects so to avoid running down the bottom of gullies, or crossing the main stream channel of the watershed. In addition to the general watershed area, archeological sites and ungulate exclosures were also monitored using the same methodology, but with different transect lengths (see additional details under **Methods: CULTURAL SITES and UNGULATE HERBIVORY**).

Baseline, pre-treatment, measurements were taken in Fall 1996 within both control and treatment watersheds with subsequent measurements taken each fall through 2000. Transects were read continuously (at centimeter resolution) recording both basal intercept and overstory. Basal intercept was recorded by ground cover type (i.e. plant species, soil, litter, wood) and totaled 100 percent. Overstory measures included multiple aerial layers recorded per species (or cover type) and could exceed 100 percent. Data was also summarized by growth form (trees, shrub, sub-shrub, grass, and forb) and cover type (i.e. soil, litter.). Similar plant species (i.e. bluegrama and hairy grama grasses) which could not be reliably distinguished in vegetative condition were lumped (as bluegrama) for purposes of this study. Likewise, microbiotic crusts could not be reliably distinguished from bare soil across different soil moisture conditions, and thus were included within the bare soil component. New crews were provided training in the line transect method, and calibrated with previous years measures (i.e. of aerial cover and resolution of basal intercept) to minimize year to year variability due to observer bias.

PLANT BIOMASS

In addition to cover, standing biomass and subsequent production of grasses and forbs, were also sampled. Perennial grasses produce new growth annually from a basal crown and this allows the above ground biomass to be harvested at the end of each growing season without inducing mortality or decreasing vigor. Forbs are more problematic in this regard, but we harvested these as well in 1997 and 1998, although with some likely impacts on 1998 response due to harvest in 1997. Selected sub-shrubs were also sampled in 1998 across both watersheds. Six, one-half meter (0.5 m^2) quadrats were placed at 20 meter intervals (parallel to and offset one meter from the line transects and beginning at the 0-m stake and ending at the 100-m stake) along each of the twenty vegetation transects per watershed (figure 7a). Plants within each quadrat were destructively sampled by clipping above ground biomass per species. The biomass samples were subsequently bagged by species in number 12 paper bags, oven dried at 60°C (140°F) for 24 hours, and weighed to closest .001g. Dry mass is expressed as kg/ha dry weight and allows us to estimate production by species across the two watersheds. Data were collected in 1996 (pre-treatment), and at one (1997), and two years (1998) post-treatment. Data were summed across the six (0.5m) quadrats sampled per transect and means per watershed calculated ($n=20$). In 1998, some loss of samples did occur due to delays in processing and subsequent molding; these molded samples were simply discarded given relatively few number and absence of any obvious bias toward either watershed.

CULTURAL SITES

Vegetative cover associated with pre-historic cultural sites was assessed using the line transect method described above for the general watershed area (VEGETATION AND SOIL COVER). Two, twenty five meter, transects were established on each of five cultural sites within each of the two, paired watersheds (figure 7b).

Bandelier National Monument has one of the highest densities of cultural sites in the National Park Service and the great majority of these are located within the woodland zone (Mozzillo, 1999). Within the watershed study area alone there are over fifty recorded sites with the majority being structural (i.e. abandoned stone structures) and the balance lithic (i.e. surface scatter of archeological material). The soils on structural sites, and to a lesser extent the associated vegetation, are often different from the adjacent watershed area in being elevated mounds of eolian deposited material which has filled in and buried stone wall alignments since abandonment; in addition, many of these structural sites located atop shallow, rocky soils typical of topographic highs favored for building locations. Lithic scatters (miscellaneous cultural materials, i.e. pot shards, rock tools, and worked volcanic glass on otherwise unaltered terrain) by contrast, are essentially identical to the immediately adjacent watershed area with regards to both soil and vegetation.

As a modification to the general thinning prescription imposed on the treatment watershed, tree cover was completely removed from all cultural sites within the treated watershed with heavier trunk and branch sections removed to perimeter areas as a preventative measure to mitigate heat effects in the event of fire. Within each watershed three structural and two, non-structural, lithic scatters were selected for sampling. Transects were oriented perpendicular to the long axis of each site and spaced five to ten meters apart depending on the size of the cultural site. Baseline data were collected in Spring 1997, immediately following restoration treatment, but prior to onset of the growing season; subsequent year measurements were collected during the same time period through 2001. Photos were taken along the line transects using the methodology described below (see: PHOTO DOCUMENTATION) for the larger watershed area, except that quadrat-delineated photos were taken at intervals of five meters.

ARTHROPODS

Restoration treatment was anticipated to affect productivity and diversity of selected arthropod functional groups. Forty arthropod pitfall traps were installed within each of the two watersheds to provide estimates of diversity and relative abundance per species. Two pitfall traps were installed along each of the twenty vegetation line transects per watershed, for a total of forty traps in each of the two watersheds (figure 8a, Methods Overview). Placement of the sampling points utilized a stratified, randomized sampling scheme. Along each transect, one trap was assigned to the upper transect segment (0 to 50 m) and one to the lower (60 to 100); traps were offset 10m from transects at randomly selected meter marks. Traps were constructed of 3" (O.D.) PVC pipe, cut in six inch lengths. These were used to line holes in the ground, and set flush with the soil surface. Each PVC trap was lined with a 10oz. Solo plastic cup. Traps are provided with a small plywood roof (elevated several inches above the ground and trap on six inch nails) to keep out rain and prevent disturbance by ungulates (which are attracted to the glycol). Selected traps required small rock or wood diversions to prevent runoff and sediment from filling in the cups. To begin a sampling period, cups were filled with four to five ounces of propylene glycol (Sierra Antifreeze) and placed in the PVC traps. After a sampling window of ca. eight weeks, pitfalls were collected using protocols developed by Dr. David Lightfoot at University of New Mexico (Brantley et al., 2000). Only those taxa that are known to be ground-surface active, and appropriately sampled by pitfall traps, were tabulated. Target taxa and functional group include: spiders (Araneae)- predators, bristletails (Microcoryphia)- detritivores, crickets (Orthoptera)- omnivores, and darkling beetles (Coleoptera: Tenebrionidae)- detritivores.

Pitfall samples were collected a total of five times across two growing seasons in 1998 and 1999: spring 98, summer 98, fall 98, spring 99, and fall 99. For each sample date, all 80 pitfall traps were collected and data per trap represent arthropods captured during the six to eight week sampling window. Resulting samples were sent to the University of New Mexico for processing. Bugs from each individual sample (trap x date) were identified, sorted and counted by taxa by staff under the direction of Dr. David Lightfoot.

SOIL MOISTURE AND TEMPERATURE

Soil moisture is easily the most limiting resource to plant establishment and growth in semiarid systems; its availability strongly influences the composition and structure of plant communities (Noy-Meir, 1973). In semiarid woodland systems, competition for water and other resources escalates with increasing tree density, and results in lower productivity of the associated herbaceous understory (Clary, 1986). Breshears et. al. (1997) observed that while woodland intercanopy soil moisture was higher as compared with tree canopy soils, due to interception of precipitation by the canopy (Johnsen, 1962), the extensive root systems of these trees effectively harvested much of the available intercanopy water and nutrients. For herbaceous plants, degraded intercanopy sites can be a harsh environment and a challenging place to establish and grow. In summer, solar radiation striking exposed soil surfaces can rapidly evaporate moisture and allow soil temperatures to reach levels lethal to plants. During colder months, exposed soils are subject to freeze-thaw action, which can heave and sever delicate seedling root systems.

Restoration treatment can improve micro-site conditions for herbaceous plant establishment by increasing available soil moisture, while moderating temperature extremes, and reducing moisture losses due to evaporation and transpiration. Thinning trees reduces canopy interception and losses from transpiration, although decreased canopy shade and a robust herbaceous response and would consume a portion of the gain. Dispersal of slash mulch on exposed soil surfaces intercepts runoff and increases infiltration, resulting in lower temperature maximums and dampening diurnal temperature fluxes, while mitigating losses from evaporation.

Time Domain Reflectometry (TDR) was used to sample soil conductance at 24 points (three arrays of eight points each) within each of the two, paired watersheds (figure 8b). Arrays were established at three locations on west facing slopes within each watershed and the eight probes were equally divided between canopy (former canopy in the treatment area) and intercanopy settings. Spacing between probes was kept to a minimum distance of 4m, and sampling from intercanopy settings was restricted to non-pumice soils. Probes are 30cm long, but soil depth is frequently less than 15cm; therefore probes were inserted at a steep angle to consistently sample only the top 10 to 15cm of soil profile. Soil temperature was also sampled from one point in each of the arrays; a soil temperature probe was buried in the top two centimeters of the soil profile in an intercanopy setting. Temperature measurements were taken concurrently with soil moisture measurements.

Soil conductance (i.e. the propagation velocity of an electromagnetic energy pulse) can be used to provide an indirect measure of soil moisture, since velocity varies with the amount of water in the soil. Repeated measurements were taken for five years (1996-2000) from the same 24 points periodically throughout the monsoon season that occurs from July through September. Measurements were taken at least once a month; as well as after significant moisture events (i.e. snowmelt, significant rain, or extended drought). No results for soil moisture and soil temperature are presented here.

SOIL NUTRIENTS

Second only to soil moisture, nutrient availability (and specifically nitrogen) is generally the most limiting resource to plant growth in arid and semiarid ecosystems. A number of authors have examined the spatial distribution and cycling of soil nutrients in piñon-juniper communities (Padien and Lajtha, 1992 ; Barth, 1980) and they consistently find that canopy soils have greater available nitrogen than the associated inter-canopy soils. Both biotic and abiotic mechanisms work individually or in concert to promote the creation of nutrient concentrations beneath woody plant canopies. Shrubs and trees acquire soil resources beyond the radial extent of their canopy via the lateral extension of their root systems into the intercanopy. These resources are subsequently deposited and accumulate as litter beneath the tree canopy, where it is readily available for recycling. Over time this results in a net increase in nutrient concentration in canopy soils over adjacent intercanopy soils. In addition, the above ground physical structure of shrubs and trees captures and holds nutrient resources (i.e. soil and organic matter) as they are dispersed by wind and water from adjacent intercanopy areas. Wind and water erosion then can progressively redistribute nutrient resources from intercanopy to canopy locations. Consequently, bare soil patches expand become increasingly connected to other patches creating pathways for sediment and nutrients to leave the plant community (Davenport *et al.* 1998).

Ion-exchange resin bags as described by Lajtha (1988) for semiarid ecosystems were employed to quantify in-situ availability of soil nitrogen in intercanopy and canopy soils in thinned and unthinned PJ woodlands. Three and a half grams of a cation and anion exchange resin were placed into groups of three per sample local to increase the probability of retrieving at least one bag from each observation point. Nutrients were sampled in association with soil moisture measurements over the course of the 1999-2000 growing season, with a set of (three) resin bags placed in close proximity to each of the soil moisture probes (≤ 0.5 m) and stratified by canopy/ intercanopy locations. Resin bags were placed in the soil to a depth of 2cm beginning in late spring, prior to the onset of the summer monsoon season, and extracted at thirty day intervals, with fresh bags emplaced through fall. Resin bags were subsequently processed at Colorado State University to determine availability of soil nitrogen (NH_4 and NO_3) in samples. No results for soil nutrients are presented in this report.

SOILS AND EROSION

Soil erosion rates were quantified and compared across treatment and control watersheds using a soil erosion bridge technique. Each soil erosion bridge samples changes (at mm resolution) in soil surface elevation for 31 points along a two meter bridge. Four sampling strata were identified relative to potential sources of sediment production; locations beneath trees (i.e. canopy) were excluded, since minimal erosion had been observed from these locals. Sample locations were initially stratified by channel versus inter-channel, and subsequently by either soil type or slope position as follows:

- 1) *upper slope, between-channel*: all inter-channel sample points were located on the upper slope positions (i.e. above the inflection point). These upper slope, non-channel surfaces were further sub-divided into two soil types on the basis of surface cover/ texture: pumice and non-pumice.
- 2) *upper and lower slope, within-channel positions*: Within-channel positions were further separated into upper (i.e. above the inflection point) and lower slope positions (figure 9b).

Within each watershed, soil erosion bridges were grouped into sets of five bridges located between each of the twenty vegetation transects for a total of eighteen groups (n=90) per watershed. Non-pumice sites received two of the five total bridges per set, with the other three strata receiving one bridge per set each. Upon implementation of mechanical thinning and slash mulch treatment it was decided not to put slash on or in the immediate vicinity of bridge locations. An additional four sets of four bridges (n=16 bridges per enclosure per watershed) were placed with each ungulate enclosure (figure 9a), Methods Overview). Soil erosion bridge sets (n=4) within enclosures included only one bridge per strata.

The erosion bridge is a, two-meter long, aluminum bar, with 31 precisely machined holes to accommodate milled aluminum pin rods of corresponding diameter. This portable apparatus is set onto and supported by two permanently installed stakes at each sample location. Initial installation requires the bridge to be level when set onto the two supporting stakes; wooden shims are used to make minor level adjustments and to secure the bridge in place while measurements are taken. Thirty-one aluminum pin rods are carefully lowered through each of holes in the bridge until the point of each rod contacts the soil surface (or other basal cover type). They are then clamped in place, suspending them from the bridge, to allow for precise measurements. Measurements are taken, at millimeter resolution, from the upper surface of the horizontal bridge to the top of each pin. One of the 31 pins is a reference point, which measures the distance to a permanent metal stake set near to ground surface. Changes in the soil surface profile or elevation from previous measurements, correspond to soil loss, accumulation, or no change, as averaged across the 31 pins of the erosion bridge. Measurements were taken twice a year, once in spring, prior to summer rains, and once in fall, post-monsoon and prior to soil freezing temperatures. Ideally this is before and after the summer monsoons, during which the most of the sediment moving rain events occur, as well as before freezing temperatures can lift and fluff soils.

Baseline soil surface elevations were taken in the fall of 1996 across 212 soil erosion bridges (n=106 bridges per watershed). Post-treatment measurements were taken beginning in late spring of 1997 (prior to the onset of monsoonal precipitation) and again in fall 1997, spring 1998, fall 1998, spring 1999, and fall 1999. Following measurements in the fall of 1998, a modification was made to the original experimental design. Fresh cut slash was distributed across half (i.e. one meter) of each erosion bridge within the treatment watershed. The rationale for this modification was based on the realization that raindrop splash was likely a major factor in mobilizing sediment for transport at the meter scale the erosion bridges were sampling (particularly for inter-channel locations); without slash, bridge locations within the treatment watershed were, in part, untreated islands. The presence of slash is thought to affect erosion locally by intercepting raindrop splash,

initially as a coarse mulch and subsequently as ground litter. In addition, slash can slow runoff and reduce sediment transport. The first measurements with slash present on treatment bridges were taken during the 1999 field season.

UNGULATE HERBIVORY

Ungulates, particularly elk and mule deer, are of interest because of the potential for grazing to impact sites undergoing restoration. Restoration treatment may create desirable foraging habitat for these animals, focusing ungulate use (particularly in winter), and selectively limiting the response of preferred vegetation. A two-hectare (5 acre) enclosure was erected within each of the paired watersheds, and sited to maintain comparability with respect to landscape position, soils, and vegetative cover. Enclosures are constructed of an eight stranded, eight-foot tall, electric fence, using existing trees for support, and powered by a solar charged 12-volt marine battery (see Methods Overview). Within each of the two enclosures, five, 60 meter vegetation line transects were established. These transects were established and read using the same methodology described above for the general watershed (see: **Methods:** VEGETATION AND SOIL COVER). Four soil erosion bridges were also established between each of the six transects (n=16 per enclosure) and measured per methods described for larger watershed (see: **Methods:** SOILS AND EROSION)

PHOTO DOCUMENTATION

Repeat photos were taken each fall in conjunction with the sampling of both vegetation transects and soil erosion bridges. Along vegetation transects, photo points were established at ten meter intervals, beginning at 0-m and going downhill to 100m (figure 7a). A 50cm x 100cm quadrat frame delineated the photo plot, oriented with the long side parallel to the transect and placed on the left hand side (facing downhill). Photos were also taken looking down the transects from both end points. At soil erosion bridge locations, a general photo was taken looking upslope which framed the bridge and immediate vicinity. Prepared site/year labels were placed within the field of view for later photo identification.

DATA COLLECTION, MANAGEMENT, AND ANALYSIS

Data were collected in the field using HP Palmtops 200LX, typically in DOS text, MS Notepad or Lotus 123 formats. Field data were subsequently downloaded, processed using a variety of customized (i.e. Permpilot) and MS Office applications (i.e. Excel), and stored on the project desktop computer. MS-Excel and Statistical Package for the Social Sciences were used to analyze the datasets and generate graphics. GPS data were downloaded to the park's GIS system and used to generate a variety of site maps (e.g. showing locations for sampling points, relation of sampling points to other data layers like soil type, or to calculate size and relative percent of features such as watersheds and exclosures). Data were regularly archived on portable media (i.e. zip and compact disks), as well as on the park server; backups were made at regular intervals to avoid data loss.

For purposes of this report, data were generally summed to obtain means for each parameter x treatment x sample date combination, plotted with standard error bars, and tested for significant differences (at 90% and 95% confidence levels as noted for individual graphs) among and between summed means (rounded to nearest whole number). Significant results were used to infer meaningful responses to treatment for each parameter and sample date. Cover data were collected per species (i.e. juniper) and cover type (i.e. bare soil), and aggregated to obtain estimates per growth form (i.e. tree= juniper + pinyon) or cover form (i.e. soil = bare soil + rock + pumice). Because the line transect method employed here allowed for measures of multiple, overlapping, aerial canopies, cumulative canopy cover can potentially exceed 100%. Basal cover was measured independently of aerial cover and cumulatively they are reported as total cover. Relative cover as used here, is component cover divided by the total cover for a particular cover type or growth form.

Methods Overview



Sampling vegetation and soil surface cover along line transect



Two hectare, solar electric ungulate exclosure within treatment watershed



Erosion bridge used to measure changes in surface elevation



Arthropod pit fall traps constructed of 3" PVC pipe with plastic cup liner (filled with antifreeze) and plywood roof.

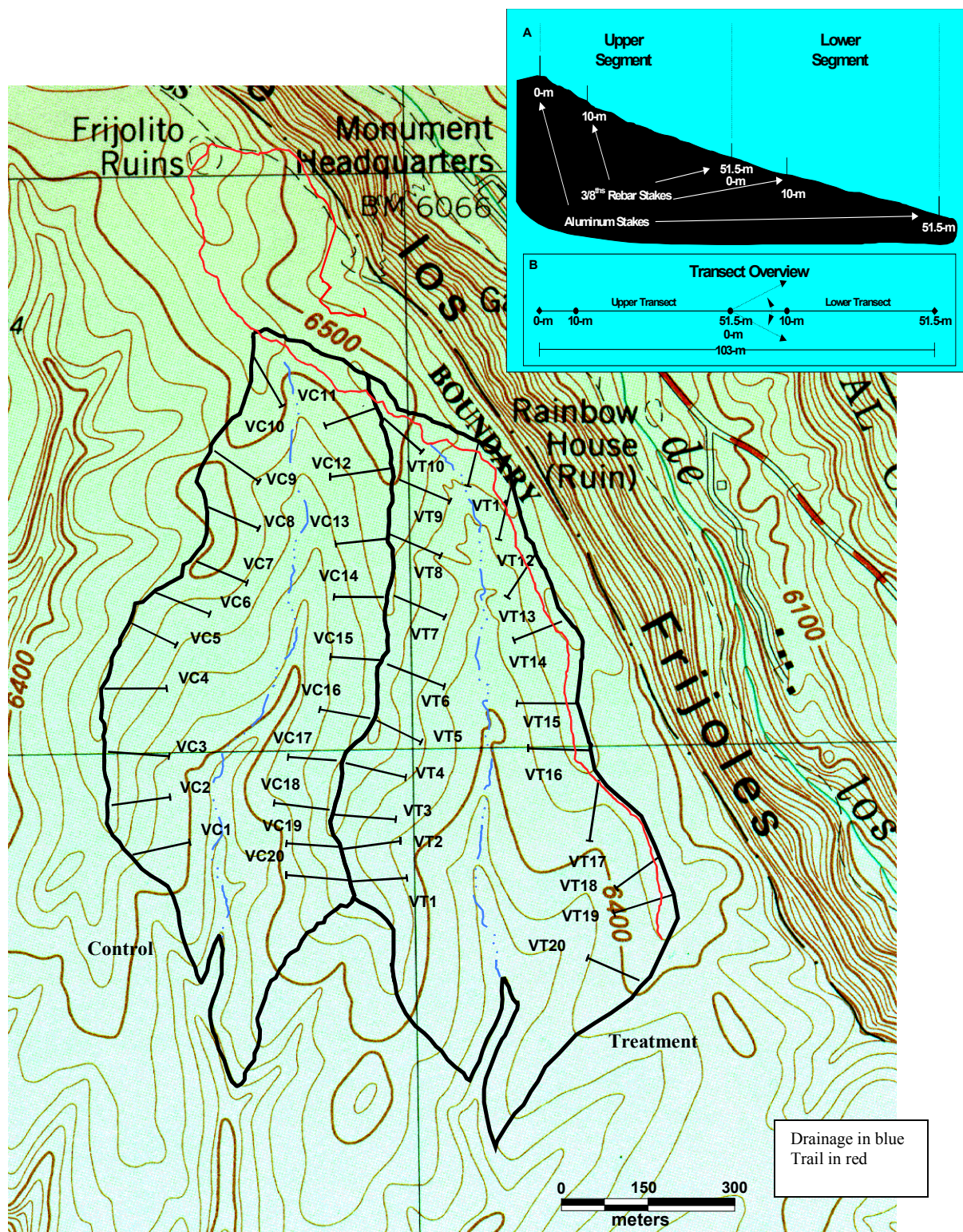


Figure 6. Location of vegetation transects (n=20) within the control and treatment watersheds. Each transect is divided into two, fifty meter segments (upper and lower slope). *INSET:* Two 50m transect segments comprise each of the 20, 100m transects per watershed. Each 51.5m segment includes a 1.5m unread leader, for total transect length of 103m.

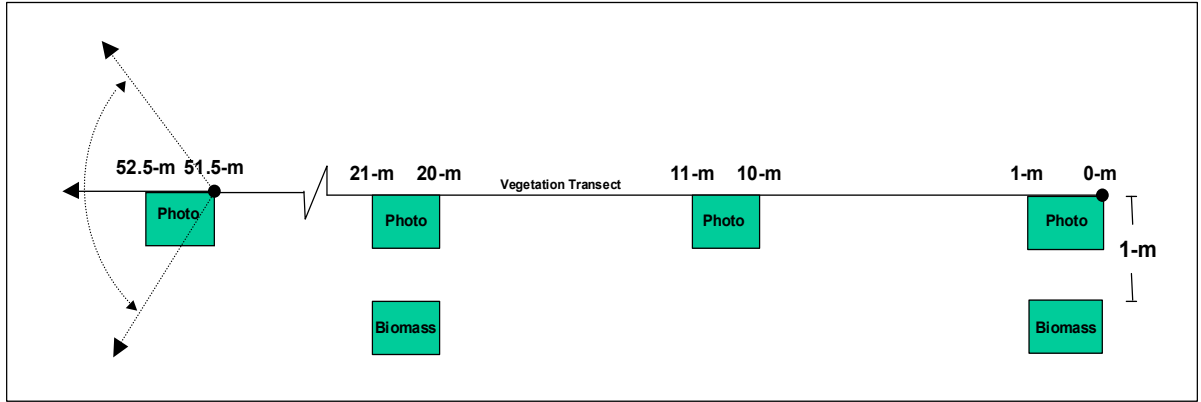


Figure 7a. Sampling layout for collecting biomass and quadrat photos. Biomass was collected at 20-m intervals, including end points, one meter offset of transect. Photos were taken at 10-m intervals along transect A .5m quadrat frame delineated the photo plot, oriented with the long side parallel to the transect and placed on the left hand side (facing downhill).

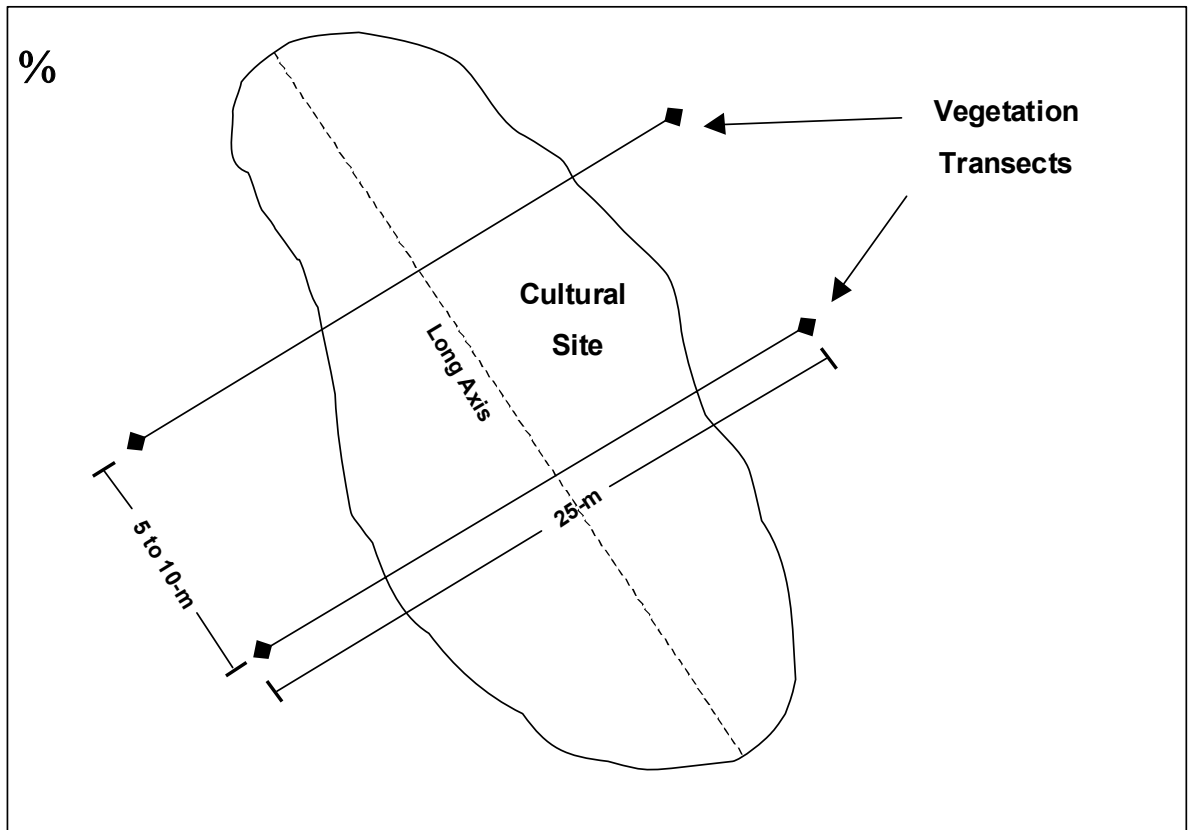


Figure 7b. General layout of vegetation transects on cultural sites. Transects are 25m in length, aligned parallel, and placed 5 to 10m apart.

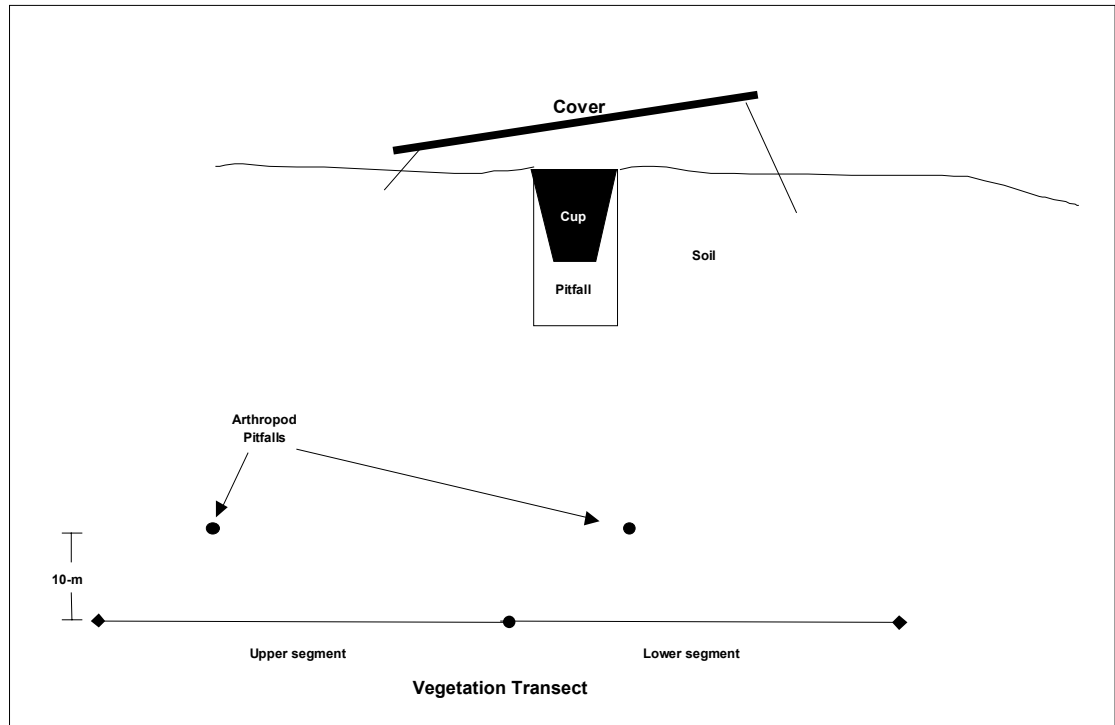


Figure 8a. Schematic of the arthropod pitfall trap sampling design. Pitfall trap locations are offset from the vegetation transect 10m.

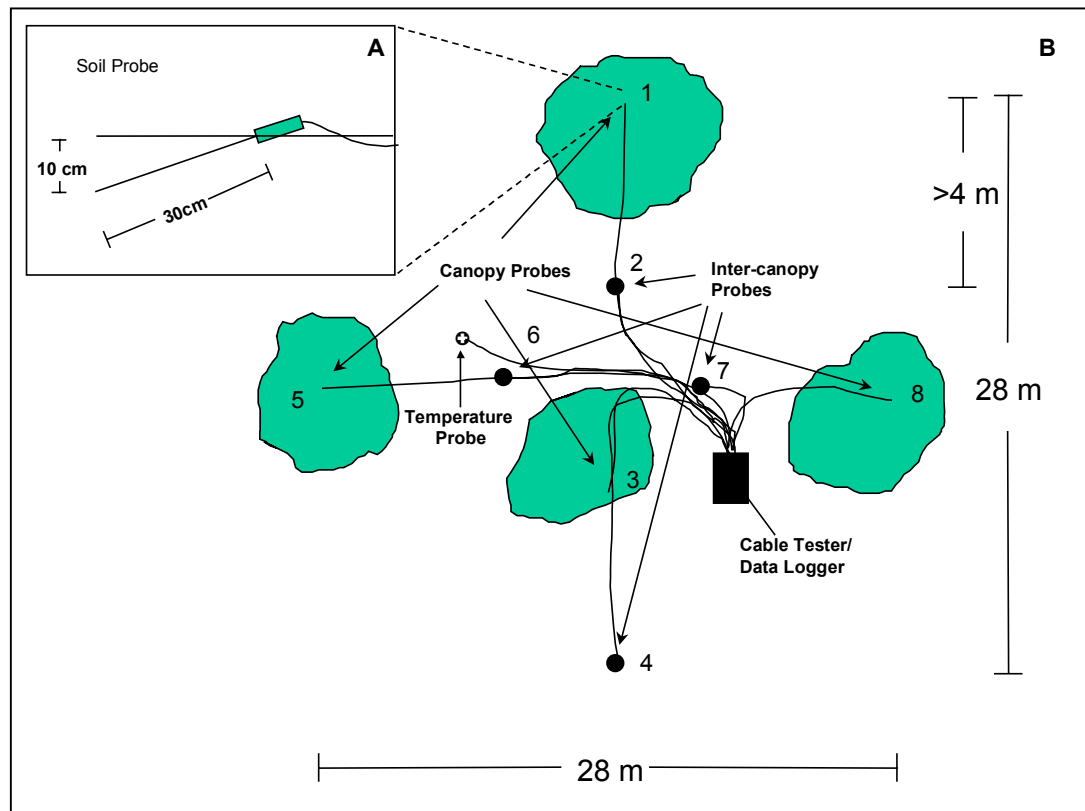


Figure 8b. General layout of a soil moisture probe array.

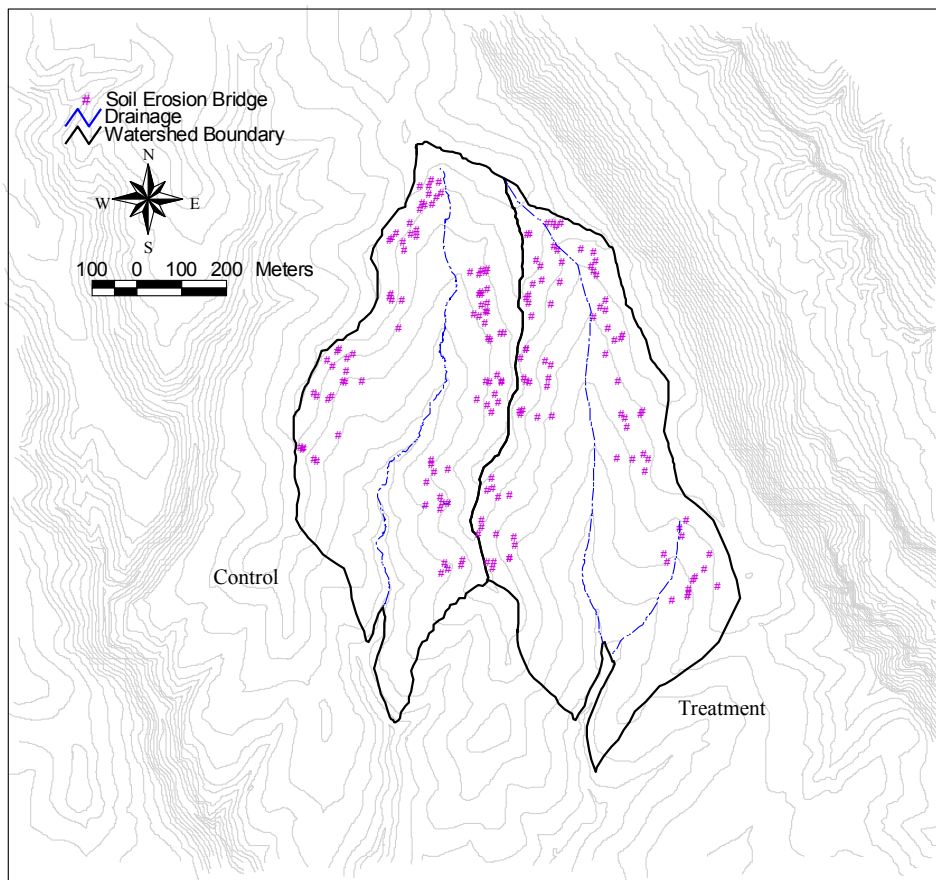


Figure 9a. Distribution of permanently located soil erosion bridges in the treatment and control watersheds (n=90).

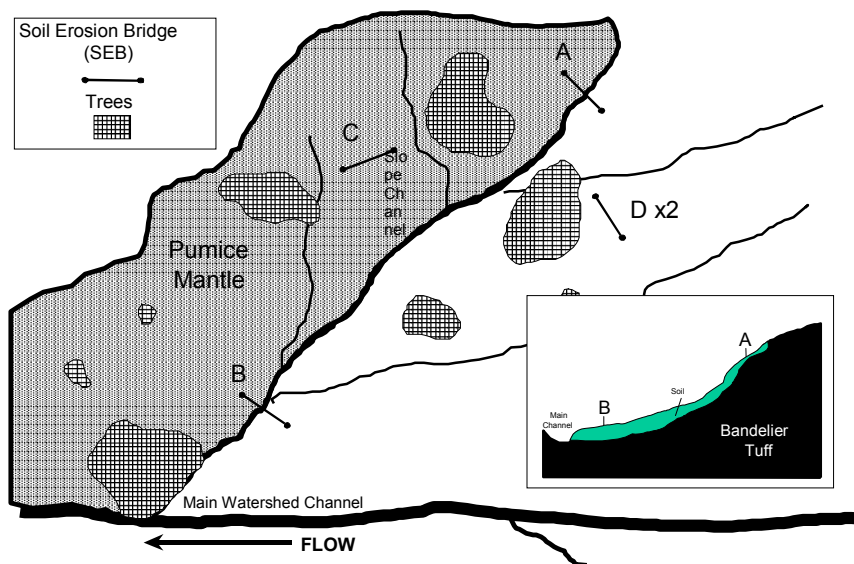


Figure 9b. Sampling design for placement of soil erosion bridges; bridges were grouped into eighteen sets of five bridges (n=90 per watershed); with an additional four sets of four bridges per enclosure (n=16). Bridge sets were stratified across four locales: upper and lower slope, non-canopy, within-channel positions (one bridge is located at the upper portion, above the inflection point, of channel (A), the second (B) below the inflection point), and upper slope, non-channel, inter-canopy positions divided into two soil types on the basis of surface cover; the third bridge (C) is located on on the pumice soils; the fourth and fifth bridges (D) are located on non-pumice soils.

Results

This report presents summary results for selected vegetation, soil, ungulate, arthropod, and cultural site components, from the paired watershed restoration study for the 1996-1999 timeframe. In addition to results for individual components, the project was designed to enable integration of response data from across multiple biotic and abiotic parameters. Although this objective is still largely unattained, it remains a compelling need to fully realize the value of the information contained in the separate datasets.

BASELINE VEGETATIVE CONDITIONS

Prior to mechanical treatment, vegetation and associated ground cover were measured to document baseline conditions (figure 10), and provide a reference to assess change over time and relative to treatment. The initial measures also provided some indication of similarity between the control and treatment watersheds.

Soil (including pumice, bare soil and rock components), comprised 51% of total ground cover (basal intercept) in the control watershed and 54% in the treatment watershed. Litter (including wood) was the second most important basal cover type with 46% in the control watershed and 43% in the treatment watershed.

Tree cover was the most common aerial cover type with 42% in the control watershed and 34% in the treatment watershed (tree cover, as reported here, is the sum of the individual means for individual species, i.e. live piñon and juniper canopy cover). Despite eight percentage point difference in total tree cover between the treatment and watersheds, the variability in this component suggests the two watersheds were not significantly different (figure 10). At the individual species level, within the control watershed, piñon had 24% cover and juniper 17%. Within the treatment watershed, juniper cover was similar to the control watershed at 18%, however piñon cover was significantly lower at 15%. Although there is abundant evidence (i.e. dead and down material) that Ponderosa Pine was previously a significant component of tree cover across the study site, prior to the mid-1950's drought, only a few, widely scattered individuals still persisted within the sample area pre-treatment (Allen and Breshears, 1998). Within the treatment watershed, tree cover was positively correlated ($r=0.66$) with litter cover, as needle fall accumulates directly beneath tree canopies. Consequently, tree cover was significantly negatively correlated with exposed soil ($r=-0.68$).

The other major vegetative cover types (grasses, shrubs, sub-shrubs, and forbs) did not exceed mean aerial cover values of 10%, pre-treatment, in either watershed (figure 10). Grass canopy cover was the most abundant, at 5% cover within the control and 6% within the treatment watershed. Shrub canopy cover was 4% in the control and just over 2% in the treatment watershed. Forb and sub-shrub canopy cover did not exceed 2% in either watershed.

CHANGES IN THE CONTROL WATERSHED

Within the control watershed, there were no significant changes for any of the measured parameters from pre-treatment (in 1996) through three years post-treatment in 1999 (figure 11). Herbaceous plant canopy cover fluctuated, year to year, following initial measurements made in 1996 largely in response to annual weather patterns and moisture induced pulses of annual/ biennial forbs. The most notable change was an temporal increase in forb cover from less than 2% in 1996 to 7% in 1997, but given high variance, this increase was not significant at the 90% confidence level. This increase was largely in response to above normal precipitation in May and June of that year. Control watershed forb cover values in 1997 were primarily attributable to three species: deervetch, yellow ragweed, and purple wallflower. Cumulatively, these three species contributed about three-quarters of total forb cover. After 1997, forb cover steadily declined, returning to 1996 levels by 1999. The small year-to-year fluctuations observed in the other cover types can be largely attributed to observer error, including the use of different field crews each year, imprecise placement of measurement tapes, and other sampling errors associated with the line-intercept method (also see **Results: UNGULATE EXCLOSURES:Control Exclosure**).

CHANGES IN THE TREATMENT WATERSHED

Tree cover

In response to the mechanical thinning treatment applied in the winter and early spring of 1997, mean tree canopy cover within the treatment watershed was reduced 71%, from 33% to 9% (figure 12, 13). Piñon cover was reduced from 15% to 4%, and juniper cover from 18% to 5%, effectively maintaining the pretreatment cover ratio between the two species (figure 13). At three years post-treatment (1999), tree reproduction (i.e. released seedlings and 're-growth' from persistent branches at the base of cut trees) was evident on casual observation, but was not detectable in the transect data given the variability inherent in estimating tree canopy and relatively small contribution of re-growth to total tree cover. A several percent increase in tree cover attributable to reproduction was documented on similar woodland sites at five years post-treatment, where 100% of the original tree canopy was removed (Jacobs, unpublished data).

Slash and litter cover

Slash, generated by thinning, was treated as two separate cover layers: portions of branches in direct contact with the soil surface were designated as ground slash, while those branch portions that were suspended above the surface were counted as aerial slash. Following treatment in 1997, aerial slash was measured at 19% and ground slash was at 1% (figure 14). By 1998 this was reversed, with aerial slash declining to about 2% while ground slash increased to 23%. We attribute these changes to the progressive breakdown of the slash, with smaller branches and needles falling to the soil surface and contributing to ground slash and eventually to litter. Mean litter cover increased by 2% from 39% to 41% between 1996 and 1997 and declined to 35% in 1998. In 1999, while aerial slash remained relatively unchanged from the previous year, ground slash declined

dramatically from 23% to 4%, and was associated with an equally dramatic increase in litter cover from 35% to 62% (figure 14).

Soil and pumice ground cover

Similar to litter cover, soil cover (composed of bare soil, rock, and pumice components) remained relatively unchanged with a decrease of 3% from 54% to 51% between 1996 and 1997. Pumice cover declined from 21 to 17%, rock cover from 3% to 2%, while bare soil increased slightly from 30 to 32%. After 1997, both pumice and bare soil generally decreased, while rock cover remained essentially unchanged at 3%. Pumice cover decreased to 17% in 1997, 13% in 1998, and 6% in 1999. Bare soil increased to 32% in 1997, decreased to 21% in 1998, and to 19% in 1999. Cumulatively, soil cover decreased by nearly half at three years post-treatment, from 54% in 1996 to 29% in 1999 (figure 14).

Cumulative Vegetation Response

Cumulative plant canopy cover within the treatment watershed was about 44% prior to treatment (figure 15). Tree cover made the largest contribution, making up over three-quarters of total relative cover (78%), followed by grass (14%), shrub (5%), forb (2%), and sub-shrub (1%).

In direct response to mechanical thinning in 1997, tree canopy cover was reduced by almost three-quarters (from 33% to 9%), exceeding the targeted two-thirds reduction. Piñon cover was reduced from 15% to 4% and juniper from 18 to 5%, effectively maintaining the pre-treatment cover ratio between the two species. While cumulative plant canopy cover remained essentially unchanged at the end of the first growing season, post-treatment (1997), relative contributions changed dramatically with tree cover decreasing from 78% to 23% of the total, while relative forb cover increased from 2% to 53%. Relative grass cover increased from 14% to 17%, sub-shrub from 1% to 2% relative cover, and shrub remaining essentially unchanged at 5% relative cover.

Cumulative plant canopy cover in 1998 at the end of the second growing season post-treatment (1998) was unchanged from the previous two years, but without the dramatic changes in relative cover among component growth forms experienced the previous year. In 1998 relative forb cover declined to about 44%, while relative grass cover (22%) and relative tree cover (27%) increased, with no change in shrub and sub-shrub relative cover values.

In 1999 at three years post-treatment, cumulative plant canopy cover remained stable. Relative cover of forbs (41%) and relative cover of trees (21%) declined, while relative grass cover (27%) continued to increase. Relative cover of sub-shrubs increased significantly from 2% to 5% of total plant canopy cover; increases in relative shrub cover (5% to 6%) however were not significant over previous year measures.

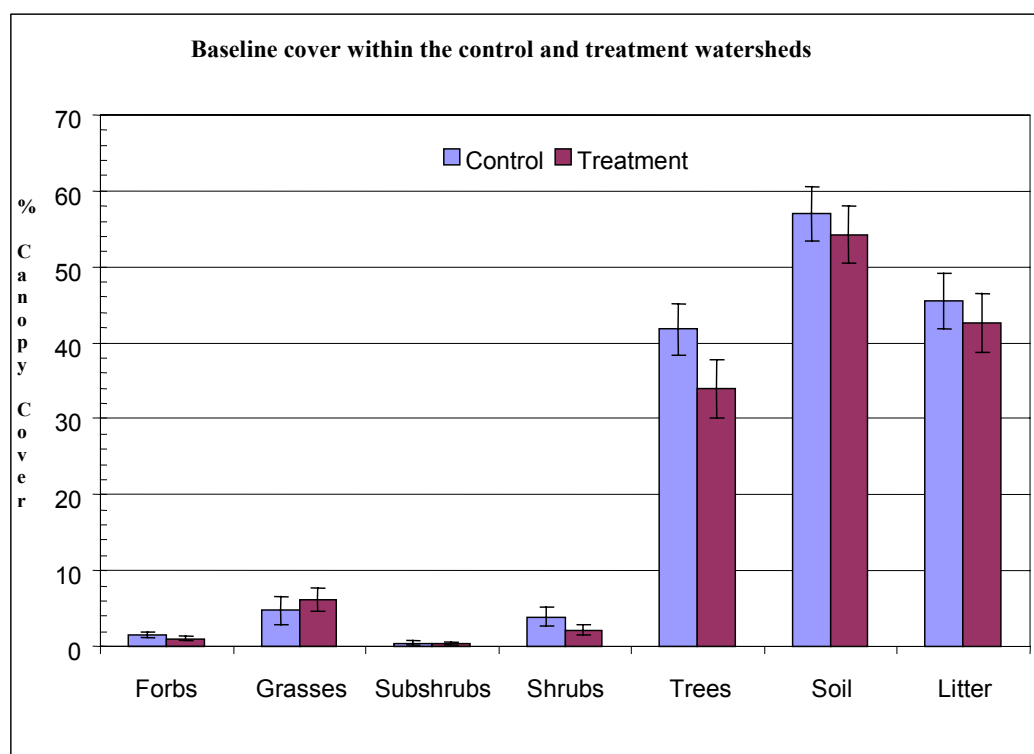


Figure 10. Comparison of pre-treatment vegetation and ground cover within the control and treatment watersheds. Error bars are 90% confidence intervals; none of growth form components are significantly different between control and treatment watersheds.

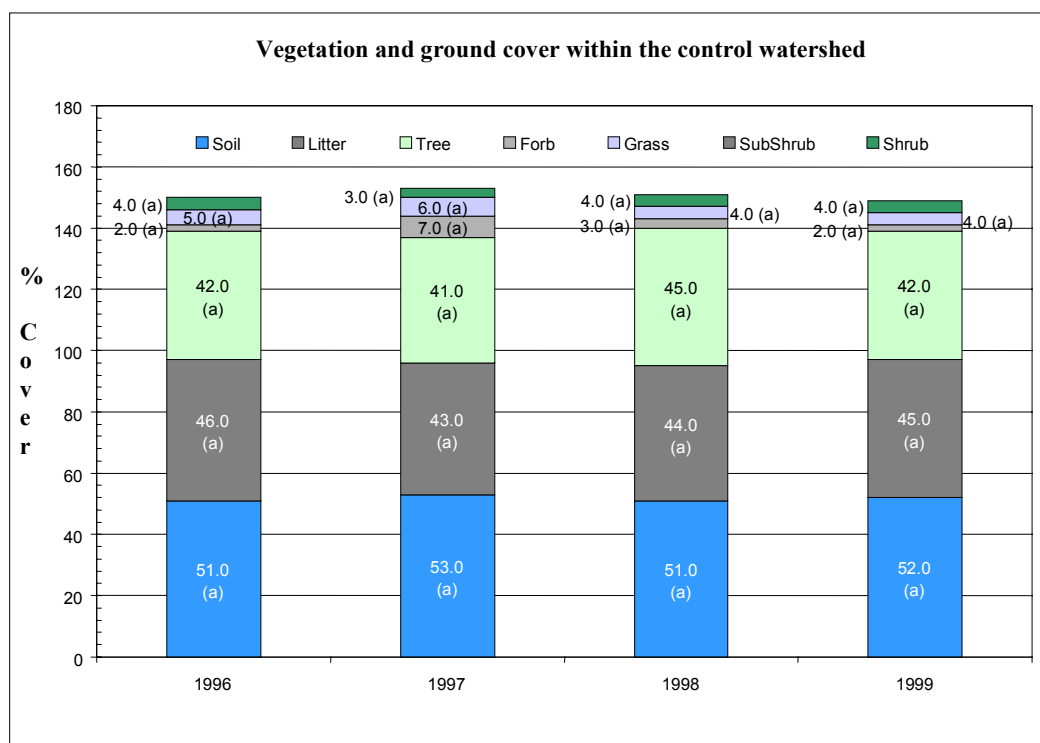


Figure 11. Cumulative cover values for the seven cover types measured within the control watershed from 1996 through 1999. Contrasts are made among years for each cover type. Bars with the same letter are not significantly different at the 90% confidence level.

Aerial perspective of mechanical tree thinning

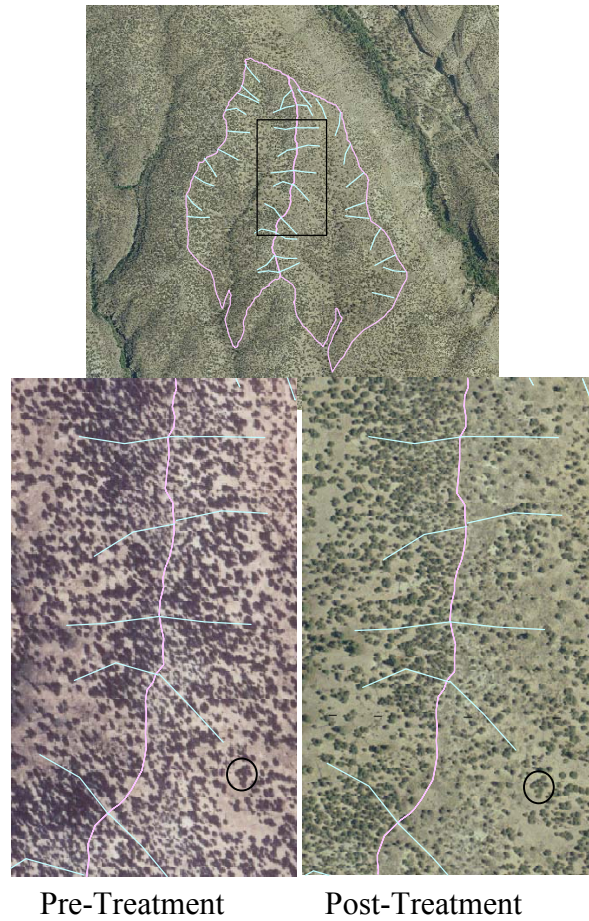


Figure 12. Effects of tree thinning on treatment watershed relative to control from aerial photos taken pre-treatment and post treatment.

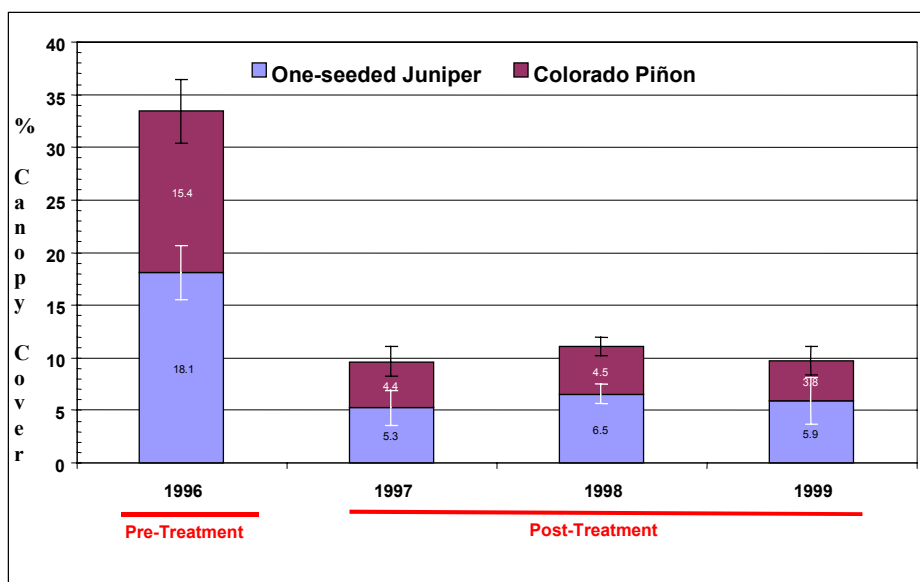


Figure 13. Effects of thinning within the treatment watershed on canopy cover of piñon and juniper. Error bars are 90% confidence intervals. Overlapping error bars are not significantly different.

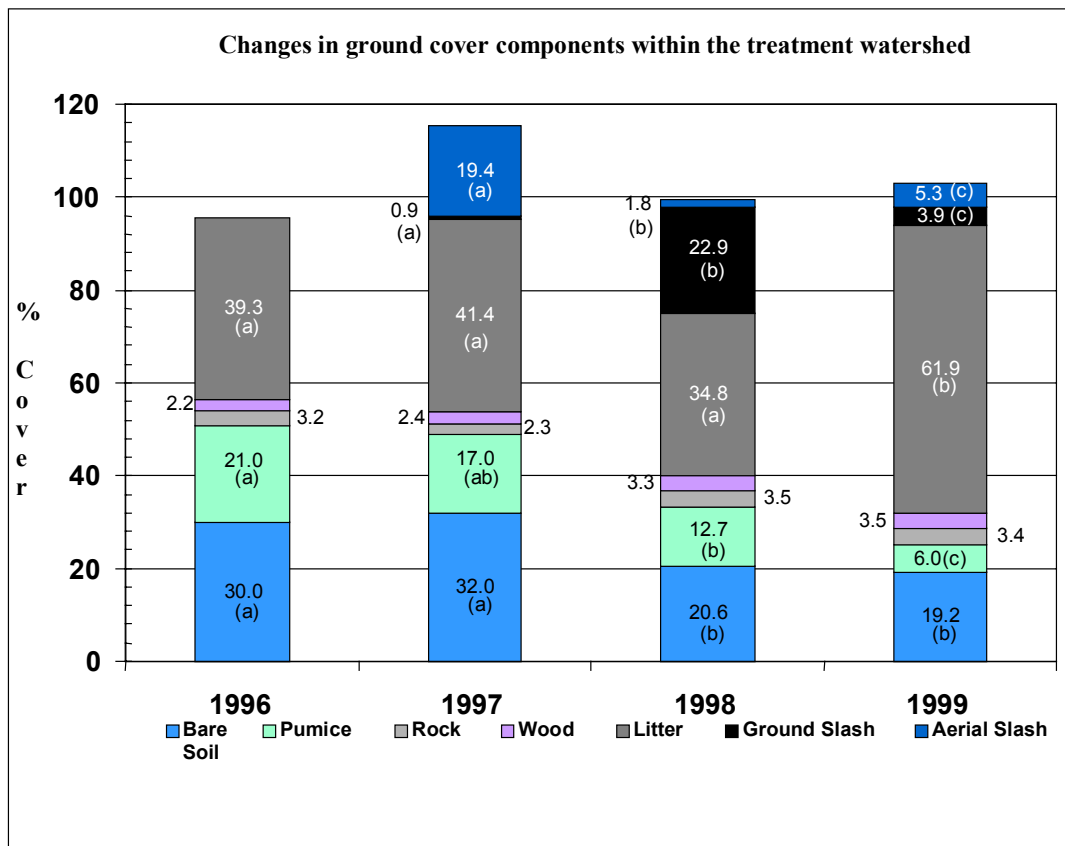


Figure 14. Changes in ground cover within the treatment watershed (1996 to 1999). Comparisons are made among years for each cover type; values are absolute mean cover percentages contributed by each cover type. Bars with same the letter not significantly different at the 90% confidence level ($p \leq 0.1$).

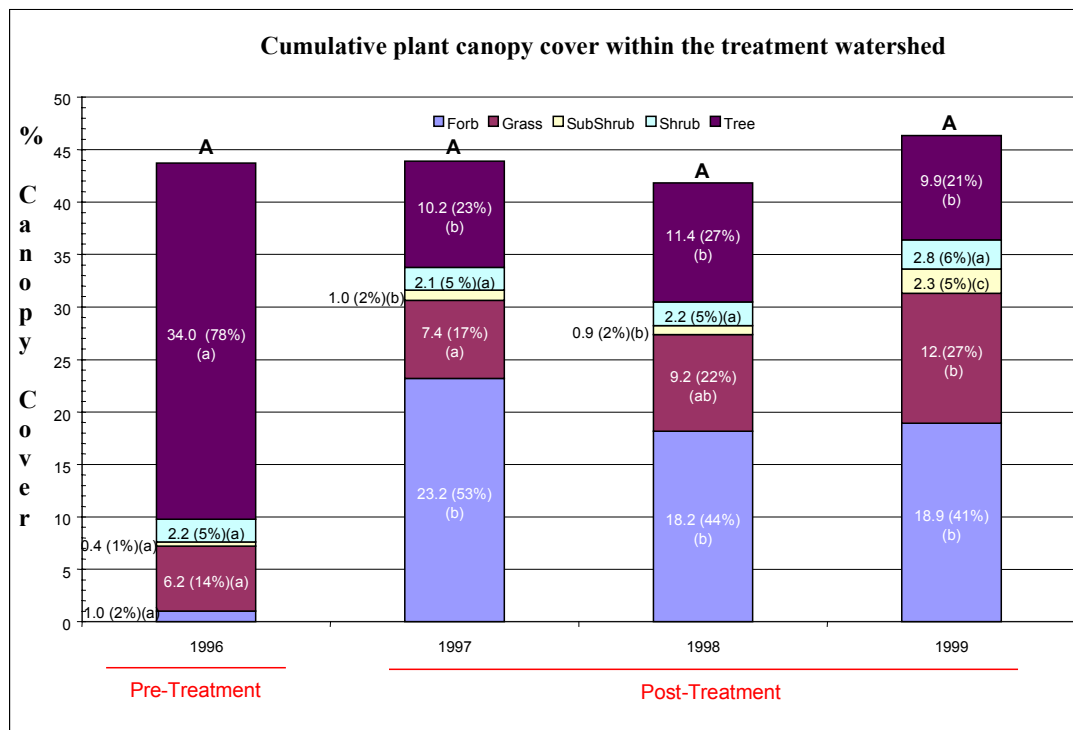


Figure 15. Changes in cumulative plant canopy cover within the treatment watershed from 1996 through 1999. Contrasts are made among years for each cover type. Values in parentheses are percentages of relative plant cover for that year. Columns with the same upper case letter, and similarly colored bars with the same lower case letter, are not significantly different ($p \leq 0.1$).

Forb Response

In 1997, at one year post-treatment, total mean forb cover increased from ~1% to 23% (figure 16). The increase was largely attributable to the response of two short-lived species, deervetch and yellow ragweed; each increased to about 8% cover, and collectively provided 16% cover, or almost 70% of the total forb cover. In 1998, total forb cover decreased to 18%; this was due in large part to the decline in cover of yellow ragweed, a biennial species, to less than 1% cover. The decline in yellow ragweed was offset by continued increases in deervetch, a short-term perennial, while the cover of other forb species remained relatively unchanged. In 1999, total forb cover remained relatively unchanged from the previous year, despite a decline in deervetch cover by nearly half. The deervetch decline was offset by increases in a wider diversity of other forb species. Over the three growing seasons following treatment, the number of forb species exceeding 0.5% (an arbitrary threshold for separating out species response from 'other spp.' category) remained about the same with eight in 1997, seven in 1998, and eight in 1999. Three forb species exceeded 0.5% cover all three years: deervetch and groundsel, both perennials, and mullein, a biennial. Nine of the twelve forb species which exceeded the 0.5% cover reporting threshold, persisted at this level for more than one growing season.

Annual non-native forb species, including lamps quarters and mullein, showed responses to a combination of mechanical disturbance and above average spring precipitation on sterile pumice soil sites (figure 16). However, it has been observed that increases in these annual/ biennial species are temporal responses to moisture and disturbance and that will be increasingly constrained by increasing cover of native, perennial species.

Shrubs and Sub-shrubs

Mean canopy cover of shrubs changed little following treatment (figure 15,17). At a little over 1% aerial cover, wavyleaf oak was the dominant shrub prior to treatment and throughout the study period. Total sub-shrub cover prior to treatment in 1996 was negligible (<0.5%); however, by 1999 at three years post-treatment sub-shrub cover had increased to >2%. The cover of a single sub-shrub species, broom snakeweed, reached nearly 2% in 1999, exceeding the individual cover of all of the most common shrubs, wavy-leaf oak, mountain mahogany, and apache plume. Broom snakeweed increased in cover every year following treatment, with the increase in total shrub/ sub-shrub cover to 5% during the third growing season largely attributable to snakeweed. Although snakeweed is a native species and a component of grasslands throughout the Southwest, it is of concern because of its tendency to increase on grazed rangeland and displace more palatable species.

Grass Response

In 1996, pre-treatment cover for perennial grasses was 6% aerial and 1% basal (figure 18). Blue grama was the dominant grass species with 0.6% basal and 3.2% canopy cover, followed by little bluestem with 1.3% canopy cover and mountain muhly with 0.7% canopy cover. These three warm season species collectively account for about 80% of total grass canopy cover and 95% of total grass basal cover. None of the remaining

grass species (i.e. mutton grass, june grass, little seed rice grass, galleta, three-awn, bottle brush squirrel tail, etc.) exceeded half a percent in canopy cover; collectively they accounted for only about 1% of total plant canopy cover.

In 1997, at one year post-treatment, there was no significant change in grass cover with only small increases in cumulative canopy cover and small decreases in cumulative basal cover. Mutton grass, a cool season species and considered desirable winter forage for ungulates, increased its canopy coverage to about 0.5%. In 1998, at two years post-treatment, both cumulative canopy and cumulative basal cover increased, but the increases were not significant. Blue grama, little bluestem, and mutton grass increased in both canopy and basal coverage, while mountain muhly remained relatively unchanged. By 1999, at three years post-treatment, total grass basal cover had increased to 2.1% (from 0.95% in 1996) and canopy cover nearly doubled to 12.4% (from 6.2% in 1996). Six species exceeded 0.5% aerial cover, with bottle-brush squirrel tail and three-awn joining the previously listed species. Although canopy cover had increased for most species, only little bluestem had increased significantly over its pre-treatment value (from 1.3% to 4.2%). This increase was sufficient for little bluestem to become a co-dominant with blue grama, which had increased aerial cover by only a third from its initial measurement. Conversely, increases in basal cover for little bluestem were not significant, while basal cover increases for blue grama were. Mutton grass, like blue grama, also experienced a significant increase in basal cover, but not in canopy coverage. These species level differences in aerial versus basal increases suggest individualized responses to improved site conditions, and may reflect differences in propagation by seed, rhizome, or vegetative spreading above ground.

CULTURAL SITES

As noted under Methods: Cultural Sites, tree cover was completely removed from within the perimeter of cultural sites as a modification to the thinning prescription applied to the larger watershed area. Herbaceous response (i.e. forb+grass) to restoration treatment, measured across a subset of five cultural sites (three structural and two lithic) within each watershed, mirrored the general response documented for the larger watershed areas (figure 19) at one, two, four, and five years post-treatment (1997, 1998, 2000, 2001). Only a subset of the vegetation transects were read in 1998. The cultural response data need to be interpreted with respect to differences in soil (and vegetation) between the larger watershed area, as noted under Methods: Cultural Sites, as well as the small sample size (n=5), ridge top location of structural sites, and more intensive thinning prescription. More intensive thinning on deeper soils could be expected to generate a more robust herbaceous response, however much of the slash was carried off to the perimeter of the site, to mitigate fire heat effects, and may have reduced the benefits of slash mulch locally. In addition, transects sampled both the deeper soils of the core structural site as well as adjacent shallow soils more typical of the ridge location. Although the unique setting and response of cultural sites sets them apart from the larger study area, they can be expected to benefit indirectly from the increased landscape stability provided by the larger watershed response.

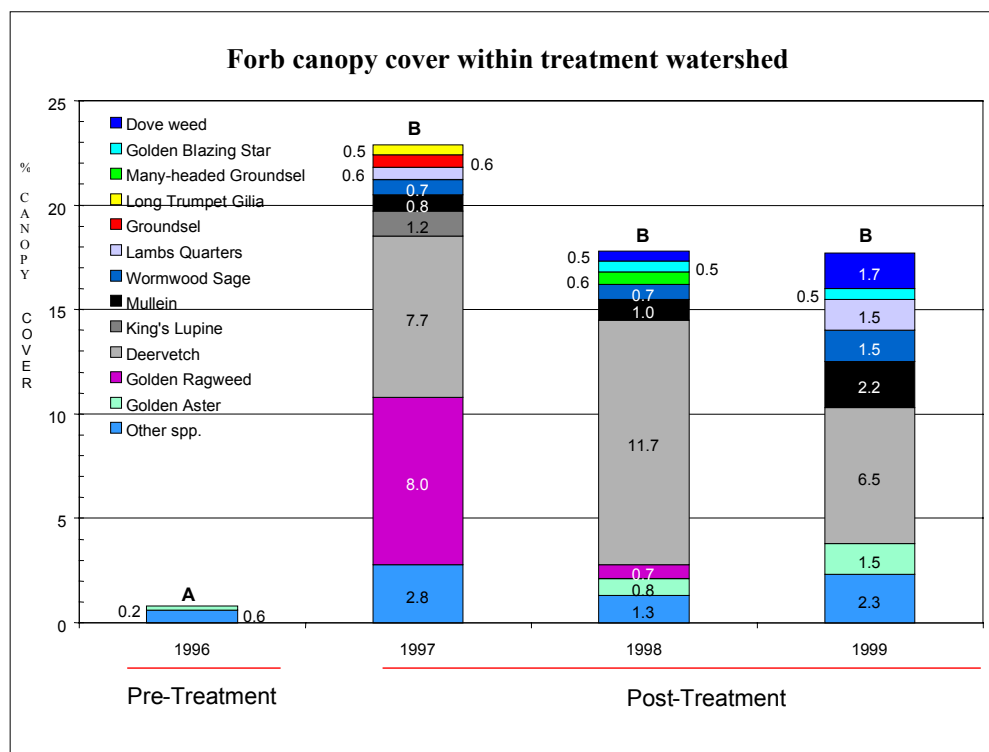


Figure 16. Cumulative mean forb canopy cover within the treatment watershed from 1996 (pre-treatment) to 1999 (three years post-treatment). After 1996, species < 0.05% were combined under ‘Other spp’. Comparisons are made across years and among for species. Columns with same letter are not significantly different ($p \leq 0.1$).

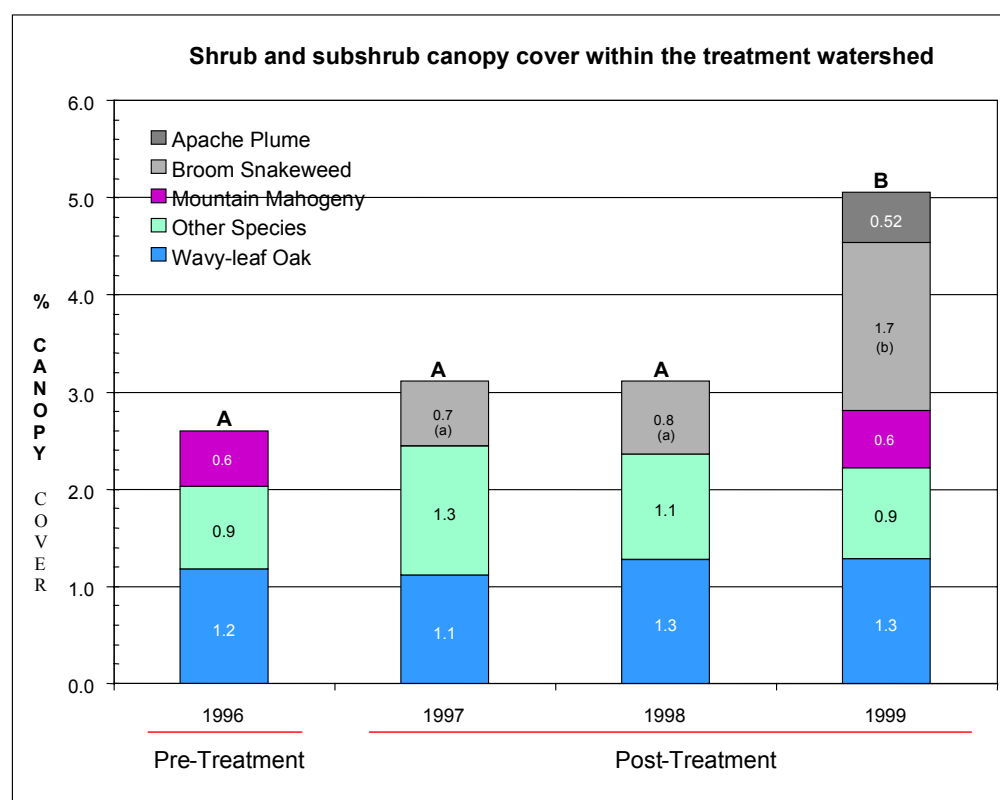


Figure 17. Cumulative canopy cover of shrubs and sub-shrubs from 1996 (pre-treatment) to 1999 (three years post-treatment). Individual species are differentiated at > 0.5% of total shrub cover. Columns with the same upper case letter, and similarly colored bars with the same lower case letter (or without a letter), are not significantly different ($p \leq 0.1$).

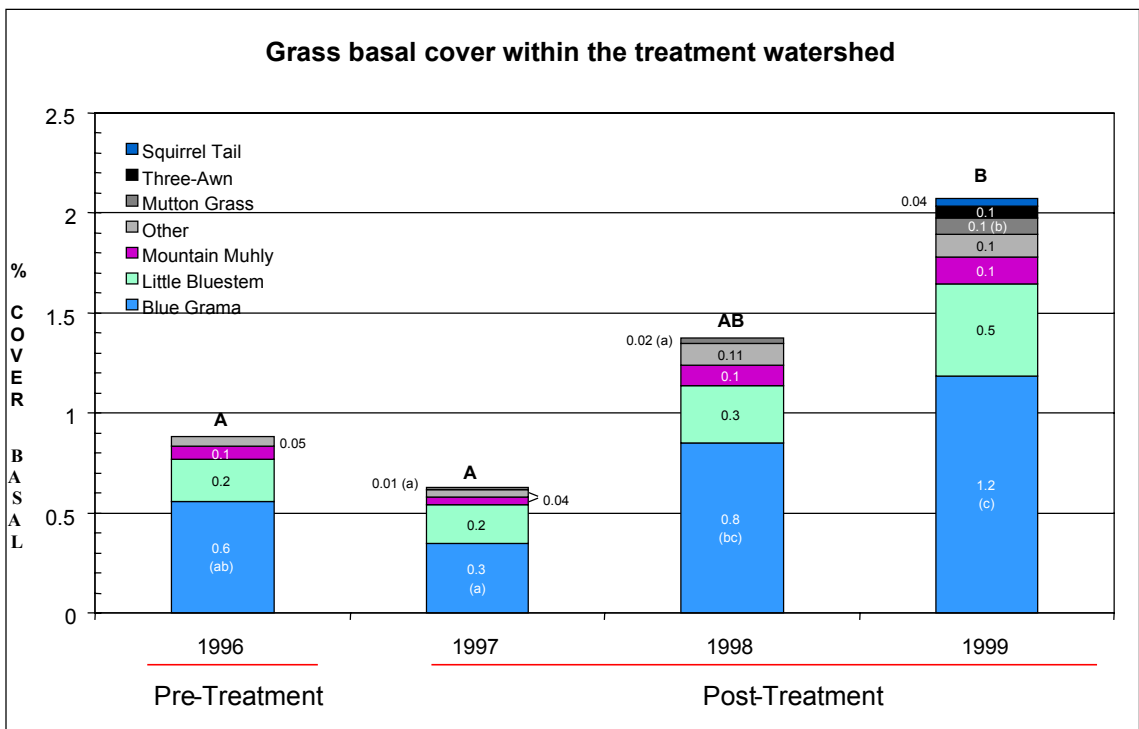
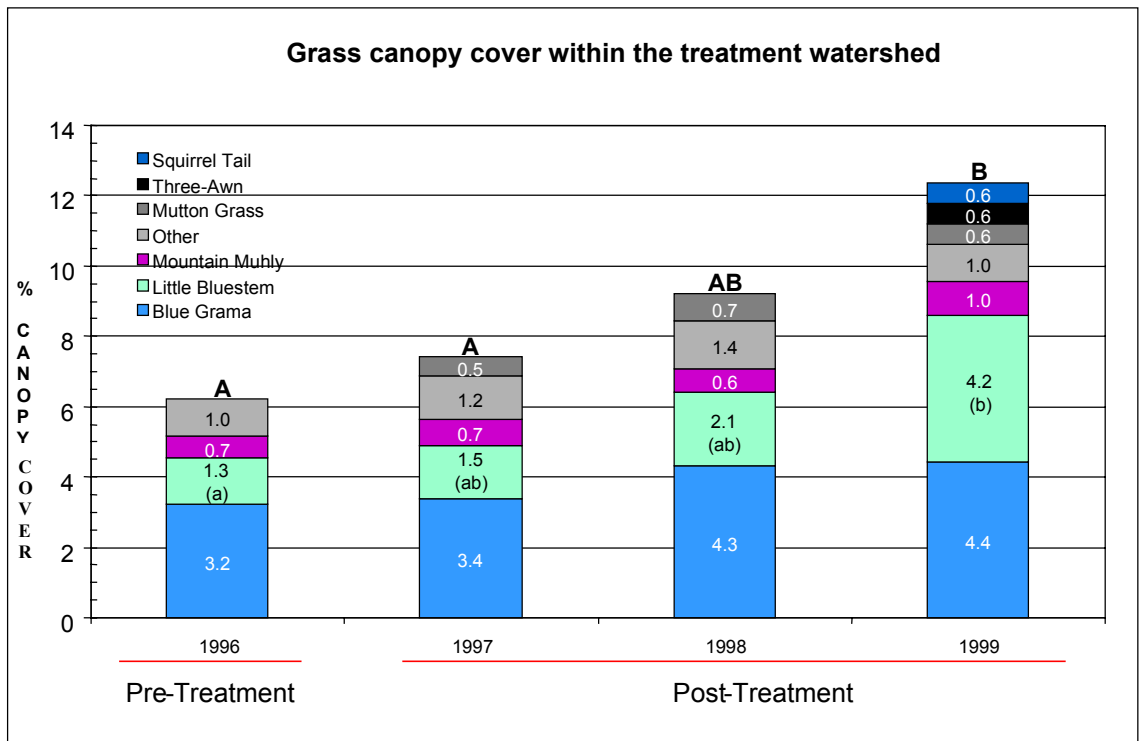


Figure 18. Cumulative mean canopy and basal grass cover from 1996 through 1999. Columns with same upper case letter are not significantly different ($p \leq 0.1$). Similarly colored bars with the same lower case letter, or without letters, are not significantly different ($p \leq 0.1$). Individual species were differentiated from 'Other sps' when their mean canopy cover values was $> 0.5\%$.

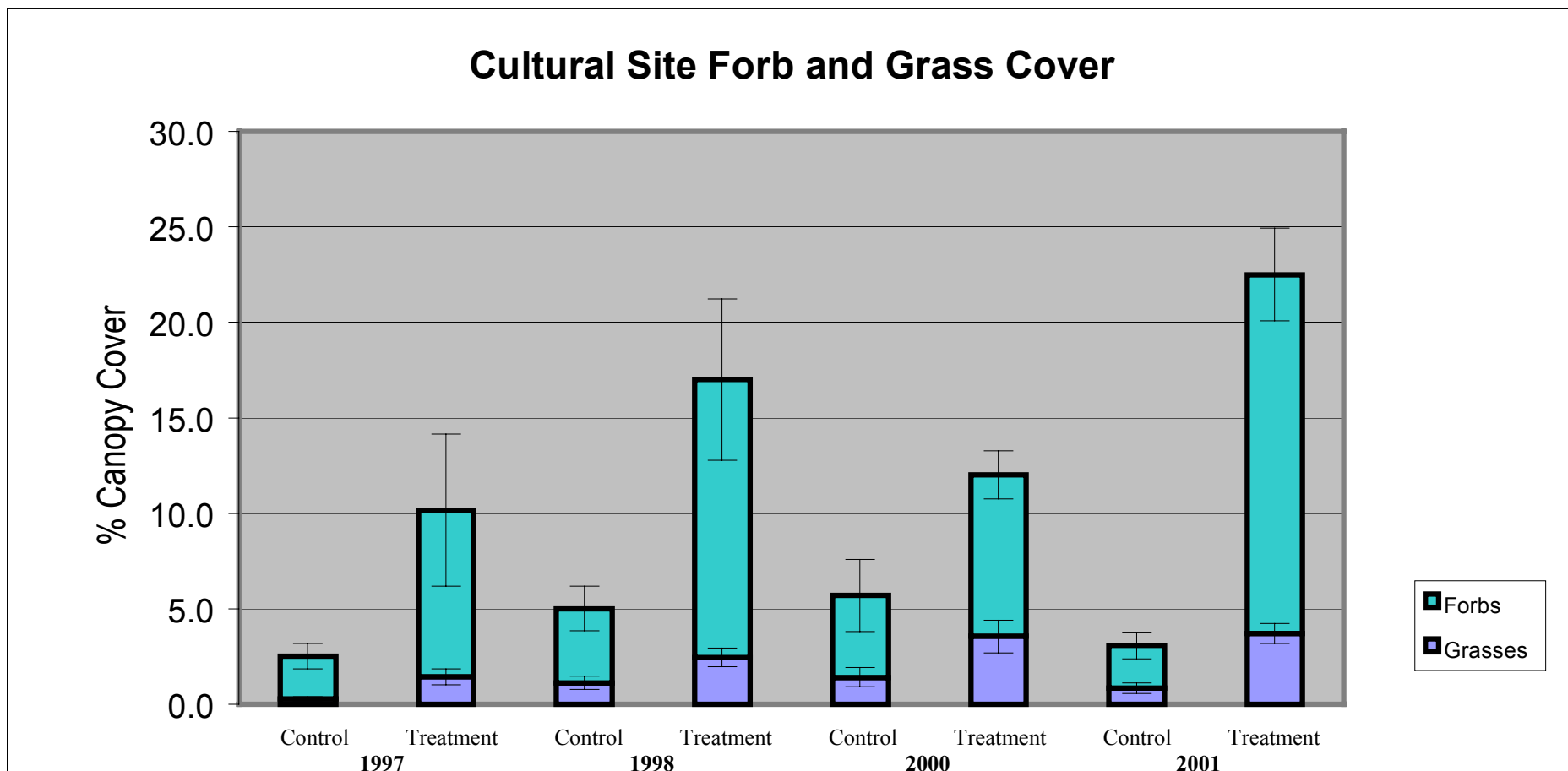


Figure 19. Changes in herbaceous cover (forb + grass aerial components) across treatment and control cultural sites (n=5) from 1997 (one year post-treatment through 2001 (five years post-treatment). Shown with standard error bars.

BIOMASS

Cumulative biomass response (384 kg/ ha) to restoration treatment was significant ($p < .01$) relative to control (110 kg/ ha) at one year post-treatment, largely due to a robust forb response (304 kg/ ha) released by thinning. At two years post-treatment, the treatment watershed biomass response (460 kg/ ha) was again significantly ($p < .01$) greater than the control watershed (154 kg/ ha) with relative grass response contributing 38% of total biomass sampled compared with only 20% the previous year (figure 20). Biomass response to treatment followed the same general pattern as found for cover estimates, with forb response dominating the first year followed by a more gradual increases for grass at two years post-treatment. The same forb and grass species noted for their contributions to cover estimates, were also largely responsible for the observed biomass response. Some reduction in measured forb response in 1998 may be attributable to harvest from the same quadrats as the previous year (1997).

ARTHROPODS

Measures of diversity and abundance, across all taxa and for total number of individuals, were found to vary both seasonally and annually, presumably as a function of life cycle, habitat, and weather patterns. Correlation of strong season and annual patterns in arthropod abundance with moisture has been previously documented in woodlands at Bandelier (Brantley et. al., 2000). Individual taxonomic groups, arranged at order, family, and genus levels, also showed variable seasonal and annual responses. Given the strong influence of climatic patterns on arthropod populations, (and only five sample periods across two years) we limit our interpretations to treatment effects per sample period and assume uniform climatic influences across the two watersheds.

Arthropod abundance, being cumulative number of individuals summed across all traps ($n=40$ per watershed) for all species, was consistently higher for the treatment watershed relative to the control (figure 21a). Arthropod species diversity, by contrast, showed no consistent patterns, either between sample dates or for treatment. Several of the most abundant arthropod groups show some patterns relative to treatment: Orders Microcoryphia (Bristletails) and Coleoptera (Beetles) are generally more abundant in the treatment watershed while Order Araneae (Sac Spiders) are generally more abundant in the control watershed (figure 21a). Abundance, calculated as mean number individuals per trap, was usually higher for Order Orthoptera (Grasshoppers) and for Family Tenebrionidae (Darkling Beetles) within the treatment watershed relative to control (figure 21b).

SOIL EROSION

Mean changes in soil surface elevation between the treatment and control watersheds diverged significantly at two years post-treatment in fall 1998 (figure 22a). Within the treatment watershed, the soil surface elevation trends were generally stable on an annual basis (following an initial negative change). There was a significant negative change in soil surface elevation between fall 1996 and spring 1997 measurements, however mean

changes between treatment (-5mm) and control (-6mm) watersheds were not significantly different. Apparent elevation changes for this time period may have been influenced by an early season snowfall; however, data from all bridges were collected after this event occurred. Soil surface elevation changes in subsequent years were less dramatic, until the fall 1998 when the control watershed experienced a mean change of -5 mm. By 1998, mean change in soil surface elevation from baseline (fall 1996) levels was significantly different between the control (-12 mm) and treatment (-7mm) watersheds (n=90). In 1999 the divergence increased, with the treatment watershed (-3mm) apparently gaining elevation since the 1998 measurement, while the control watershed (-14mm) continued to lose ground (figure 22a). As described under **Methods**, 1999 measurements incorporated a change in methodology which added slash to one-half of each bridge within the treatment watershed instituted after the fall 1998 measurement. Looking at the data stratified by bridge location suggests most of the measured differences between treatment and control areas can be attributed to changes at the within-channel locations (figure 22b). This can be explained, in part, by the absence of slash on upper slope, inter-channel bridge locations, as well as by an inferred reduction in velocity of surface runoff (and sediment transport capacity) due to the slash mulch treatment.

UNGULATE EXCLOSURES

Baseline data collected in 1996 suggested that vegetative cover was not significantly different between the treatment and control exclosures, or between the two exclosures and the treatment watershed (figure 23a). There were significant differences however, between the two exclosures and the larger treatment watershed area for ground cover. Soil (bare soil+pumice+rock) cover for the two exclosures was similar, with 41% cover for the control and 42.5% for the treatment; but both were significantly different from the treatment watershed (54% soil cover). In addition, exclosure areas did not have significant amounts of pumice or rock cover, in comparison with the larger watershed areas. Litter cover differed significantly among all three sites, with the treatment watershed having 39%, the treatment exclosure 48%, and the control exclosure 54% litter cover. Wood, usually included as a component of litter, was significantly higher in the treatment exclosure (6% cover) relative to the treatment watershed (2%), with the control exclosure intermediate.

Control Exclosure: Soil , Litter, and Vegetation Cover Changes

No significant changes in plant cover were observed within the control exclosure (figure 23b) between 1996 (pre-treatment) and 1998 at two years post-treatment (data were not collected in 1997). What appears to be a sizeable reallocation in ground cover between litter and soil (with litter >10% and soil <9%) between 1996 and 1998 appears more likely due to differences in observer resolution (i.e. recording multiple litter intercepts of ~1 cm) in 1998) relative to when the data was previously collected in 1996. Litter cover in the larger control watershed remains essentially unchanged over the same time period.

Treatment Exclosure: Soil and Litter Cover Changes

Thinning, and the application of the slash mulch, increased the amount of litter cover

relative to bare soil cover in 1998 as compared with 1996 (figure 24). Before treatment in 1996, litter cover was 54%, and by the end of the second growing season in 1998, it had decreased to 37.5%, although the decrease was in large part a result of a significant increase in ground slash (29.5%) which likely concealed the underlying litter. As documented for the larger watershed in 1999, ground slash eventually breaks down and contributes directly to the litter component. Bare soil cover (excluding rock and pumice) prior to treatment in 1996 was 40%, and at the end of second growing season in 1998 had been reduced to 23.5%. Changes in litter and bare soil cover for the treatment enclosure mirror those occurring in the larger treatment watershed over the same time period.

Treatment Enclosure: Vegetation Changes and Comparison with Treatment Watershed

Cumulative plant canopy cover within the treatment enclosure was 47%, prior to treatment in 1996, with relative tree cover constituting about 79% of the total (figure 25a). Relative cover of grass (11%), shrub/ sub-shrub cover (6%), and forb cover (3%) were also roughly equivalent to pre-treatment values within the larger treated watershed. Treatment reduced absolute tree cover from 37% to 7%, an 80% reduction; this was slightly higher than the 71% reduction achieved for the larger treatment watershed area (figure 25b). It should be noted that a consistent application of the thinning prescription as described under methods will yield variable tree reductions depending on stand age-structure (i.e. initial density of trees in smaller versus larger size classes). Within the treatment enclosure, thinning altered the juniper to piñon ratio from 2.3:1 to 0.79:1, whereas the ratio in the larger treatment watershed remained at 1.2:1 post-treatment. The change in tree species ratio resulting from thinning suggests the enclosure area had relatively higher number of small diameter junipers as compared with the larger watershed.

FORB RESPONSE

Forb response within the treatment enclosure was generally similar to that of the larger treatment watershed; however, there was a greater diversity of species contributing to total forb cover within the enclosure (figure 26a). Yellow ragweed, a biennial, increased the first year following treatment, but by year two had decreased substantially. Deervetch and golden aster increased in both years (1997-1998) following treatment. Spreading fleabane was abundant in 1997, and wormwood sagebrush was common in 1998. The relatively small differences observed for forb cover and diversity within the treatment enclosure, as compared with the larger treatment watershed, were more likely due to intrinsic site differences (including sample size) than a response to presence/ absence of grazing pressure.

SHRUB AND SUB-SHRUB RESPONSE

Prior to treatment, cumulative canopy cover of shrubs and sub-shrubs was less than 4% within both the treatment enclosure and the treatment watershed (figure 26b). Wavy-leaf oak, the dominant shrub/ sub-shrub pre-treatment on both sites, remained dominant within the enclosure site, while in the larger watershed area broom snakeweed, a sub-shrub, gained dominance over wavy-leaf oak. Again, while these data do not reveal any

effects of ungulate herbivory on shrub or sub-shrub canopy cover, several mild winters and the relatively short duration of this study may have contributed to a lack of significant findings.

GRASS RESPONSE

Grass canopy and basal cover within the treatment exclosure responded positively to the restoration treatment (figure 27a,b), similar to the response measured in the larger treatment watershed, but somewhat accelerated. In 1998, at two years post-treatment, cumulative grass canopy cover within the treatment exclosure doubled, going from 5.2% to 10.8% (figure 27a); in comparison, the larger treatment watershed area did not exceed 10% grass canopy cover until the end of the third growing season (1999). Within the treatment exclosure, blue grama was the only species to have increased significantly in aerial cover by 1998. The increase in cover for species like blue grama, little bluestem, and mutton grass contributed to a significant increase in cumulative grass canopy cover within the treatment exclosure at two years post-treatment, individual changes in cover for many of the contributing grass species were not significant.

Cumulative grass basal cover (figure 27b) within the treatment exclosure also increased following restoration treatment, with the size and pattern of the cumulative response almost identical to that of the treatment watershed. Casual observations in the larger treatment watershed area suggested cool season grasses, including mutton and June grass, were being preferentially grazed to the root crown in spring and fall, with some mortality resulting. While mutton grass experienced greater increases in both canopy and basal cover within the exclosure than outside, the increase was not statistically significant. Again, the data remain inconclusive on the effects of ungulate herbivory, but this may be due in part to mild winters, sample size and duration of the study.

Comparison of soil surface elevations inside and outside ungulate exclosure

As with soil surfaces in the larger watershed areas, there was a significant decline in elevation measured across both the control and treatment exclosure bridges (n=18 per exclosure) for the fall 1996 to spring 1997 measurements. Changes over this period within the treatment and control exclosures were not significantly different from each other or from those observed in either the control or treatment watersheds. Through fall 1998, treatment and control exclosure bridge measurements generally tracked those of the respective larger watershed area. Although the treatment exclosure had slightly higher elevation loss than the corresponding watershed, and the control exclosure slightly lower elevation loss, differences between each exclosure and its respective watershed area were not significant. Although sample size is small (n=16) and variability high, our data do not detect any effects of ungulate herbivory or trampling on changes in surface elevation during the sampled time period.

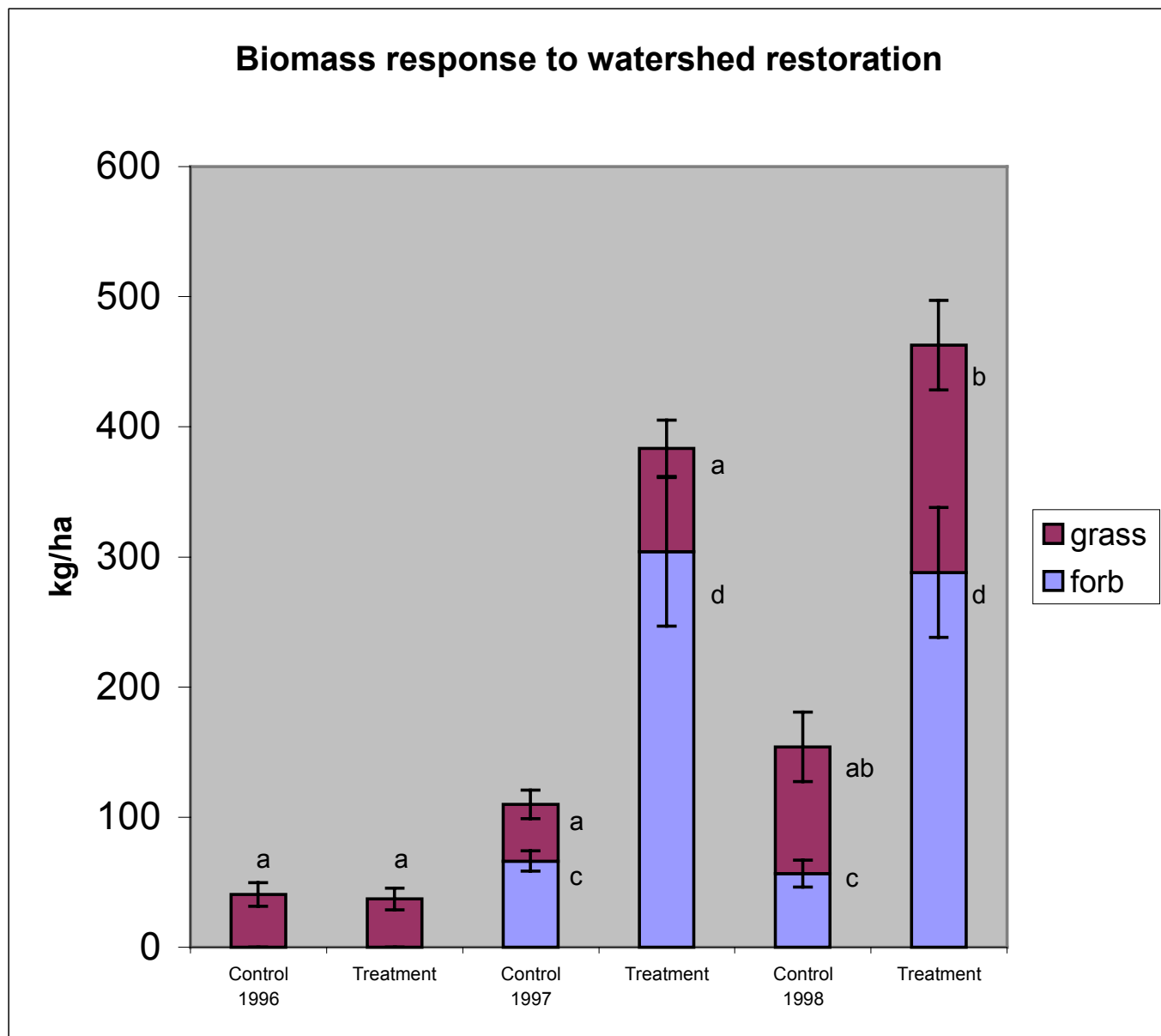


Figure 20. Cumulative mean biomass (kg/ ha) across the control and treatment watersheds (n=20) from 1996 (pre-treatment) to 1998 (two years post-treatment). Forbs were not harvested in 1996; forb response in 1998 may have been impacted by 1997 harvest from the same quadrats. Columns with same letter (or which share a common letter) are not significantly different ($p < .01$).

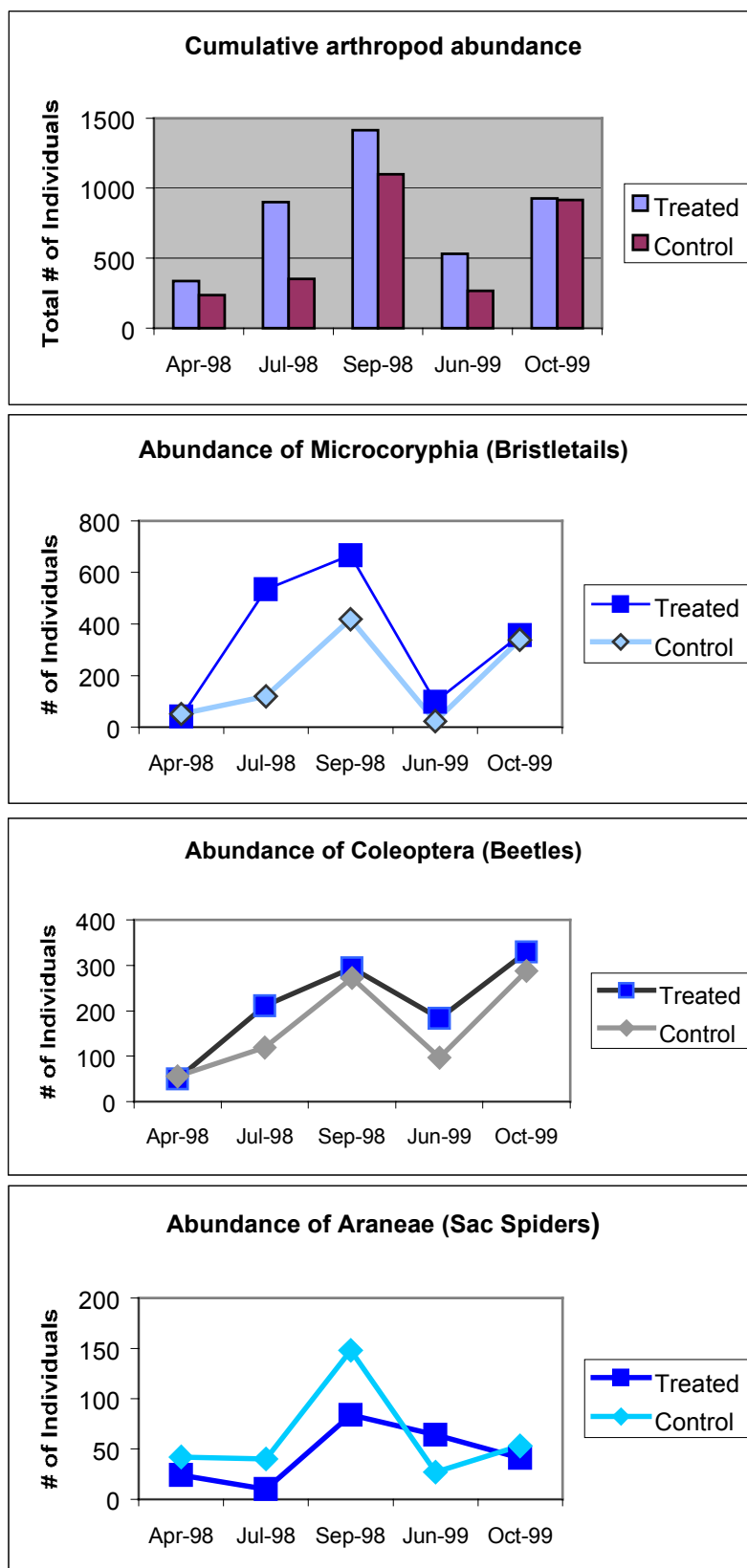


Figure 21a. Abundance of arthropods (total number of individuals) across all taxa and for selected orders for treatment and control watersheds (n=40)

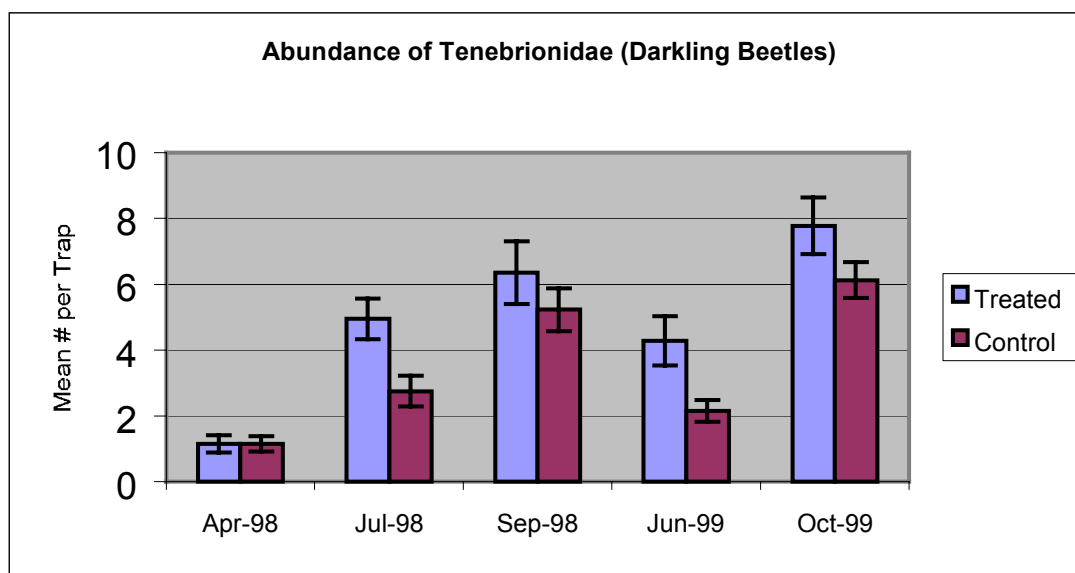
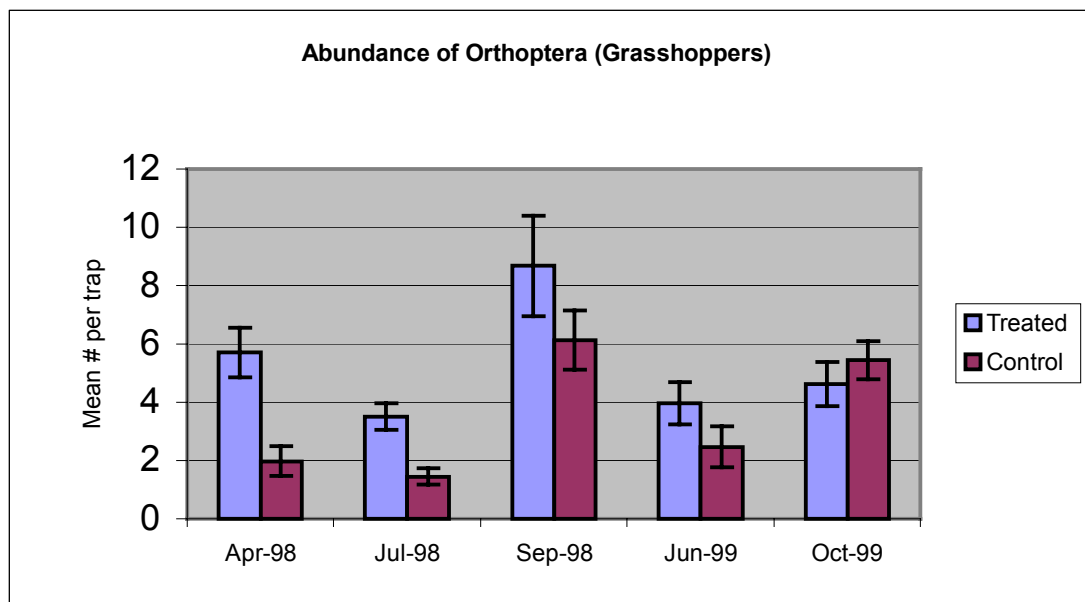


Figure 21b. Arthropod abundance (mean number individuals per trap with standard error) for grasshoppers and darkling beetles (n=40).

Changes in soil surface elevation

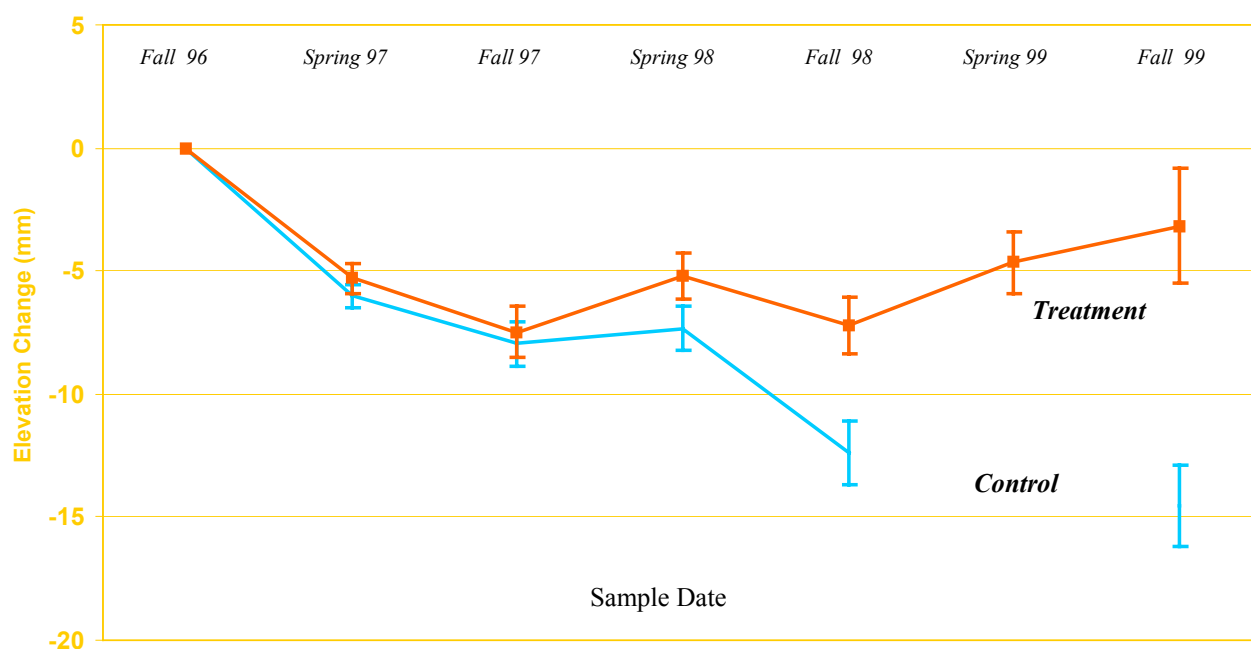


Figure 22a. Mean changes (mm) in soil surface elevation from fall 1996 (pre-treatment) to spring 1999 (two and one-half years post-treatment) for treatment and control watersheds (n=90). No measurements were taken for the control watershed in spring 1999. Error bars are 95 percent confidence intervals.

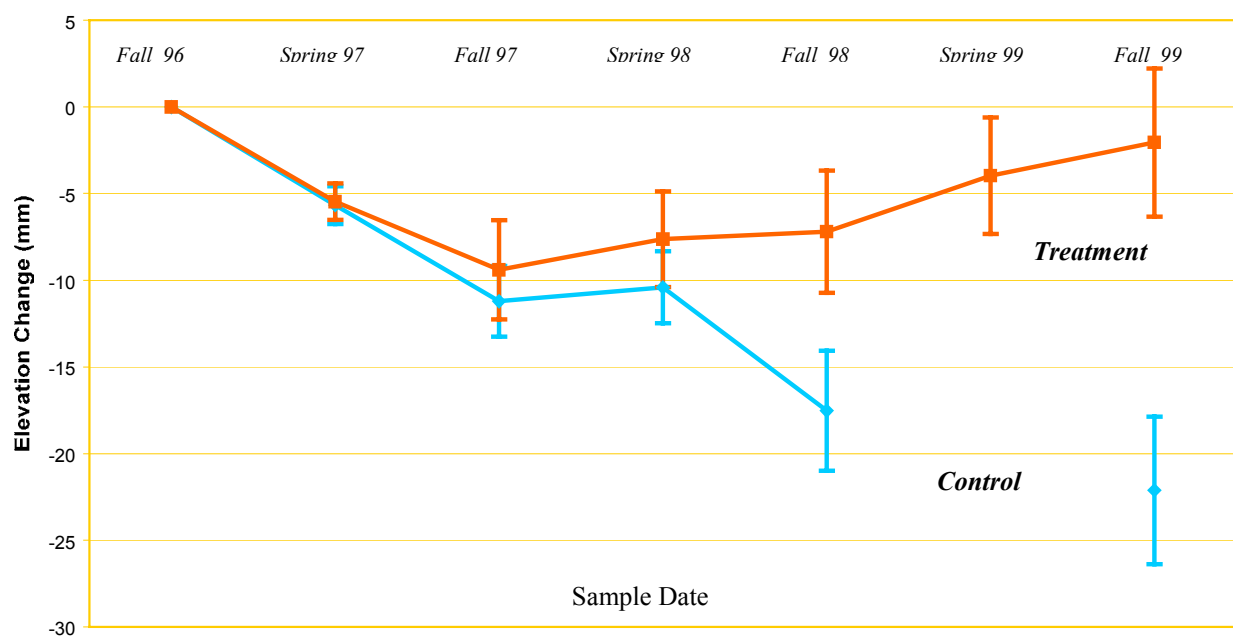


Figure 22b. Mean changes (mm) in soil surface elevation from fall 1996 (pre-treatment) to spring 1999 (two and one-half years post-treatment) for lower channel location in the treatment and control watersheds (n=18). No measurements were taken for the control watershed in spring 1999. Error bars are 95 percent confidence intervals.

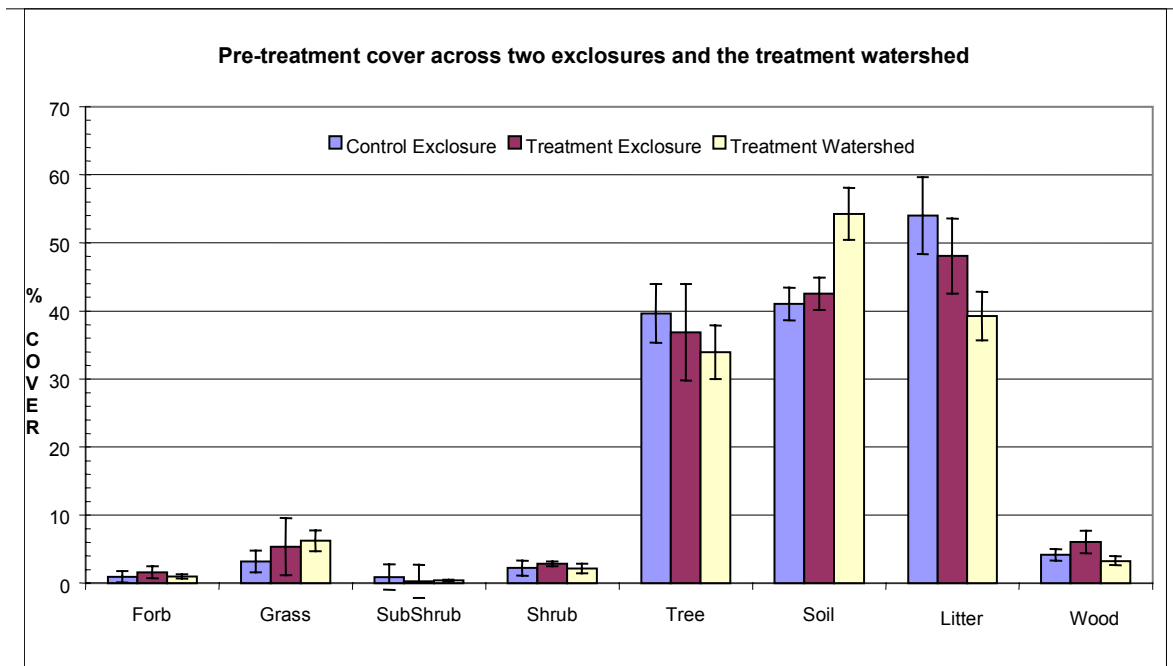


Figure 23a. Pre-treatment (1996) vegetation and ground cover for the control and treatment exclosures (n=5) as compared with the pre-treatment (1996) watershed (n=40). Error bars are 90% confidence intervals. Overlapping bars are not significantly different.

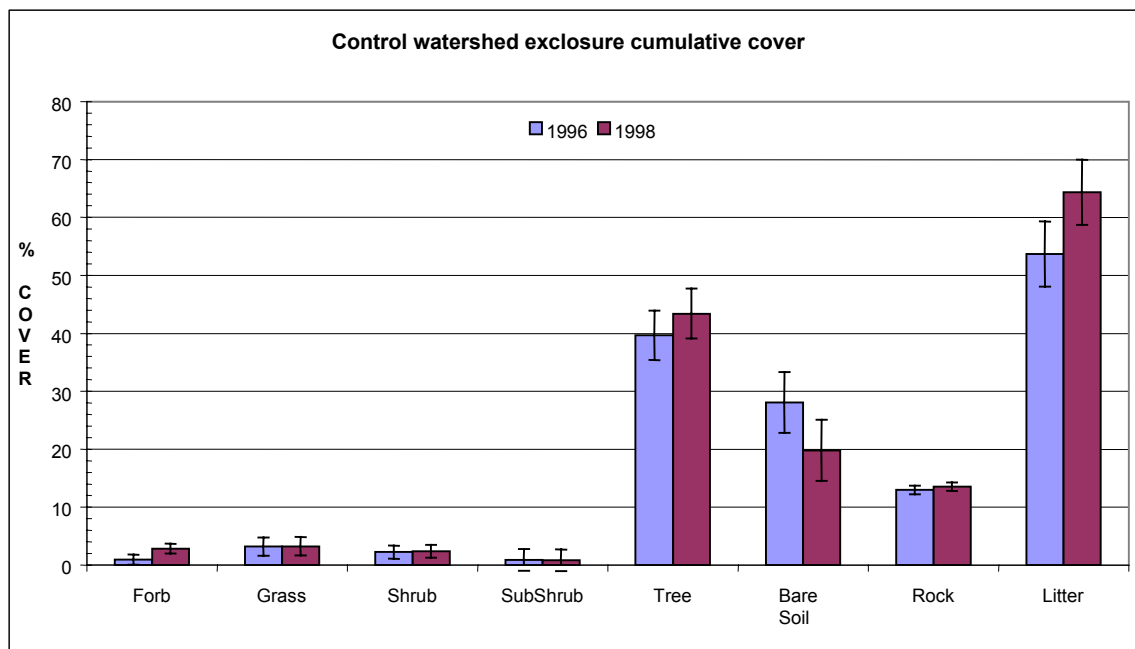


Figure 23b. Status of vegetation and ground cover within the control exclosure (n=5) in 1996 (pretreatment) and two-years post-treatment (1998). Data was not collected for 1997. Error bars are 90% confidence intervals. Overlapping bars are not significantly different.

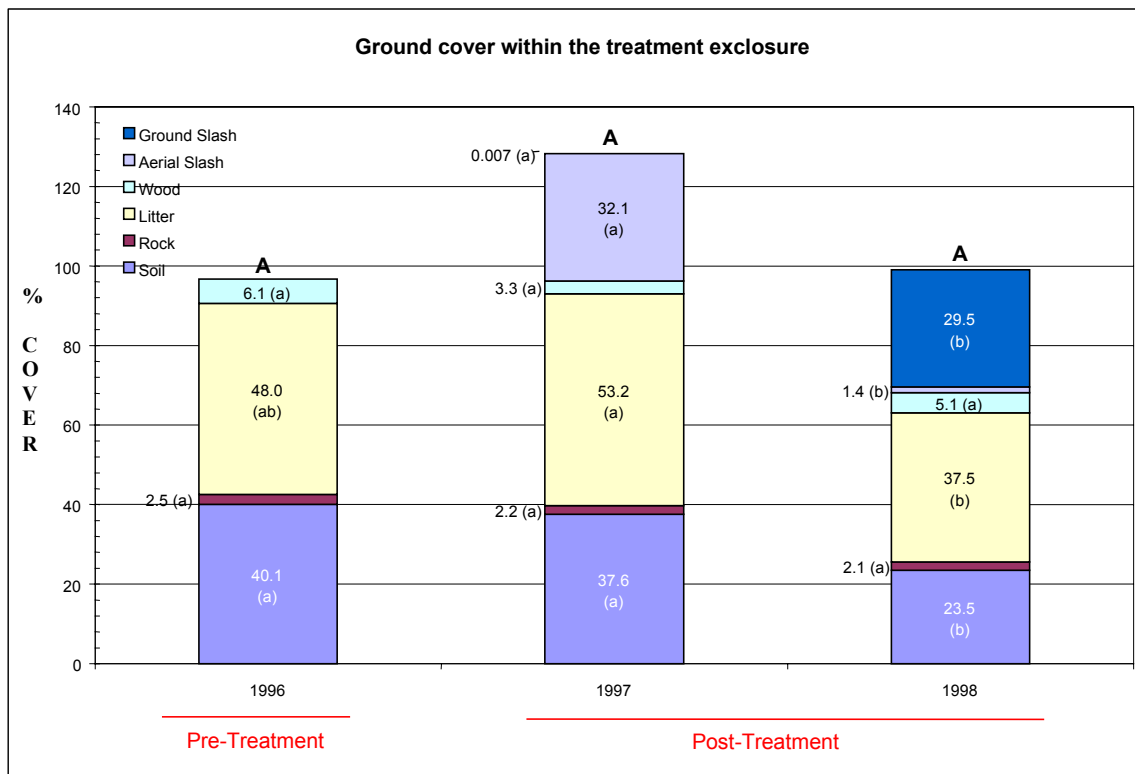


Figure 24. Ground cover within the treatment enclosure. Comparisons are made among years. Columns with the same upper case letter, and similarly colored bars with the same lower case letter, are not significantly different ($p \leq 0.1$).

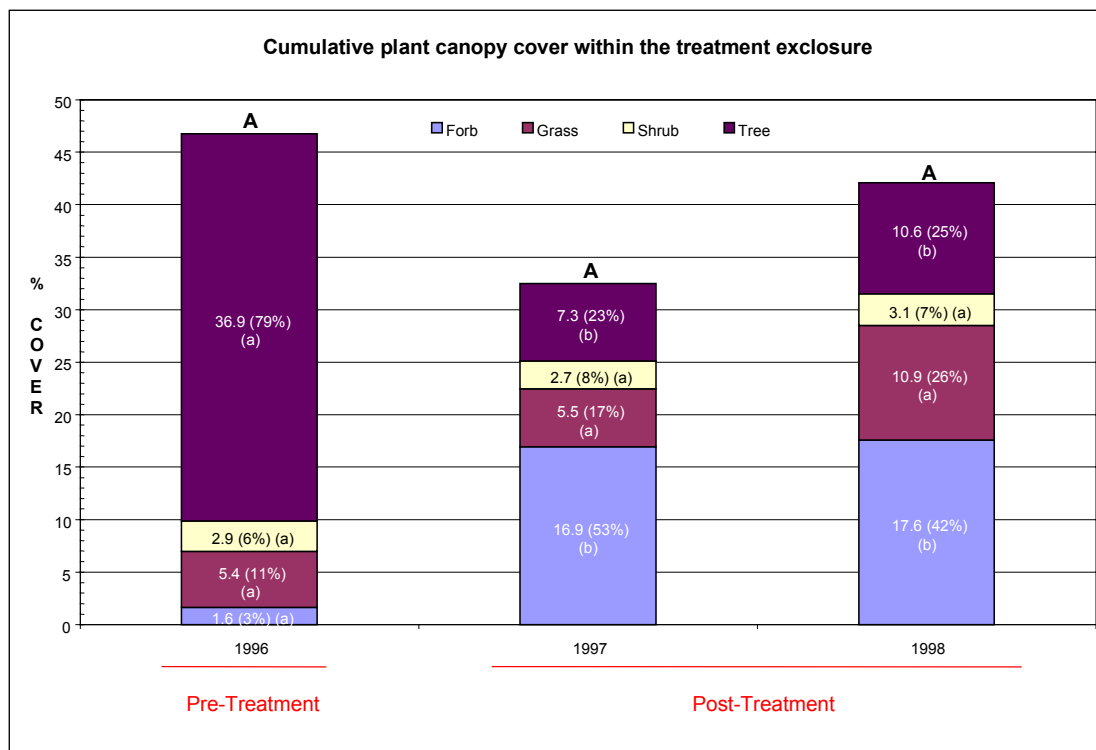


Figure 25a. Changes in cumulative vegetation canopy cover within the treatment enclosure from 1996 (pre-treatment) through 1998 (two years post-treatment). Contrasts are made among years for each cover type. Values in parenthesis are percentages of relative plant cover for that year. Columns with the same upper case letter, and similarly colored bars with the same lower case letter, are not significantly different ($p \leq 0.1$).

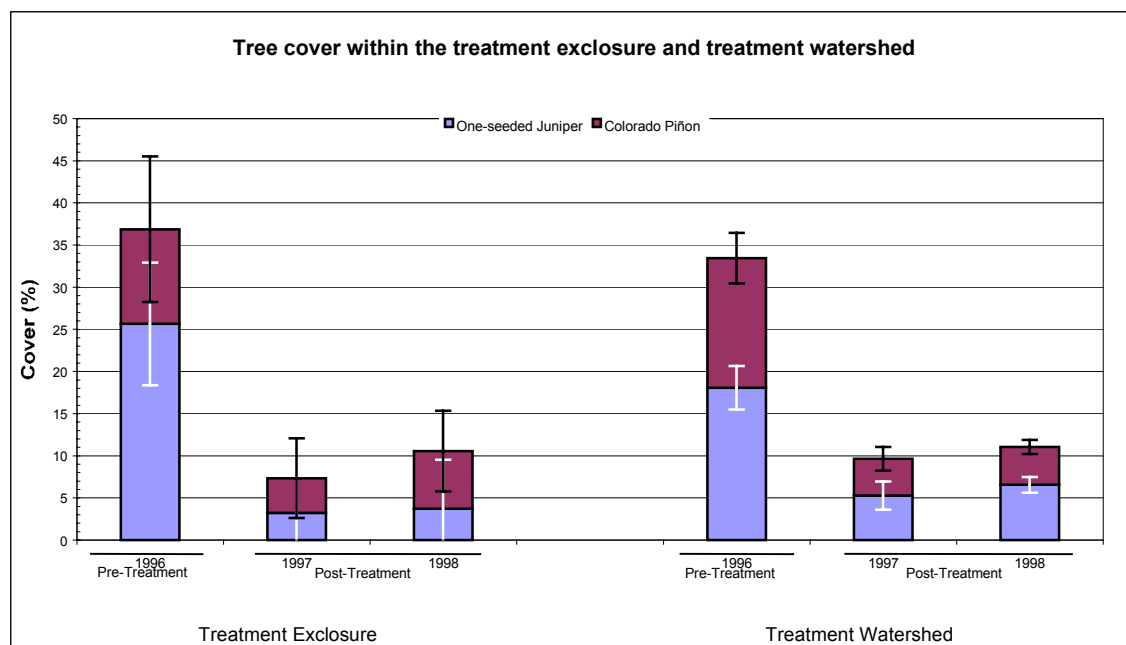


Figure 25b. Changes in tree cover within the treatment enclosure ($n=5$) as compared with the treatment watershed ($n=40$). Error bars are 90% confidence intervals. Overlapping error bars are not significantly different.

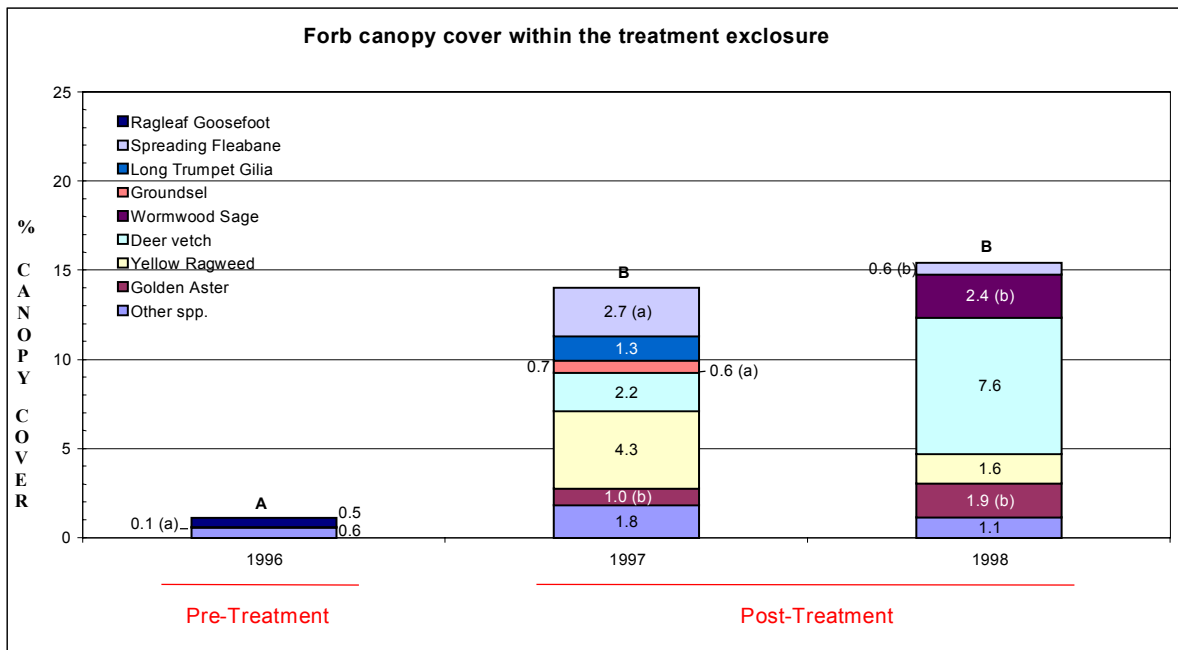


Figure 26a. Forb response to restoration treatment within the treatment enclosure (n=5) from 1996 (pre-treatment) through 1998 (two years post-treatment) with comparisons made across years and among species. Columns with the same upper case letters, and similarly colored bars with the same lower case letter, are not significantly different ($p \leq 0.1$).

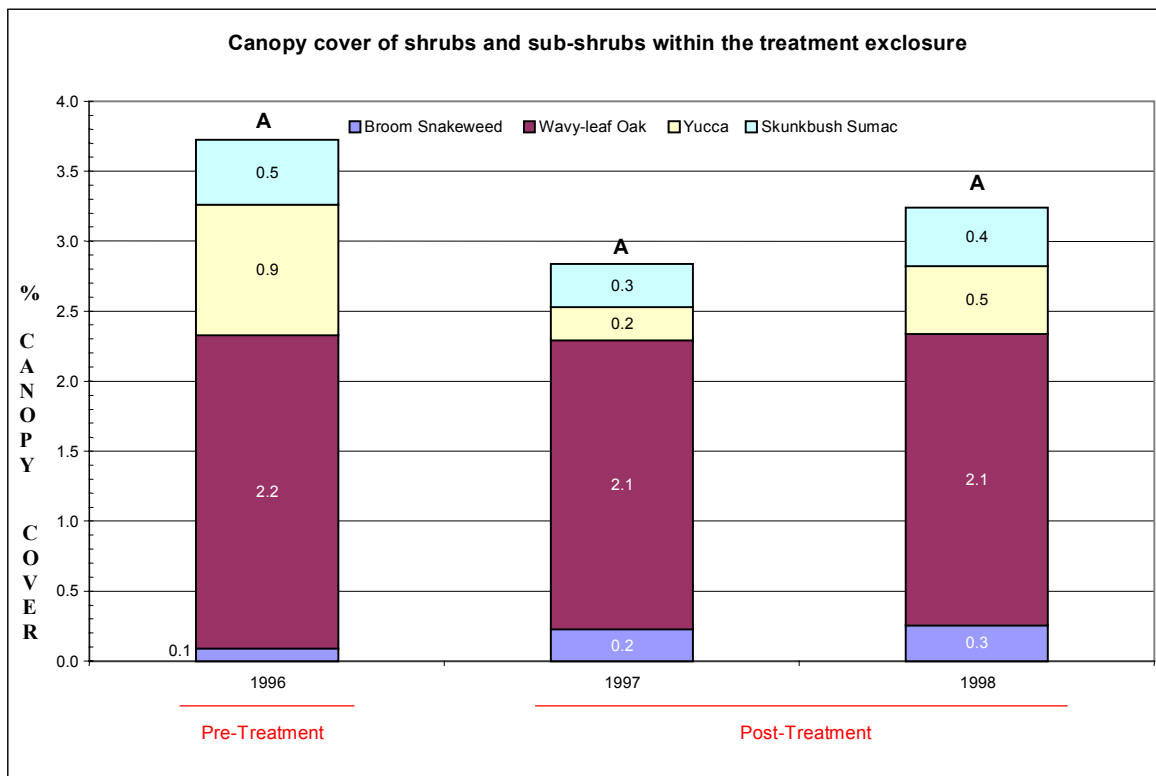


Figure 26b. Shrub and sub-shrub cover within the treatment enclosure (n=5) from 1996 (pre-treatment) through 1998 (two years post-treatment) with comparisons made across years and among species. Columns with the same upper case letters, and similarly colored bars without letters, are not significantly different ($p \leq 0.1$).

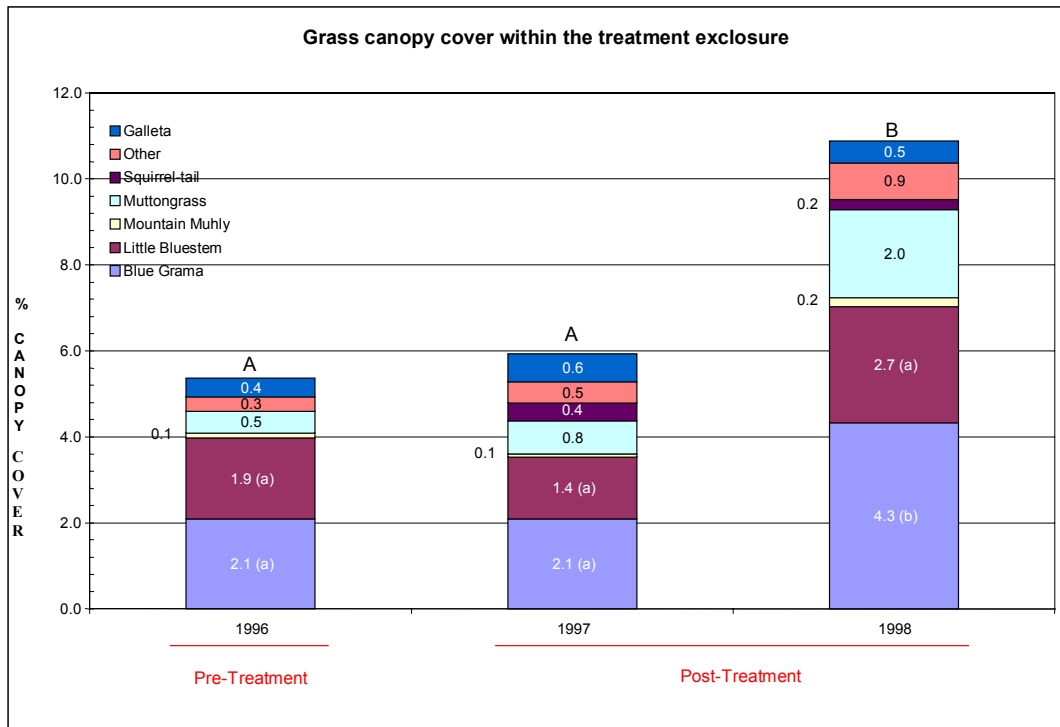


Figure 27a. Grass canopy cover within the treatment enclosure from 1996 (pre-treatment) through 1998 (two years post-treatment). Comparisons are made for total grass canopy cover across years, and among species within and between years. Columns with the same upper case letter, and similarly colored bars with the same lower case letter (or no letter), are not significantly different ($p=0.1$).

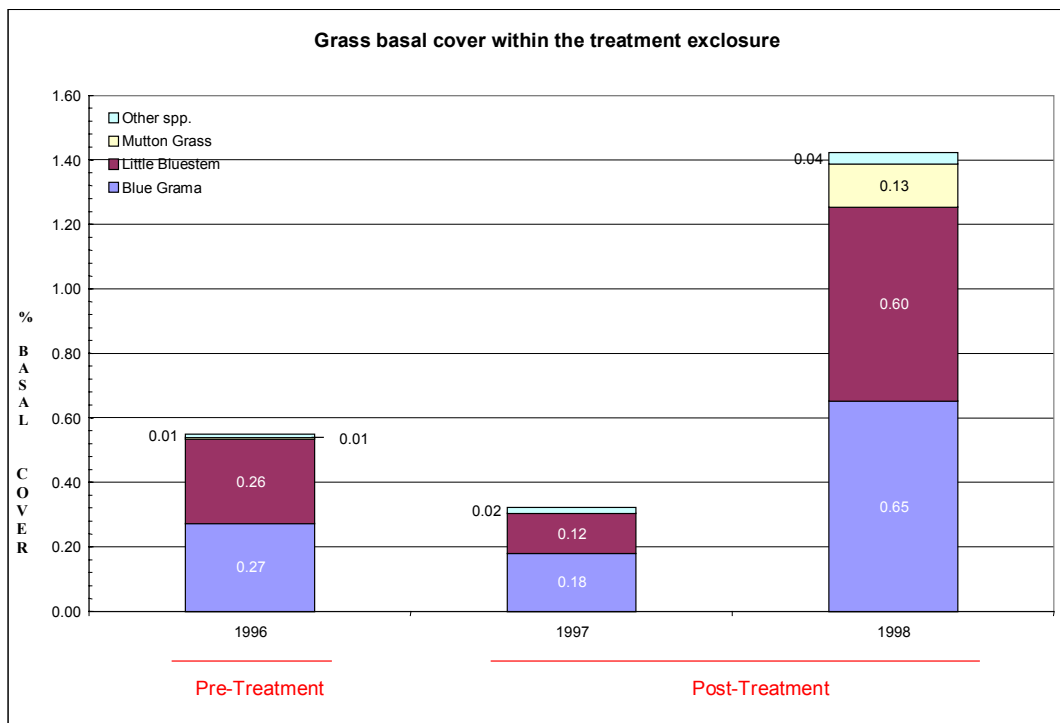


Figure 27b. Grass basal cover within the treatment enclosure from 1996 (pre-treatment) through 1998 (two years post-treatment). Comparisons are made for total basal grass cover across years, and among species within and between years.

Discussion

VEGETATION AND SOIL SURFACE

Plant biomass contributes much of the organic component to soils while aerial plant cover, surface litter, and root biomass effectively mitigate processes that drive soil erosion. Aerial plant cover dissipates the energy of raindrop impact, while litter cover and root biomass facilitate infiltration, minimizing runoff velocities and decreasing sediment transport. (Wood et. al. 1987). The condition of the vegetation has a direct bearing on erosion rates, with degradation of the plant community often viewed as the primary contributor to accelerated erosion. In managing these eroding ecosystems a primary objective is to restore a plant community capable of abating erosive forces and reducing the continuity of bare hillslope patches (Davenport et al., 1998). Ideally this would be achieved through the restoration of the structure and function of the original plant community. In degraded piñon-juniper woodlands erosion is spatially asymmetrical in that erosion occurs primarily in the inter-canopy, whereas soils beneath trees are left relatively intact (Wilcox, 1994).

Reductions in herbaceous plant cover are often the primary initiator of accelerated erosion in previously stable systems. In grasslands and savanna systems, fire, drought and ungulate herbivory are episodic disturbance regimes that often cause ephemeral reductions in plant cover and increased soil erosion. Ecosystems that have evolved under these disturbance regimes are resilient and recover within a short period following disturbance (Milchunas and Lauenroth, 1993). Disturbances that exceed tolerance levels of the of the plant community and the associated landscape can induce longer term structural and functional changes from which the ecosystem cannot rebound (Davenport et al., 1998). The state and transition ecosystem model (Westoby et al. 1989; Friedel, 1991) is often used to describe the failure of these disturbed arid and semiarid ecosystems to return to a pre-disturbance condition. Without intervention on the part of land managers, these ecosystems will continue in transition to a new steady state. The transition to a lower state of productivity is often viewed as undesirable because of the loss of biological function, diversity and/or sustainability. This model may be appropriate in describing Bandelier's degraded piñon-juniper woodlands (Gottfried et al., 1995).

Much of the piñon-juniper woodland area at Bandelier is thought to have been affected by historic livestock grazing, where consumption of the fine fuels (i.e. grasses) resulted in loss of fire process and eventually transformed open woodlands and savannas to closed woodlands with sparse herbaceous cover. This supposition of a significant grassland component as part of the pre-erosional community is in part predicated on the reconstruction of fire chronologies, indicating the presence of communities capable of supporting recurrent surface fire (Allen, 1989). Multiple lines of evidence including age structure, remnant understory communities and associated soils lend additional support to the idea that grass dominance has been replaced by tree dominance in absence of fire disturbance (Julius, 1999). The transition to coarse, from fine grained, distribution of nutrient resources is well documented in arid and semi-arid ecosystems, where it is

thought to be a negative feedback mechanism in the desertification processes preventing re-establishment of herbaceous vegetation (Schlesinger et al., 1990, Davenport et al., 1998). This may be a contributing factor to the continued poor range conditions even after cessation of the initial disturbance (i.e. intensive grazing) nearly sixty years ago.

Alteration of the soil resource by erosion is manifested by reductions of soil organic matter, plant nutrient availability, water storage capacity, and plant rooting volume that culminate in reduced net primary productivity. The loss of organic matter and soil volume has greater implications for coarse textured soils than for fine textured; this is because the relatively low plant available water coarse textured soils provide, as compared with finer textured soils. Consequently, coarse textured soils are more sensitive to a reduction in soil volume than are finer textured soils. Further, since organic matter content serves to increase water-holding capacity, its loss is a greater detriment to coarse textured soils than to finer textured soils.

The degree to which productivity has been impaired is a function of the degree erosion has progressed. Additionally, continued erosion inhibits the establishment of perennial vegetation attempting to re-colonize, due to instability of the rooting substrate. In order for significant plant establishment to occur, soils must be stabilized and in some instances, plant propagules supplemented. The spreading of slash on intercanopy soil serves to stabilize the soil and enhance conditions for plant germination and establishment.

The presence of woody plants on the landscape alters the surrounding environment in such a way as to influence herbaceous plant growth. One way this is manifested is through competition for below ground resources, principally soil moisture. This is believed to be an additional mechanism maintaining poor range conditions and intensifies with increasing tree density (Scholes and Archer, 1997). By thinning the tree overstory, herbaceous vegetation increases in part through reduced water and nutrient competition. Trees also alter their environment in ways that persist beyond their removal. The most widely recognized is the concentration of soil nutrients beneath the canopy of woody plants relative to the adjacent intercanopy. The mechanisms involved are not clearly understood, but there are three primary hypotheses. The first is the acquisition of nutrients beyond the tree canopy via the root system, and subsequent concentration of litter beneath the tree canopy. The second is the capture of atmospheric dust in the tree canopy, which is subsequently flushed by precipitation onto the litter mound. The third is by birds and other animals, which roost or seek food and shelter in or under the canopy where they then defecate and urinate, depositing nitrogen and other nutrients.

The degree to which these processes enrich canopy soil is time dependent. In western Colorado, Barth (1980) found that differences between woodland canopy and intercanopy soils were positively correlated with piñon age. In contrast, Davenport et. al. (1996) found only minor differences between canopy and intercanopy soils in woodlands on the Pajarito Plateau of north central New Mexico. They concluded that the woodland was not old enough to have significantly altered the horizontal distribution of soil resources, other

than for litter accumulation beneath tree canopies. If this is the case, then erosion plays prominently in causing the heterogeneous distribution of nutrients in the local landscape. Thus, soils found beneath trees establishing prior to the initiation of accelerated erosion may be fairly representative of pre-erosional inter-canopy soils. Soils beneath trees establishing after the initiation of accelerated erosion likely experienced some degree of degradation before the canopy developed sufficiently to protect both the remaining profile and contribute additional soil building material (i.e. needle litter drop and wind deposition). Work by Julius (1999) within the treatment watershed documents exponential tree establishment accelerated after 1880 following loss of fire process locally. His data also suggest a general pattern of increasing tree density beginning around 1700; this earlier establishment pattern may reflect both recovery from pre-historic occupation ending ca. 1550 (which is thought to have significantly deforested the local landscape) as well as longer term climatic influences (Allen, in press).

While the tree canopy may have preserved or halted soil erosion beneath their canopies, and litter accumulation may influence what species establish, the removal of the tree canopy should also enhance soil moisture conditions and increase available nutrients, especially nitrogen. Tree removal will reduce interception of precipitation to the former canopy mound, improve moisture and temperature conditions of the canopy soil, promote breakdown of accumulated litter, and increase mineralization and available nitrogen.

SOILS AND EROSION

The current model for soil erosion in degraded piñon-juniper woodlands suggests that intense and sufficiently large rainfall events are responsible for most of the measured sediment movement in the landscape (Wilcox et. al., in press). However, multiple small events may set the stage in part, for fewer, larger events, by mobilizing and staging sediment packets gleaned from inter-channel surfaces, into more centralized channel networks. Sediment aggrades in the channel network, until large events, which generate runoff of sufficient magnitude, transport staged sediment to sinks at lower positions in the landscape.

Because rainfall events can and do vary in terms of intensity and duration over short distances (<50 to 100 m), the differences in soil surface elevation observed between the two watersheds by fall 1998 could have been due, in part, to inequitable distribution of large, intense events. Over longer periods of time however, point rainfall measurements at proximal locations would be expected converge towards the same long-term mean reducing the spatial variation. By continuing to make soil surface elevation observations within these two watersheds, we can take advantage of this temporal convergence in spatial variation of rainfall events to increase confidence in our soil elevation changes. In addition, during the 2000-2001 time period, six automated rain gauges were operated within each of the two watersheds as part of a study designed to characterize sediment production (Hastings et al. In Press). The data for these two years, while highly variable between individual rain gauges, suggests there is no systematic bias towards one or the other watershed (Hastings et. al., In Review) for either event intensity or total precipitation.

The current project was not designed to assess the relative effects of slash mulch versus tree thinning on herbaceous response, although this question has been addressed in separate studies. Anecdotal evidence from our sites suggests slash mulch and thinning effects are synergistic, with herbaceous response most pronounced on sites dominated by exposed soils, where mulching could be expected to have the greatest benefits. Javed (2001) found significantly less runoff and sediment, at two years post-treatment, on plots receiving slash, as compared to plots without slash. Conversely, Brockway et. al. (2002) measured changes in soil surface elevation, and reported no significant difference in soil loss between dispersal of slash, complete slash removal, and a control, at one growing season post-treatment. However, a single year may be inadequate to assess differences between slash treatments, given the variability of precipitation or absence of threshold events to move sediment, as may have been the case for the current study during 1997. In addition to the short timeframe, the Brockway et. al. (2002) data are also limited by the high spatial variability inherent with the soil erosion bridge method. While this approach provides precise, high resolution at individual sample points, it necessarily requires extremely large sample sizes to provide representative data or yield significant results at landscape scales. Finally, Brockway et. al. (2002) note that cattle were introduced back into the site during the first year post-treatment, and it could be expected the animals spent relatively more time in those portions of the study area that had an herbaceous response to treatment, potentially confounding their erosion results.

While an effect from slash and herbaceous cover was not apparent on inter-channel surfaces within the treatment watershed at two years post-treatment, it was reflected in the channel system. The inter-channel surfaces serve as sources for sediment transported to and through the channel system. Some portion of this sediment may be temporarily stored and later transported away by a subsequent event or series of events. The dynamics of the sediment aggrading and degrading within the channel system likely magnifies the erosional processes occurring across the much larger inter-channel surfaces. In 1998, aggrading sediment in the channels of the treatment watershed, and conversely, scouring and incising in channels of the control watershed, were responsible for the significant differences observed between the two watersheds.

From a fire effects perspective, slash dispersal onto intercanopy soils may minimize the risk of creating sterile and potentially hydrophobic soils on canopy litter mounds where slash might otherwise be concentrated. From a fire management perspective, dispersing slash increases surface fuel continuity and risk of fire spread across the treated area, but this practice can also facilitate eventual fire maintenance of mechanically restored sites. Research is currently underway to assess the use of prescribed fire to maintain mechanically restored woodland savannas (Jacobs and Gatewood, 2002).

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